M.L. 2013 Project Abstract
For the Period Ending June 30, 2016
PROJECT TITLE: Sustaining Lakes in a Changing Environment - Phase II
PROJECT MANAGER: Jeffrey R. Reed, Research Scientist II
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FUNDING SOURCE: Environment and Natural Resources Trust Fund
LEGAL CITATION: M.L. 2013, Chp. 52, Sec. 2, Subd. 05a
APPROPRIATION AMOUNT: $1,200,000

Overall Project Outcome and Results

Phase 2 of the Sentinel Lakes Long-Term Monitoring Program comprised a wide variety of monitoring and research activities on the 25 Sentinel Lakes selected to provide representation of Minnesota's major lake-types. During 2013-2016, the Sentinel Lakes Program continued to integrate the activities of key, collaborative agencies and partners (e.g. DNR, MPCA, USGS, and universities) which focus on determining the effects of large-scale ecological stressors (e.g., eutrophication, invasive species, and climate changes) on lake ecosystems. Highlights include:

- Detailed summaries of fish and aquatic plant sampling activities were prepared to guide future data analyses and monitoring activities.
- High-resolution water column temperature and dissolved oxygen tracking reveals progression of oxythermal-habitat changes for important fish species in Elk Lake.
- Continuation of specialized sampling of Cisco population in 3 Sentinel Lakes further enhanced our understanding of the relationships between Cisco and climate change and the presence of invasive species.
- Evaluations of biological indicators of lake status including pupal skins of aquatic midges, and White Sucker biology. Results indicate aquatic fly composition reflects lake nutrient status, while assessing White Sucker biology proved difficult. At least 141 species of midge were detected.
- A detailed report of phytoplankton and zooplankton composition, seasonal cycling, and interactions in 13 Sentinel Lakes was completed.
- Food web research conducted to understand impacts of zebra mussels finds that lake biota (insects and fish) shifted to alternative food sources.
- Long-term water quality and baseline aquatic plant surveys at Shaokotan Lake detected a major shift in 2015 from algae-dominance to clear water, aquatic plant-dominance due to watershed restoration and BMP implementation.
- New biophysical lake models were developed Pearl and Madison lakes, while previous models were used to simulate impacts of future climate conditions in Elk and Trout lakes. Surface water temperatures increase dramatically under future climate scenarios and oxygen depletion dynamics differed between lakes.

Project Results Use and Dissemination

The information gathered during the second phase of Sentinel Lakes sampling continues to provide insights useful to lake managers. The continued ability to collect water quality, zooplankton, fisheries, aquatic vegetation, and land use data over consecutive years from a set suite of lakes has added to the strong foundation of long-term monitoring that was established during the first phase of the project (2009 to 2013). Refining metrics and more fully developing our understanding of how they react to specific ecological stressors will continue to assist managers faced with developing management strategies and practices in lakes. The value to fisheries and lake managers is perhaps most evident in the Department of Natural Resource's commitment to hire and fund a full-time Sentinel Lakes coordinator position. That internally funded position was filled in May of 2016 and will provide project continuity going forward.

As was the case in Phase 1 we again included partner institutions with different areas of expertise, thus the project was able to gain valuable insights into 1) how lake systems in agricultural zones function (USGS), 2) how Chironomid (midge) populations may serve as important indicators of trophic status (University of Minnesota), and 3) how stable isotope analysis can lead to fuller understanding of the effects invasive species such as zebra mussels have on lake food webs (University of St. Thomas). The techniques developed by partners as well as their final results should provide valuable tools and information for some time to come.

Continued, consecutive, sampling of Cisco populations has not only furthered our understanding of their population dynamics and their vulnerability to climate change and invasive species but has also added to the development of specific methods for monitoring this important climate- and land-use sensitive species in lakes across the state.

Finally, the project has become an excellent training tool for undergraduates, graduate students, and professionals. More than a dozen undergraduates have been able to gain valuable field experience and mentoring from research staff over the course of the project. The project has also served as a valuable entry point into fisheries for early-career professionals.

Much of the focus of disseminating information gathered during the project has been focused on a scientific audience but with an emphasis on making that information relevant to lake and fisheries managers. To that end, all collaborators past and present were invited to attend and present their findings at a Sentinel Lakes Summit which was held in Brainerd in 2015. Over 50 managers from DNR and PCA attended the event. A similar event is being planned for 2017. Additionally, a number of manuscripts covering a wide variety of topics are currently being prepared for submission to peer-reviewed publications. Already a number of presentations have been made at national, regional, and state-level professional meetings.

For general audiences the Department of Natural Resources maintains a series of Sentinel Lakes-related pages on their website (http://www.dnr.state.mn.us/fisheries/slice/index.html) and the Pollution Control Agency hosts Sentinel Lake Assessment reports on their website (https://www.pca.state.mn.us/water/sentinel-lakes).

Trust Fund 2013 Work Program Final Report

Date of Status Update: September 8, 2016

Date of Next Status Update Report: Final Report

Date of Work Plan Approval: June 25, 2013

Project Completion Date: June 30, 2016

PROJECT TITLE: Sustaining Lakes in a Changing Environment (SLICE): Phase 2

Project Manager: Jeffrey R. Reed, Research Scientist II

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Location: Statewide. See map in Section IX.

Total ENRTF Project Budget:	**ENRTF Appropriation:**	**$1,200,000**
	Amount Spent:	**$1,197,454**
	Balance:	**$2,546**

Legal Citation: M.L. 2013, Chp. 52, Sec. 2, Subd. 05a

Appropriation Language:
$1,200,000 the first year is from the trust fund to the commissioner of natural resources in cooperation with the United States Geological Survey, the University of Minnesota, and the University of St. Thomas to continue development and implementation of monitoring, modeling, and reporting protocols for Minnesota lakes to be used in water and fisheries management. This appropriation is available until June 30, 2016, by which time the project must be completed and final products delivered.

I. PROJECT TITLE: Sustaining Lakes in a Changing Environment (SLICE): Phase 2

II. PROJECT STATEMENT:

As Minnesota's population grows, increased demands are being placed on our resources. Hunting and fishing related activities fuel $3.6 billion in annual expenditures and a vibrant Minnesota economy demands on effective and efficient lake habitat and fisheries conservation. Thanks to the initial investments from the ENTF, multiple partners' in-kind contributions, and motivated citizen volunteers, DNR Fisheries successfully launched and is near completion of a 4-yr pilot effort in 24 sentinel lakes (SLICE Phase 1). Sentinel systems are specific ecosystems (in this case lakes) chosen for focused monitoring in order to better capture important changes or trends through time, and to give us the information to better understand the mechanisms to explain those changes. Thus, in Phase 1, we have established a suite of 24 lakes across Minnesota that capture the full breadth of the basic conditions (e.g. fertility, climate) that shape our lake resources, and in large part determine the goods and services (e.g. fishing and water recreation) that our lakes provide for our citizenry. Phase 1 identified baseline conditions in a wide variety of Minnesota lake types, their initial responses to various environmental stressors, and outlined some expectations for future conditions given various changing scenarios. Phase 1 also fortuitously (and perhaps unfortunately) gave us the opportunity to set up a rigorous system to understand pending impacts from zebra mussels following the invasion of this pest in Lake Carlos, near Alexandria, in 2009. Phase 1 will be completed in June 2013.

Phase 2 (2013 - 2016), proposed here, will take lessons learned from the first phase to develop and implement rigorous monitoring, modeling, and reporting protocols that will deliver timely information on lake trends, reduce uncertainty about potential causes, and result in more precise conservation approaches. Phase 2 also includes applying lake models to predict ecosystem impacts of major environmental and ecological stressors (e.g., possible future invasive species introductions, changing land use, and climate changes) in six Tier 1 sentinel lakes. In addition to continuing to develop and test our biophysical models of oxy-thermal habitat in Elk, Carlos, and Trout lakes, we propose new modeling efforts focused on the state's shallow, agriculturally-impacted lakes in Phase 2. Thus, we propose developing three new lake models (Shaokotan, Madison, and Pearl lakes) to enhance understanding of nutrient loading from watershed sources, in-lake nutrient and food web dynamics, and resulting fish habitat conditions related to current and changing land use practices, dynamic hydrological inputs, and physical processes such as evapotranspiration and wind mixing of water column and underlying sediments. In order to implement proactive lake conservation measures, we must acquire information about baseline habitat conditions (the past), long-term changes to that baseline (the present), and models that forecast the risk of various impairments (the future).

Amendment Request (4/15/2016)

This amendment request is to reallocate funds from the Budget Items (Under Equipment/Tools/Supplies) (total $10,193) and Budget Item Travel Expenses (total $15,567) to Budget Item Personnel – Budget Item Long-term Monitoring Fisheries Specialists ($17,567) and Student Interns ($8,193) totaling $25,760. The shift in equipment costs reflects savings from DNR supplied equipment (e.g., personal gear, water level gauges, transducer calibration, and invertebrate sampling equipment) and the termination of the contract with Contour Innovations due to completion of work ahead of schedule. Furthermore, travel savings were realized when staffing was reduced from the funding of three biologists to two. The total reallocation requested is $25,760. The proposed amendment will enable staff, including interns, to complete and continue data collection on fish populations, aquatic plant abundance, and temperature and oxygen monitoring on the Sentinel Lakes through June of 2016 as well as providing assistance with the final report preparation.

Amendment Approved by LCCMR 5-6-2016.

Amendment Request (10/30/2015)
This amendment request is to reallocate funds from Budget Item Student Intern Salaries (Under Personnel) in the amount of $34,943 to Budget Item Long-term Monitoring Fisheries Specialists. The proposed amendment will allow the Long-Term Monitoring Specialists to continue monitoring work, data analysis, and associated research as well as assist with the preparation of the final report which is due to the LCCMR in June of 2016.

Amendment approved by the LCCMR 11-6-2015.

Amendment Request (10/30/2014)
This amendment request is to reallocate funds from Budget Item Dissolved Oxygen Sensors ($11,000) to Misc. Survey Equipment and Repairs ($16,000) and a proposed Professional/Technical/Service Contract with Dr. Brian Gelder of Iowa State University for automated DEM Hydro-modification within 3 Sentinel Lake watersheds. Prior to the implementation of Phase II of SLICE, the Section of Fisheries purchased these sensors and they have been deployed for two field seasons. We are requesting that money earmarked for those purchases be reallocated for use in survey equipment and repairs in the amount of $4,000 and $3,000 to the P/T/S contract with Dr. Gelder. This contract work will supplement work that has been completed for Activity 1, Outcome 2 (‘*Inventory watershed land cover and uses (incl. BMP’s) & drainage features and archive in DNR’s GIS database’*). This work has been updated in the October 2014 Progress Report. Specifically, we are pursuing output from Dr. Gelder’s automated process for hydro-modification of the LiDAR-derived DEM for the Shaokotan, Madison and Pearl lake watersheds. We envision that these final, hydro-modified DEMs will be used in watershed models for these lakes (estimation of sediment and nutrient loads, and lake water budgets and thermal conditions given different forecasting scenarios). Specific deliverables from the Automated DEM Hydro-modification work include: 1) pit-filled input DEM (one-cell sinks removed), 2) enforced DEM (flow obstructions, as defined by ISU enforcement process, will be removed), 3) depression geodatabase (a database of all flow obstructions and their characteristics used in the enforcement program), 4) enforcement line geodatabase (a database of all enforcements applied to the input DEM, which can be modified according to future project needs and re-applied to the input DEM to create a new enforced DEM), and 5) consultation on the results of the model run.

Amendment approved by LCCMR on 12-17-14 with the requirements that no funds will be used for Iowa State University overhead.

Amendment Request (4/15/2014)
This amendment request is to replace Dr. Donald Pereira as Project Manager with Jeffrey Reed as Project Manager. Dr. Pereira left the Fisheries Research Unit to become Chief of the Section of Fisheries effective November 20th, 2013. Due to the change in job responsibilities Dr. Pereira will no longer be able to devote time to administering this project. This change will enable more efficient communication between LCCMR staff and Sentinel Lakes Program staff. The following changes are requested for administration of the grant:

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Amendment Request 04/15/2014:
This amendment request is to reallocate funds from 'Travel expenses in Minnesota' to the contract with the University of St. Thomas (Dr. Kyle Zimmer). This is needed to cover travel expenses (lodging and meal expenses) incurred by the University of St. Thomas researchers and student interns associated with field sampling at Carlos, Ida, and Elk lakes. Funds totaling $3,675 are reallocated from 'Travel expenses in Minnesota' to Professional/Technical contracts to University of St. Thomas (Dr. Kyle Zimmer).

Amendment Approved by the LCCMR April 21, 2014

Amendment Request 09/06/2013:
This amendment request is to reallocate funds from reduced Direct and Necessary Support Services costs and from reduced lake monitoring equipment costs anticipated to be incurred during the grant period to increase funding to the contract with the University of St. Thomas (Dr. Kyle Zimmer). Lower equipment costs are from DNR pre-purchasing equipment necessary for the lake modeling aspect of the monitoring program. This enabled us to get monitoring hardware in the lakes at the beginning of the growing season, thus allowing a full year of data collection, and was necessary because the appropriation of the current grant did not occur until July 1. Equipment pre-purchased with non-LCCMR end-of-biennium Game and Fish funds totaled $24,783.41.

Funds totaling $10,000 are reallocated from Direct and Necessary Support Services ($8,654 in reduced DN costs) and Equipment ($1,346 from Water Level Pressure Transducers) to Professional/Technical contracts to University of St. Thomas (Dr. Kyle Zimmer) to incorporate new, technical advances (^{34}S and ^{2}H isotopes) and to add one non-sentinel lake (Lake Ida) to strengthen the basic experimental design as it relates to understanding zebra mussel impacts in Lake Carlos. Lake Ida is in the same chain of lakes and watershed as Lake Carlos, but zebra mussels have not yet been observed there. Zebra mussels are likely to be detected there in the future, thus this lake provides a critical point of comparison and future change. Direct and necessary costs totaling $61,614 are for Activity 1.
Amendment Approved: 09/11/2013

Project Status as of 10/30/2013:
Sampling resumed in 2013 on several of the Sentinel Lakes included in the SLICE long-term monitoring program following a one year break in sampling (none in 2012) after the completion of Phase 1 (2008-2011). We have now implemented a tiered sampling approach with 8 Tier 1 lakes receiving more frequent sampling (annual or biannually), and 16 Tier 2 lakes receiving reduced monitoring frequency (minimum of once every 5 years). Among the most notable program accomplishments in this first reporting period was the hiring of three long-term monitoring biologists stationed at the Hutchinson, Glenwood and Tower area fisheries offices to help coordinate and conduct program monitoring and research work. The hiring of the long-term monitoring biologists allowed us to complete fisheries population assessments on 13 of the sentinel lakes, install water monitoring equipment on 4 of the

lakes, and begin research that will enable us to further refine our fisheries sampling program and identify the most promising metrics of change within lakes.

Preceding the 2013 field season, six subject expert committees were formed to make comprehensive long-term sampling recommendations for Phase 2 of the long-term monitoring program based on lessons learned from Phase 1, peer-reviewed literature, other past experiences and previous data, and group expert opinion. Subject expert committees included Watersheds, Water Quality, Physical Structure (including aquatic plants and woody habitat), Zooplankton, Macroinvertebrates, and Fish Communities. Each expert group developed and delivered proposals for program consideration and implementation. The fish community committee recommended the consideration of adding a Tier 2 status lake with a stable resident lake trout population. The addition of a lake with a lake trout population will better enable trend detection across lakes containing this temperature-sensitive species. To this end, we added Greenwood Lake (near Grand Marais) as a 25th sentinel lake. This brings the number of sentinel lakes with lake trout to two, and provides the program with a lake trout population that is thought to be more stable compared to the population resident in Trout Lake that is thought to be declining. Project coordinators, the three long-term monitoring biologists, and project partners have begun implementing and refining each of the subject committee plans. Some refinements to the plans were made in 2013 in an attempt to balance collection of key lake parameters with a reasonable and achievable level of field effort that can be sustained in the future. As of this first reporting period, final field data is being or was just recently collected (fall water chemistry and fish sampling), water chemistry and zooplankton samples are being processed, and data entry is currently in progress with the first complete data sets yet to be assembled.

Project Status as of 4/15/2014:
Work continues to progress for both Activities 1 and 2. Among the notable accomplishments for Activity 1, fall fish sampling surveys were conducted in several of the Sentinel Lakes, age and growth information was gathered and analyzed from bony structures collected from target fish species, and 10-year sampling plans and fish sampling protocols have been decided for the 25 Sentinel Lakes. LTM Biologists have also been working with DNR Division of Ecological and Water Resources (EWR) scientists to develop an aquatic plant sampling program for the Sentinel Lakes, and most sampling protocols have now been decided. All zooplankton samples collected from the Sentinel Lakes during 2013 have been identified, counted, and summarized. Additionally, fish stomachs were collected with the help of Trout Lake Resort owners and Grand Marais Area Fisheries staff to gather baseline data on feeding habits prior to the establishment of a spiny waterflea population in Trout Lake. We have now hired a GIS Research Analyst to develop watershed land cover, drainage feature, and other GIS data sets in support of the overall project. The stable isotope project (Dr. Zimmer, Univ. of St. Thomas) has focused on processing samples collected during 2013 for analysis of stable isotopes of hydrogen, carbon, nitrogen, and sulfur. To date 90% of the samples have been processed and the first round of samples has been submitted to the UC-Davis Stable Isotope Lab for analysis. The chironomid pupal exuvia project (Dr. Ferrington, Univ. of Minnesota) is also progressing, with a M.S. student now recruited to conduct the work, and fieldwork planning is now well underway. Activity 2, to build or adapt biophysical lake ecosystem models (Dr. Kiesling, USGS) is progressing according to schedule.   Two recent meetings between USGS and DNR staff have been held, to review 2013 data, and to define and coordinate open water fieldwork during summer 2014. LTM biologists have continued outreach and dissemination efforts as well, presenting results at scientific meetings as well as to groups of internal and external stakeholders.

Project Status as of 10/30/2014:

Field sampling and data collection were the primary foci during this reporting period. Fishery populations assessments (n=11), IBI sampling (n=10) and pelagic fish sampling (n=3) were completed during the 2014 field season. 2013 data have been entered into the DNR's lake survey database. Water quality parameters and zooplankton were sampled monthly from each of the Tier 1 lakes as well as 3 Tier 2 lakes. Remote sensing equipment, including water level transducers, water temperature loggers, and weather stations, were deployed in April and May.

A GIS Research Analyst completed work on the compilation of data sets on watershed land cover, drainage features as well as other GIS data that will support the overall project in the long-term. Several of the results from this work are detailed in the outcomes section of this progress report.

Work continued on the collection of samples for stable isotope analyses and UST researchers are summarizing 2013 data. We expect to be able to present some results in the next progress report. Similarly, the chironomid pupal exuvia project work being done through the UofM is also progressing well. UofM researchers note that they have identified over 50 genera from 11 of the Sentinel Lakes.

Activity 2 continues to collect data (including inflow/outflows, water temperature, and weather) for the compilation of biophysical lake ecosystem models. USGS and DNR have collaborated on these efforts.

Project Status as of 4/15/2015:
Long-term monitoring biologists continued work on fisheries-related aspects of Activity 1, including completion reports for population assessments conducted in 2014. LTM biologists have also undertaken research on White Sucker population dynamics and aging as well as aging young of the year Largemouth Bass and Smallmouth Bass. Data from 2014 has been entered into DNR databases. Remote sensing equipment is ready to be deployed as the ice leaves the lakes. Stable isotope work (Dr. Kyle Zimmer, University of St. Thomas) continues on schedule and some results are presented in this progress report. Similarly, chironomid pupal research (Dr. Len Ferrington, University of Minnesota) continues on schedule and samples continue to be collected and processed.

A contract was signed with Dr. Brian Gelder, Iowa State University, to complete an automated process for hydro-modification of the LiDAR-derived DEM for the Shaokotan, Madison and Pearl lake watersheds. This work is scheduled to be completed by May of 2015. We expect to present results in the next progress report.

Over 70 people attended the Sentinel Lakes Summit in early February in Brainerd. The event featured presentations from past and present collaborators and their work on Sentinel Lakes. A full program listing as well as select presentations (in PowerPoint format) can be obtained by contacting the project manager.

Project Status as of 10/30/2015:
Spring and summer fisheries assessments were completed on schedule and biologists are currently working on survey write-ups and processing aging structures. Data generated from these assessments are housed in DNR data bases. White Sucker population dynamic research has been completed and a summary of that work will be included in final report submitted to the Commission in June of 2016. Biologists also completed vegetation (submerged and emergent) surveys and those data are currently being analyzed. In preparation for ice cover and winter, we are in the process of removing remote sensing equipment. Stable isotope work is continuing on schedule and full results will be presented in

the final report. Additionally, chironomid pupal research is nearing completion and final results will be presented in the final report as well.

The work by Dr. Brian Gelder, Iowa State University, was completed for the Shaokotan, Madison, and Pearl lake watersheds.

Project Status as of 4/15/2016: All parties involved with the project are currently wrapping up their respective contributions for the final report.

Overall Project Outcome and Results:
Phase 2 of the Sentinel Lakes Long-Term Monitoring Program comprised a wide variety of monitoring and research activities on the 25 Sentinel Lakes selected to provide representation of Minnesota's major lake-types. During 2013-2016, the Sentinel Lakes Program continued to integrate the activities of key, collaborative agencies and partners (e.g. DNR, MPCA, USGS, and universities) which focus on determining the effects of large-scale ecological stressors (e.g., eutrophication, invasive species, and climate changes) on lake ecosystems. Highlights include:

- Detailed summaries of fish and aquatic plant sampling activities were prepared to guide future data analyses and monitoring activities.
- High-resolution water column temperature and dissolved oxygen tracking reveals progression of oxythermal-habitat changes for important fish species in Elk Lake.
- Continuation of specialized sampling of Cisco population in 3 Sentinel Lakes further enhanced our understanding of the relationships between Cisco and climate change and the presence of invasive species.
- Evaluations of biological indicators of lake status including pupal skins of aquatic midges, and White Sucker biology. Results indicate aquatic fly composition reflects lake nutrient status, while assessing White Sucker biology proved difficult. At least 141 species of midge were detected.
- A detailed report of phytoplankton and zooplankton composition, seasonal cycling, and interactions in 13 Sentinel Lakes was completed.
- Food web research conducted to understand impacts of zebra mussels finds that lake biota (insects and fish) shifted to alternative food sources.
- Long-term water quality and baseline aquatic plant surveys at Shaokotan Lake detected a major shift in 2015 from algae-dominance to clear water, aquatic plant-dominance due to watershed restoration and BMP implementation.
- New biophysical lake models were developed Pearl and Madison lakes, while previous models were used to simulate impacts of future climate conditions in Elk and Trout lakes. Surface water temperatures increase dramatically under future climate scenarios and oxygen depletion dynamics differed between lakes.

IV. PROJECT ACTIVITIES AND OUTCOMES:

ACTIVITY 1: Monitoring a comprehensive suite of important lake and watershed indicators in 25 Sentinel Lakes to gage status and trends of lake health (see map in Section IX).

Description:

Several scientific and programmatic lessons learned from the Phase 1 pilot will be applied to this activity. Three dedicated field staff will provide regional expertise regarding status and trends of important water and fisheries resources in each of the State's four major land types. A programmatic adaptation proposed in Phase 2 is a tiered classification of sentinel lakes with comprehensive, intensive monitoring in 8 Tier 1 Sentinel Lakes (see also activity 2) and reduced monitoring schedules in the remaining 16 sentinel lakes. Thus all lakes will receive regularly scheduled monitoring of the following parameters: Water quality (according to established MPCA protocols), Fisheries population assessments (primarily game fish), IBI surveys (index of biotic integrity) of all fish (thus including non-game species) and aquatic rooted plants, zooplankton sampling, sampling of deepwater and nearshore macroinvertebrates, and continuous water temperature monitoring. The specific frequency for conducting the individual monitoring components will be determined from rigorous statistical analysis soon to be completed as a key part of the Phase 1 project. Automated sensors will also be installed in the eight Tier 1 lakes. We will measure naturally-occurring stable isotope abundances in lake biota to track food web and ecosystem responses to expanding zebra mussel populations in Lake Carlos, and to characterize baseline food web linkages, energy flow, and ecological niches of piscivore species in Elk Lake. Additionally, chironomid pupal skins will be sampled to assess chironomid communities in each of the Tier 1 sentinel lakes, with the goal of developing species or community indicators that reflect lake trophic and thermal conditions. Chironomids are sensitive indicators of lake conditions, and probably respond rapidly to lake degradation or improvement, thus will be receiving increased research attention in Phase 2.

Summary Budget Information for Activity 1:	**ENRTF Budget:**	**$ 992,456**
	Amount Spent:	**$ 989,910**
	Balance:	**$ 2,546**

Activity Completion Date: June 30, 2016

Outcome	**Completion Date**	**Budget**
1. *Conduct standard lake surveys; data QA/QC & database management*	*Ongoing*	$583,265
2. *Inventory watershed land cover & uses (incl. BMP's) & drainage features and archive in DNR's GIS database*	*June 30, 2016*	$18,480
3. *Establish automated temperature sensors in 8 Tier 1 Lakes*	*May 2014*	$37,454
4. *Use carbon and nitrogen stable isotopes to examine food web responses to expanding zebra mussel populations in Lake Carlos and to characterize the Elk Lake food web*	*June 30, 2016*	$65,312
5. *Relate chironomid (pupal exuvia) communities to sentinel lake trophic and thermal characteristics*	*June 30, 2016*	$82,633
6. *Reports on status and trends for 4 major landtypes, with stressors or BMP influence*	*Annual to June 2016*	$205,312

Activity Status as of 10/30/2013:
Outcome 1 – Fisheries surveys were conducted on the following lakes in 2013: Artichoke, Bear Head, Belle, Carlos, Carrie, Cedar, Elk, Madison, Pearl, Peltier, Shaokotan, Tait, Ten Mile, Trout, and Greenwood. Surveys to calculate an Index of Biotic Integrity (IBI) score were conducted on Bear Head, Belle, Carlos, Carrie, Elk, Madison, Pearl, Shaokotan, Tait, Trout, and Greenwood lakes. Additionally, vertical gill netting surveys were conducted to assess the coldwater fisheries of Carlos, Elk, and Ten Mile lakes.

Monthly water quality and zooplankton samples were collected from each of the Tier 1 SLICE lakes, and the following six Tier 2 lakes: Tait, South Center, Portage, St. James, Belle, and Carrie. Deepwater macroinvertebrates were sampled from the following seven SLICE lakes: Shaokotan, Madison, Pearl, Ten Mile, Bear Head, Trout, and Greenwood. No mid-summer point intercept plant surveys were conducted during 2013, but spring to early summer curly-leaf pondweed surveys were conducted on Madison, St. James, Carrie, Belle, and Pearl lakes. Division of Ecological and Water Resources staff assessed the human impacts on shorelines, using their Score the Shore methodology on Belle, Carlos, Cedar, Elk, Hill, Pearl, Portage, Red Sand, South Twin, Tait, and Trout lakes. As of this progress report data collected during these surveys is still being entered into appropriate databases for subsequent summarization and analysis.

Outcome 2 – We will be developing an internship posting, conducting interviews, and hiring an intern to begin watershed land cover, drainage feature, and other GIS work beginning in early 2014. The intern will be stationed and supervised out of the Bemidji Area Fisheries office.

Outcome 3 – We purchased and deployed automated water temperature sensors in the 8 Tier 1 sentinel lakes. In each of the 6 biophysical lake system model lakes (Activity 2) we have purchased and/or deployed the following additional remote monitoring equipment: 1) pressure transducers that continuously record lake level, 2) weather stations that continuously record and store air temperature, relative humidity, and wind speed and direction information, and 3) in Elk Lake only, an initial set of 6 combined dissolved oxygen/temperature loggers that continuously record and store these two variables. Initial tests using the dissolved oxygen/temperature loggers were promising, and we will likely expand our use of these loggers on the six Activity 2 lakes in future years.

Outcome 4 – A proposal guiding the stable isotope research has been developed and submitted to the DNR. The proposal was expanded to incorporate new, technical advances (^{34}S and ^{2}H isotopes) and to add one non-sentinel lake (Lake Ida) to strengthen the basic experimental design as it relates to understanding zebra mussel impacts in Lake Carlos. Lake Ida is in the same watershed as Lake Carlos, but zebra mussels have not yet been observed there. Zebra mussels are likely to be detected there in the future, thus this lake provides a critical point of comparison for future change. Contracting is currently in progress and expected to be completed soon. Preliminary biological samples, representing producers at the base of the food web and consumers at all trophic levels, were collected at Elk and Carlos lakes in July 2013. Once the contract with University of St. Thomas is executed, laboratory work and sample preparation will continue, followed by samples being sent to an external lab for determination of isotopic composition.

Outcome 5 – A research proposal for work relating chironomid pupal exuvia work to lake conditions has been developed and submitted to DNR. Contracting is currently in progress, and is nearing completion. The next important step will be recruiting and selecting a graduate student to conduct the work and produce a M.S. Thesis from the study.

Outcome 6 – Because 2013 data sets are still currently in development we cannot offer any presentation of trends as of this update.

Activity Status as of 4/15/2014:
Outcome 1 – In October of 2013, boat electrofishing surveys were conducted on Bear Head, Carlos, Elk, Madison and Pearl Lakes. These surveys targeted young-of-the-year Largemouth Bass, Yellow Perch and Bluegill to assess the effectiveness of this gear for evaluating pre-winter lengths of these three species.

Pelagic fish communities, consisting mainly of Cisco and Yellow Perch were assessed using vertical gill nets (VGN) and hydroacoustics on Carlos, Elk, and Ten Mile lakes. Analyses of age and growth information as well as relative abundance of target species is currently being completed and will be entered into the DNR Section of Fisheries lake survey database. Data will be available later in 2014.

During the last 6 months long-term monitoring (LTM) biologists were actively engaged with Section of Fisheries managers and biologists in developing 10-year sampling schedules for each of the 25 sentinel lakes. Sampling plans for 2014 include fish population assessments on Carlos, Echo, Elephant, Madison, Pearl, Shaokotan, South Twin, and Ten Mile lakes. Additionally, pelagic fish assessments (VGN and hydroacoustics) will be conducted on Carlos, Elk, and Ten Mile lakes. Bass sampling (electrofishing) and targeted young-of-the-year sampling will be conducted on Bear Head, Elk, Carlos, Madison, Pearl, Shaokotan, Ten Mile, and Trout.

Furthermore, LTM biologists have also been involved with implementing the sampling protocols recommended by the Sentinel Lakes Fish Advisory Committee. These recommendations reflect an effort to identify important indicator species as well as the development of a sustainable long-term monitoring effort. Target species identified by the committee for long-term monitoring consists of primary (Yellow Perch, Cisco, Largemouth Bass and Smallmouth Bass) and secondary (Bluegill, White Sucker, Northern Pike and Lake Trout) species. Sampling efforts will focus on these species.

Age and age at maturity were determined to be important indicators of change from environmental stressors. To obtain adequate representation for all age classes present, the number of Bluegill from which aging structures, sex, and maturity data will be collected was increased to 10 fish/cm length group. Similarly, on Ten Mile Lake, the number of Cisco from which aging structures, sex, and maturity data will be collected was increased to 20 fish/cm length group. Because age estimates from Northern Pike structures are unreliable, we will no longer collect structures for aging; however, scales will be collected for archival purposes.

Fall daytime electrofishing will be used to collect samples of Bluegill, Yellow Perch, and black bass (Largemouth Bass and Smallmouth Bass), which have not recruited to trap or gill nets. Otoliths, scales, and sex/maturity data will be collected from 10 fish/cm length group for Bluegill and Yellow Perch and from 5 fish/species/cm group for black bass (<150 mm). This data will be used, along with data from fish caught in trap and gill nets, to construct growth profiles to better understand growth of these species during all life stages.

LTM Biologists are also engaged with DNR Division of Ecological and Water Resources (EWR) scientists in developing a robust, sustainable aquatic plant sampling program. Sampling will include stratified point-intercept sampling, emergent and floating-leaf plant mapping, and plant voucher sample collections. Additionally, we will be incorporating EWR's Score the Shore protocol on 9 of the Sentinel Lakes during 2014. Score the Shore provides a scoring system to evaluate the level and degree of shoreline development on lakes. EWR staff are leading this sampling and it should be noted that these efforts do not affect the LCCMR budget, i.e., plant sampling is largely an in-kind, cooperative effort.

A total of 171 zooplankton tows were collected from all Tier 1 lakes and 7 Tier 2 lakes during the 2013 open water season. Tows were collected once a month from May through October using both 13cm and 30cm mouth diameter nets. In addition, 3 replicate tows were taken monthly from Trout and Greenwood lakes, using a 50cm mouth diameter, 250μm mesh net to estimate spiny waterflea (*Bythotrephes longimanus*) densities. All Sentinel Lake zooplankton samples have been processed and analyzed, and entered in Excel spreadsheets. Zooplankton density and biomass estimates collected with the two different nets (13cm versus 30cm) from all 8 Tier 1 lakes were statically compared, and regression equations calculated. All samples collected with the large 50cm net have been processed and spiny waterflea densities for both Trout and Greenwood lakes have been calculated. Data from the three sentinel lakes that have either spiny waterfleas (Trout and Greenwood) or zebra mussels (Carlos) were analyzed further and these data incorporated into a long term monitoring presentation given at the Lake of the Woods Water Forum in International Falls, MN in March.

Trout Lake fish stomachs were collected from angler caught trout throughout the open water angling season. A total of 16 rainbow trout and lake trout stomachs were collected with the help of Trout Lake Resort owners to gather baseline data on feeding habits prior to established spiny waterflea population in Trout Lake. In addition, 12 lake trout and 11 rainbow smelt stomachs were collected from Trout Lake in July by MNDNR-Grand Marais Fisheries personnel while conducting routine small-mesh gillnet sampling. All fish stomach contents were enumerated and identified and sent to MNDNR-Grand Marais Area Fisheries office for their records.

Outcome 2 – We have hired Jacqueline Amor to fill a Research Analyst position, stationed in Bemidji to develop watershed land cover, drainage feature, and other GIS data sets in support of the overall project. It should be noted that because of the high level of GIS work that this position requires we have filled this position at the Research Analyst level. This does not affect the budget for the project and reflects consultation and conversations with Minnesota Information and Technology staff, DNR Human Resources staff, as well as LCCMR staff.

Outcome 3 – Remote sensing equipment (DO monitors, weather stations, etc.) were deployed last summer and into the fall. Data collection and downloads will occur once weather and ice conditions allow. Pressure transducers (designed to collect data on water levels) were retrieved in November, however problems were encountered with the data that were collected. We are currently working with USGS staff and the manufacturer to address the issue.

Outcome 4 – Work since the initiation of this project in January has focused on processing samples for analysis of stable isotopes of hydrogen, carbon, nitrogen, and sulfur. Sample processing consisted of freeze drying, grinding, weighing, and wrapping samples of fish, plants and aquatic invertebrates, as well as drying and wrapping phytoplankton and seston samples. Additionally, samples to be analyzed for carbon stable isotopes were subsampled for lipid extraction in order to develop a correction factor to account for the fractionation effect of fats on whole-body isotope signatures of carbon. To date 90% of the samples have been processed and the first round of samples has been submitted for analysis. Preparation for field work in the upcoming summer is commencing.

Outcome 5 – During the current report period we have recruited and trained personnel that will assist in field, laboratory and identification components of the work plan. We have written new standard operating procedures (SOPs) for project-specific tasks, revised existing SOPs for tasks that are common for general field and lab practices and reviewed each SOP for comprehension and adherence. All persons that will be involved in the project have completed or updated their GLP trainings, Lab Safety trainings, and Responsible Conduct of Research on-going educational and training requirements. We have discussed Quality Assurance and Quality Control procedures that are integral to this project, and have created the appropriate tools for tracking QA/QC. We have refined a data management plan that is consistent with plans required for federal grants related to environmental assessment projects. We have reviewed on-line water quality and physical/chemical data summaries for each lake to be investigated, have searched for and read/discussed relevant publications related to how our selected sentinel organisms are assessed in other biological monitoring programs in lakes across the United States and elsewhere. We have developed a list of expendable supplies that will be purchased, and have discussed potential timing of field assessments. We are attempting to finalize a list of sample sites within each lake that are considered to be appropriate for achieving our project goals.

Outcome 6 – No presentation of trend data is available as of this update. Data sets from 2013 are still being compiled and are yet to be merged with Phase 1 results.

Activity Status as of 10/30/2014:
Outcome 1 – Despite a late spring LTM biologists completed spring black bass surveys on Bear Head, Carlos, Echo, Elephant, Elk, Madison, Pearl, Portage, St. Olaf, South Twin, Ten Mile and White Iron lakes. Comparison of day and night catch per effort (CPUE) and size structure of black bass were made on Bear Head, Carlos, Echo, Elephant, Elk, Pearl, Ten Mile, Trout and White Iron lakes. Assessments of fish populations (following DNR standardized sampling protocols) were completed on Carlos, Echo, Elephant, Madison, Pearl, Portage, St. Olaf, Shaokotan, South Twin, Ten Mile, and White Iron lakes. Aging structures were collected from target species as per the Sentinel Lakes fish sampling protocols and are currently being analyzed. Age at maturity data were also collected from target species. Data summaries from these assessments, as well as previous surveys conducted on Sentinel Lakes, can be accessed through DNR's Lake Finder.

Pelagic fish communities in Carlos, Elk, and Ten Mile were assessed during the reporting period with vertical gill nets and hydroacoustics.

Nearshore fish communities were assessed using DNR IBI methods on Bear Head, Carlos, Elk, Pearl, Portage, Madison, St. Olaf, Shaokotan, Trout, and White Iron Lakes. Voucher ID samples were collected and are being processed. IBI scores from each of the lakes will be available in 2015.

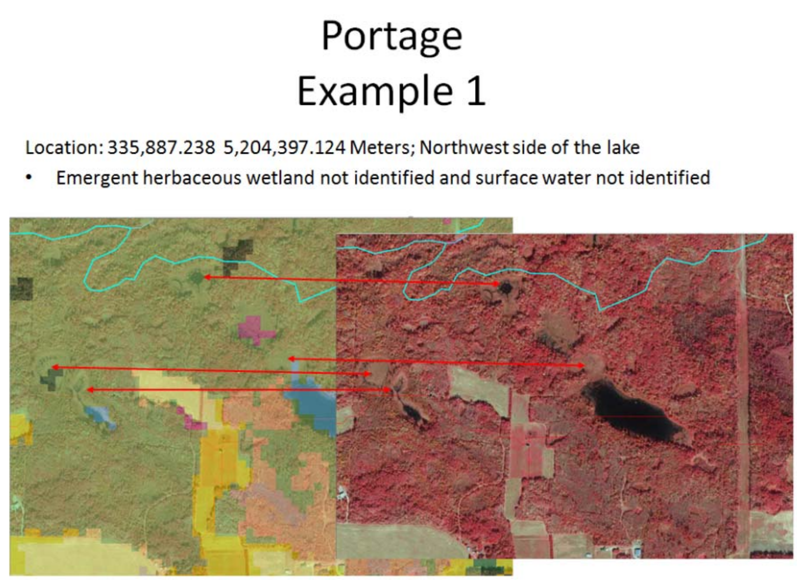
Point-intercept surveys of submersed aquatic vegetation were conducted on Bear Head, Elk, Madison, Pearl, Portage, South Twin, St. Olaf, Trout, and White Iron lakes. Emergent vegetation was mapped on Bear Head, Madison, Pearl, St. Olaf, and White Iron lakes. Additionally, Score the Shore assessments were completed on Bear Head, Madison, Pearl, St. Olaf, and White Iron. These surveys were completed mainly by DNR EWR staff and represent an in-kind, cooperative effort between Sentinel Lakes and EWR staff.

Zooplankton were collected from all Tier 1 lakes as well as Greenwood, South Twin, and White Iron lakes (all Tier 2) during the 2014 open water period. Samples will be processed and analyzed over the winter.

Outcome 2 – The health of the Sentinel Lakes reflects land use and hydrologic transport of sediments, nutrients, and other pollutants, and hydrologic connectivity has implications for potential colonization by invasive species. Phase 2 GIS work focused on updating watershed land cover data as well as using LiDAR data to understand water flow, thus supporting future modeling of nutrient and pollutant transport, and movements/invasions by non-native species.

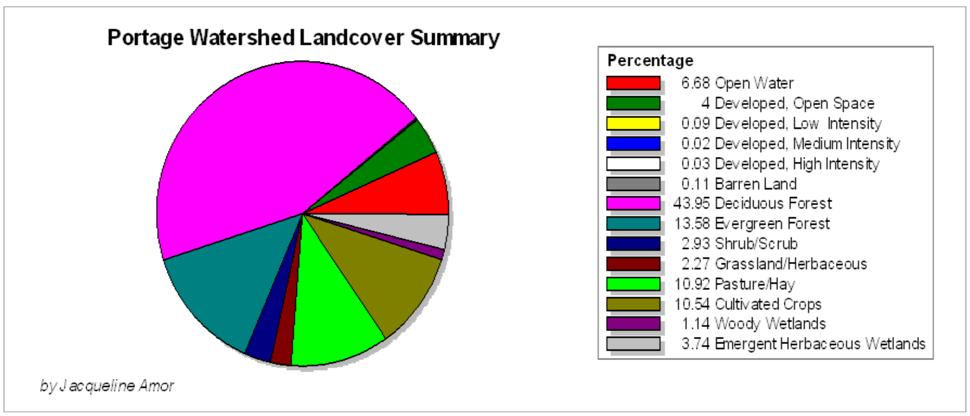
As noted in our previous progress report, Jacqueline Amor was hired as a Research Analyst to work on various Sentinel Lakes GIS projects during the period from April 9 to October 7, 2014. Here we make a final report of her accomplishments in the general area of watershed land use and LiDAR data development as described above. GIS work focused on 5 major tasks: 1) the inventory, organization, and digital archival of Phase 1 GIS datasets, 2) development of updated land cover for the 25 sentinel lake watersheds using the 2011 National Land Cover Dataset, 3) compilation of LiDAR-derived 1-meter and 3-meter Digital Elevation Models (DEMs), 4) topographic position indexes (TPIs) created from the 3-m DEMs; TPI is a focal statistics tool that ultimately helps accentuate, thus help visualize depressions and probable areas of concentrated flow in the landscape (e.g., ditches, intermittent streams, and connections present only in high water conditions), 5) hydro-modification of the DEM in a small portion of the Lake Shaokotan watershed; here “digital dams” (e.g., roads, bridges, water control structures) act as barriers to actual water flow paths (e.g., in reality culverts pass water under roads) and were removed from the DEM to reflect actual water flow within a watershed. Below we provide additional detail regarding tasks 2-5 above, primarily methods and some example products and findings from the work new to Phase 2.

Land Cover Data for Sentinel Lake Watersheds. National Land Cover Dataset 2011 (NLCD 2011) was used to update land cover data for each of the 25 Sentinel Lake Watersheds. NLCD 2011 data was reviewed and assessed for accuracy relative to land cover and land use visible on digital aerial photography. Watersheds were Level 8 catchments identified using the upstream watershed tool available in DNR Quick Layers Watershed Suite. The watershed for each lake was then clipped to the NLCD 2011 raster data, which depicts remotely-sensed land cover types across the United States. We compared the accuracy of the land cover data to the most recent land imagery available (2011, 2012, or 2013), depending on the watershed’s location within the state. Our process for reviewing the 2011 NLCD involved creation of a PowerPoint that catalogued inconsistencies found within each Sentinel Lake watershed. Infrared land imagery (best at showing water features) was layered in the background and a snap shot was taken of the upstream watershed surrounding each Sentinel Lake (either the entire watershed or as much of the watershed as possible was clipped) at 1:30,000 zoom. For large watersheds, a snap shot was taken at a farther zoom to capture the entire upstream watershed. The NLCD 2011 was layered on top of the land imagery at 50% transparency for best data comparison. Snap shots were taken of misidentified land cover features suspected within each upstream watershed at 1:6,000 zoom, and up to 3 examples of misidentified land cover features provided per watershed. While we fully accepted that more misclassifications could be present, our goal instead was to look at the big picture and quickly determine if there were repeating and consistently misidentified land covers. An example of misclassified land covers is provided below.



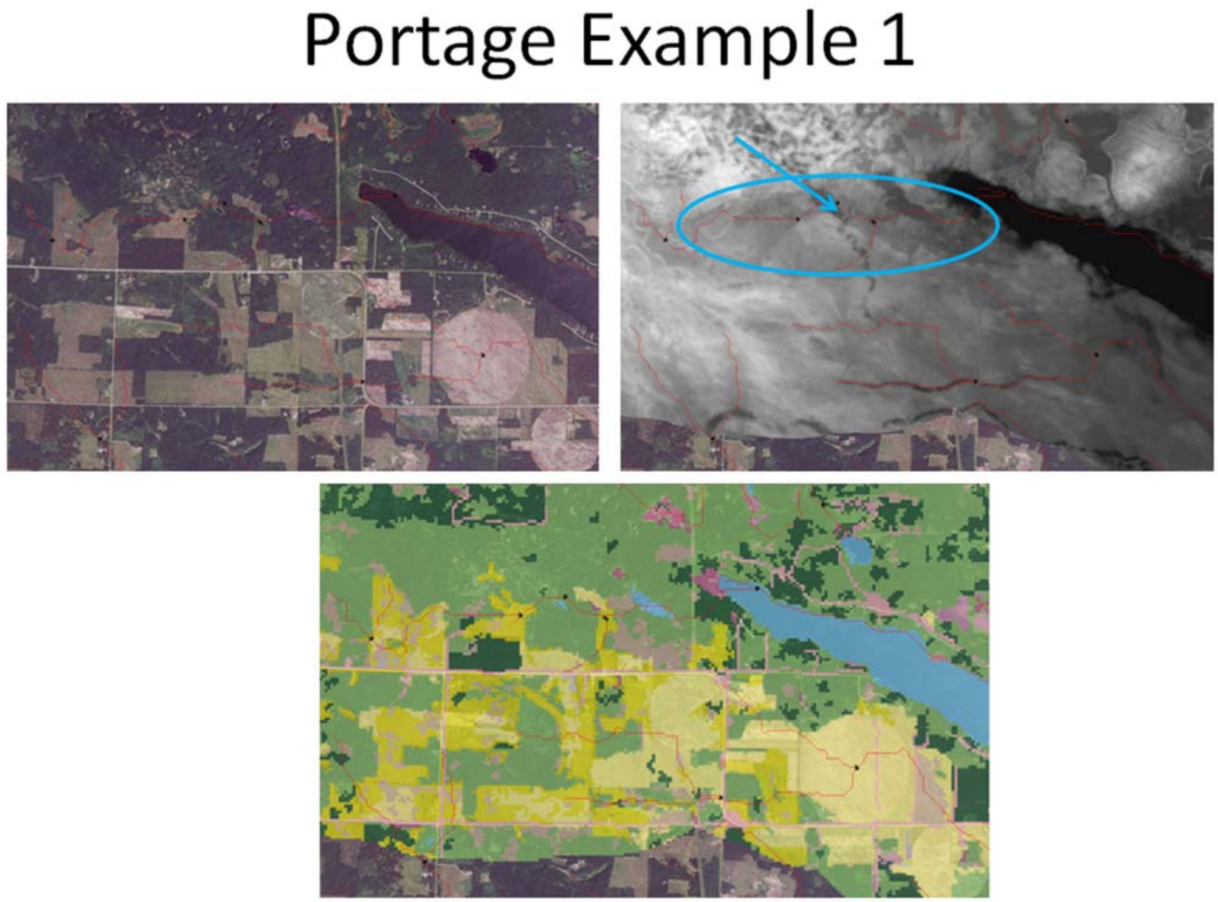
The NLCD appears to be very accurate, with the exception of some misclassification of small aquatic habitats (wetlands and certain open water features). We determined that these discrepancies were minor and that additional delineation and reclassification would not be pursued. Below is an example 2011 NLCD summary graph for Portage Lake, which clearly shows that forested land cover dominates within this watershed.

Portage



Compile LiDAR-derived 1-meter and 3-meter Digital Elevation Models (DEMs) for Sentinel Lake Watersheds. Digital Elevation Models (DEM) of 1m and 3m distance were clipped to each watershed. DEMs were used to determine if flow networks were accurate, and 3-m DEMs were used to create topographic position indexes (see below). The flow networks are accumulation lines previously made for each hydrologic unit code (HUC) for all lakes in the state of Minnesota that have a surface area of 100 acres in size or larger. The flow network represents the theoretical volume and placement of surface water run-off. This data was created from hydrologically-corrected digital elevation models (ANUDEM) in concert with on-screen delineation in ArcView.

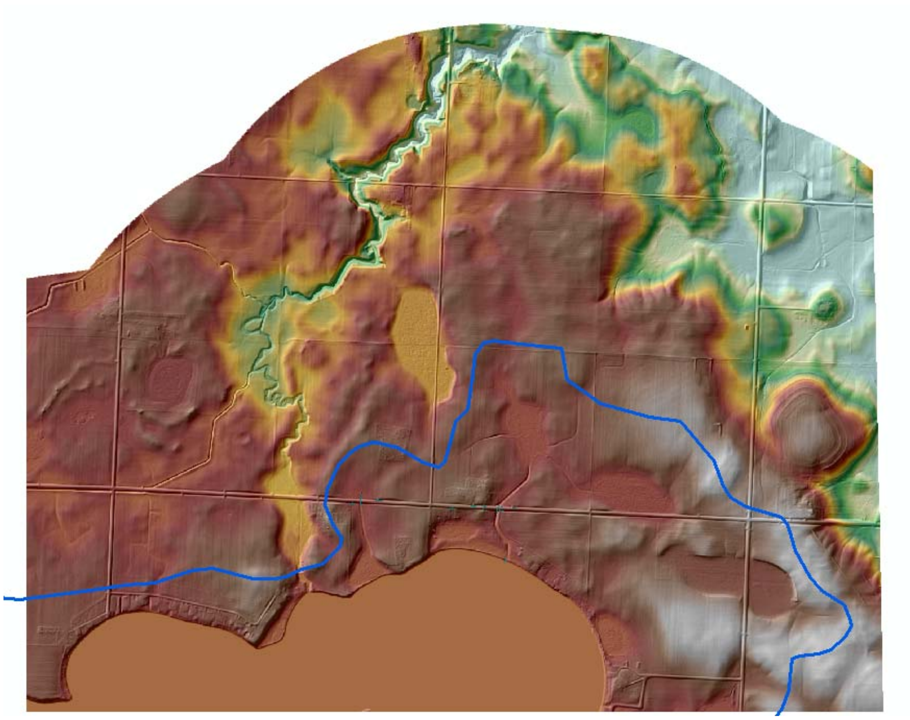
The DEM 1m data came from the dem01m data service from DNR Quick Layers and dem03m data service from DNR GIS Support Specialist, Katie Rossman (dem03m is no longer on Quick Layers). Flow networks were compared to 2011, 2012, and 2013 land imagery and DEMs. Some flow networks did not match up with the LiDAR data, which better shows water flow features across elevation changes. This prompted us to pursue additional analyses for detecting water conveyances over the landscape (i.e., creation of TPIs). Below is an example of probably connectivity apparent in LiDAR, but not aligning well with existing flow networks.



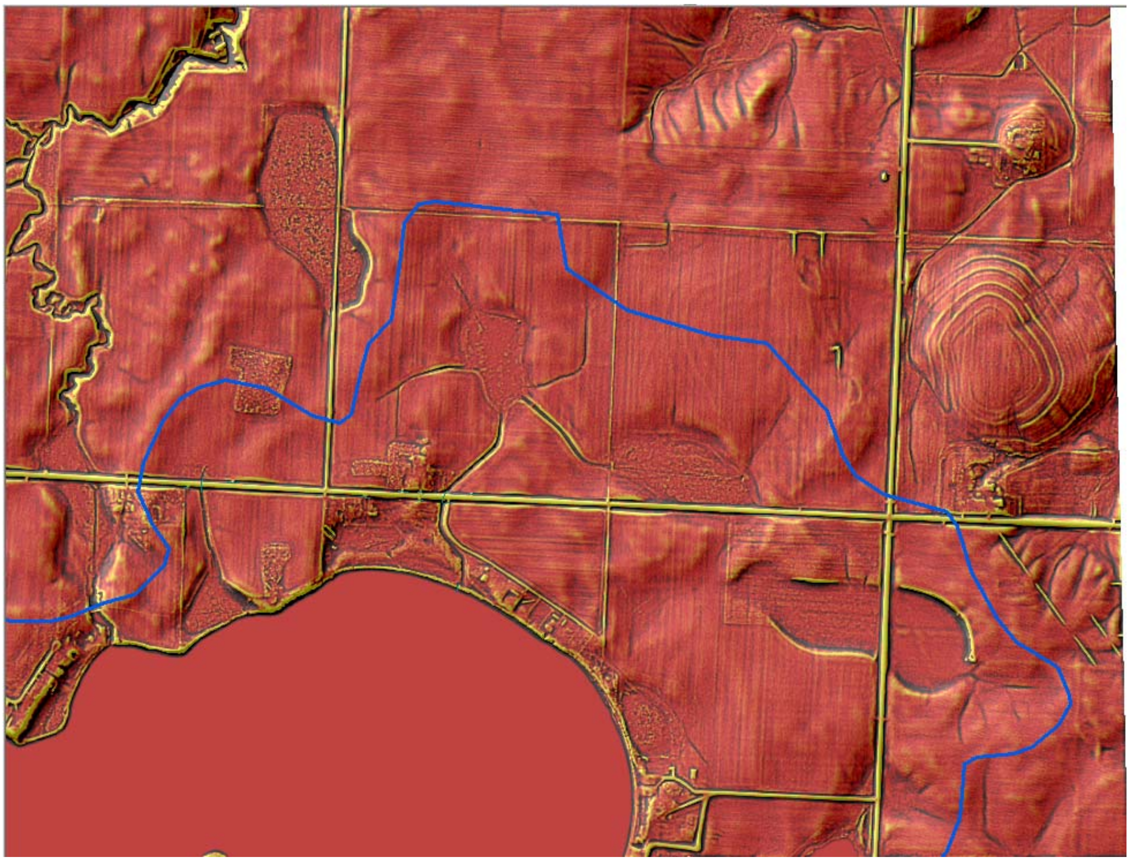
Picture Caption: The arrow points to a flow network that does not align with the LiDAR-derived DEM, which captures very small elevation differences that help to reveal water flow paths. LiDAR-indicated water flow was considerably more accurate than indications of water flow and surface water interconnectivity from 2011 NLCD and aerial photography.

Topographic position indexes (TPIs) created from the 3-m DEMs. TPIs allow visualization and identification of hydrological connectivity within lake watersheds. TPI's have been created for each Sentinel Lake watershed. The 3m-DEM (dem03m data service) was acquired from DNR GIS Support Specialist, Katie Rossman (dem03m is no longer on Quick Layers) and clipped to the watershed. The clip area was created from the watershed suite layer in Quick Layers, using the upstream flow tool and level 8 catchments. A 1-mile buffer was then added to the watershed, and constituted the area of the 3-m DEM clip. The dem_3m_m clip that was created was then plugged into the hillshade tool to create a hillshade-visualization of the data. Next, TPIs were created in Model Builder using the focal statistics tool. Topographic position index (TPI) was created using the rectangle and 3x3 cell options to create the output file fcst_dem3_r33, which was subtracted from the first input dem_3m_m. In a nutshell, TPI is a focal statistics tool that ultimately helps accentuate, thus help visualize depressions and probable areas of concentrated flow in the landscape (e.g., ditches, intermittent streams, and connections present only in high water conditions). The composition of water conveyances and hydro-connectivity across a landscape is important information that can be used to understand how and when aquatic habitats are connected, nutrient and pollutant loads to lakes, and movements/invasions by non-native species. This can also help us understand native species distribution.

Researching hydro-connectivity within the Sentinel Lake watersheds has shown that corrections need to be made to show actual water conveyances that aren't captured in current data. The TPI's are remarkable at revealing ditches, depression, and areas of concentrated flow, as well as various "digital dams" that prevent the correct analysis of water flow throughout a landscape. Digital dams are actual on-the-ground features captured during LiDAR data collection (e.g., roads, bridges, fish barriers, and actual water control structures). The most prevalent examples of digital dams in Minnesota are roads and driveways which appear as digital dams; whereas in reality we know culverts pass water under roads. Consequently only when these digital dams are removed do we have most accurate representation of hydro-connectivity and water conveyances available with current technology.

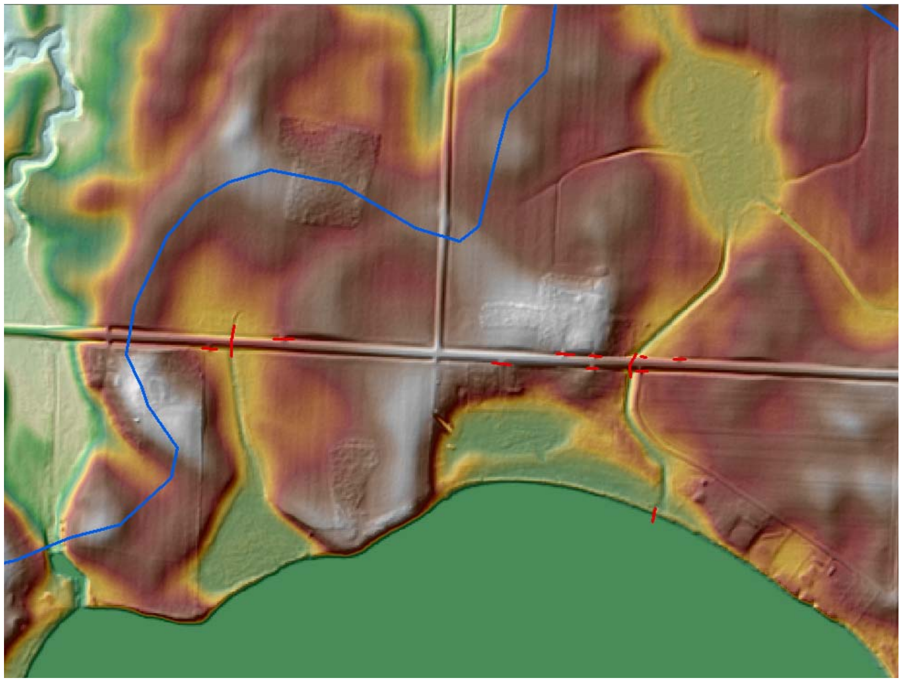


3m-DEM for a portion of the Lake Shaokotan watershed



TPI for a portion of the Lake Shaokotan watershed

Hydro-modification of the DEM in a Small Portion of the Lake Shaokotan Watershed. Using TPIs, contours, and air photos, we identified flow direction and performed hydro-modification within a small portion of the Lake Shaokotan watershed. This was done as an exploration and demonstration of the amount of accuracy that could be achieved if digital dams were systematically removed throughout a lake watershed. Specifically, hydro-modification involves determining where the water flows according to elevation contours and visual aid such as land imagery and TPIs, which aids in the identification of digital dams. Digital dams are then "breached" to reflect that actual water flow on the landscape. A modified raster data set is ultimately created that incorporates these breaches, and ultimately reflects actual water flow paths. We are currently exploring an automated process for hydro-modification of the LiDAR-derived DEM for the Shaokotan, Madison and Pearl lake watersheds to be performed by Dr. Brian Gelder of Iowa State University. We envision that final, hydro-modified DEMs will be used in watershed models for these lakes (estimation of sediment and nutrient loads, and lake water budgets and thermal conditions given different forecasting scenarios). These efforts will be led by Dr. Richard Kiesling of USGS and/or Dr. Jim Almendinger of the St. Croix Watershed Station.



Digital dam breaks are shown in red, and indicate where digital dams occur and need to be corrected for accurate hydro-connectivity

Deliverables. All GIS data created in Phase 2, and described above, has helped us better analyze the movement of water within Sentinel Lake watersheds, and will be used in current and future watershed, lake, and invasive species modeling efforts. The TPIs for SLICE lake watersheds are expected to be added to DNR Quick Layers. All other data described above is available upon request by contacting Brian Herwig at brian.herwig@state.mn.us.

Outcome 3 – Remote sensing gear (water temperature, weather, water level, etc.) were deployed over the open water period and data were routinely downloaded over the course of the summer. Retrieval of gear prior to ice cover is currently ongoing. Data will be stored in appropriate data bases and shared with research partners as needed.

Outcome 4 - Work since our previous progress report has consisted of lab analysis of samples collected in 2013, collection of samples in the field in 2014, and lab analysis of samples collected in 2014. For our field work, we expanded our sampling this year and added a third lake (Lake Ida) to serve as a control lake for Lake Carlos. Lake Carlos has a well-established population of zebra mussels, while zebra mussels were first detected in Lake Ida this past summer. Given their close proximity to each other, comparing isotope characteristics of Ida and Carlos will allow us to assess impacts of zebra mussels on lake food webs. We also expanded our sampling this year such that each of the three lakes (Carlos, Ida, and Elk) were each sampled twice, once in June and once in August, to provide an estimate of temporal variability in isotope characteristics. On each date we collected samples of aquatic invertebrates, submerged aquatic plants, floating aquatic plants, emergent aquatic plants, periphyton, metaphyton, terrestrial plants in the watershed, phytoplankton (both above and below the thermocline), zooplankton and other planktonic invertebrates, benthic invertebrates, and littoral invertebrates. Additionally, our DNR collaborators provided samples of numerous species of fish. Each sample will be analyzed for $\delta^{13}C$, $\delta^{34}S$, $\delta^{2}H$, and $\delta^{15}N$ values. On each date we also collected samples from the epilimnion and hypolimnion that were analyzed for δ^2H of lake water, $\delta^{13}C$ in the dissolved inorganic C pool (DIC), and $\delta^{13}C$ in the dissolved organic C pool (DOC). We also collected a 20L water sample from each lake on each date to incubate phytoplankton for measuring the "pure" phytoplankton δ^2H value.

In the lab all solid tissue samples were dried and crushed. A subsample of each was weighed out for analysis of $\delta^{13}C$, $\delta^{34}S$, δ^2H, and $\delta^{15}N$. We also weighed out a subsample of tissue to be analyzed for lipid content in order to correct our $\delta^{13}C$ estimates. For 2014 our total sample collection numbers by category consist of 266 fish, 305 invertebrates, 128 periphyton, 198 macrophyte, 36 DIC , 36 DOC, 54 δ^2H for water, and 18 "pure" δ^2H samples for phytoplankton. Pure δ^2H samples for phytoplankton were collected by filtering 19L of the 20L of water collected from each lake, then adding the remaining 1L as an inoculum of phytoplankton. The phytoplankton were then allowed to grow to the same density observed in the original 20L, and a sample was collected by filtration. This process produces an estimate of nearly "pure" δ^2H of phytoplankton, as well as the δ^2H value of detritus floating in the water.

All tissue samples have been dried and ground and only ~15% still need to be weighed out for analysis. Three trays of $\delta^{13}C$ and $\delta^{15}N$ samples (288 samples) and one tray of $\delta^{34}S$ have been mailed for analysis to the UC-Davis stable isotope lab, but we have only received the results for one $\delta^{13}C$ and $\delta^{15}N$ tray. One tray of δ^2H samples (70 samples) has been mailed to the University of Wyoming stable isotope lab, but we have not received the results. All of the δ^2H samples for water and $\delta^{13}C$ for both DIC and DOC have been submitted for analysis to the UC-Davis lab, and we have received the results for all the water samples, but only half the DIC and DOC samples.
Our lab work has progressed to the point where we will be submitting large numbers of samples for analysis. Depending on the turn-around time from the Davis and Wyoming lab (which can be several months at times), we should have a substantial amount of results in by mid-winter.

Outcome 5 – From April through October 2014 we have made progress on this project by establishing sample sites on all 12 lakes, sampling the lakes for surface floating pupal exuviae (SFPE), sorting all samples that were collected, quantifying the number of exuviae per sample, and identification to subfamily, tribe and genus for all taxa collected. The number of SFPE varied by sample, ranging from as few as 23 to an estimated number of more than 5,000 per sample. Samples with 500 or fewer SFPE were prepared for species-level identification by slide-mounting the exuviae. For very large samples of SFPE, we established a protocol for subsampling the SFPE in the samples with more than 500 exuviae. Species-level identifications will be completed during the upcoming progress report period.

During this report period we also designed and started to build a web site devoted to this project. The web pages can be accessed through our ***Chironomidae Research Group*** lab web site, which is on-line at: http://midge.cfans.umn.edu/. To navigate to the project pages from our home page click the "**Biodiversity**" link on the pull-down menu tool bar under the heading "**Research.**" On the "Research" tab click on "**Chironomidae in SLICE Lakes**" link to go to the pages describing aspects of this project. The web site will continue to be developed during the upcoming progress report period.

Sample Sites: We have established at least two sampling sites for each of the 12 lakes. The sites were established so that they are located on roughly opposite side of the lake or are in areas with differing depth profiles or aquatic vegetation characteristics if these two parameters differ markedly across one or more bays of the lake. In lakes that are relatively round or oval and without well-developed bays (e.g., Pearl Lake) sampling is conducted on the upwind and downwind shore lines. When possible, our sampling is near public access points or parklands.

Sample Collection: We used our Standard Operating Protocol for collecting all samples. Sampling effort was equivalent at all sites except effort was reduced when large numbers of exuviae were obvious at a given sample site and would result in tens of thousands of exuviae being collected. In cases where the effort was reduced because of the abundances of SFPE, the actual amount of time spent sampling was recorded in field notes. Samples were collected in May, June, July and early September. Based on the outcomes for 2014 we will either sample using the same approach or increase the number of sample events in 2015.

Results: SFPE of species in the subfamilies Orthocladiinae, Tanypodinae and Chironominae were collected in 2014. Table 1 (below) is a list of genera for each of the subfamilies. The subfamilies of Tanypodinae and Chironominae are further sub-divided into Tribes and the tribes and genera are listed alphabetically in the table.

Table 1: Genera of Chironomidae detected in Sentinel Lakes during 2014.

Orthocladiinae
- *Acricotopus*
- *Corynoneura*
- *Cricotopus*
- *Epoicocladius*
- *Heterotrissocladius*
- *Hydrobaenus*
- *Limnophyes*
- *Nanocladius*
- *Orthocladius*
- *Parakiefferiella*
- *Psectrocladius*
- *Synorthocladius*
- *Thienemanniella*
- *Zalutschia*

Tanypodinae

Tribe Coelotanypodini
- *Clinotanypus*
- *Coelotanypus*

Tribe Macropelopiini
- *Psectrotanypus*

Tribe Pentaneurini
- *Ablabesmyia*
- *Conchapelopia*
- *Guttipelopia*
- *Labrundinia*
- *Larsia*
- *Nilotanypus*
- *Paramerina*
- *Thienemannimyia*

Chironominae

Tribe Chironomini
- *Chironomus*
- *Cladopelma*
- *Cryptochironomus*
- *Cryptotendipes*
- *Dicrotendipes*
- *Einfeldia*
- *Endochironomus*
- *Glyptotendipes*
- *Harnischia*
- *Lauterborniella*
- *Microtendipes*
- *Omisus*
- *Parachironomus*
- *Paracladopelma*
- *Paratendipes*
- *Polypedilum*

Tribe Pseudochironomini
- *Pseudochironomus*

Tribe Tanytarsini
- *Cladotanytarsus*
- *Micropsectra*
- *Paratanytarsus*
- *Tanytarsus*

Tribe Procladiini
Procladius

Tribe Tanypodini
Tanypus

Outcome 6 - No presentation of trend data is available as of this update. Data sets from 2013 are still being compiled and are yet to be merged with Phase 1 results.

Activity Status as of 4/15/2015:
Outcome 1 – Due to the departure of one of the LTM biologists, we were unable to complete YOY and young fish sampling as planned on Carlos, Ten Mile, and Shaokotan lakes. This sampling was completed on Elk, Pearl, and Madison lakes. Plans are to continue this sampling in 2015.

2014 fish population assessment data have been entered into the DNR database. IBI scores for the 8 lakes sampled in 2014 have been calculated and voucher samples archived as per DNR protocols. Fisheries data and IBI scores are available upon request. We expect LTM biologists will be initiating trend analyses, which will include data collected in Phase 1, on fisheries data in the coming months.

LTM biologist Matt Hennen has initiated research on young of the year Largemouth Bass and Smallmouth Bass growth and is making progress refining aging techniques. White Sucker were identified in Phase 1 as a potential indicator species and LTM biologist Eric Katzenmeyer is examining population dynamics and aging techniques of this species in Pearl Lake.

Outcome 2 – Jacqueline Amor has completed her appointment with the DNR and all GIS data was handed off to the project coordinators on April 3, 2015. This hand-off included a data dictionary of key files and data sources, as well as a methods/metadata document. All GIS data is contained within geodatabases, including a geodatabase that was created for each lake, which stores three key data layers: NLCD 2011 land cover, LiDAR, and LiDAR-derived data. Results from Jacqueline's work were reported in the October 30, 2014 progress report. All GIS work has been completed, with the exception of hydro-modification of the LiDAR-derived DEM for the Shaokotan, Madison and Pearl lake watersheds, which is being performed by Dr. Brian Gelder of Iowa State University.

The contract for professional/technical services with Iowa State University was executed on February 24, 2015. Automated DEM hydro-modification deliverables from this work are to include: 1) pit-filled input DEM (one-cell sinks removed), 2) enforced DEM (flow obstructions, as defined by ISU enforcement process, will be removed), 3) depression geodatabase (a database of all flow obstructions and their characteristics used in the enforcement program), 4) enforcement line geodatabase (a database of all enforcements applied to the input DEM, which can be modified according to future project needs and re-applied to the input DEM to create a new enforced DEM), and 5) consultation on the results of the model run with a designee of the State. In email correspondence with Dr. Gelder on April 15, 2015, he informed me that he has an enforcement process up and running, and work is proceeding according to schedule. We should have results before the next progress update. We envision that final, hydro-modified DEMs will be used in watershed models for these lakes (estimation of sediment and nutrient loads, and lake water budgets and thermal conditions given different forecasting scenarios), but these efforts may not occur until after the current project has ended.

Outcome 3 – Weather stations and temperature loggers were monitored on an as needed basis. Data are currently being examined for errors and will be stored in appropriate databases. Deployment of pressure transducers is currently occurring as lakes become ice-free. Data has been shared with partners as requested.

Outcome 4 – Outcome 4 - Work conducted from our October 2014 progress report through April 2015 has consisted of continuing our lab processing of collected isotope samples, analysis of the limited amount of results we have received back from the isotope laboratory, and training ourselves on the use of mixing models to analyze our isotope data.

All samples collected in both 2013 and 2014 have been cleaned, washed, dried, and ground. We continue to work on weighing out samples, the last step before they can be submitted to either the UC Davis or University of Wyoming isotope labs. We anticipate finishing the weighing process in early summer. We have also continued lipid extractions that are needed for correcting $\delta^{13}C$ samples. To date all $\delta^{13}C$ in DIC, $\delta^{13}C$ in DOC, and $\delta^{2}H$ in water have been submitted and results have been received back from the UC Davis Isotope lab. We have also received results for tissue samples for 3 trays of $\delta^{13}C$ and $\delta^{15}N$ and three trays of $\delta^{34}S$ from the UC Davis lab and one tray of $\delta^{2}H$ from the University of Wyoming. We anticipate having all samples submitted for analysis by the end of the summer.

Our plan for initial submission of samples was to submit a broad range of samples types in regards to both types of tissue (fish, zooplankton, etc.), as well as type of isotope ($\delta^{13}C$, $\delta^{15}N$, etc.). Submitting a broad range of samples initially will provide information on which types of tissue and which types of isotopes are most variable, thus requiring more samples to get accurate estimates relative to less variable types of samples. To date the number of samples we have received results for are too few to say anything specific about the lake food webs, but the limited results are encouraging in regards to demonstrating trophic relationships and differences among taxa in food web positioning and patterns of energy flow. For example, preliminary results for Lake Carlos show a gradient of $\delta^{13}C$, with $\delta^{13}C$ increasing from hypolimnetic seston (most depleted), through epilimnetic seston, to emergent macrophytes, and then submerged macrophytes (most enriched). Based on the $\delta^{13}C$ gradient, benthic chironomids and zooplankton are getting their C primarily from seston, while Bluegill, Black Crappie, and Northern Pike C inputs are more balanced between seston and littoral sources. $\delta^{15}N$ also illustrated patterns in Lake Carlos, with herbivorous chironomids and zooplankton falling 3.5‰ below Bluegill, indicating Bluegill are feeding primarily on herbivorous invertebrates and not on other fish. In contrast, Black Crappie $\delta^{15}N$ is 2.5‰ higher than Bluegill and similar to Northern Pike, indicating Black Crappie consumed greater amounts of fish or carnivorous invertebrates compared to bluegill.

Preliminary results for Elk Lake are more limited, but $\delta^{13}C$ suggests Cisco are more reliant on seston-derived C (pelagic) than Bluegill, Walleye, or Northern Pike (Fig. 2). For Lake Ida, $\delta^{13}C$ again indicates Cisco acquire most of their C from pelagic sources, while other fish utilize more littoral C (Fig 3). $\delta^{15}N$ indicates Bluegill and Yellow Perch occupy a similar trophic level, while Walleye are more piscivorous. $\delta^{34}S$ should be depleted in organisms deriving their energy from sediments, and our preliminary results from Lake Ida follow this pattern. $\delta^{34}S$ is more elevated in pelagic zooplankton and Black Crappie, while levels are depleted in bottom-orientated chironomids, amphipods, and Common Carp. All of these results are extremely preliminary and may change with additional results. But the results do indicate that our field and lab techniques work and we are producing data that will allow us to assess patterns of energy flow and predator-prey relationships in these lakes.

Lastly, we have worked on using the MixSIAR program (Semmens et al. 2009) to analyze our stable isotope data. MixSIAR is a relatively new software package that allows a user to estimate diet composition with confidence intervals based on multiple stable isotopes. This software will be used to estimate the relative importance of different prey types for each trophic level in our study lakes.

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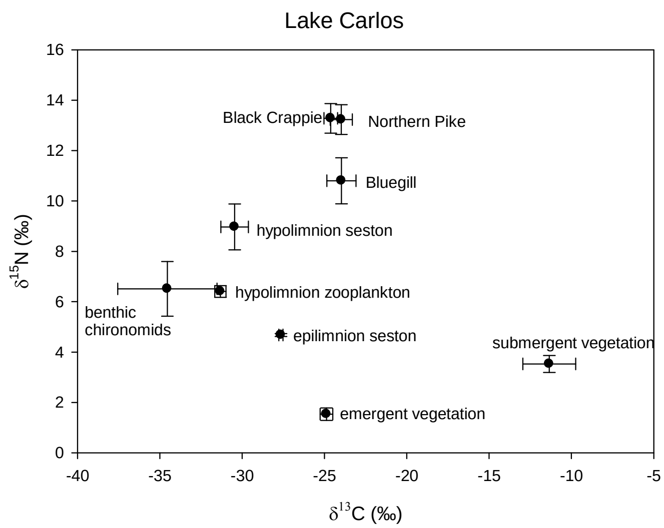

Figure 1. Bi-plot of δ^{13}C and δ^{15}N in several components of the food web in Lake Carlos.

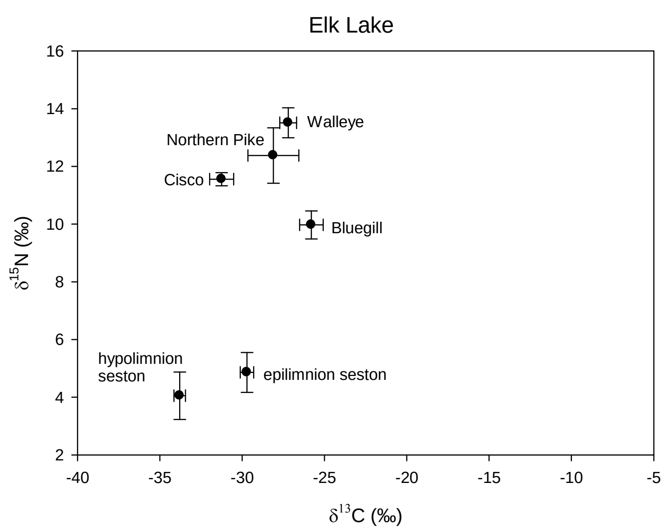

Figure 2. Bi-plot of δ^{13}C and δ^{15}N in several components of the food web in Elk Lake.

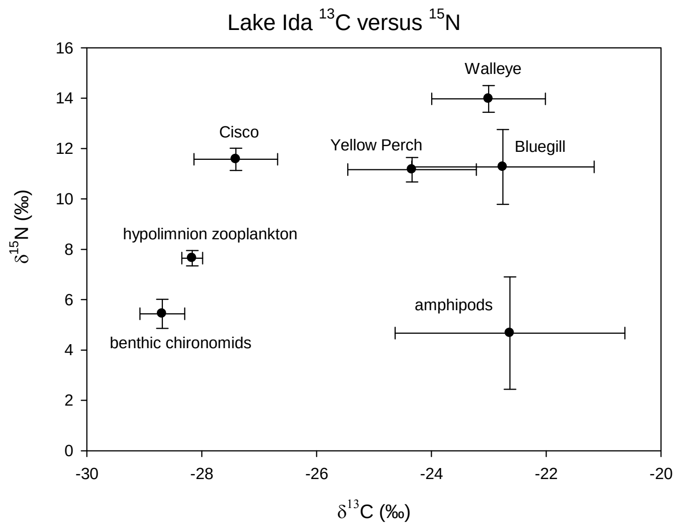

Figure 3. Bi-plot of $\delta\ ^{13}C$ and $\delta^{15}N$ in several components of the food web in Lake Ida.

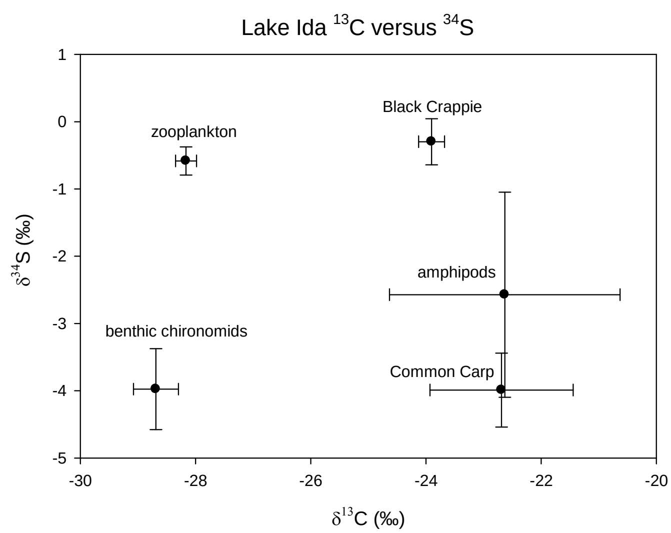

Figure 4. Bi-plot of $\delta\ ^{13}C$ and $\delta^{34}S$ in several components of the food web in Lake Ida.

Outcome 5 – During the last report period I have continued to work on species identification, differentiation and enumeration. At this time 80 species of Chironomids have been documented (Table 2). Progress has also been made on development of data base with specimens-level entries. Plans for field work in 2015 have also be tentatively set, but will likely need to be revised if the warm weather this spring results in earlier than average ice-off dates in the lakes being studied. During this report period I also met with one of the project supervisors to discuss progress & concepts for longer-term assessments of aquatic insects in Sentinel Lakes.

Table 2 follows, with the most-up-to-date estimates of species encountered in the project, organized by genus. I will continue to use this format for reporting future estimates of species in the Sentinel Lakes.

Table 2: Estimates of Chironomidae species detected in Sentinel Lakes during 2014.

Orthocladiinae		Chironominae	
Acricotopus	(one species)	*Tribe Chironomini*	
Corynoneura	(three species)	*Chironomus*	(three species)
Cricotopus	(four species)	*Cladopelma*	(two species)
Epoicocladius	(one species)	*Cryptochironomus*	(two species)
Heterotrissocladius	(one species)	*Cryptotendipes*	(one species)
Hydrobaenus	(one species)	*Dicrotendipes*	(two species)
Limnophyes	(one species)	*Einfeldia*	(one species)
Nanocladius	(two species)	*Endochironomus*	(one species)
Orthocladius	(two species)	*Glyptotendipes*	(two species)
Parakiefferiella	(two species)	*Harnischia*	(one species)
Psectrocladius	(two species)	*Lauterborniella*	(one species)
Synorthocladius	(one species)	*Microtendipes*	(one species)
Thienemanniella	(three species)	*Omisus*	(one species)
Zalutschia	(one species)	*Parachironomus*	(two species)
		Paracladopelma	(one species)
Tanypodinae		*Paratendipes*	(one species)
Tribe Coelotanypodini		*Polypedilum*	(four species)
Clinotanypus	(one species)		
Coelotanypus	(one species)	*Tribe Pseudochironomini*	
		Pseudochironomus	(two species)
Tribe Macropelopiini			
Psectrotanypus	(one species)	*Tribe Tanytarsini*	
		Cladotanytarsus	(two species)
Tribe Pentaneurini		*Micropsectra*	(two species)
Ablabesmyia	(three species)	*Paratanytarsus*	(two species)
Conchapelopia	(two species)	*Tanytarsus*	(four species)
Guttipelopia	(one species)		
Labrundinia	(one species)		
Larsia	(one species)		
Nilotanypus	(one species)		
Paramerina	(one species)		
Thienemannimyia	(one species)		
Tribe Procladiini			
Procladius	(three species)		
Tribe Tanypodini			
Tanypus	(one species)		

Outcome 6 - We have downloaded 2013 and 2014 water chemistry data from MPCA's EQUIS database and are currently working on summarizing these data. No presentation of trend data is available as of this update. Once these data from 2013 and 2014 are summarized they will be merged with Phase 1 results.

Activity Status as of 10/30/2015:
Outcome 1 – LTM biologists completed spring black bass surveys on Bear Head, Carlos, Echo, Elephant, Elk, Greenwood, Madison, and Pearl lakes. An assessment of the protocols used by LTM biologists for assessing black bass populations is ongoing.

Pelagic fish communities in Carlos, Elk, and Ten Mile lakes were assessed during the reporting period with vertical gill nets and hydroacoustics. The frequent assessments of these communities, particularly in respect to Cisco populations, has allowed biologists to track the growth and abundance of these populations. These data provide managers unique insights into an important component of the lake food web that is largely missed by traditional sampling methods. A full report is in preparation.

Research exploring the utility of using White Sucker as an indicator species (i.e., a species vulnerable to changes in water quality and increases in water temperature due to climate change) was completed on Pearl Lake. A full report on this work is also in preparation.

Nearshore fish communities were assessed using DNR IBI methods on Bear Head, Carlos, Elk, Echo, Elephant, Greenwood, Pearl, Portage, Madison, Shaokotan, and Trout lakes. Voucher ID samples were collected and are being processed. IBI scores from each of the lakes will be available in 2016.

Point-intercept surveys of submersed aquatic vegetation were conducted on Carlos and Shaokotan lakes. Emergent vegetation was mapped on Belle, Carlos, Carrie, St. James, and Shaokotan lakes. Additionally, Score the Shore assessments were completed on Elephant, Greenwood, Shaokotan, South Center, St. James, and Ten Mile lakes. As in previous years, many of these surveys were completed by DNR EWR staff and represent an in-kind, cooperative effort between Sentinel Lakes and EWR staff. Additionally, LTM staff conducted surveys, using the commercial product BioBase, to assess submerged plant biovolume on Belle, Bear Head, Carlos, Carrie, Cedar, Elk, Portage, and South Twin lakes. To test the repeatability of the technique, biologists conducted three surveys on Elk Lake. Biologists are currently analyzing the data from these surveys and will be making recommendations regarding the future use, including sampling protocols, of the BioBase product.

Zooplankton were collected from all Tier 1 lakes as well as Echo, Elephant, Hill, Portage, St. James, South Center, South Twin, Tait, and White Iron lakes (all Tier 2) during the 2015 open water period. Samples will be processed (counted and species identified) and analyzed over the winter.

Outcome 2 - Dr. Brian Gelder provided final geodatabase (GIS) files to DNR containing hydro-modified digital elevation models (hDEMs) and hDEM-derived products for the Shaokotan, Madison, and Pearl lake watersheds. One important hDEM-derived product that was provided in fulfillment of the contract was line work identifying landscape water conveyance features using an automated proprietary process. This automated process is designed to remove digital dams from LiDAR data (e.g., where water actually flows thru depressions in a landscape or thru man-made structures such as culverts beneath a roadway. Such conveyances, particularly man-made ones, are typically represented incorrectly in LiDAR data as dams; beaver dams would be one such example where digital dams are created naturally. Another product provided to DNR was water flow path representations (line work of intermittent and permanent surface water flows) at three scales of spatial agregation: 160 ac, 25 ac, and 5 ac. DNR will now begin reviewing the data to determine how well the process did at removing digital dams and representing water conveyance on the landscape. It may be determined that further manual editing is required; this remains unknown at this point. The ultimate goal is to have available, and to use, best-possible hDEMs for future applications of watershed hydrological models to estimate nutrient loads to Pearl, Madison and Shaokotan lakes.

Outcome 3 - Weather stations and temperature loggers were monitored on an as needed basis. Downloaded data are continually being examined for errors and stored in appropriate databases. In preparation for ice cover and winter, water level pressure transducers are being removed from lakes. We continue to share data with partners as requested.

Outcome 4 - Work conducted April 2015 through October 2015 has been focused on sampling our study lakes to increase sampling coverage of all food web components, processing isotope samples in the laboratory and submitting samples for analysis by external isotope facilities, and preparing for data analysis using MixSIAR software.

This past spring we checked our inventory of samples to identify species in each lake lacking samples, or species with low numbers of samples. During the summer we resampled our study sites to fill in these gaps, plus sampled food web components that likely show temporal variability such as periphyton and phytoplankton. These efforts filled all of our sampling needs. Particularly important was our success in sampling benthic insects in Lake Carlos, which we had been unable to find sufficient quantities for analysis the past two summers.

Our main effort in this time period was to process and prepare samples in our laboratory for subsequent stable isotope analysis by facilities at the University of Wyoming and University of California – Davis. We exceeded our goals for this time period. We submitted 441 H isotope samples, 576 C and N isotope samples, and 576 S isotope samples. We are now prioritizing which species to prepare additional samples for analysis, and we will check the variability in results as they come in to prioritize further supplementation with additional samples. We have just started to receive the results of submitted samples, so we have very little new results to report.

Lastly, we have continued to explore the MixSIAR software package for analyzing our results. Our work to date has been focused on creating a database compatible with the MixSIAR package.

Outcome 5 - Samples collected during 2014 were enumerated and results computerized during the current summary period. Copies of EXCEL-based files were saved in PDF format and have been linked to our project web site for public access and distribution. Copies of results by lake and taxonomic group in PDF format can be accessed via links at
http://midge.cfans.umn.edu/research/biodiversity/chironomidae-slice-lakes/data-based-results/

During current summary period (April-October 2015) a more intensive sampling protocol was implemented, with each lake being sampled on a 4-5 week repeating interval, starting with the southern-most lakes and ending with Bear Head Lake and Trout Lake. In addition, supplementary resources were used to take samples from two-additional northern lakes, White Iron Lake and Mink Lake on several occasions. Initial processing of samples collected during 2015 has resulted in the discovery of additional taxa in several lakes, including the northern-most lakes. Very large, and unexpected, emergences were documented during the first sample event after the early ice-out this year in the southern-most lakes.

The emergences of species in the southern-most lakes during the first sample event coincided closely in time with the initial reports of Avian Influenza Virus (AIV) outbreaks on turkey farms in the state.

Although not our intent, these data have turned out to be extremely critical for developing a conceptual model of possible vectoring of virus from water bodies to the poultry-production facilities. We are now in a position (as a consequence of having early emergence data) to developed testable hypotheses related to the species of aquatic insects that are likely candidates to test as vectors, and are preparing additional grant requests to other agencies that, if funded, will be able to determine if any early spring-emerging taxa could be responsible for the initial out breaks of the AIV in Minnesota. This is an unplanned, but highly significant, application of our monitoring results to a very economically important animal health and food security issue in our state economy. Our grant planning includes input from and use of expertise and facilities of collaborators in the Veterinary School of the University of Minnesota.

We have also started to interpret our results for 2014 in relation to previously developed models of lake trophic state (Saether 1978). The models were intended for use in either littoral zones or profundal zones of lakes based on quantitative or semi-quantitative data derived from dredge sampling of larvae in the benthos. Our methods are different and rely on collections of surface floating pupal exuviae that cannot be readily divided into taxa that are from littoral versus profundal zones. However, our results compare favorably with those reported in Saether (1978) if we create a mixed model from his results. We will continue to work on model development, refinement and validation using data from 2015. At present, however, we would like to continue our field assessments into 2016 and select additional northern Sentinel Lakes to enhance our data base for refining and validating.

Outcome 6 - We are continuing work on summarizing the 2013 and 2014 water quality data from MNPCA with the ultimate goal of merging those data with those collected during Phase 1. At this time, no presentation of trend data is available.

Activity Status as of 4/15/2016:
Outcome 1 - Biologists are currently working on summarizing work in this section for inclusion in the final report. This includes, but is not limited to, Cisco population trends, White Sucker assessments, black bass electrofishing results, Bluegill netting comparison, and use of sonar to assess aquatic plant communities. All other fisheries-related data collections have been summarized and entered into the Section of Fisheries' (MNDNR) statewide database.

Outcome 2 – Work related to inventorying watershed land cover and land uses, and drainage features has been completed and final reports for these activities have been provided in the progress summaries above.

Outcome 3 - Weather stations and temperature loggers were monitored on an as needed basis. Downloaded data are continually being examined for errors and stored in appropriate databases. With the exception of NE Minnesota lakes which remained ice-covered at the time of this report, water level monitoring equipment has been deployed. We continue to share data with partners as requested.

Outcome 4 - Work during the past few months on this project has focused on submitting our last rounds of isotope samples for analysis to the UC-Davis isotope facility. We have all begun preliminary analyses of our isotope results using the MixSIAR program in order to assess patterns of energy flow in our study lakes. We also presented some of our preliminary results at the state meeting of the American Fisheries Society this past winter, and we also submitted two abstracts for presenting this work at the annual meeting of the Ecological Society this August in Florida.

Outcome 5 - Most samples from 2015 were sorted and specimens identified to genus. We are now completing sorting of the remaining samples and anticipate having identifications completed to genus. Several genera have two or more (some >6) species. Some taxa that were encountered cannot be effectively identified to genus and may represent undescribed species. Some efforts this spring will be made to capture adults related to these unidentifiable pupal exuviae. We will use taxa that can be identified effectively to genus and species to validate or refine Saether's (1978) model. At this point it looks like the early spring and mid-spring samples will be most informative as we validate/refine the model.

Outcome 6 – A summary report of algal community and zooplankton community dynamics in the Sentinel Lakes is currently undergoing final edits and will be presented in the final report to LCCMR. Water quality data are continually being uploaded, edited, and stored in EQUIS. Merging these data with those collected during all phases of the project is also ongoing. A summary of trends in water clarity and plant growth in Shaokotan Lake in response to watershed phosphorus load reductions is being developed for the final report.

Final Report Summary:
Outcome 1 - LCCMR funding allowed us to hire three professional-level biologists to continue monitoring efforts on the Sentinel Lakes. A comprehensive list of those monitoring efforts is listed in Tables 1 and 2 in the Overview of Sampling. Data were collected in large part by biologists Matt Hennen, Eric Katzenmeyer, and Bill McKibbin and a number of student interns under their direction. The data collected from those efforts are stored in MN DNR databases; those wishing to access them should contact Jeff Reed (jeffrey.reed@state.mn.us).

In addition to the continuation of the monitoring efforts biologists were able evaluate aspects of our sampling program and make recommendations for future sampling efforts. While largely a collaborative effort by a number of people, certain sections of this report were led by the following staff: Bear Head Lake History and Bluegill Sampling – Matthew Hennen; White Sucker Evaluation, Plant Sampling with BioBase, and Lake Shaokotan Case Study – Eric Katzenmeyer. The summary of pelagic fish sampling was completed by Derek Bahr and Beth Holbrook, both research staff with MN DNR.

Overview of Sentinel Lakes Fish Sampling 2008-2015

An intensive long-term lake monitoring program designed to better understand historic and current lake conditions amid changing environmental conditions across Minnesota began in 2008. During Phase 1 of the project (2008-2011), twenty-four lakes were selected as *Sentinel Lakes* to represent statewide gradients in nutrients, climate, ecoregion, and land use. Phase 2 of the Sentinel Lakes Program (SLP), took a tiered approach to sampling that included 8 *tier 1* lakes and 16 *tier 2* lakes from the original 24 lakes (Tables 1 and 2). Tier 1 lakes were selected to be more intensively monitored than tier 2 lakes and an additional lake, Greenwood Lake, was added as a tier 2 lake to represent cold water, Canadian Shield lakes containing Lake Trout and Cisco populations.

The Minnesota Department of Natural Resources (MNDNR) uses standardized gill netting and trap netting during summer and electrofishing in spring or fall as the primary sampling methods to assess fish populations (MNDNR 1993; McInerny 2014). A unique aspect of the SLP included increased fish sampling frequency in addition to exploring experimental and alternate sampling techniques and methods in the 25 sentinel lakes. The fish sampling methods that have been employed include: spring electrofishing, fall electrofishing, summer gill netting, hydroacoustic surveys, nearshore index of biotic integrity (IBI) surveys, spring ice-out trap netting, summer trap netting, vertical gill netting, and fall quarter-inch trap netting (Tables 1 and 2). Electrofishing is an active sampling technique that uses electricity to temporarily stun fish, allowing netters to identify and capture targeted fish species. Spring electrofishing generally occurs in May and June and was designed to target Largemouth Bass and Smallmouth Bass (collectively referred to as black bass) in nearshore areas using a boom electrofishing boat. Fall boat electrofishing has been used on sentinel lakes to target young-of-the-year (YOY) black bass, Bluegill, Walleye, and Yellow Perch in nearshore areas. Additionally, it was used on White Iron Lake to target both YOY Walleye and all ages of black bass. This survey technique generally occurred from late August through October. Summer gill netting and trap netting are passive sampling techniques involving overnight net sets typically deployed sometime between June and mid-September. Gill netting involved the use of standardized experimental gill nets consisting of five 50' panels with sequential 0.75-, 1.0-, 1.25-, 1.5-, and 2.0-in bar mesh webbing across the length of the net (MNDNR 1993; McInerny 2014). The nets are set in offshore habitats and capture fish by entanglement. Additionally, gill netting at Trout Lake included both deep and shallow sets with the standardized experimental nets, as well as specialized small-mesh gill nets (0.37-, and 0.5-in mesh). Short-term gill netting was also explored at Trout Lake as method for assessing Lake Trout populations using both monofilament standardized experimental gill nets and single mesh (0.75-,1.0-, or 1.25-in) gill nets. Trap netting uses standardized double frame nets consisting of two 3 x 6 ft frames and five 2.5 ft diameter hoops wrapped with 0.75-in mesh, and a single 40 ft lead of 0.75-in mesh that directs fish into the net frame, where fish further funnel into the hoops and cod end of the net. This gear is used to sample fish in shallow, shoreline habitats (MNDNR 1993; McInerny 2014). Ice-out trap netting was used to capture Northern Pike in several lakes and Lake Trout in Trout Lake using either standard double-frame trap nets or single-frame 0.75-in bar mesh nets (McInerny 2014). The timing of ice-out trap netting occurred in early spring while ice was receding or immediately following ice-off. Nearshore Index of Biotic Integrity (IBI) surveys, occurring during the same time frame as gill netting and trap netting, target populations and sizes of fish generally not sampled effectively using the standard gears (Drake and Valley 2005; MNDNR 2014). These surveys combine the use of backpack electrofishing units and shoreline seining at a specific number of equally spaced sites, based on surface acreage, along the shoreline. Additionally, 0.25-in mesh trap nets are used for IBI surveys in Canadian Shield lakes (MNDNR 2014). These 0.25-in mesh trap nets were also used at varying times (i.e., July through September) for various sampling goals (e.g., added species diversity, Yellow Perch size at maturity, YOY fish sampling). Recently, hydroacoustics has been paired with vertical gill net surveys to sample the pelagic fish community focusing primarily on Cisco (Ahrenstorff et al. 2013). Hydroacoustics is the use of sound pulses to detect fish throughout the entire water column via echo return imaging. Vertical gill netting is used as a complementary method to verify the fish species that are detected during hydroacoustic surveys (Ahrenstorff et al. 2013).

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Table 1. Tier 1 Sentinel Lakes fish sampling 2008-2015. EF = spring electrofishing, FEF = fall electrofishing, GN = summer gill netting, GST = Short-term gill netting, HA = hydroacoustic surveys, IBI = nearshore fish surveys, STN = spring ice-out trap netting, TN = summer trap netting, VGN = vertical gill netting, TQU = 0.25-inch trap netting.

Lake	Year							
	Phase 1					Phase 2		
	2008	2009	2010	2011	2012	2013	2014	2015
Bear Head	EF, GN, IBI, TN	TN, IBI	EF, TN	TN	IBI	FEF[a], GN, IBI, TN	EF, IBI, TN	EF, FEF[a], IBI, TN
Carlos	EF, GN, IBI, TN	EF, GN, IBI, TN	EF, GN, IBI, TN	EF, GN, IBI, TN	IBI	FEF[a], GN, HA, IBI, TN, VGN	EF, GN, HA, IBI, TN, VGN	FEF[a], HA, IBI, VGN
Elk	EF, GN, IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, STN, TN	IBI, STN		FEF[a], GN, HA, IBI, STN, TN, VGN	EF, FEF[a], HA, IBI, VGN	EF, FEF[a], HA, IBI, VGN
Madison	EF, FEF[b], GN, IBI, TN	EF, GN, IBI, TN	FEF[b], GN, IBI, TN	EF, FEF[b], GN, IBI, TN	FEF[b], IBI	FEF[a], GN, IBI, TN	EF, FEF[ab], GN, IBI, TN	EF, FEF[ab], IBI
Pearl	EF, GN, IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, GN, STN, TN	IBI	FEF[a], GN, IBI, TN	EF, FEF[a], GN, IBI, TN	EF, FEF[a], GN, IBI, TN
Shaokotan	FEF[b], GN, IBI, TN	FEF[b], GN, IBI, TN	FEF[b], GN, TN	GN, IBI, TN		GN, IBI, TN	EF, GN, IBI, TN	FEF[a], GN, IBI, TN
Ten Mile	EF, GN, TN	EF, IBI, TN	EF, GN, HA, IBI, TN, VGN	EF, HA, IBI, TN, VGN	IBI	GN, HA, TN, VGN	EF, GN, HA, IBI, TN, VGN	FEF[a], HA, VGN
Trout	IBI, STN, TN, TQU	EF, GN, IBI, STN, TN, TQU	EF, TN, TQU	GN, TN, TQU		GN, IBI, TN, TQU	EF, GST, IBI, TQU,	GN, IBI, TN

[a]Fall electrofishing targeting young-of-the-year Bluegill, Largemouth Bass, Smallmouth Bass, and Yellow Perch
[b]Fall electrofishing targeting young-of-the-year Walleye

Table 2. Tier 2 Sentinel Lakes fish sampling 2008-2015. EF = spring electrofishing, FEF = fall electrofishing, GN = summer gill netting, HA = hydroacoustic surveys, IBI = nearshore fish surveys, STN = spring ice-out trap netting, TN = summer trap netting, VGN = vertical gill netting, TQU= 0.25-inch trap netting.

Lake	Year							
	Phase 1					Phase 2		
	2008	2009	2010	2011	2012	2013	2014	2015
Artichoke	EF, FEF[a], GN, IBI, TN , TQU	EF, FEF[a], GN, IBI, STN, TN, TQU	EF, FEF[a], GN, IBI, TN, TQU	EF, FEF[a], GN, IBI, TN, TQU	FEF[a], GN, TQU	FEF[a], GN, TQU	FEF[a], GN, TN, TQU	EF, FEF[a], GN, TQU
Belle	EF, FEF[a], GN, IBI, STN, TN	EF, FEF[a], GN, IBI, STN, TN	EF, FEF[a], GN, IBI, STN, TN	EF, GN, IBI, STN, TN	FEF[a]	EF, FEF[a], GN, IBI, TN	FEF[a]	FEF[a]
Carrie	EF, FEF[a], IBI, STN, TN	EF, FEF[a], IBI, STN, TN	EF, FEF[a], IBI, STN, TN	EF, FEF[a], GN, IBI, STN, TN		FEF[a], GN, IBI, TN	FEF[a]	
Cedar	EF, IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, STN, TN		GN, IBI, TN		EF, STN
Echo	EF, IBI, STN, TN, TQU	EF, GN, IBI, STN, TN	EF, STN, TN	EF, STN, TN			EF, GN, TN	EF, IBI
Elephant	EF, IBI, STN, TN, TQU	EF, GN, IBI, STN, TN, TQU	EF, STN, TN	EF, STN, TN			EF, GN, TN, TQU	EF, IBI
Greenwood[b]		GN, TN				GN, IBI, TN		EF, GN, TN
Hill	EF, IBI, STN, TN	EF, GN, IBI, STN, TN	EF, GN, IBI, STN, TN	EF, IBI, STN, TN				EF, GN, IBI, TN

Table 2 – continued.

Lake	Year							
	Phase 1					Phase 2		
	2008	2009	2010	2011	2012	2013	2014	2015
Peltier	EF, IBI, STN, TN	EF, GN, IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, STN, TN		GN, IBI, TN		
Portage	EF, FEF[a], IBI, STN, TN	EF, FEF[a], GN, IBI, STN, TN	EF, FEF[a], IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, STN, TN	FEF[a], STN	EF, FEF[a], GN, IBI, STN, TN	EF, FEF[a], STN, TN
Red Sand	EF, GN, IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, STN, TN	EF, IBI, STN, TN				
St. James	EF, IBI, STN, TN	IBI, STN, TN	EF, STN, TN	EF, GN, IBI, TN, STN		EF, GN, TN		EF, GN, TN
St. Olaf	EF, IBI, STN, TN	EF, IBI, STN, TN	IBI, STN, TN	EF, GN, IBI, STN, TN		STN	EF, GN, IBI, TN	
South Center	EF, GN, IBI, STN, TN	EF, IBI, TN	EF, GN, IBI, TN	EF, IBI, TN	IBI			EF, GN, TN
South Twin	EF, GN, IBI, TN	EF, GN, IBI, TN	EF, GN, IBI, TN	EF, IBI			EF, GN, IBI, TN	
Tait		EF, GN, IBI, STN, TN, TQU	EF, STN, TN, TQU	GN, STN, TN, TQU		GN, IBI, STN, TN, TQU		GN, STN, TN
White Iron	FEF[c], GN, IBI, TN	IBI, STN, TN	FEF[c], GN, STN, TN	FEF[c], TN			EF, GN, IBI, TN	

[a]Fall electrofishing targeting young-of-the-year Walleye
[b]Added as a Sentinel Lake in 2013
[c]Fall electrofishing targeting young-of-the-year Walleye and all ages of Largemouth Bass and Smallmouth Bass

Sentinel Lake Highlights: Abbreviated History of Bear Head Lake, Its Fish and Fisheries Surveys and a Trap Net Survey Timing Evaluation for Sampling Bluegill

History

Bear Head Lake, located in St. Louis County approximately 10 miles east of Tower, MN, lies within the Rainy River Headwaters watershed. The lake itself has a relatively small watershed, 2,270 acres, consisting of primarily coniferous forest land and to a lesser extent wetlands and water (Anderson et al. 2012). Bear Head Lake covers approximately 662 acres and is separated into three distinct basins (Anderson et al. 2012; Figure 1). It has a mean depth of 13.8 feet and maximum depth of 45.9 feet and tends to thermally stratify during the summer months. Historically, logging occurred on the forested land around the lake. A sawmill operated on the south side of the lake until 1911 when forest fires consumed much of the forest in the area. Large quantities of wood slabs were noted primarily in shallow water bays during the first lake survey conducted in 1961 and can still be found present day. In the 1920's, a few seasonal cabins were located on the southern shore of Bear Head Lake and the 1961 lake survey noted 3 trailer houses and 2 cabins situated on the west and south sides of the lake. The area around Bear Head Lake, including the majority of the watershed and the entire shoreline of the lake, was designated as a state park in 1961. A campground and associated recreational facilities were established in the 1970's near the northwest basin which now represents the only development on the lakeshore.

Fish and Fisheries Surveys

Currently, Bear Head Lake supports a relatively simple fish community composed of a mixture of cold, transitional, and warm water species (Table 1). According to historical documents, Bear Head Lake was stocked with primarily Walleye *Sander vitreus* fry and some "bass" fingerlings between the years of 1928-1947 (Table 2). From 1961 to present day, only Walleye have been stocked in the lake and since 1981 Walleye fry have been stocked biennially at roughly 1,000 fish per littoral acre. Fish sampling by the Minnesota Department of Conservation (now Minnesota Department of Natural Resources) Division of Game and Fish (now Division of Fish and Wildlife) first occurred in November 1960 as two, 250' overnight gill net sets captured one Walleye and four "suckers". In June 1961, the initial fisheries lake survey included watershed, habitat, and development assessments in addition to fish and aquatic vegetation surveys. The stated purpose of the full survey was to obtain information on the Walleye population likely initiated in response to the newly developing Bear Head Lake State Park. The fish survey consisted of gill netting, trap netting, and seining that captured hybrid sunfish *Lepomis* spp., Largemouth Bass *Micropterus salmoides*, Northern Pike *Esox lucius*, Walleye, common sucker (now referred to as White Sucker *Catostomus commersonii*), and Yellow Perch *Perca flavescens* (Table 1). From 1961 until 2007, various fish surveys were conducted at Bear Head Lake using gill nets, trap nets, boat electrofishing, and seining and added nine additional fish species (Table 1).

In response to increasing evidence that suggests human development and ecological changes are cumulatively affecting fish populations and their habitats in Minnesota lakes, an effort was undertaken in 2008 to establish a long-term ecological monitoring program for lakes across the state (McInerny 2014). The goal of the Sentinel Lakes Program is to gain a better understanding of complex ecological interactions and how they affect lake ecosystems, thereby helping managers make better and more informed management decisions concerning lakes. This requires the development of a series of metrics that can be tracked over time and analyzed to determine the role and extent of various environmental stressors on Minnesota's aquatic resources. Potential stressors include global climate change, shifts in land-use, angling, or introduction of aquatic invasive species. For example, warming climate trends leading to warmer water temperatures may affect composition of the fish community, potentially increasing the abundance of warm water fish such as Bluegill *Lepomis macrochirus* and Largemouth Bass while causing declines in cool water species such as Yellow Perch and cold water species like Cisco *Coregonus artedi*.

Twenty-five sentinel lakes were specifically chosen to represent the gradient of lake ecosystems across Minnesota. Bear Head Lake was selected as a sentinel lake representing lakes of the Canadian Shield, a subdivision of the Laurentian Mixed Forest ecoregion (Figure 2). To better detect trends in the fish community, increased frequency in traditional gill net, trap net, and boat electrofishing surveys in addition to newly developed near-shore sampling using backpack electrofishing, one-quarter inch trap nets, and seining has occurred on Bear Head Lake, and other selected sentinel lakes (McInerny 2014). The newly implemented near-shore surveys captured several new fish species previously undocumented in Bear Head Lake (Table 1). Furthermore, the focused sampling has allowed us to evaluate fisheries assessment techniques for use in long-term monitoring (McInerny 2014).

Bear Head Lake remains a good selection for continued long-term monitoring as a representative of Canadian Shield lakes for various reasons. The small lake watershed, lack of surface flow inlets or outlets, and limited potential for future shoreline development may allow researchers to more easily identify specific drivers of change (e.g., climate) compared to more complex aquatic systems. Additionally, the presence of intolerant fish species (i.e., Iowa Darter *Etheostoma exile*) and cold-water species (i.e., Burbot *Lota lota*) provide potential monitoring opportunities. In addition to fish population monitoring, Bear Head Lake would provide a unique opportunity to explore invasive rusty crayfish *Orconectes rusticus* dynamics and their interaction with a rare and diverse aquatic plant community should they become established here. Various benefits to fisheries management at Bear Head Lake have been observed through the Sentinel Lakes Program including discovering previously undocumented fish species, increased frequency of fisheries and vegetation surveys, and additional analysis of data.

Trap net surveys

Trap nets are a passive sampling gear set perpendicular to shore overnight with a lead extending to shore (Figure 3). They are designed to entrap fish using shoreline habitats and are used to target species such as Bluegill and Black Crappie *Pomoxis nigromaculatus* (MNDNR 1993). Twenty trap nets of unknown mesh size were used in the initial fish survey in 1961. Trap nets were not used on Bear Head Lake again until a 1979 young-of-the-year assessment of Walleye using 0.25-in mesh trap nets. Six (0.75-in or 1-in mesh) trap net sets were used during assessments in September 1983 and August 1993. From 1995 to 2015, nine trap net surveys were conducted in June and three in August following protocols described in the MNDNR lake survey manual (MNDNR 1993; Table 3).

Trap Net Timing Evaluation for Sampling Bluegill

Introduction

Bluegill are a warm water fish species in the sunfish family (Centrarchidae) first documented in Bear Head Lake during a 1969 gill net survey (Figure 4). However, during the initial fish survey in 1961, a total of 22 fish identified as hybrid sunfish were captured in gill nets and trap nets. There is a potential that at least some of these fish were misidentified leaving Bluegill undocumented until the following survey. Bluegill were not captured in gill nets again until 1991; however, Green Sunfish *Lepomis cyanellus*, Pumpkinseed *Lepomis gibbosus*, and hybrid sunfish were all captured in the seven gill net surveys conducted between 1974 and 1989 providing some potential for misidentification given the similar appearance of these species and potential for hybridization. Despite trap net surveys in 1961 and 1983, Bluegill were not caught in trap nets until 1993. From the historical netting data it appears that the Bluegill population began to expand in the early 1990's, however this observation could simply be a byproduct of several factors including survey timing, net selectivity, sampling protocols, and misidentification.

The Sentinel Lake Program selected Bluegill as a potential ecological change indicator. The species was selected for several reasons including their widespread distribution, importance as prey, angling popularity, and potential sensitivity to ecological changes in land use, climate, and angler exploitation (McInerny 2014). Numerous metrics including abundance, size structure, growth, sex, and maturity of indicator species are being used to monitor potential ecological changes. However, links between these metrics and environmental stressors need to be determined. Ideally, these metrics could be examined for trends over time to make inferences on the population and potential mechanisms of change. Indices of abundance are commonly used in fisheries management to estimate the relative abundance of fish in a particular system. For example, the number of Bluegill sampled per trap net lift (i.e., catch per unit effort [CPUE]). Also, aging structures (e.g., otoliths and scales) are collected to estimate metrics describing age and growth of populations of interest. Age and growth metrics include mean age of fish captured, mean lengths of age classes at capture, mean back-calculated lengths at age, and growth patterns. Lastly, gonads of target species can be examined to determine sex and maturity, thus, metrics segregated by sex can also be estimated as well as length and age at maturity.

In Minnesota, trap netting generally during summer months (i.e., June, July, and August) is the standard method to sample Bluegill populations (MNDNR 1993). From 1995 to 2011, seven trap net surveys were conducted in June at Bear Head Lake (Table 3). However, in 2013 trap netting was done in August to meet specific objectives of Phase 2 of the Sentinel Lakes Program. This led to concerns regarding the comparability of historical catch and size structure data from June surveys with recent data from August surveys. To examine potential differences, the Sentinel Lakes Program conducted standardized trap netting during both June and August in 2014 and 2015 to evaluate potential differences in Bluegill catch and size structure.

<u>Methods</u>

Standard trap nets in Minnesota consisting of two 3 x 6 ft frames connected to five 2.5 ft diameter hoops wrapped with 0.75-in mesh, and a single 40 ft lead of 0.75-in mesh were used to sample the Bluegill population in Bear Head Lake (MNDNR 1993, McInerny 2014). Six trap nets were set overnight in two consecutive nights to sample all 12 standardized locations in both June and August of 2014 and 2015 (Figure 1). The surveys occurred June 18th-20th and August 25th-27th in 2014 and June 8th-10th and August 10th-12th in 2015. Fish collected in individual trap net locations were counted and total length (mm), sex, maturity, and any presence of disease were recorded for a subsample. Sex and maturity were determined for a subsample of fish during June surveys in 2014 and 2015 via manually squeezing the abdomen in attempt to expel eggs or milt present during the spawning period. In June 2013 and August 2015, sex and maturity were determined via visual inspection of the gonads of a subsample of fish.

Sagittal otoliths were removed from a subsample of fish (n = 122) collected via trap netting in August 2015. The goal was to obtain structures from 10 fish per 10-mm length group (e.g., 140-149 mm). However, fish measurements obtained in the field sometimes differed from the lab where the otoliths were removed; therefore, some length groups did not have a sample size of 10. Otoliths were placed into micro centrifuge tubes and allowed to air dry prior to processing. The structures were cracked in half transversely to expose the nucleus, and the broken edge was placed next to a candle flame until singed. The unsinged edge was then embedded in clay and immersion oil was added to the singed surface to increase clarity. Finally, the otolith was magnified under a stereomicroscope to estimate age. Annuli (i.e., yearly growth rings) were identified as alternating light and dark zones with the dark zones being counted (Figure 5).

Historical trap net catch and size data from 1995 to 2015 were used to place the contemporary data pattern in a broader temporal context. Linear models ($\alpha = 0.05$) were used to analyze potential differences in Bluegill catch and size structure between the years and months from sampling done in 2014 and 2015. Pairwise t-tests ($\alpha = 0.05$) were used to examine differences in individual trap net catch within a year (i.e., June trap net 1 – August trap net 1) for 2014 and 2015 data. Welch two-sample t-tests ($\alpha = 0.05$) were applied to compare mean length of fish captured in June and August for 2014 and 2015 combined and both years separately.

Results and Discussion

Historical Comparisons of Bluegill Catch and Size Structure

Since 1995, nine trap net surveys were conducted in June and three in August capturing relatively high numbers of Bluegill ($n \geq 139$; Table 3). The number of Bluegill sampled per trap net (i.e., catch per unit effort [CPUE]) has fluctuated in June surveys from 11.6 to 38.5 (Table 3; Figure 6). A linear trendline displays a slightly negative slope, however the line poorly fits the data ($r^2 = 0.08$) suggesting no discernable trend. The CPUE of Bluegill collected in August from 2013-2015 fell within the range of June CPUE estimates. The total length of fish sampled in June and August surveys ranged from 59 to 233 mm and 82 to 244 mm, respectively (Table 3). The mean length of fish captured in June surveys ranged from 145.1 to 165.0 mm and there is no detectable trend (Table 3; Figure 7). The mean length of Bluegill in the August 2015 survey was slightly smaller (141.4 mm) than observed in any of the June surveys, but the samples from August 2013 and 2014 fell within the range of June surveys. The sex ratio was skewed towards males in 2 of 3 June surveys and in both August surveys where sex was recorded (Table 3).

2015 Age, Size, and Sex

Of the 122 fish collected in August 2015 where age was estimated from otoliths, 53% (n = 64) were age-5 (Figure 8). The oldest fish captured was age-12 and no fish were sampled from ages 0-2, ages 6-7, and age-11. It is not surprising that no fish were captured from ages 0-2 given that these fish were likely not large enough to be effectively sampled by trap nets. The lack of age-6 and age-7 fish and the low number of age-8 fish (n = 3) indicate 3 consecutive years of relatively low recruitment to the population. In contrast, relatively large year classes appear to be present from 2006 (age-9), 2010 (age-5), and 2011 (age-4). Bluegill year-class strength can be partially explained by water temperatures, food availability, and predators (Tomcko and Pierce 2005; Spotte 2007). It appears that Bluegill began to recruit to our gear at around 90 mm which occurred for some fish at age-3. McInerny (2014) suggested that adding gears (e.g., electrofishing and smaller-mesh trap nets) would sample a wider size range and potentially more age-classes of Bluegill because trap netting captures this species relatively late in their lives (McInerny and Cross 2004). This makes it more difficult to understand and track mechanisms regulating recruitment and growth because of the amount of time elapsed by time of first capture.

Sex was determined from all but two fish suggesting gonads were well defined by age-3 or age-4 and were distinguishable during visual inspection. Most of the age-5 fish sampled were male (73%) and were on average larger (mean = 150.4 mm) than the females (mean = 133.6 mm). The relatively large range in sizes for specific age classes, for example age-4 (96-187 mm) and age-5 (101-197 mm) fish, and the potential for limited sample sizes for some year-classes may prevent the ability to effectively track and analyze certain Bluegill metrics in Bear Head Lake. Specific sample size requirements are needed to estimate age structure metrics with adequate precision and accuracy. McInerny (2014) recommended that the sample size for a given lake should be 3 to 4 times the total number of age classes sampled per 10-mm length group. In Bear Head Lake this would equate to 12 to 16 fish per 10-mm length group to account for the four age classes sampled within the 180 mm and 190 mm length groups in 2015. However, in this study we only attempted to attain approximately 10 fish per 10-mm length group. McInerny (2014) also suggested that mean length at capture of age-5 fish and mean back-calculated length at ages 1 through 5 calculated from 5-year olds were potentially useful Bluegill growth metrics for Sentinel Lakes, but cautioned of potential low-sample sizes. For example, if age data had been collected from this population of Bluegill during the previous year (2014) there would likely be a low sample size of age-5 fish given the lack age-6 fish in the 2015 sample. Given the sporadic recruitment observed, potential temporal comparisons could be limited to specific sampling years. Additionally, quality control measures that ensure age estimates are reasonably accurate and application of consistent sampling procedures for collecting age structures are both needed (McInerny 2014).

Comparison of June and August Bluegill Catch in 2014 and 2015

During the 2014 and 2015 surveys conducted in both June and August, we found that individual net catches (n = 48) were extremely variable (mean = 20.1; SD = 21.8), ranging from 0 to 85 fish, and three-quarters of the catches were less than 25 fish (Figure 9). The linear model examining Bluegill catch as a function of the year, month, and individual trap net displayed no significant difference in Bluegill catch between 2014 and 2015 ($p = 0.57$). Therefore, the years were combined and the linear model comparing Bluegill catch between June and August also displayed no differences ($p = 0.15$). The median catch was slightly higher in August (n = 13.5 fish/net) compared to June (n = 13.0 fish/net), however the overlapping confidence intervals of a notched box-and-whisker plot provided further evidence that the medians did not differ (Chambers et al. 1983) (Figure 10). Furthermore, no paired differences were observed in individual trap net catch within a year ($p > 0.18$). From this exercise, it appears that June and August surveys are likely to produce similar catches of Bluegill in Bear Head Lake, but the high variability in the data potentially warrants the collection of additional years of data to further evaluate any bias in survey timing. Trap netting during summer months (i.e., June, July, and August) is the standard method to sample Bluegill populations in Minnesota (MNDNR 1993). However, McInerny (2014) recommended that summer trap netting be discontinued for long-term monitoring purposes because of poor precision and accuracy of CPUE metrics that are strongly affected by spawning behavior and associated movement patterns (Paukert et al. 2004; Spotte 2007). Cross et al. (1995) observed changes in estimates of trap net CPUE and size structure of Bluegill from June through August in three Minnesota lakes and attributed these changes to reproductive behavior. Paukert et al. (2004) suggested sampling Bluegill with passive gears (e.g. trap nets) during midsummer to coincide with peak activity and active gears (e.g., electrofishing) during spring and other portions of summer to coincide with reduced activity. However, pinpointing seasonal activity rates in individual lakes is likely not feasible. McInerny (2014) suggested trap netting in fall when water temperatures were between 12°C and 21°C or in spring prior to initiation of spawning and after water temperatures reach 10°C to obtain reliable CPUE metrics. These methods would need to be thoroughly evaluated for their applicability to a long-term monitoring program.

Comparison of June and August Size structure

Total length was measured for 512 Bluegills during June and August surveys in 2014 and 444 Bluegills in 2015 (Table 3). Examination of length frequency histograms displayed two different patterns between the years (Figure 11). The June 2014 sample was bimodal with relatively few fish observed between 120-160 mm while the 2014 August sample peaked at that size range. In 2015, the size distributions were skewed towards larger fish in the June survey and towards smaller fish in the August survey. The linear model used to examine potential differences in mean length of Bluegills captured in 2014 and 2015 displayed no significant difference between the years ($p = 0.07$). With years combined, mean length of Bluegills sampled in June (154.3 mm) was significantly larger than in August (144.4 mm) ($p < 0.01$), but the model explained little of the variability ($r^2 = 0.02$). Welch two-sample t-tests indicated no differences observed in mean lengths between the months in 2014 ($p = 0.74$). Therefore, length differences observed in 2015 weighted the overall statistical difference observed with the months combined. The median size was also larger in June (164.0 mm) than in August (144.0 mm) and the non-overlapping confidence intervals of the notched box-and-whisker plot provided evidence that the medians were statistically different as well (Chambers et al. 1983) (Figure 12). Again, the differences observed in the 2015 surveys are driving this result. The biological significance of the statistical differences observed between June and August samples is limited (i.e., only about a 10 mm mean length difference) and likely an artifact of the large sample size of measured fish (n = 956). In other words, the large sample size gave us more power to detect small, but less biologically meaningful differences. When examining larger fish separately, there were no differences in the catch rate of Bluegills ≥178 mm (7 in) when comparing the months ($p = 0.43$) or years ($p = 0.88$).

The dissimilar length frequency distributions of fish captured during June surveys could be a function of seasonal factors regulating Bluegill movement and spawning behavior (Cross et al. 1995; Paukert et al. 2004; Spotte 2007). In northern Minnesota lakes there is a potential that early June trap net surveys could occur prior to or very early in the spawning season potentially missing the shoreward movement and peak activity of mature Bluegill that occurs during early to midsummer (Paukert et al. 2004; Spotte 2007). Also, alternative life history strategies displayed by male Bluegill (e.g., territorial males and cuckolding males referred to as sneakers and satellites) may bias sampling (Spotte 2007). For instance, during the spawning period territorial males tend to exhibit greater site fidelity due to nest guarding behavior (Paukert et al. 2004; Spotte 2007).

McInerny (2014) and Cross et al. (1995) suggested that length-based metrics obtained from summer trap netting are strongly affected by spawning behavior. In particular, Cross et al. (1995) observed declines in catches of Bluegill >149 mm from early June through August and attributed the differences, in part, to seasonal changes in spatial distribution. Additionally, it has been shown that larger, mature Bluegill use offshore and open water habitats in mid- to late-summer following spawning, thus making them less vulnerable to trap nets (Cross et al. 1995; Paukert and Willis 2002; Weimer et al. 2014). Alternatively, smaller, immature Bluegill may be more susceptible to trap netting throughout the summer due to their tendency to use nearshore areas with macrophyte coverage and their increased probability of recruitment as they grow during this period (Hall and Werner 1977: Cross et al. 1995). Thus, passive capture techniques (e.g., trap nets) alone may not efficiently sample the population (Paukert et al. 2004; McInerny 2014). To augment catches by standard summer trap netting, McInerny (2014) suggested adding sampling gears (e.g., electrofishing and smaller-mesh trap nets) and/or deploying gears during spring and fall to obtain wider size ranges of fish.

Conclusions

Bluegill were chosen as a target species for the long-term monitoring program because of their widespread distribution, importance as prey, public value, and the belief that they would be affected directly or indirectly in some measureable way by ecological system changes (McInerny 2014). Traditionally, Bluegill populations have been targeted using summer trap netting in Minnesota in an attempt to gain information about important population parameters (e.g., relative abundance, growth, etc.). However, Bluegill movement and spawning behavior, including alternative life history strategies, may limit the applicability of certain metrics (e.g., length at maturity, length at age) collected via summer trap netting in displaying trends over time.

Overall, the trap net survey timing study determined that due to high variability, more years of data collection may be needed to confidently evaluate any bias in catch or size structure when comparing June and August surveys. No clear trend in historical June trap net CPUE of Bluegill at Bear Head Lake was evident, and the more recent surveys completed in August appear comparable to previous June surveys in terms of catch. Direct comparisons of catch and size structure information collected from June and August surveys completed within the same year for two consecutive years displayed no observable differences in trap net catch of Bluegill at Bear Head Lake. Although the average length of Bluegill sampled in June and August surveys was slightly different (by approximately 10 mm), the biological significance of this finding is limited. Additionally, larger Bluegill were captured at the same rate in June and August. However, it is recommended that summer trap netting for Bluegill be discontinued for long-term monitoring purposes because of poor precision and accuracy of CPUE metrics and length-based metrics that are strongly affected by variable seasonal movement patterns. Furthermore, McInerny (2014) stated that trap net CPUE did not appear to reflect the density of Bluegill in several sentinel lakes. Thus, alternative gears, sampling methodology, and metrics need to be explored for usefulness in tracking potential changes over time.

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Table 1. Fish species collected in Bear Head Lake by trap nets, gill nets, shoreline seining, and backpack electrofishing from 1960-2015. The associated trophic, thermal and environmental tolerance classifications for each species are listed.

Common name	Species name	Trophic guild[a]	Thermal guild[b]	Environmental tolerance[a]	First sampled
Black Crappie	*Pomoxis nigromaculatus*	Piscivore	Warm	Neutral	1989
Bluegill	*Lepomis macrochirus*	Insectivore	Warm	Neutral	1969
Burbot	*Lota lota*	Piscivore	Cold	Neutral	1989
Central Mudminnow	*Umbra limi*	Insectivore	Transitional	Neutral	2009
Golden Shiner	*Notemigonus crysoleucas*	Insectivore	Warm	Neutral	2013
Green Sunfish	*Lepomis cyanellus*	Insectivore	Warm	Neutral	1980
Hybrid Sunfish	*Lepomis* spp.	Insectivore	Warm	Neutral	1961*
Iowa Darter	*Etheostoma exile*	Insectivore	Warm	Intolerant	2009
Largemouth Bass	*Micropterus salmoides*	Piscivore	Warm	Neutral	1961
Northern Pike	*Esox lucius*	Piscivore	Transitional	Neutral	1961
Pumpkinseed	*Lepomis gibbosus*	Insectivore	Warm	Neutral	1974
Tadpole Madtom	*Noturus gyrinus*	Insectivore	Warm	Neutral	2010
Walleye	*Sander vitreus*	Piscivore	Transitional	Neutral	1960
White Sucker	*Catostomus commersonii*	Omnivore	Transitional	Tolerant	1960
Yellow Perch	*Perca flavescens*	Insectivore	Transitional	Neutral	1961

[a]Classifications from Barbour et al. (1999) and Drake and Valley (2005)
[b]Classifications from Lyons et al. (2009)
[*]Identification of hybrid sunfish is questionable from 1961 survey

Table 2. Bear Head Lake fish stocking history.

Year	Species	Total no.	Life stage
1928-1947 (intermittent)	"Bass"	950	Fingerling
1928-1947 (intermittent)	Walleye	3,535,015	Fry
1961-1976 (intermittent)	Walleye	287,830	Fingerling
1961	Walleye	2,000,000	Fry
1977	Walleye	500,000	Fry
1978-1980 (annual)	Walleye	1,000,000	Fry
1981-2013 (biennially)	Walleye	350,000	Fry
2015	Walleye	371,000	Fry

Table 3. Survey timing (Month), total catch (# BLG), catch per unit effort (CPUE), number measured, length metrics, and male to female sex ratio for Bluegills captured during Bear Head Lake trap net surveys from 1995 to 2015.

Year	Month	# BLG	CPUE	# Measured	Min. length (mm)	Max length (mm)	Mean length (mm)	M:F
1995	June	197	16.4	197	112	232	154.8	
1999	June	462	38.5	462	93	228	162.5	
2004	June	139	11.6	127	92	233	155.4	
2008	June	279	23.3	279	85	213	145.1	
2009	June	234	19.5	180	94	223	163.7	
2010	June	217	18.1	217	88	202	148.8	
2011	June	316	26.3	213	100	227	146.7	2.28:1
2013	August	268	22.3	268	91	204	146.3	1.69:1
2014	June	219	18.3	218	59	215	146.2	1.04:1
2014	August	303	25.3	294	82	235	147.2	
2015	June	167	13.9	167	81	223	165.0	0.67:1
2015	August	277	23.1	277	89	244	141.4	2.53:1

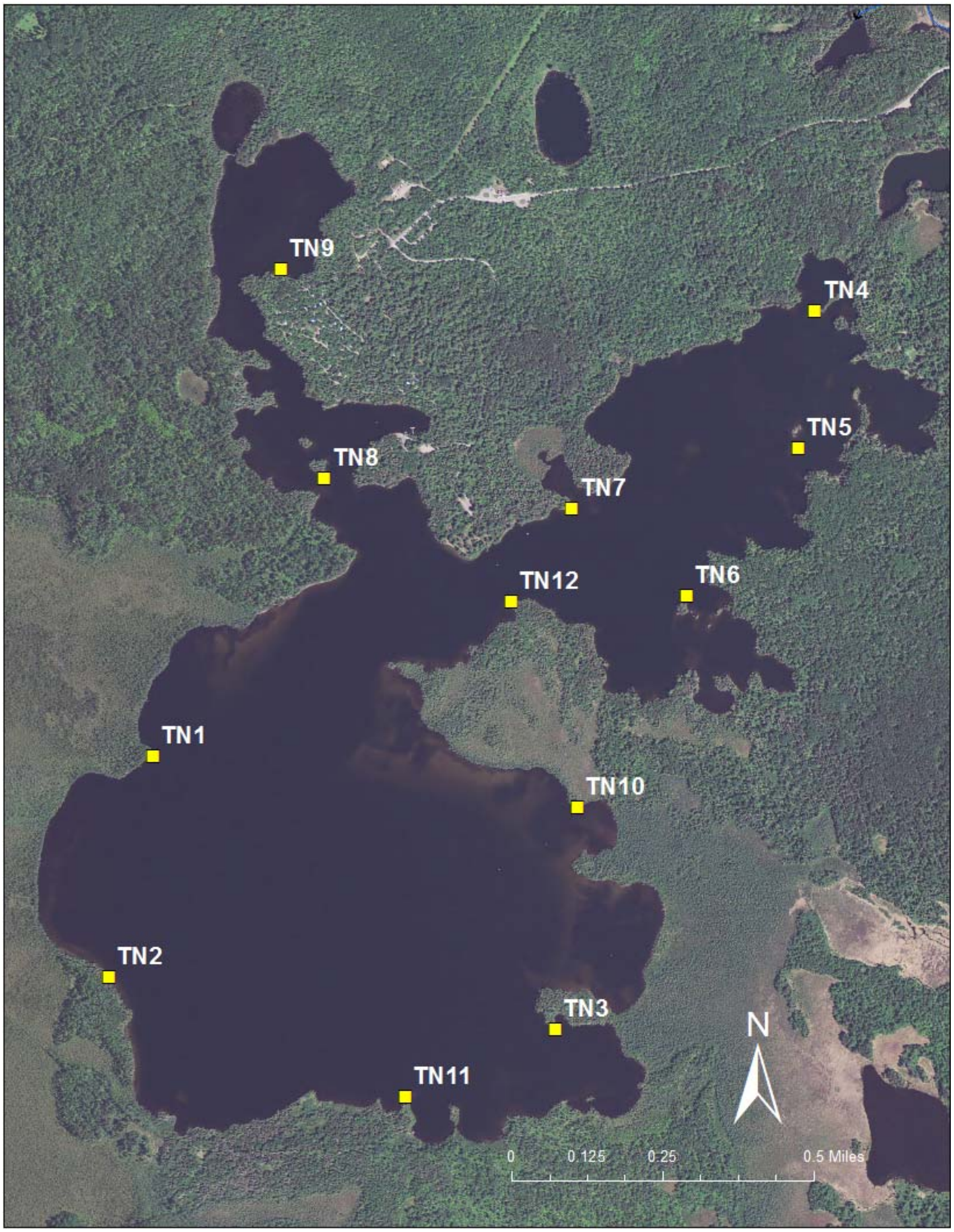

Figure 1. Bear Head Lake with 12 trap net (TN) locations used since 1995.

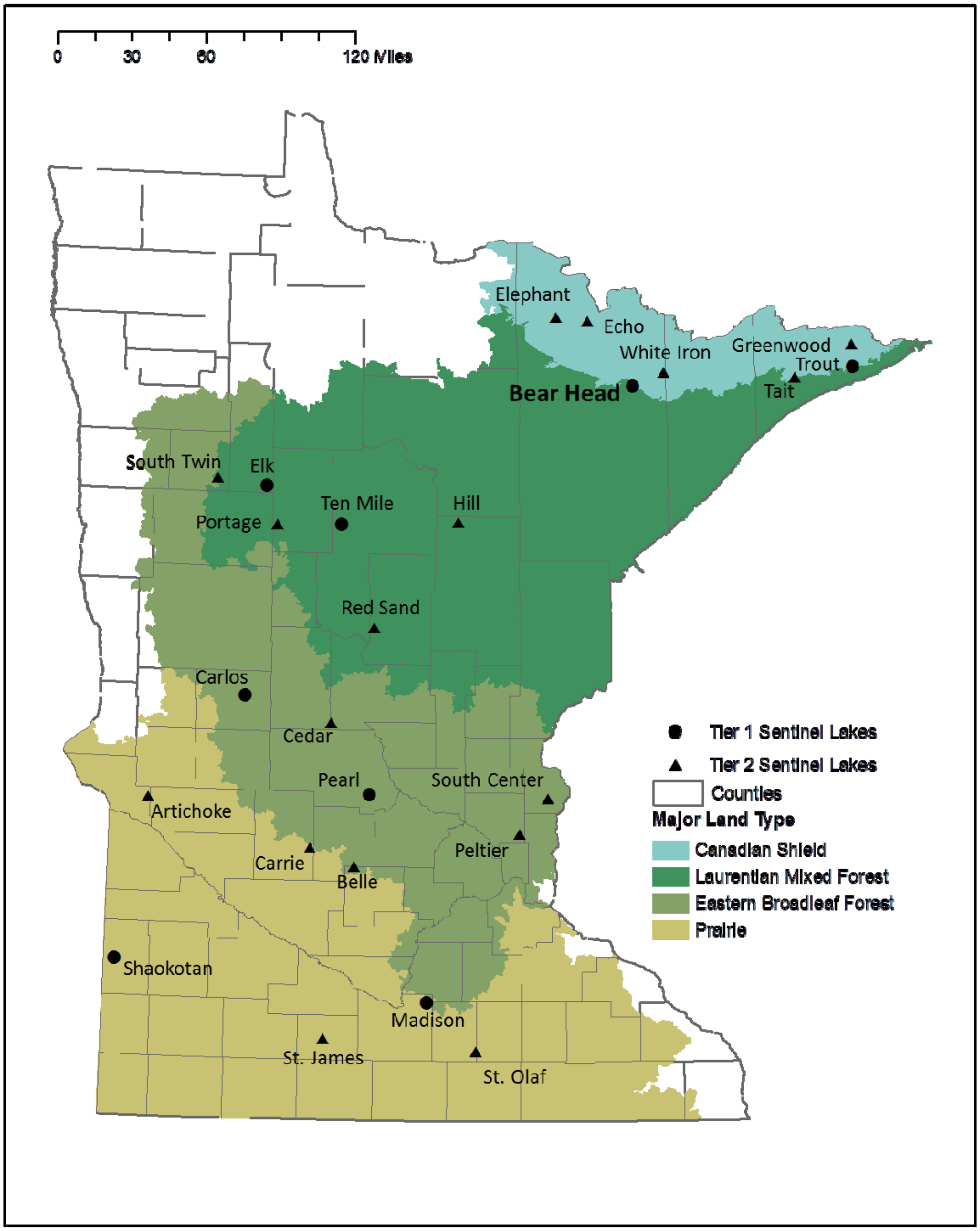

Figure 2. Map of Tier 1 and Tier 2 Sentinel Lakes located within Minnesota counties across four major land types. Lakes were designated into tiers to priortize fish sampling during Phase 2 of the Sentinel Lakes Long-term Monitoring Program.

Figure 3. Standard trap net set.

Figure 4. A Bluegill captured from Bear Head Lake.

Figure 5. Age-4 Bluegill sagittal otolith sectioned in half and singed to estimate age. The thin, dark bands represent an annuli (winter growth) and the wide light colored bands represent summer growth.

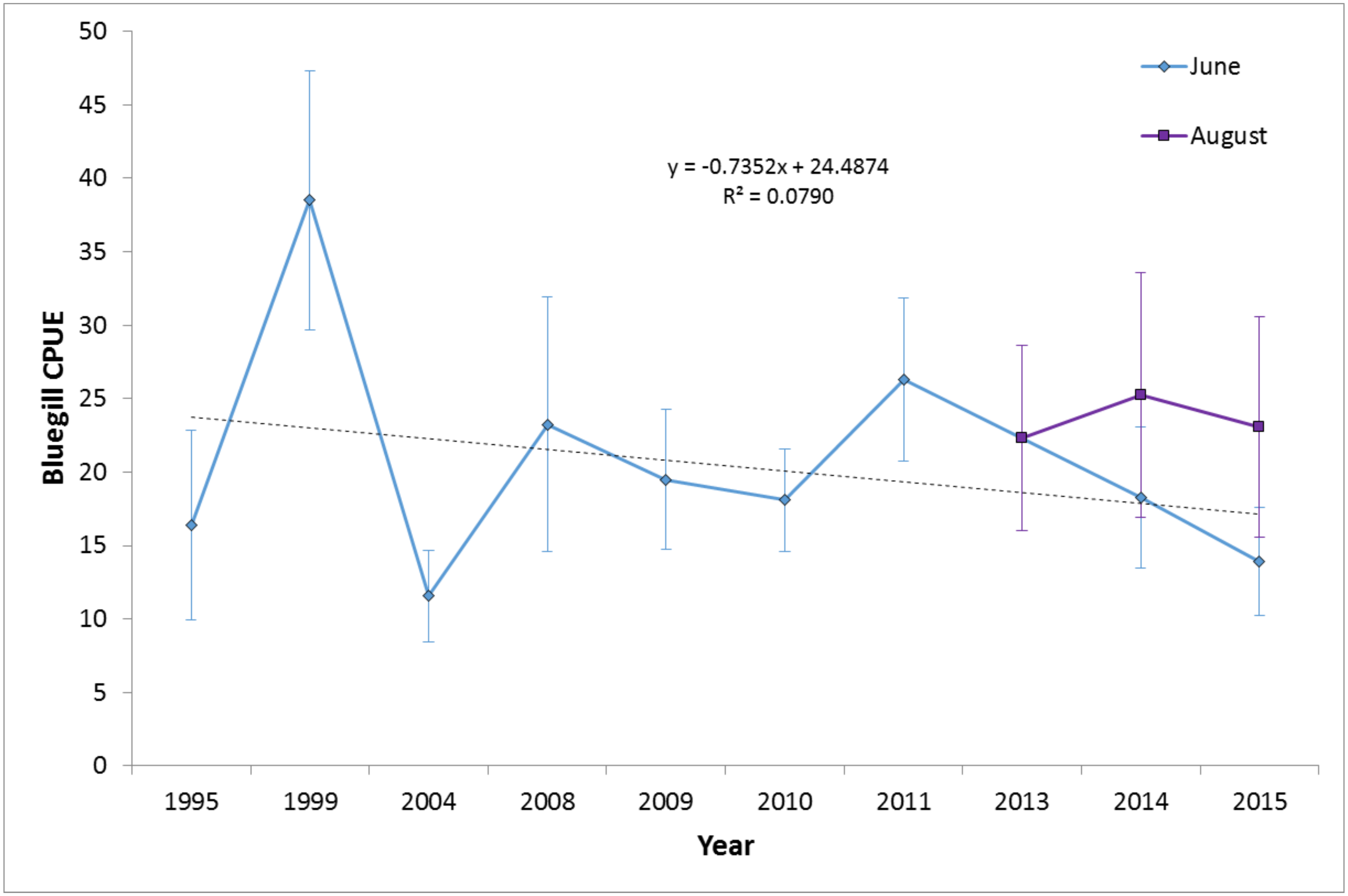

Figure 6. Mean catch per unit effort (CPUE ± SE; catch per net night) for Bluegill collected from Bear Head Lake during June (blue) and August (red) from 1995 to 2015 using standard trap nets. The trend line (dashed black) and associated equation are fitted only to the June samples.

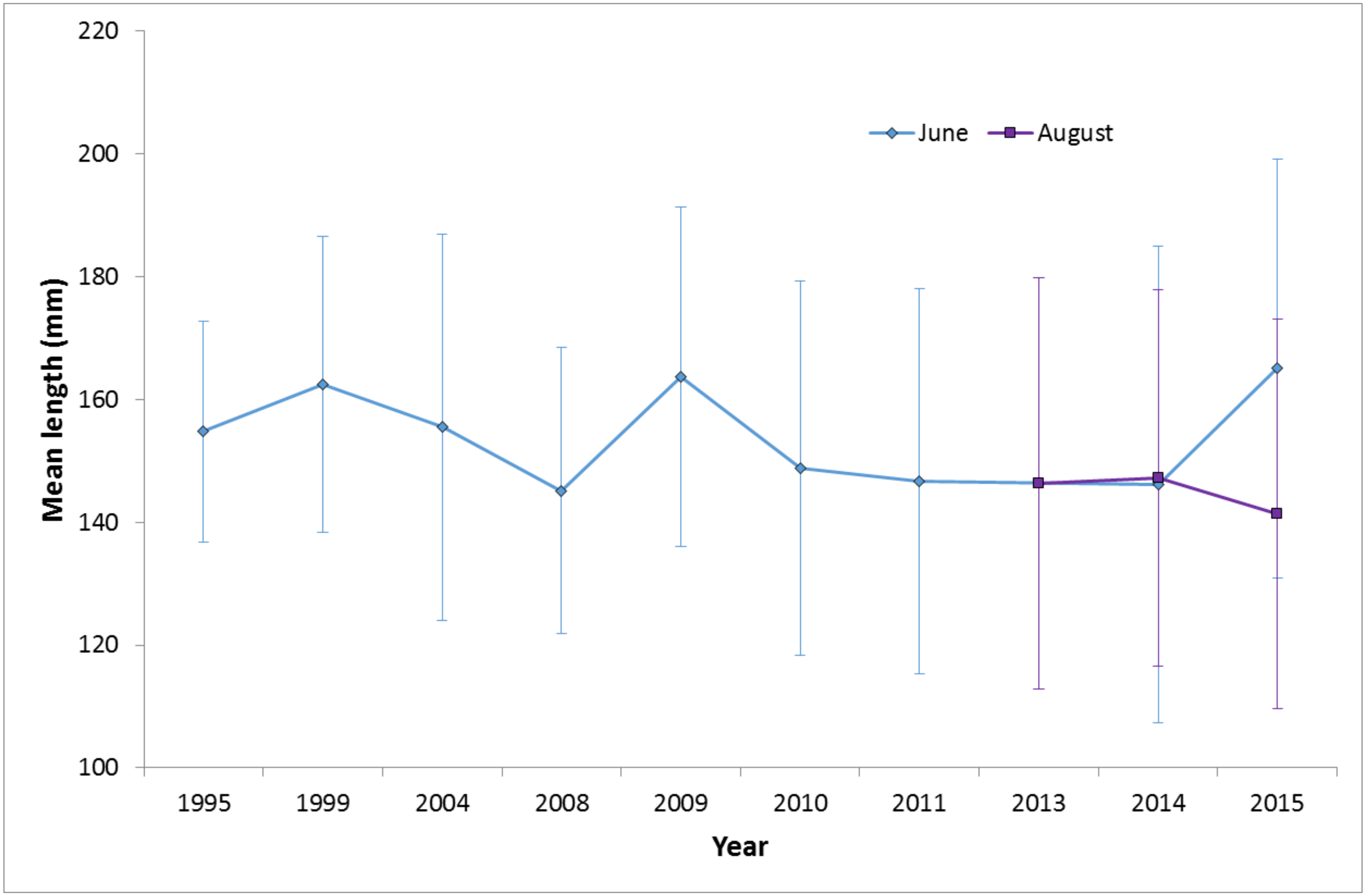

Figure 7. Mean length (mm; ± SD) of Bluegill collected from Bear Head Lake during June (blue) and August (red) from 1995 to 2015 using standard trap nets.

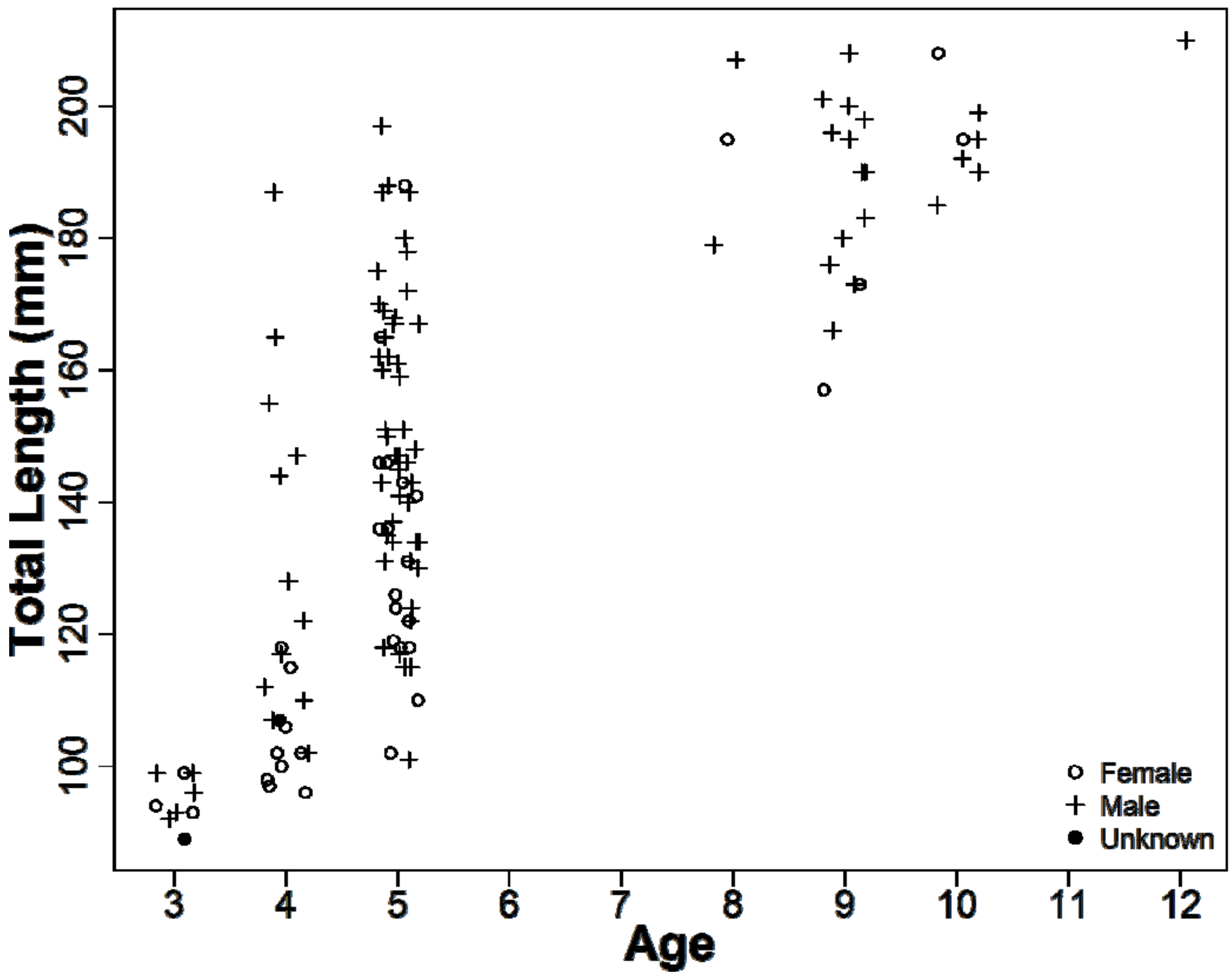

Figure 8. Total length (mm) at estimated ages for female (n = 34; black dots), male (n = 86; medium gray dots), and unknown sex (n = 2; light gray dots) Bluegill collected at Bear Lake in 2015. Data points were jittered along the x-axis at each of the discrete age assignments.

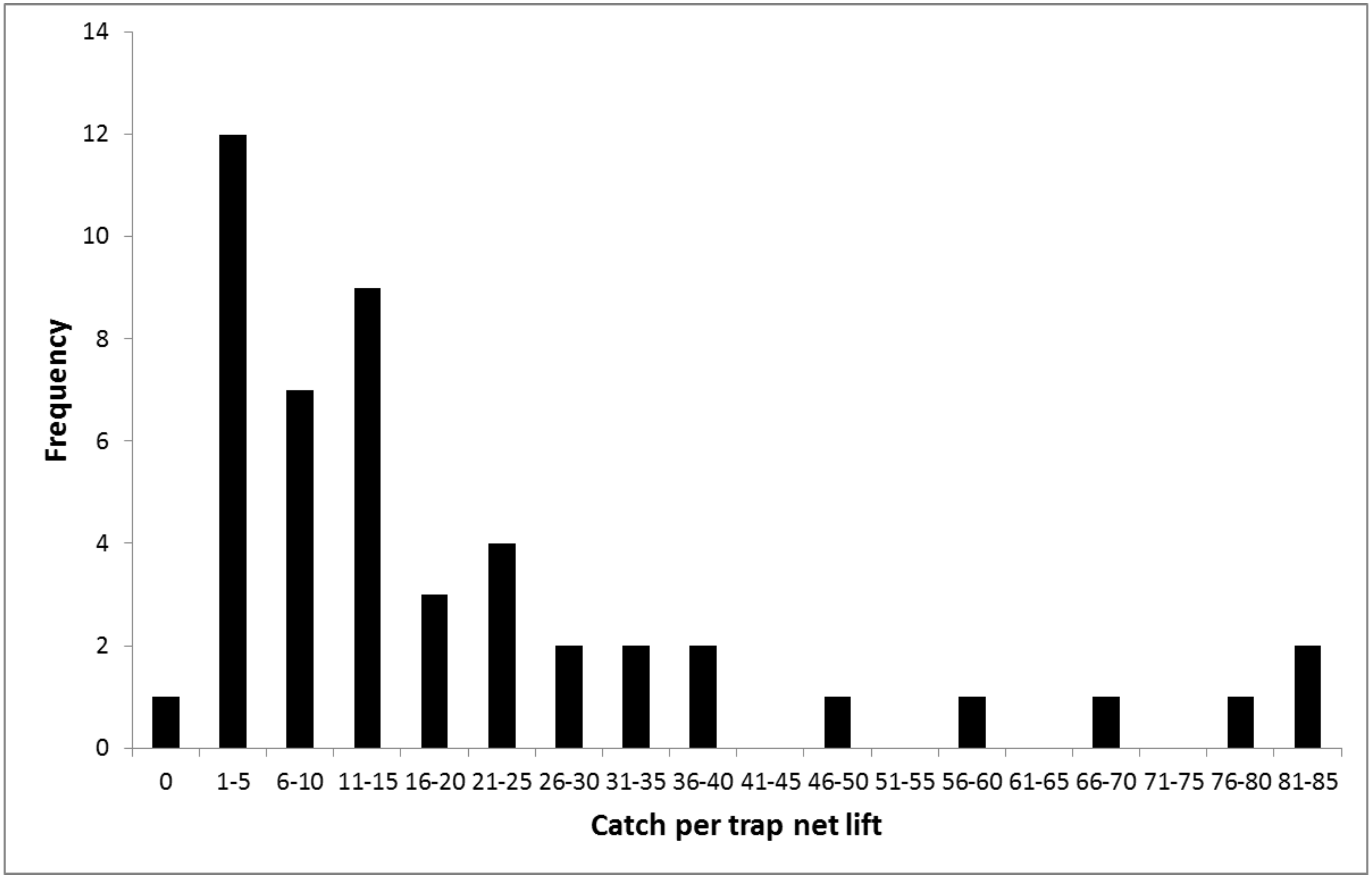

Figure 9. Frequency distribution of trap net catches of Bluegill in Bear Head Lake sampled during June and August of 2014 and 2015.

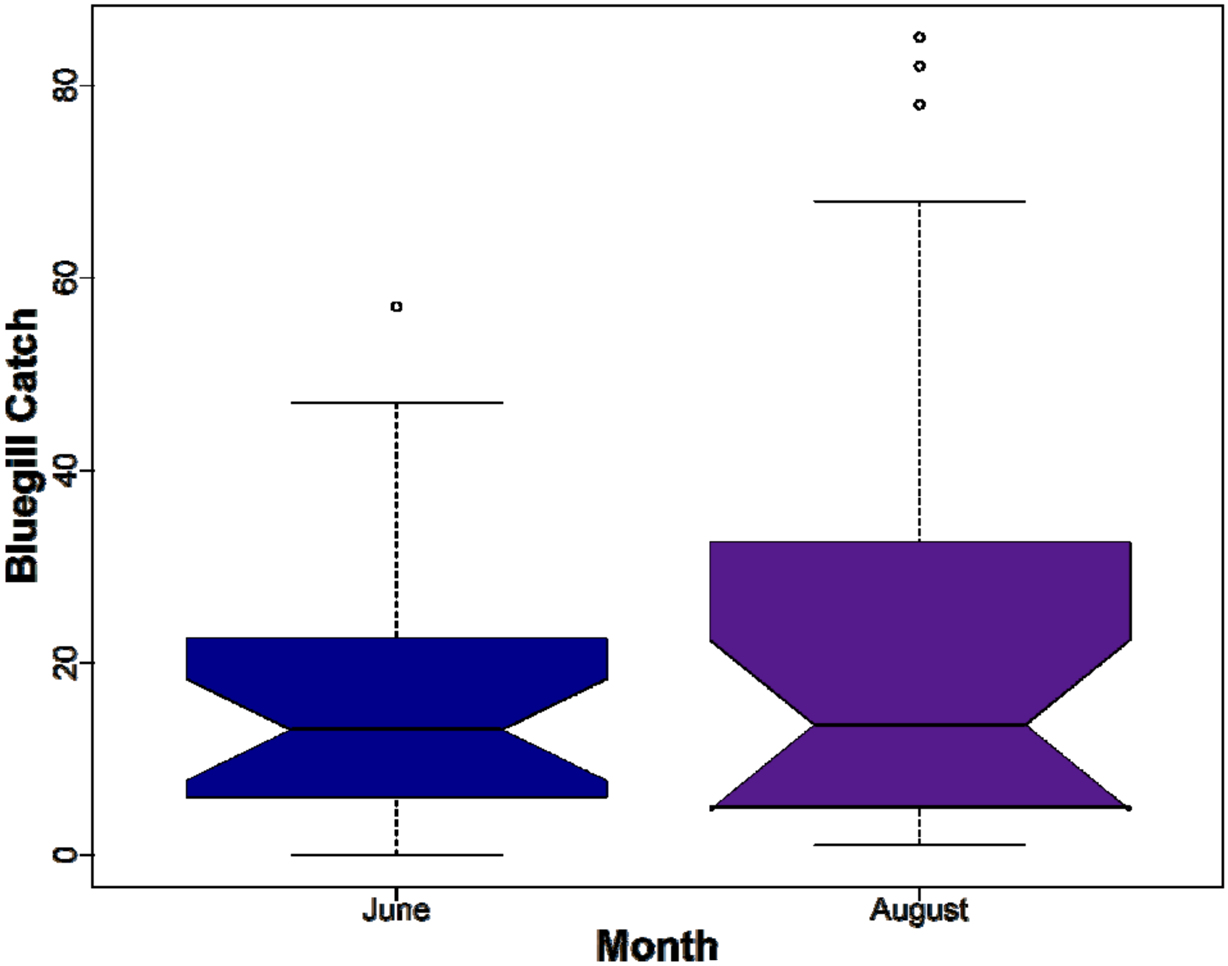

Figure 10. Notched box-and-whisker plot displaying median Bluegill catch per trap net for June and August 2014 and 2015. The box displays the interquartile range (IQR) (i.e., the 25th and 75th percentile) around the median and the whiskers are ± 1.5 times IQR. The notch in the box displays a 95% confidence interval.

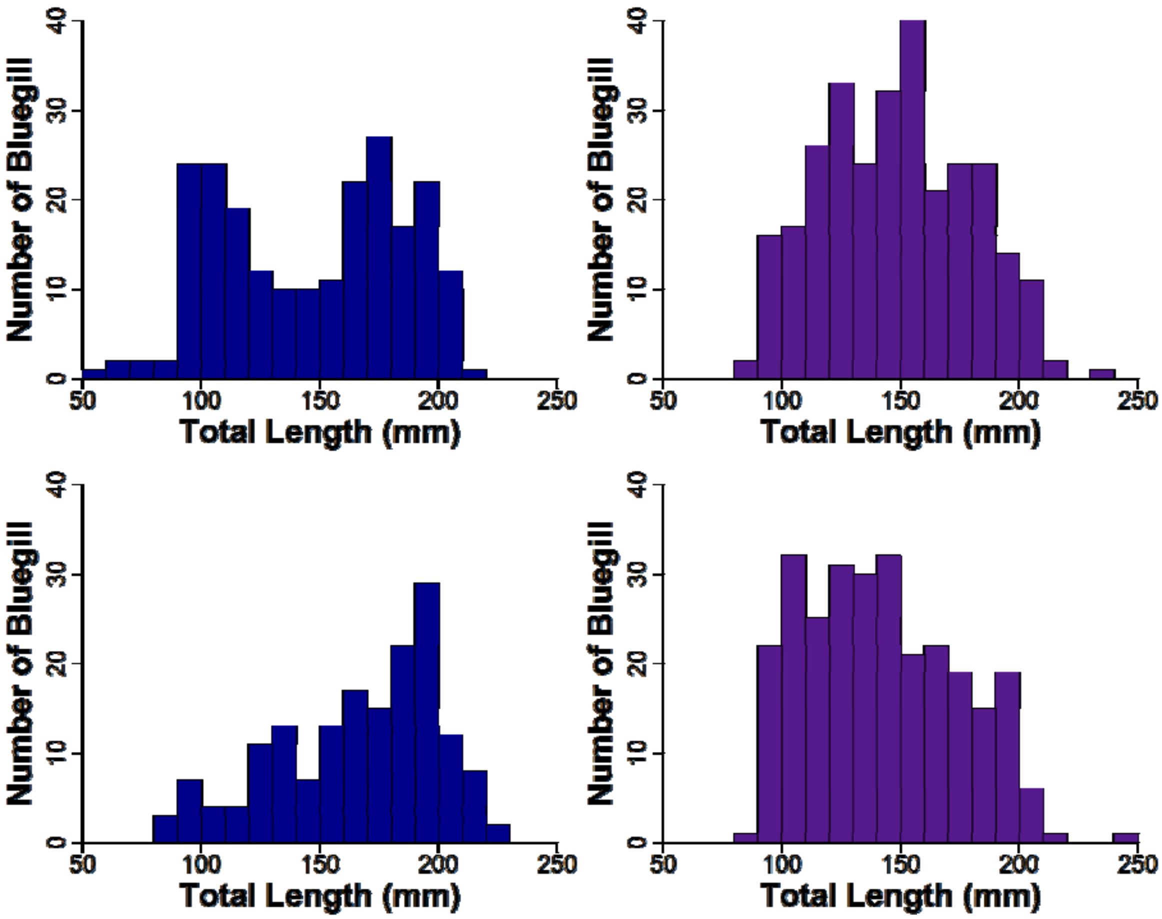

Figure 11. Length frequency histograms of Bluegill captured in trap nets in June (blue) and August (purple) of 2014 (top) and 2015 (bottom).

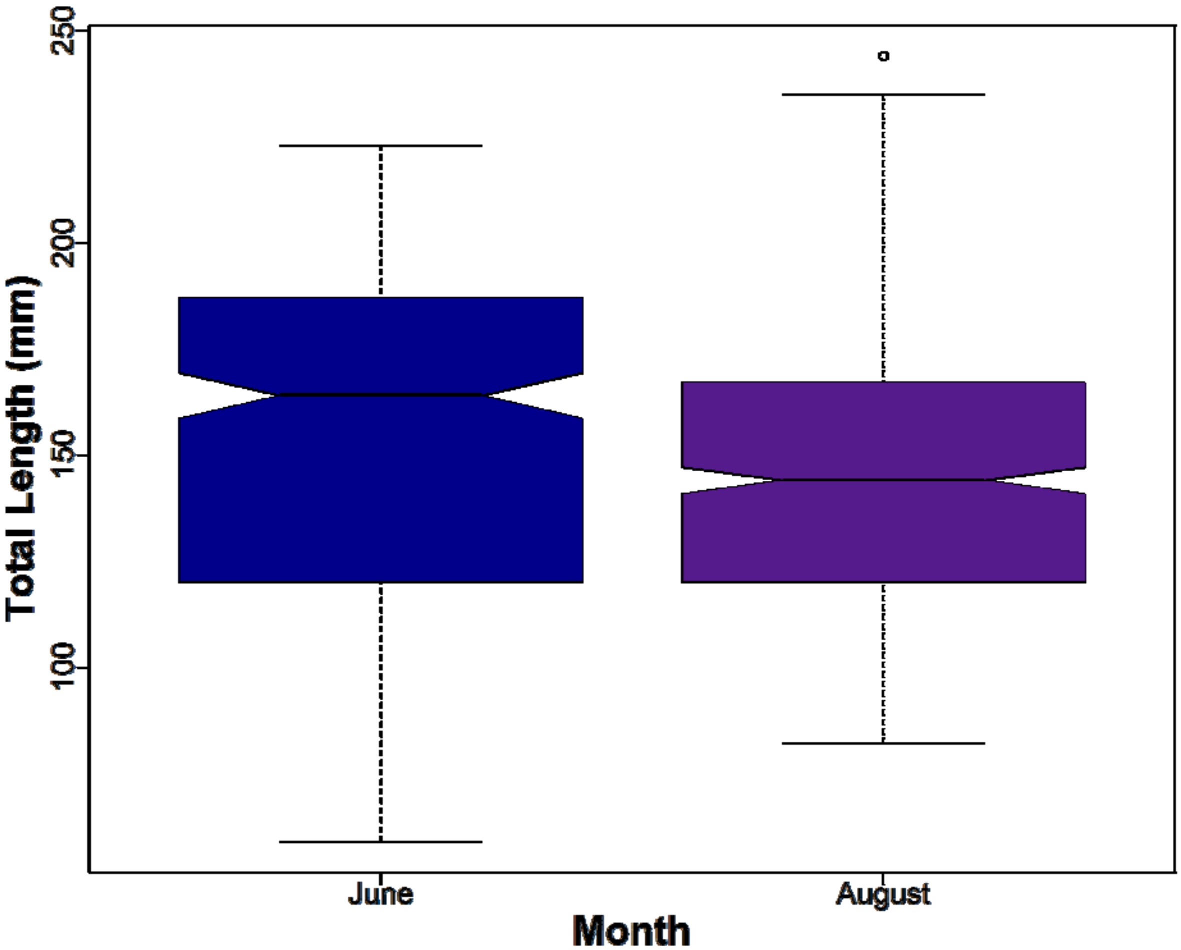

Figure 12. Notched box-and-whisker plot displaying median length of Bluegill captured in June and August 2014 and 2015. The box displays the interquartile range (IQR) (i.e., the 25^{th} and 75^{th} percentile) around the median and the whiskers are ± 1.5 times IQR. The notch in the box displays a 95% confidence interval.

Evaluation of White Sucker as an Indicator of Environmental Change

Introduction

White Sucker *Catostomus commersoni* was chosen as a species of interest for long-term monitoring for several reasons. This species exhibits flexible feeding behavior and tolerance for a wide range of environmental conditions, which may help explain their abundance and wide distribution in Minnesota lakes. They are important forage for other species, should be sensitive to warming water temperatures (Lyons et al. 2009), and are subjected to minimal angling exploitation (Scott and Crossman 1973). This species is found in 23 of 25 sentinel lakes and is resident in all but one of the 8 Tier I lakes (Trout Lake, Cook County). These factors make White Sucker a potentially excellent indicator species for sentinel lakes in Minnesota, especially in shallow, eutrophic lakes in the southern part of the state, where the effects of climate change may have a greater influence on habitat conditions (Stefan et al. 1995; Stefan et al. 1996).

Management surveys in Minnesota use standard gill nets during the months of June-August and typically capture only larger individuals (McInerny 2014). Other studies have demonstrated the effectiveness of seines and electrofishing gear for sampling White Sucker during spawning runs in rivers or tributary streams (Quinn and Ross 1982; Sylvester and Berry 2006), but these gears have not been widely used to target White Sucker in lakes. To effectively evaluate White Sucker populations, better methods for sampling this species in lakes are needed to estimate important population parameters more effectively. While other researchers have demonstrated success counting annuli on otoliths, fin rays, and scales (Thompson and Beckman 1995; Sylvester and Berry 2006), preliminary results from phase 1 of long-term monitoring found poor reader agreement when examining aging structures (J. Stewig, MNDNR, personal communication). Further assessment of these aging structures was necessary to determine their usefulness in estimating age of White Sucker.

Methods

In an effort to learn more about White Sucker population dynamics and evaluate their usefulness as an indicator species, a mark-recapture experiment was undertaken at Pearl Lake (Stearns County). In the spring of 2014 and 2015, trap nets and nighttime boat electrofishing were used to capture White Sucker at or near predicted spawning sites as water temperature approached 10^{o}C. Captured fish were measured, a fin ray was removed and, if possible, sex was determined by manually squeezing the abdomen in attempt to expel eggs or sperm. Before release, fish were marked with a combination of passive integrated transponder (PIT) tags, T-bar anchor tags and fin clips. Capture histories of tagged White Sucker were inputted in to Program MARK (http://www.warnercnr.colostate.edu/~gwhite/mark/mark.htm#Documentation) where several open and closed population models were used in an attempt to estimate abundance.

Aging structures were examined from White Sucker captured in Phase 1 of this project (2008-2012) as well as fish captured during tagging. Lapilli otoliths were mounted in epoxy, sectioned through their nucleus, polished and examined with reflected light under a dissecting microscope. Annuli were identified as alternating light and dark zones with the light zones being counted. Pectoral fin rays were mounted in epoxy and a thin section was cut at the proximal end of the structure. Sections were viewed with transmitted light under a dissecting microscope and again annuli were identified as alternating light and dark zones. Photos were taken of each aging structure using a Leica DMS 1000 microscope with an integrated digital camera (http://www.leica-microsystems.com/products/stereo-microscopes-macroscopes/macroscopes/details/product/leica-dms1000/). This made it possible to view images and estimate age at a later date and by a second reader.

Results and Discussion

During 2014 and 2015, 229 White Sucker were captured and tagged during 10 capture events. Sixteen fish were recaptured in a later capture event, resulting in a population estimate of 937 fish (SE = +/- 485). Adult White Sucker appeared to be most vulnerable to electrofishing gear for a short window prior to and while spawning but were difficult to capture following the spawn. Furthermore, juvenile White Sucker (< 300 mm total length) were captured infrequently (n=23) and in low numbers, making examining early life history traits of this species difficult.

While annuli were identifiable on some sectioned otoliths, it was often difficult to identify the first and second annuli. This makes it difficult to estimate growth in the early stages of life and leads to imprecise age estimates. It was easier to identify the first and second annuli on pectoral fin ray sections, but as the fish get older annuli crowd near the edge of the structure and become difficult to distinguish. This also leads to imprecise age estimates. Neither structure produced consistent images that are suitable for precise age estimation.

Standard summer surveys using gill nets and trap nets typically capture large White Sucker, but the sample size is usually small and may not be representative of the population (McInerny 2014). While our targeted spring sampling captured more individuals, a substantial amount effort was required and these fish are mostly older adults that are difficult to age. Furthermore, while it was possible to calculate a population estimate, the effort needed to obtain that figure was so great that it would preclude work on other, more promising indicator species. Based on the aging structures we examined it should be possible to obtain precise age estimates for fish younger than 4 years old, but fish of this age were difficult to capture. White Sucker were difficult to capture and age, thus do not appear to be a practical indicator species for long-term monitoring designed to detect influences of ecological stressors on lake biota.

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Summary of Sentinel Lakes Pelagic Forage Fish Sampling

The pelagic forage fish community was sampled annually with vertical gillnets and hydroacoustics during 2013-2015 in three long-term monitoring lakes (Carlos, Elk, and Ten Mile). Surveys occurred during peak stratification when >20°C epilimnetic temperatures caused coolwater pelagic species to move deeper in the water column thereby increasing their susceptibility to the sampling gear.

Individual gangs of vertical gillnets were comprised of seven mesh sizes (9.5mm, 12.7mm, 19.0mm, 25.4mm, 31.8mm, 38.1mm, and 44.5mm bar measure) of varying widths (0.9m, 0.9m, 1.2m, 1.8m, 3.0m, 3.0m, and 3.0m, respectively) set overnight in the deepest location(s) in each lake (Figure 1). Information recorded for captured individuals included net location, species, mesh, depth of capture (m), length (mm), and weight (g). Additionally, temperature and dissolved oxygen profiles were collected during each survey.

Hydroacoustic surveys were conducted at night ~1 hour after sunset with a Simrad 38-kHz 9.6° split-beam transducer and EK60 transceiver. Sampling transects were equally spaced across each lake at depths >4 m (Figure 1). Vertical gillnet and profile information were used to interpret hydroacoustic data, which were analyzed using the MNDNR Standardized Hydroacoustic Data Analysis Protocol for Inland Pelagic Fish Populations (unpublished document available upon request).

Age-0 cisco were removed from the hydroacoustic analysis using a minimum threshold because this age class was not equally susceptible to sampling gear due to differences in growth and the timing of surveys in different lakes. Depending on timing of the survey there may be a large pulse of age-0 cisco picked up by the gear that would greatly inflate density estimates; however, first year mortality rates may be high and variable, so if the lake was resampled the following year, or even a few weeks later, density estimates may be drastically different if including age-0 cisco. Thus, eliminating age-0 Cisco from the analysis reduced effects of abundance fluctuations, resulting in more comparable cross-lake evaluations, which are obtainable when focusing on age-1+ cisco. The minimum threshold was identified using a cluster analysis on hydroacoustically-derived length measurements of individual fish. In Elk and Carlos, the minimum threshold varied each year and corresponded well to the maximum size of age-0 Cisco captured in vertical gillnets. In Ten Mile, the cluster analysis did not detect a clear minimum threshold and age-0 Cisco could not be easily identified in vertical gillnets due to a truncated size structure, so a consistent minimum threshold of ~93 mm was applied. A maximum threshold eliminating backscatter from species such as Walleye (*Sander vitreus*), Northern Pike (*Esox lucius*), and Lake Whitefish (*Coregonus clupeaformis*) was also applied using results from the cluster analysis and vertical gillnet data.

Lake-wide abundance and biomass were estimated for each lake and year. These estimates were divided by the volume of water in each lake >6 m bottom depth to estimate standardized density and biomass.

Results and Discussion

In the three long-term monitoring lakes where the pelagic forage fish community was assessed, cisco (*Coregonus artedi*) was the dominant species located below the epilimnion (Figure 2). In Ten Mile, Lake Whitefish were interspersed at similar depths as cisco (Figure 2) but were removed from the hydroacoustic analysis due to their low abundance. In Elk, high catch rates of Yellow Perch (*Perca* *flavescens*) were observed in the epilimnion (Figure 2); however yellow perch were not included in the hydroacoustic analysis because the gear does not effectively sample shallow waters.

The size structure of Cisco differed between lakes over the three years of the study (Figure 3). In Elk, the length distribution increased as a large year class of presumably age-1 fish in 2013 grew in length and weight during subsequent years (Figure 3). Compared to Elk, the size structure of Cisco in Carlos and Ten Mile appeared truncated, with fish longer than 260 mm and 160 mm in each respective lake infrequently caught in vertical gillnets. Hydroacoustic length distributions were generally similar to vertical gillnet length distributions (Figure 3), although neither distribution may have been representative of the true population. Vertical gillnet are size-selective, so that fish with lengths between the optimum for each mesh have a lower probability of being caught (Hamley 1975). Size-selectivity issues are also compounded by increased encounter rates for larger fish that have higher swim speeds (Rudstam et al. 1984). Although hydroacoustic gear is not size-selective, the orientation of the fish relative to the transducer affects size estimates (Buerkle 1987, Foote 1980). A broader hydroacoustic size distribution will result compared to the true population because not all fish will be located perpendicular to the transducer (i.e. fish may be swimming up or down, or the transducer itself may be tilted). Additionally, separating the hydroacoustic signatures of individual fish in lakes with high densities is difficult, and overlapping fish would result in larger size estimates compared with the true population. Hydroacoustic length bias due to high fish densities may have been the case in Ten Mile, where hydroacoustic size distributions were consistently larger than fish captured in vertical gillnets (Figure 3).

Abundance and biomass estimates were generated for each lake by year (Tables 1 and 2). In general, error estimates overlapped between years, indicating that annual abundance and biomass estimates did not vary significantly. The exception was Elk Lake, where there was a significantly smaller Cisco population in 2014 compared with the other two years of the study (Table 1), possibly due to a failed year class as indicated by vertical gillnet data where only a few presumed age-1 Cisco were captured in 2014 and no presumed age-2 Cisco were captured in 2015 (Figure 3). Additionally, in Elk Lake there was a significant increase in biomass in 2015 compared with 2013 (Table 2) likely caused by an increase in size structure and mean weight of the population over time (Figure 3). When comparing standardized density between lakes, Ten Mile estimates approximately quadrupled those of Carlos and Elk (Figure 4). However, Elk had the highest standardized biomass of the three lakes, with estimates increasing from 2013-2015 (Figure 4).

Differences in cisco populations may have been related to the amount of available hypolimnetic habitat in each lake. In Elk, cisco were squeezed into warmer metalimnetic waters as a result of hypoxic conditions by mid-July each year compared with Carlos and Ten Mile that had a greater reservoir of hypolimnetic oxygen during the years of this study. Decreased growth rates of Cisco have been previously observed in an experimental lake where hypolimnetic habitat was artificially increased through oxygen injection (Aku and Tonn 1997). In the same study, Cisco density was positively correlated with increases in hypolimnetic habitat (Aku and Tonn 1997) and high Cisco densities have been found to be inversely related with length-at-age (Bowen et al. 1991; Rudstam 1984). In the three long-term monitoring lakes assessed in this study, more research could be done to evaluate Cisco population dynamics and whether differences in length distributions between lakes are potentially caused by density-dependent growth related to limited food availability, metabolically-reduced growth related to colder temperatures, or a combination of these factors.

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Table 1. Hydroacoustic population estimates (total number) of pelagic cisco for Carlos, Elk, and Ten Mile for each year of the survey, 2013-2015. The mean and 95% confidence intervals are displayed.

Lake	2013	2014	2015
Carlos	732,000 ± 160,000	650,000 ± 112,000	520,000 ± 60,400
Elk	72,100 ± 12,500	41,800 ± 7,850	60,000 ± 7,850
Ten Mile	5,680,000 ± 1,120,000	4,550,000 ± 887,000	6,500,000 ± 1,910,000

Table 2. Hydroacoustic biomass estimates (total kg) of pelagic cisco for Carlos, Elk, and Ten Mile for each year of the survey, 2013-2015. The mean and 95% confidence intervals are displayed.

Lake	2013	2014	2015
Carlos	39,400 ± 8,660	37,000 ± 6,410	33,400 ± 4,090
Elk	7,870 ± 1,660	9,180 ± 2,050	13,500 ± 1,860
Ten Mile	87,300 ± 17,100	92,500 ± 17,400	107,000 ± 29,300

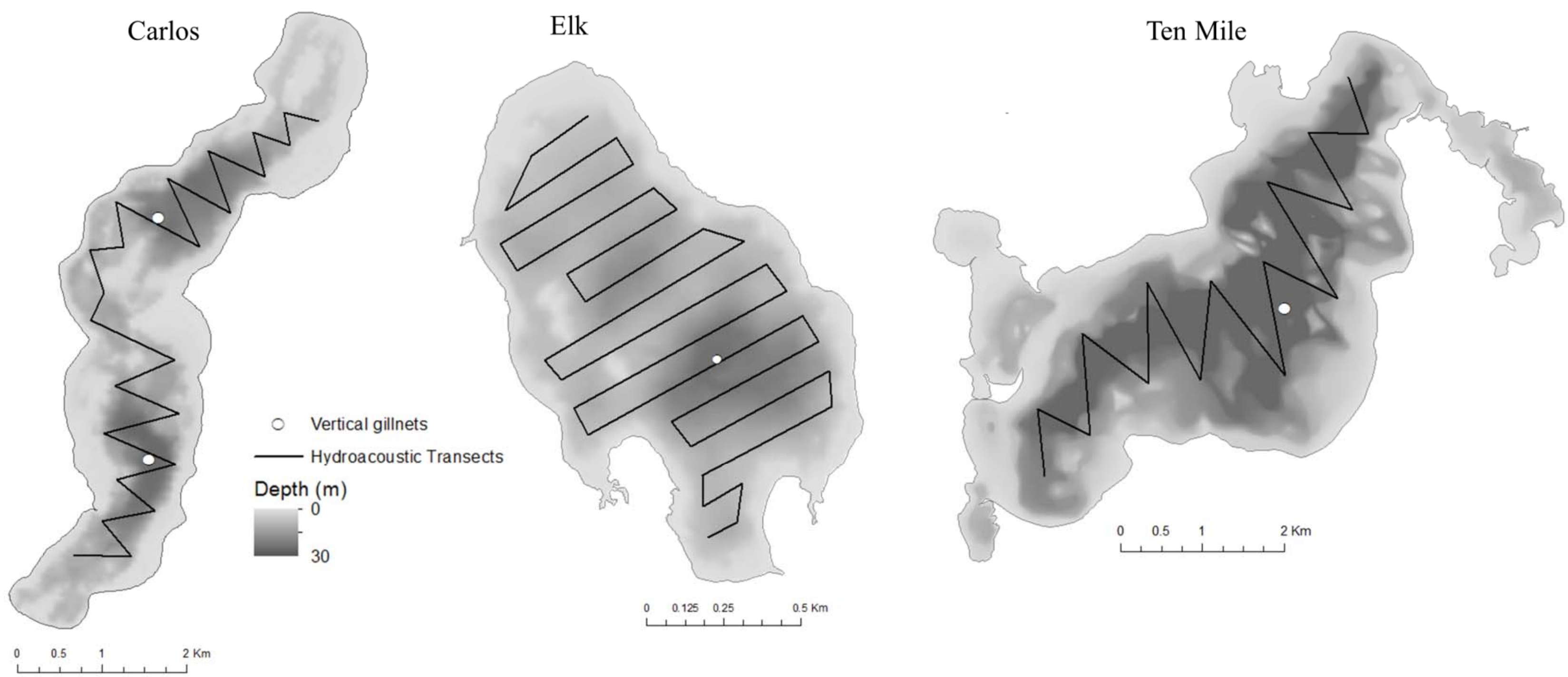

Figure 1. Maps of the vertical gillnet locations and hydroacoustic transects on the three long-term monitoring lakes where the pelagic fish community was sampled.

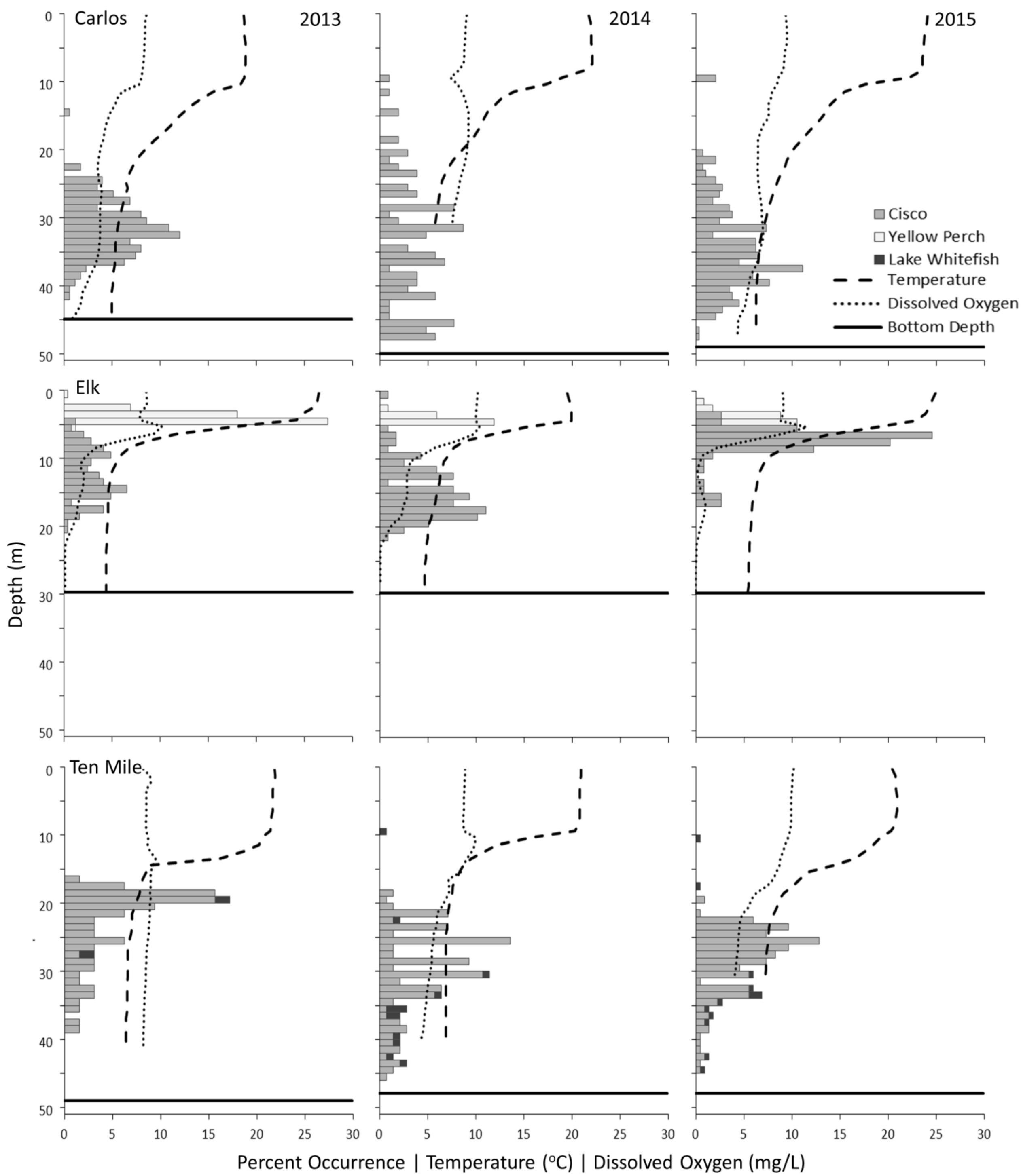

Figure 2. Vertical depth distribution of cisco in each lake by year compared with temperature and dissoved oxygen profiles collected at the time of the pelagic fish survey. Cisco were generally located below the thermocline if hypolimnetic oxygen was available.

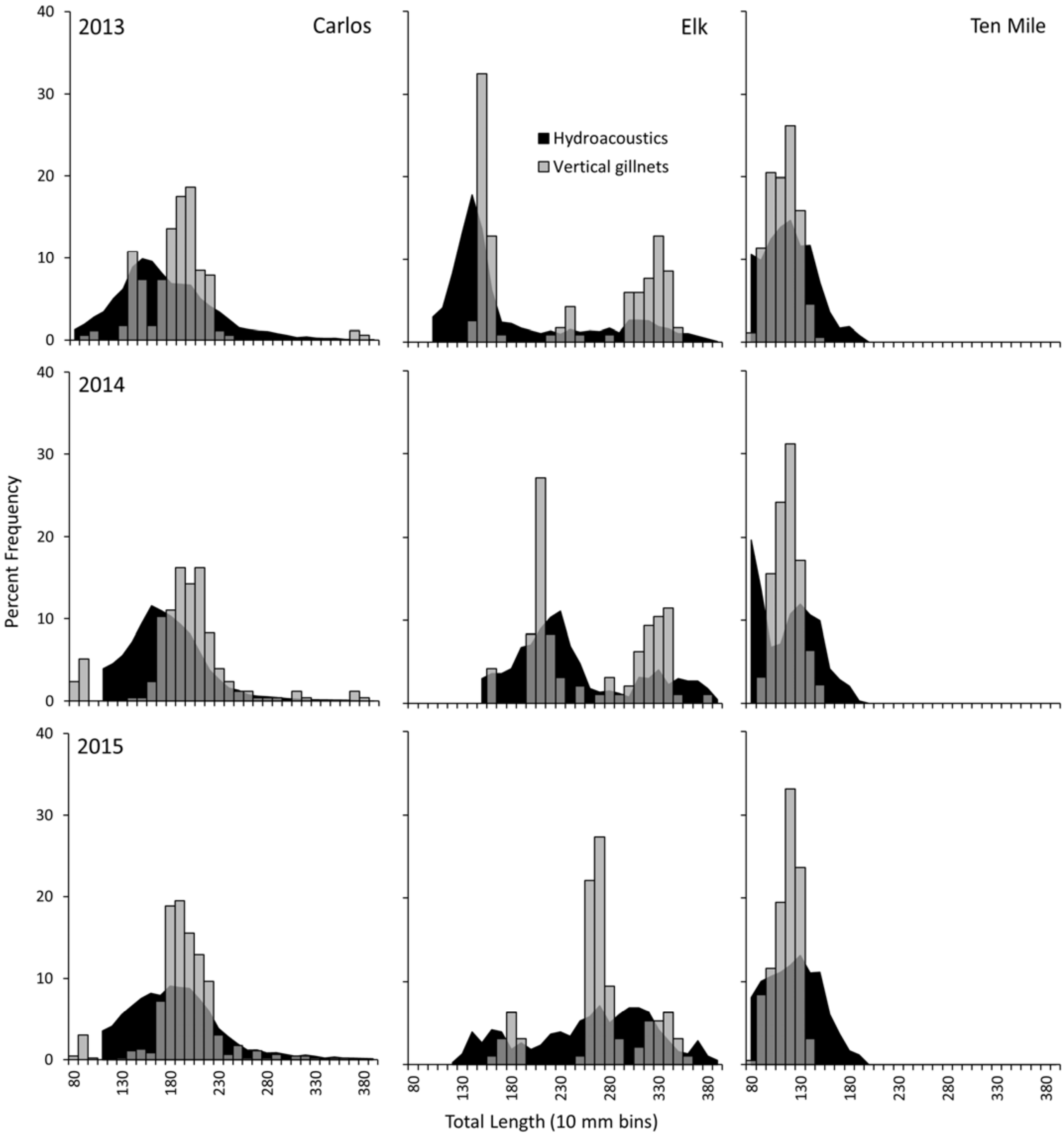

Figure 3. Length distribution of pelagic fish caught in vertical gillnets and sampled with hydroacoustics in each lake by year. The left edge of the hydroacoustic distribution represents the minimum size of analysis applied each year to eliminate age-0 cisco.

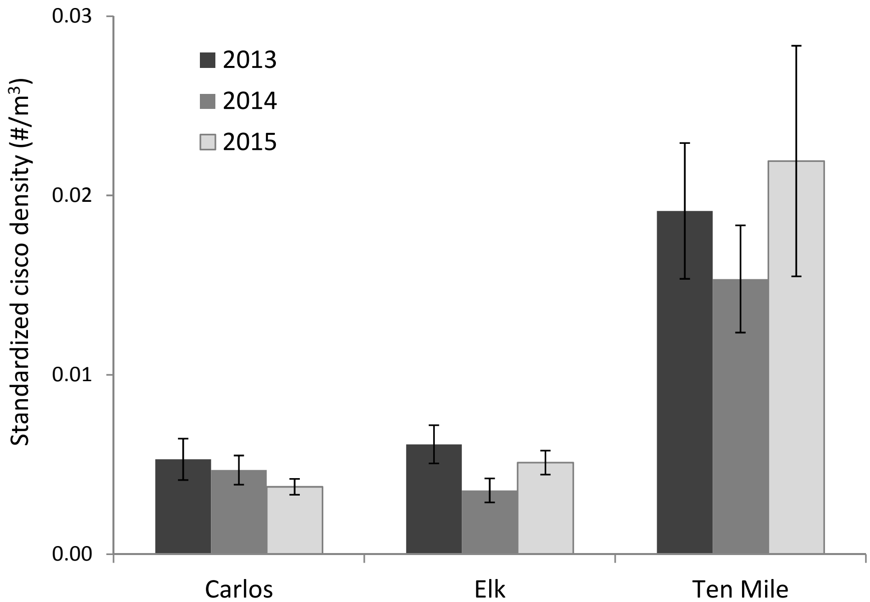

Figure 4. Standardized cisco density (total lake-wide population divided by volume of water >6 m bottom depth) for each lake and year. Error bars are 95% confidence intervals.

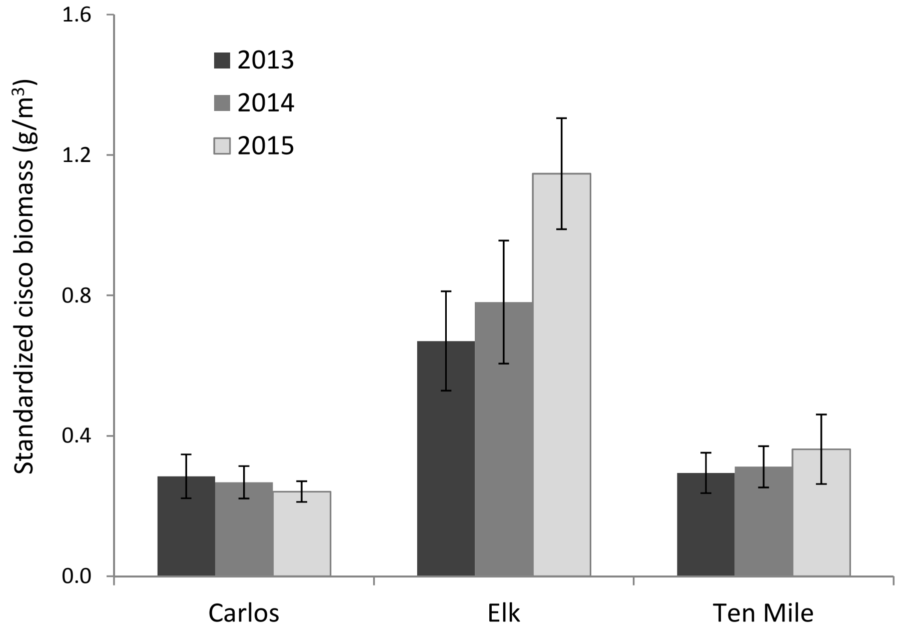

Figure 5. Standardized cisco biomass (total lake-wide biomass divided by volume of water >6 m bottom depth) for each lake and year. Error bars are 95% confidence intervals.

Bathymetry and Submerged Aquatic Plant Biovolume Mapping

Hydroacoustic surveys of depth and submerged aquatic plant biovolume (percent water column filled with plants) were conducted on ten Sentinel Lakes during summer 2015. Transects were created in ArcMap 10.2 (https://desktop.arcgis.com/en/arcmap/) following instructions provided by Navico BioBase (http://www.cibiobase.com/). Lakes were surveyed using transects spaced 50 m apart and perpendicular to the long axis of the lake, except for Carlos, where 65 m transects were used (Figure 1). Surveys were conducted by driving a boat, fitted with a Lowrance HDS sonar unit, along transects while recording the sonar log. Transects were driven at a speed of approximately 5.5 miles per hour or less. Most lakes were sampled in entirety, meaning that in addition to vegetation biovolume maps, detailed bathymetry data were also collected simultaneously during these whole-lake surveys. Carlos received a survey of the littoral-zone only, while the littoral zone of Elk lake was resampled in late August on two consecutive repeat surveys to assess repeatability of results by attempting to drive the exact same transect route. Data were then uploaded to the BioBase website as .sl2 files where they were analyzed by a cloud based signal-processing system software (BioBase 2014). The user can choose to merge several files from one lake into a single file and also choose the size of the buffer around each transect, which determines how much area is estimated by interpolation (kriging) on either side of the transect. Buffers were selected to minimize areas without estimated biovolume, while still providing precise estimates (grid cell sizes ranged from 5-6m).

Reports generated by the BioBase software produce summary statistics as well as a vegetation biovolume map for each lake (Table 1; Figures 2, 3). Summary statistics include percent area covered (PAC; i.e., overall surface area that has vegetation growing), biovolume plant (BVp; i.e., percentage of the water column taken up by vegetation when vegetation is present; areas that do not have any vegetation growing are not used in this calculation) and biovolume all water (BVw; i.e., average percentage of the water column taken up by vegetation regardless of whether biovolume exists; a zero value is entered into the calculation where no vegetation exists, thus reducing overall biovolume of the entire area covered by the survey). Vegetation biovolume maps can be recreated in ArcMap 10.2 (Figure 4) allowing users to run additional analyses and calculate a wide variety of desired statistics. For lakes where the time required to complete survey was recorded, average data collection rates were 76 acres/hr for whole lake surveys (range= 67-99 acres/hr), based on data in Table 1.

Multiple surveys were conducted on Elk Lake to assess the repeatability of these surveys. On August 25, 2015 two surveys were conducted in the same day, which yielded fairly similar results, though it is evident on the maps that there were substantial differences in locations of areas with high biovolume estimates (Figure 5). Considerable differences in BVp were observed when comparing surveys conducted on July 1, 2015 (BVP = 41.9) and August 25, 2015 (BVp range = 24.4-25.6) (Table 1). This pattern likely reflects the natural seasonal change in macrophyte biomass throughout the growing season, peaking in mid-summer and then beginning to senesce by late summer (Wetzel 2001).

While the main objective of these surveys was to quantify plant biovolume in lakes, considerable depth data were also collected, allowing the production of high resolution depth contour maps. Using Lowrance's Sonar Viewer 2.1.2 software, recorded sonar logs were viewed, and data were exported for processing (http://lowrance-sonar-viewer.software.informer.com/). The resulting files were read into ArcMap 10.2 where geostatistical tools and smoothing functions were used to produce contour lines. The resulting contour map for Belle Lake, showing 1 foot depth contours, provides much more detailed information than the map currently available on Minnesota DNR's LakeFinder, which was produced in 1975 (compare Figures 6 and 7). The same process was used to create a bathymetric map of Bear Head, which is a relatively deep lake with an abundance of complex underwater structure (Figures 8 and 9).

Several potential limitations of the BioBase system were identified upon detailed examination of the post-processed data. The kriging process appears to produce maps with unrealistic vegetation growth patterns. Other limitations include occasional false detections of vegetation and inconsistencies between the sonar-recorded and actual depths at discrete locations. We discuss each of these issues below, and make several recommendations for improving data quality.

Kriging is a method of interpolation that uses measured values to estimate values at unmeasured locations. This method, when used with transect data, produces models that predict vegetation growth with both overestimation and underestimation biases over un-surveyed areas, as can be seen in Figures 10 and 11. Vegetation is often depicted to be growing deeper (overestimated) in the un-surveyed area than at measured locations along transects (Figure 10). It also produces patchy vegetation beds with high biovolume estimates directly along transects, and lower biovolume estimates between transects (Figure 11). This pattern of vegetation growth seems unlikely, thus poorly reflecting how vegetation is actually growing. While more sophisticated and precise kriging models could be developed to interpolate this data, it would require geospatial expertise and would incur additional staff time and expense. Designing a sampling regime that included transects perpendicular to the parallel transects spaced along the lake's long axis (i.e., cross-hatched transect design) or concentric circles in the littoral zone may also produce results more representative of actual lake conditions and should be further explored.

Maximum depth of vegetation growth was a parameter of interest to the Sentinel Lakes Program, and should be easily estimable from this data. While calculating this parameter for Cedar Lake and Lake

Carlos, estimates of maximum depth of vegetation growth were 45.1 and 32.3 feet, respectively. These results seemed unrealistic, and after further investigation into this parameter, it was apparent that there were false detections of vegetation at depths where vegetation was not actually growing. It appeared that the system was identifying clouds of zooplankton or macroinvertebrates as vegetation. This issue could potentially be solved by manually deleting data points where false detections occur. Such an effort is likely to be time-intensive and objective determinations of plants versus invertebrates may be difficult, or even impossible to make.

While looking for these false detections, it was also noted that there were some inconsistencies between the sonar log-recorded and sonar log-indicated depths. More specifically, the depth depicted on the sonar log was up to 35 feet different than the depth reported by BioBase for that location (Figures 12 and 13). The most drastic inconsistencies occurred on Cedar Lake, where there is large change in depth over a short distance. It is possible that this problem could be lessened by driving the boat slower in these areas and by mounting an external GPS antenna directly over the transducer. Particularly when using a boat with a console, it is important to mount a GPS antenna above the transducer to ensure location and sonar data are collected at the same point.

After conducting these surveys and going through the post-processing procedures described in this section, it appears that BioBase may not produce the detailed results and metrics desired in a long-term monitoring study (e.g., maximum depth of submerged aquatic vegetation). Hydroacoustic methods for surveying plants may be sufficient to detect large, lake-wide shifts in aquatic vegetation, however, considering that natural seasonal changes produced biovolume estimates 64% higher earlier than later in the summer, careful attention must be paid to survey timing to hope to separate inter-annual from within-year variation. Details of transect architecture and fully understanding the biases and errors associated with sonar equipment, as well as specific BioBase algorithms for plant height determination and kriging and interpolation of biovolume remain important considerations for using this technique as a long-term monitoring method. With further effort, it may be possible to design a survey strategy or analytical methods that more precisely captures the extent of aquatic vegetation growth changes in lakes.

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Table 1. Details and summary statistics for hydroacoustic plant surveys conducted during summer 2015 and analyzed using BioBase software. Abbreviations as follows: Type = Survey Type (E= entire lake surveyed, L= survey of littoral zone-only, PAC = Percent Area Covered (i.e., overall surface area that has vegetation growing), BVp = Biovolume plant (i.e., percentage of the water column taken up by vegetation when vegetation is present; areas that do not have any vegetation growing are not used in this calculation), BVw = Biovolume all water (i.e., average percentage of the water column taken up by vegetation regardless of whether biovolume exists).

Lake	Lake Size (acres)	Survey Dates	Type	Kriging Buffer (m)	PAC	BVp	BVw	Time Required to Complete Survey (hrs)
Bear Head	662	8/17-8/18	E	30	36.6	24.8	9.1	9.5
Belle	925	7/21-7/22	E	30	10.0	18.4	1.8	na
Carlos	2,605	8/14-8/24	L	35	na	22.5	15.3	na
Cedar	236	7/9	E	30	25.3	35.6	9.0	3.5
Carrie	89	7/23	E	25	8.6	28.5	2.5	na
Elk	303	7/1	E	30	28.3	41.9	11.8	4.3
Elk	303	8/25	L	25	na	25.6	12.0	2.3
Elk	303	8/25	L	25	na	24.4	11.3	2.5
Madison	1,446	7/30-8/4	E	30	26.9	28.8	7.8	na
Pearl	753	7/23-7/27	E	30	59.6	29.4	17.5	na
Portage	429	7/8	E	30	97.3	34.1	33.1	6.0
South Twin	1,126	8/7-8/14	E	30	58.9	31.5	18.6	11.3

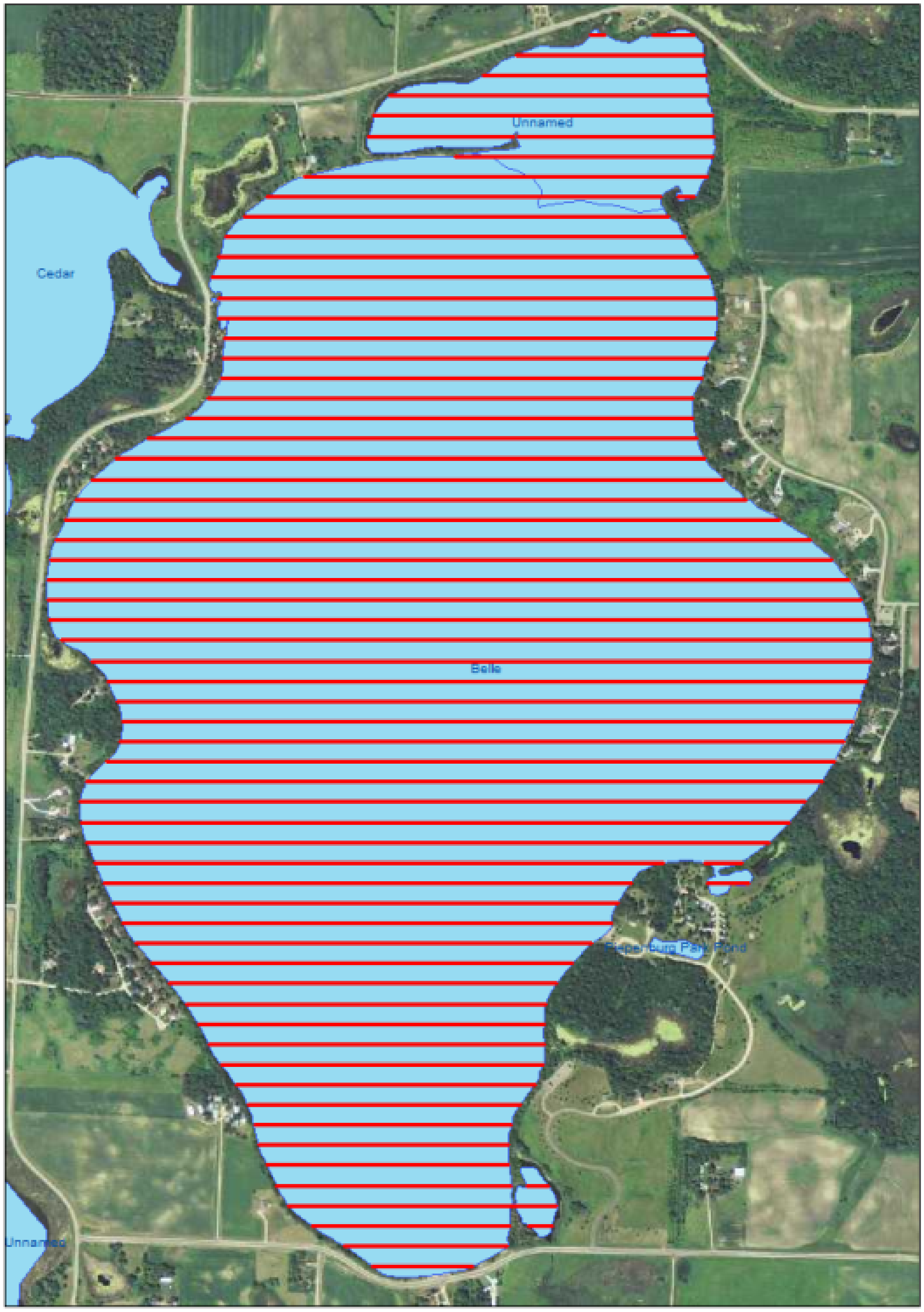

Figure 1. Biovolume survey transects with 50 m spacing created in ArcMap 10.2 for Belle Lake.

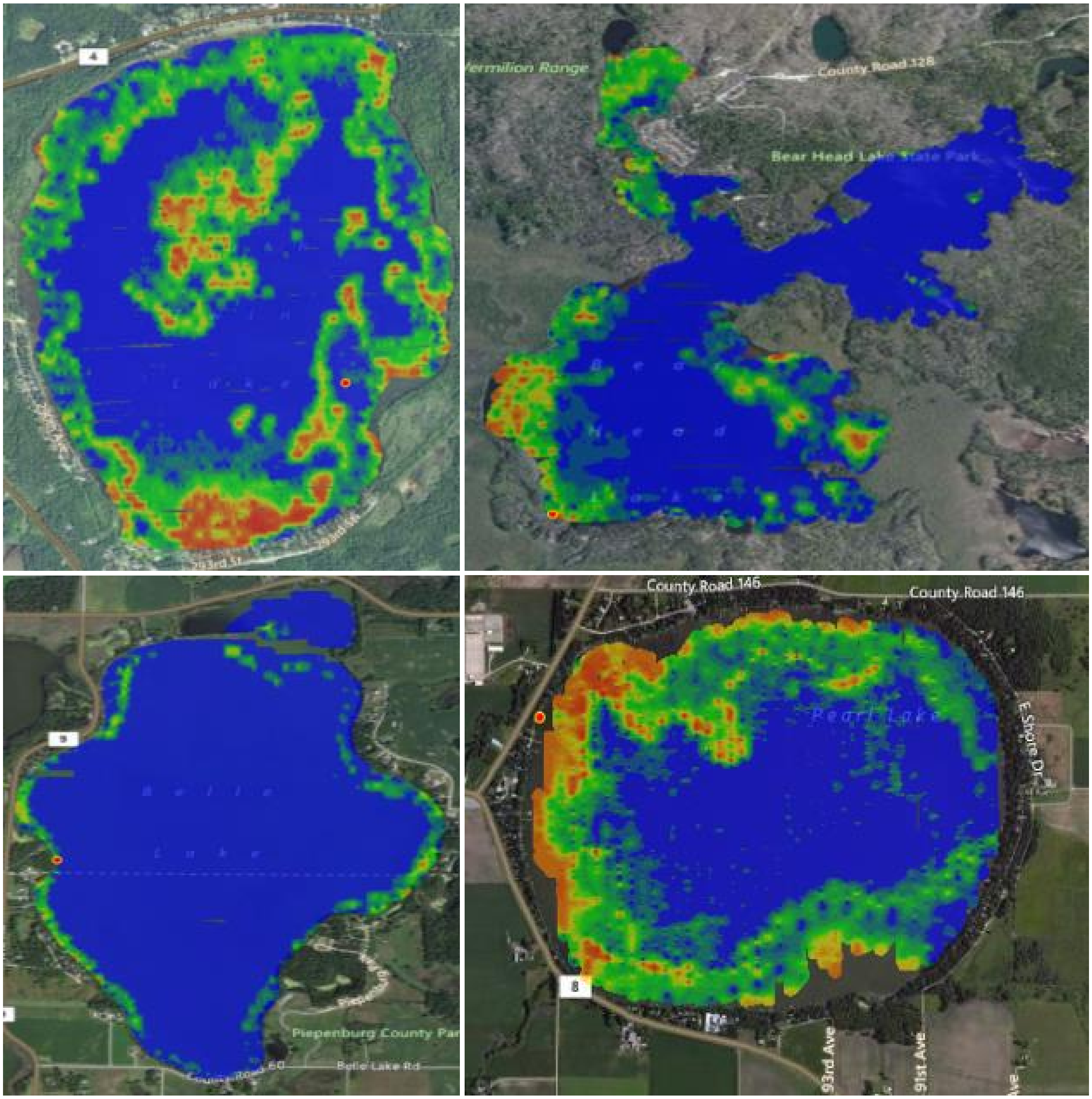

Figure 2. Vegetation biovolume maps produced by BioBase software for South Twin (upper left), Bear Head (upper right), Belle (lower left), and Pearl (lower right) lakes surveyed in summer 2015.

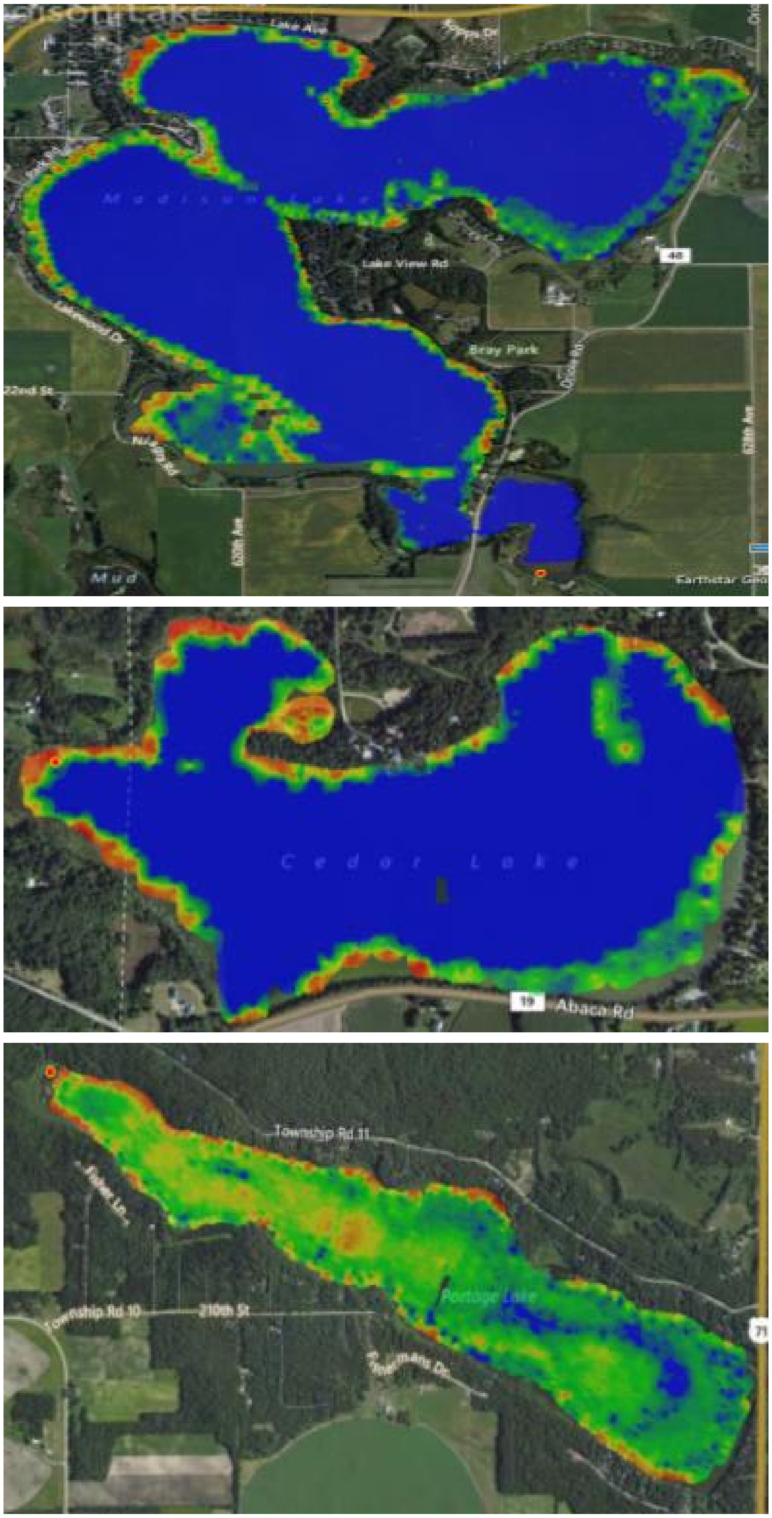

Figure 3. Vegetation biovolume maps produced by BioBase software for Madison (top), Cedar (middle), and Portage (bottom) lakes surveyed in summer 2015.

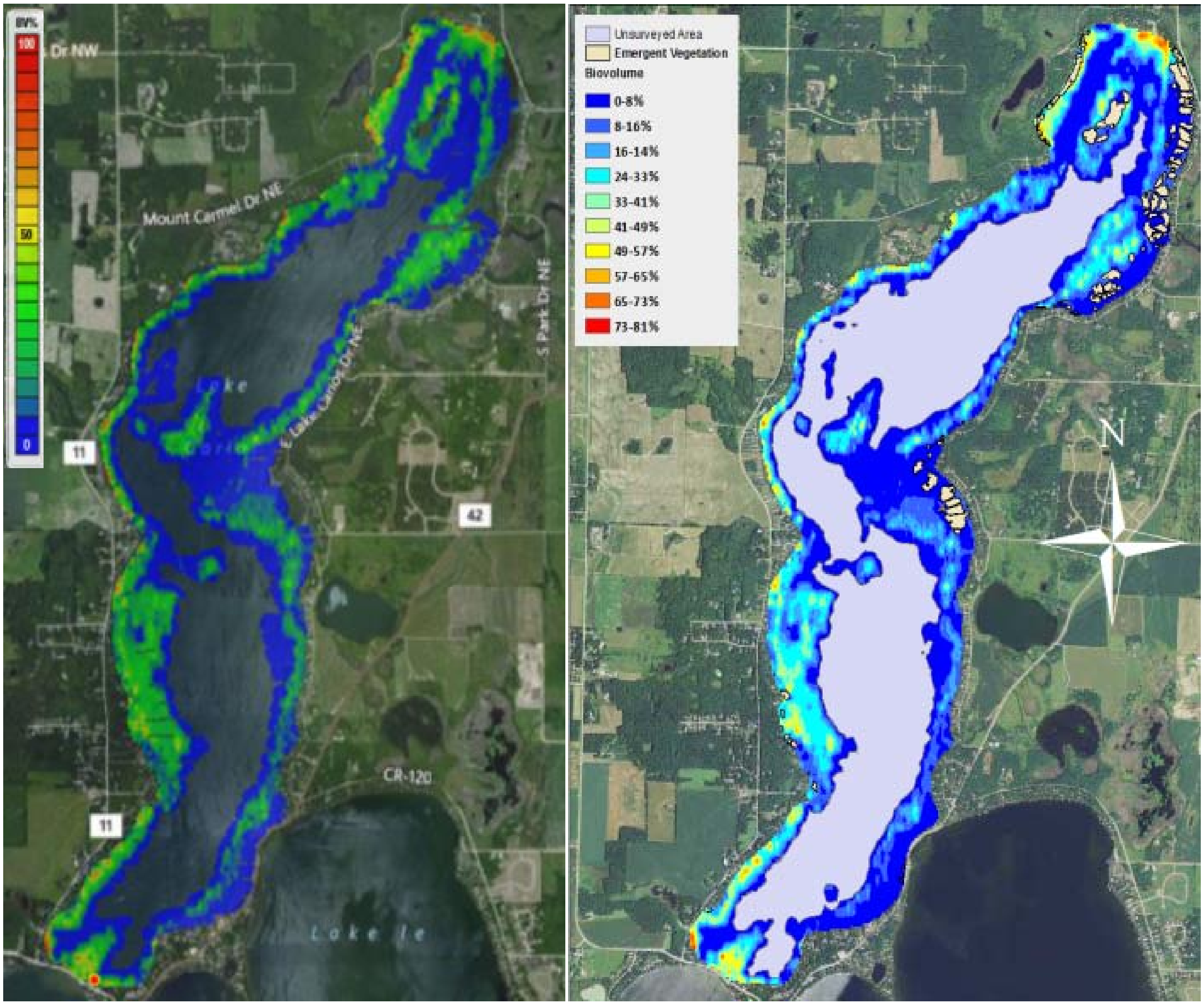

Figure 4. Vegetation biovolume map produced by BioBase software (left) and the vegetation biovolume map recreated in ArcMap 10.2 for Lake Carlos.

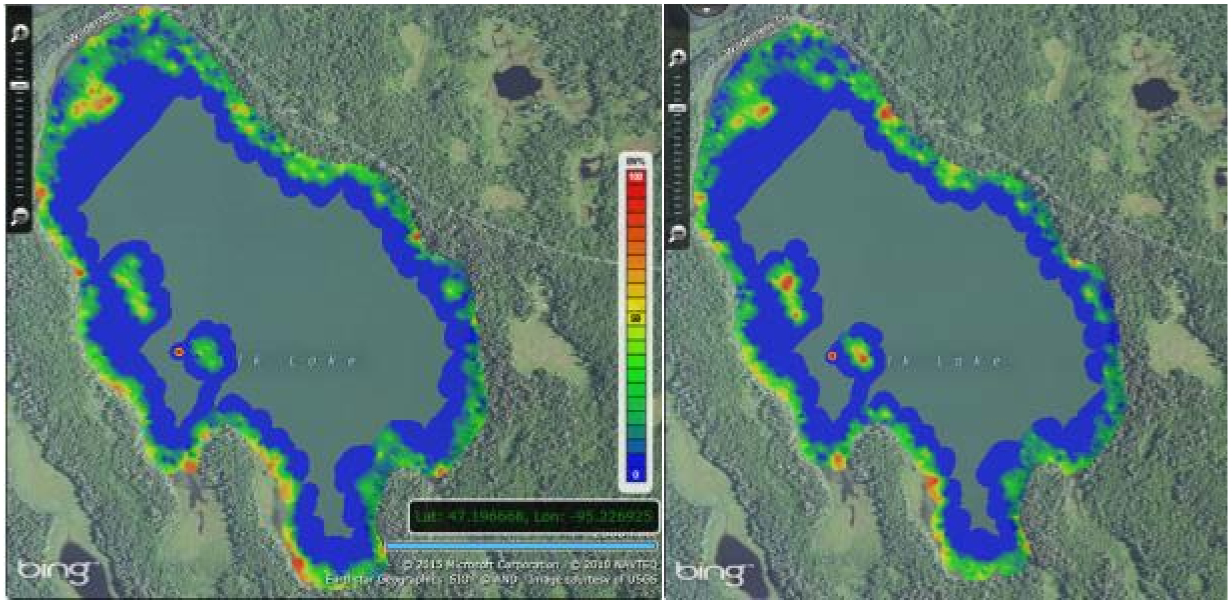

Figure 5. Vegetation biovolume maps produced from two surveys conducted on Elk Lake on August 25, 2015. While survey statistics were relatively similar for both surveys, the maps show some notable differences in locations where areas of high biovolume is estimated.

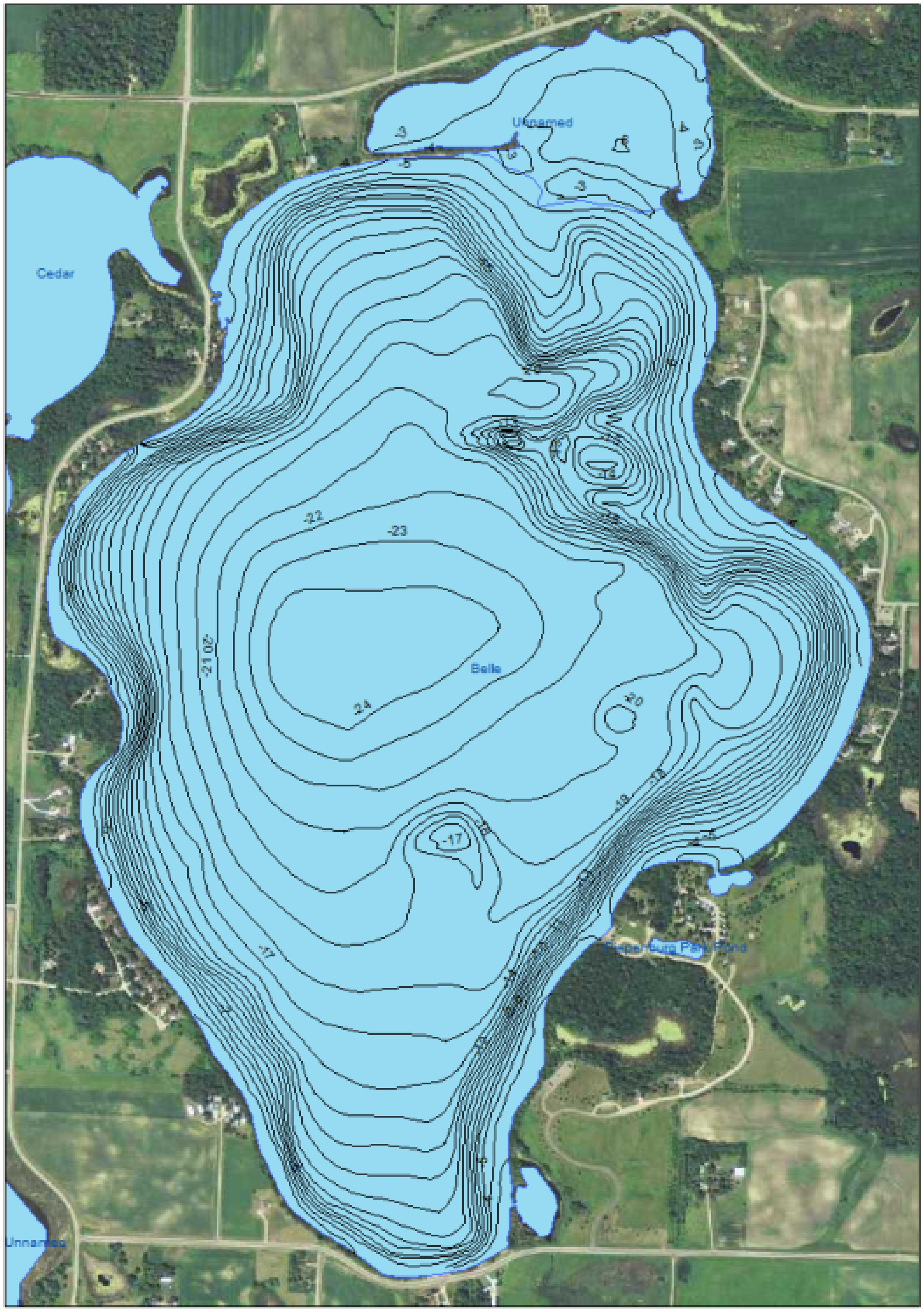

Figure 6. Belle Lake depth contour map created in ArcMap 10.2 using data collected using a Lowrance HDS in 2015.

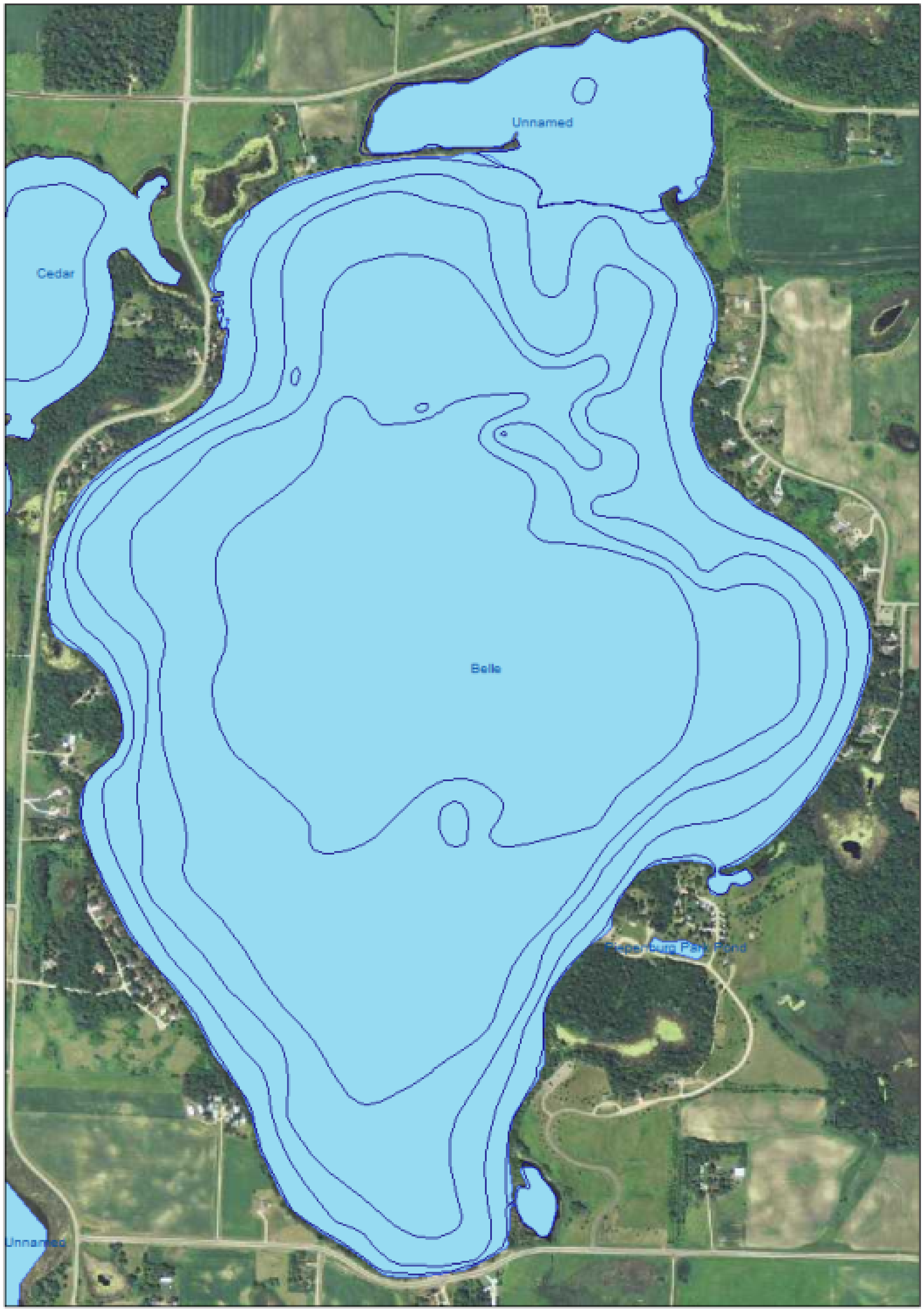

Figure 7. Belle Lake depth contour map, created in 1975, available to the public on MN DNR's LakeFinder.

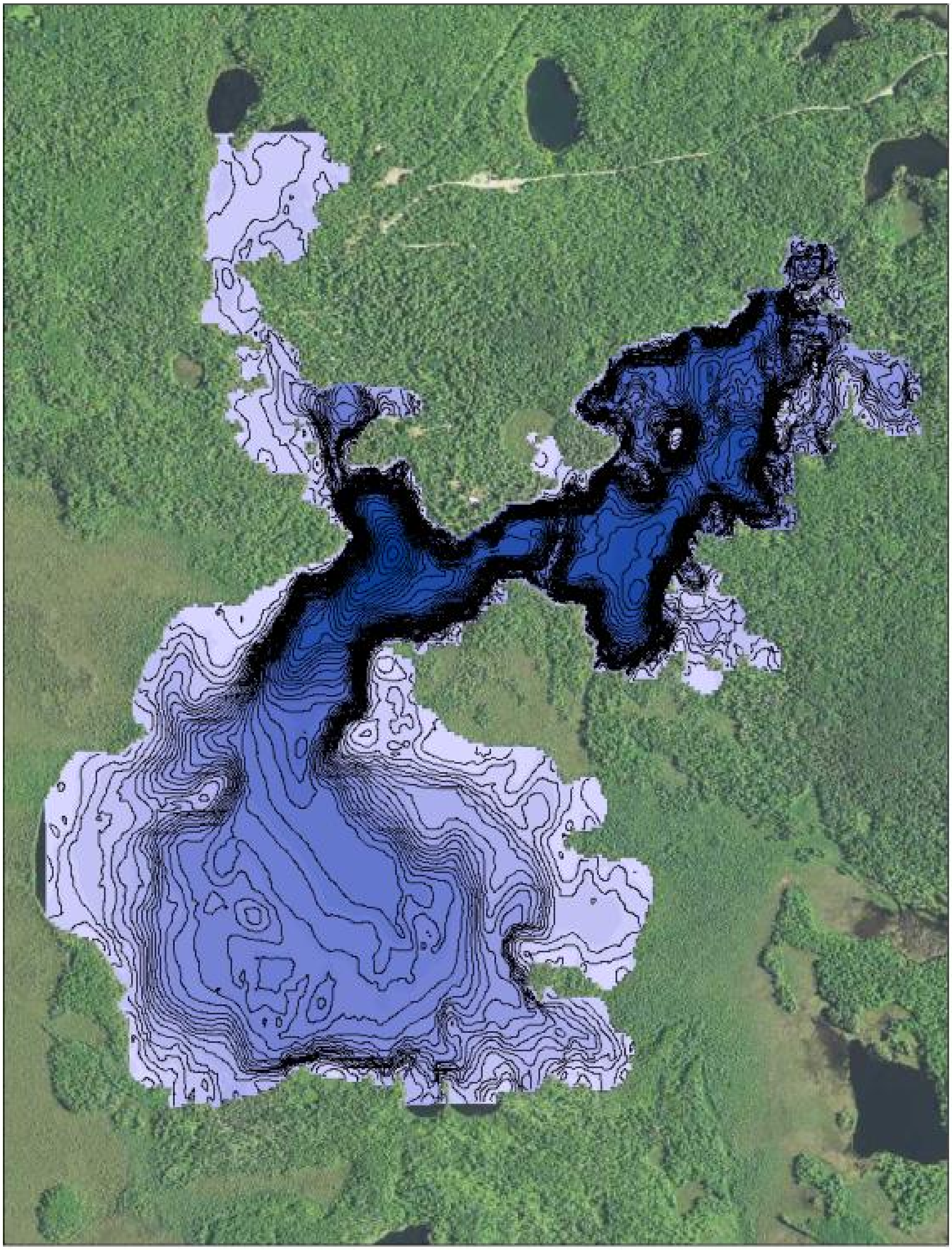

Figure 8. Bear Head Lake depth contour map created in ArcMap 10.2 using data collected using a Lowrance HDS in 2015.

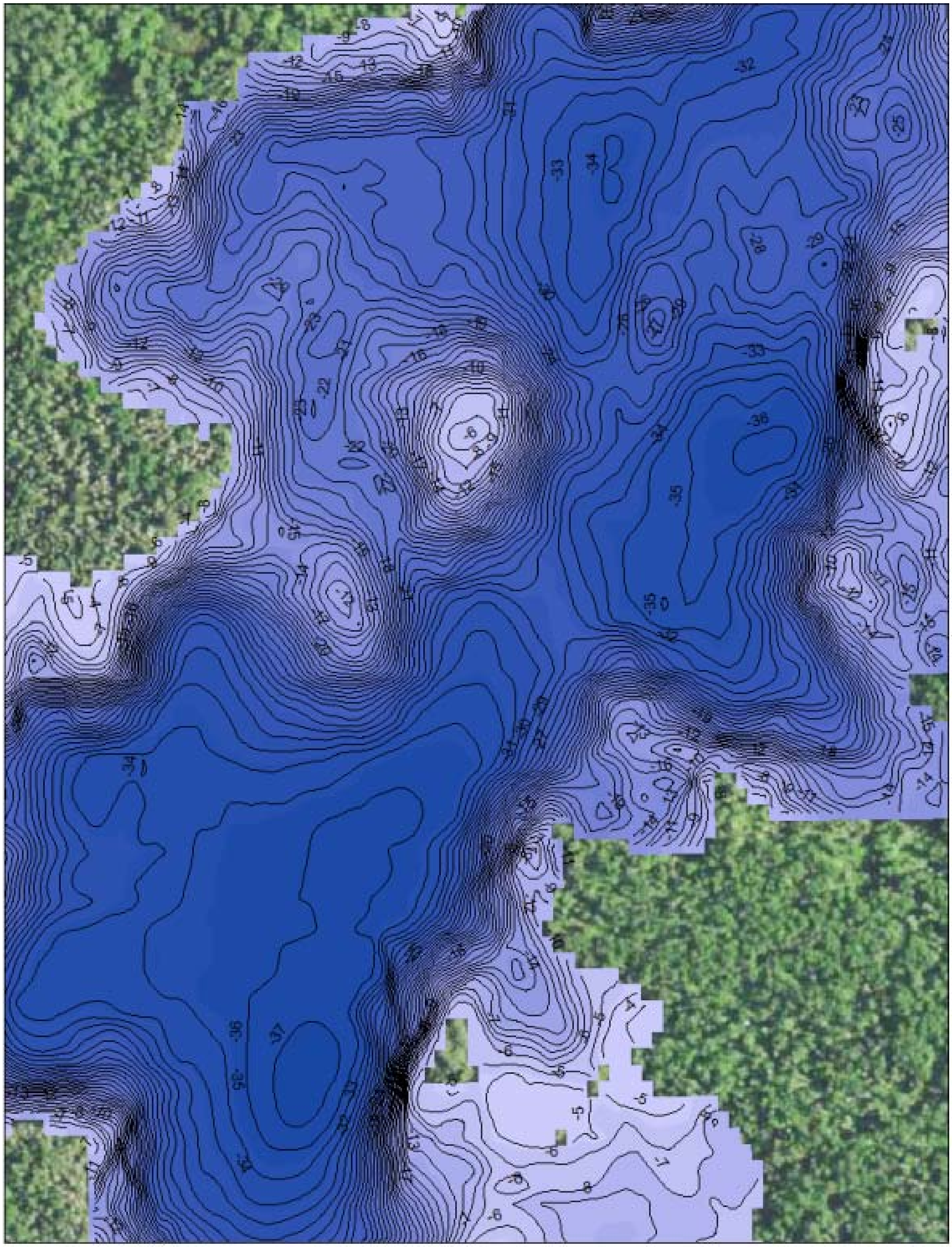

Figure 9. Depth contour map zoomed into the Northeast portion of Bear Head Lake.

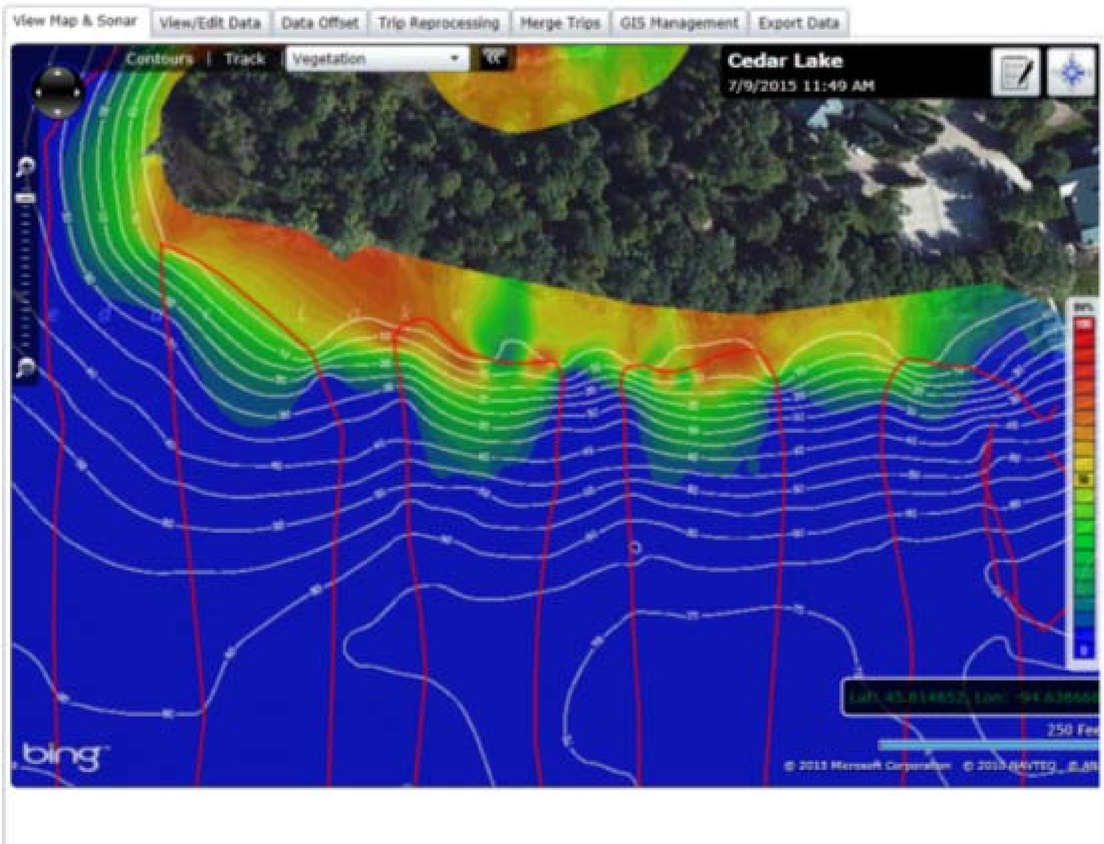

Figure 10. A zoomed in portion of Cedar Lake showing transect lines, in red, and vegetation biovolume. Krigging was used to estimate biovolume between transects, however biovolume is estimated between transects at depths deeper than where vegetation is actually present along transects. This problem overestimates biovolume and gives the illusion that vegetation is growing at depths deeper than is actually recorded.

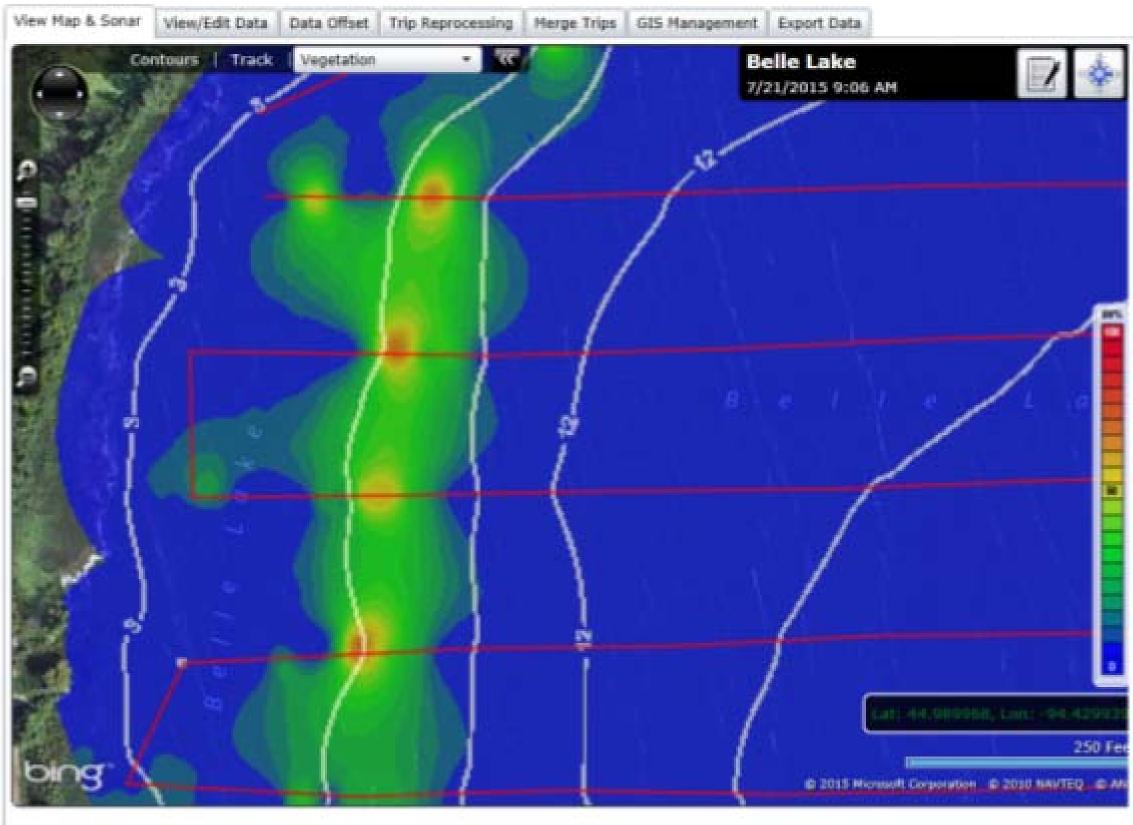

Figure 11. A zoomed in portion of the Northwest side of Belle Lake showing transects (red lines) and biovolume estimates. The highest estimates of biovolume are estimated along transects with lower estimates in between transects. This produces a patchy vegetation map which is unnatural.

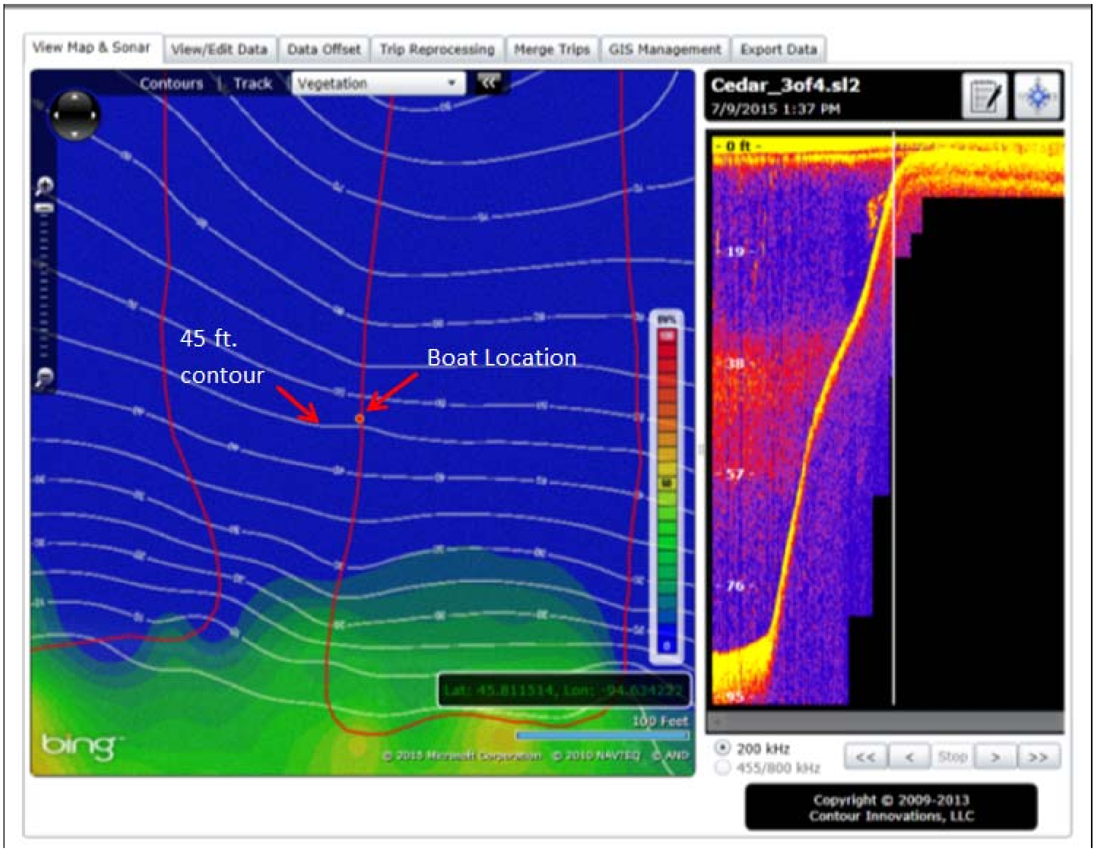

Figure 12. Example of a location on Cedar Lake where the sonar log (right) indicates a depth of approximately 10 ft and bathymetry map (left) indicates a depth of approximately 46 ft.

Depth
Vegetation
Composition

Delete	Date	Lat	Lon	MPH	Type	Z
	7/9/2015 1:51:05 PM	45.81276	-94.63466	7.989	Bottom	-86.779
	7/9/2015 1:51:08 PM	45.81268	-94.63465	7.989	Bottom	-87.058
	7/9/2015 1:51:10 PM	45.8126	-94.63465	7.892	Bottom	-86.642
	7/9/2015 1:51:12 PM	45.81253	-94.63464	7.892	Bottom	-86.586
	7/9/2015 1:51:14 PM	45.81245	-94.63464	7.892	Bottom	-86.264
	7/9/2015 1:51:17 PM	45.81236	-94.63464	7.892	Bottom	-85.481
	7/9/2015 1:51:19 PM	45.81229	-94.63464	7.989	Bottom	-85.28
	7/9/2015 1:51:21 PM	45.8122	-94.63465	8.086	Bottom	-84.437
	7/9/2015 1:51:23 PM	45.81211	-94.63465	8.086	Bottom	-83.719
	7/9/2015 1:51:25 PM	45.81205	-94.63466	8.086	Bottom	-84.471
	7/9/2015 1:51:27 PM	45.81195	-94.63467	8.086	No Data	0
	7/9/2015 1:51:29 PM	45.81188	-94.63468	7.989	No Data	0
	7/9/2015 1:51:32 PM	45.81181	-94.6347	7.892	Bottom	-68.752
	7/9/2015 1:51:34 PM	45.81172	-94.63472	7.892	Bottom	-65.448
	7/9/2015 1:51:36 PM	45.81165	-94.63473	7.581	Bottom	-60.491
	7/9/2015 1:51:38 PM	45.81158	-94.63473	6.998	Bottom	-55.968
	7/9/2015 1:51:40 PM	45.81151	-94.63474	5.384	Bottom	-49.691
	7/9/2015 1:51:41 PM	45.81148	-94.63474	4.685	Bottom	-45.938
	7/9/2015 1:51:43 PM	45.81144	-94.63474	5.287	No Data	0
	7/9/2015 1:51:45 PM	45.8114	-94.63475	5.384	No Data	0
	7/9/2015 1:51:46 PM	45.81137	-94.63475	5.093	Bottom	-38.251

Figure 13. The line highlighted in yellow shows data from the same point depicted in Figure 12. Depth is found in the "Z" column and is approximately 46 feet. There is a large discrepancy between this depth and the depth indicated in the sonar log, which is approximately 10 ft.

A Sentinel Lake Highlight: Lake Shaokotan – a prairie shallow lake responding to nutrient reductions as revealed through long-term monitoring

Background

Shaokotan Lake is a shallow and naturally productive 995-acre sentinel lake located near the town of Ivanhoe in Lincoln County, MN. Its maximum depth is just 10 feet and the lake has a small watershed to surface area ratio of 9:1. Watershed land use is dominated by agriculture, and notwithstanding recent improvements the lake has a history of serious water quality problems dating back to at least the 1980s, and probably 40 years earlier than that based on observations recorded in a 'General Duck Lake Survey' conducted by DNR in 1949. Water quality problems have included severe nuisance blue-green algae blooms, and periodically, both summer and winter hypoxic conditions have formed that have resulted in fish kills. Ultimate causes for these water quality problems were investigated in a Total Maximum Daily Load (TMDL) study and determined to be the result of excessive runoff from nearby agricultural fields, feedlots, and developed shorelines.

The lake contains a simple fish community dominated by Walleye *Sander vitreus* and Yellow Perch *Perca flavescens*. Walleye stocking and winter aeration have been used to sustain game fish populations, but significant natural reproduction by Walleye has also been documented. Historically, the lake had high abundance of disturbance-tolerant species such as Fathead Minnow *Pimephales promelas*, Green Sunfish *Lepomis cyanellus* and Black Bullhead *Ameiurus melas*. A severe winterkill during the winter of 1968-69 apparently eliminated the invasive Common Carp *Cyprinus carpio* population as no Common Carp have been observed in the lake since that event.

Previous data collection efforts include Minnesota Pollution Control Agency (MPCA) water quality monitoring, sediment core collections by the Science Museum of Minnesota and Clean Water Partnership (CWP)-funded monitoring conducted by the Yellow Medicine River Watershed. A detailed CWP Phase I diagnostic study was initiated in 1989 and restoration efforts began in 1991. Phase II implementation included rehabilitation of three animal feedlots, four wetland areas, and shoreline septic systems. By 1994, significant reductions of in-lake phosphorus (P) were realized. Summer mean P concentration was 98.75 µg/L, dramatically lower than the 200 to 350 µg/L noted in previous summers, and very near the median value (90 µg/L) for similar lakes within the Northern Glaciated Plains (NGP) ecoregion. As a result, the frequency and severity of nuisance algal blooms decreased and water transparency increased. By 1999, anecdotal evidence suggested submerged macrophyte abundance was increasing; however, plant surveys in 2000 and 2002 found essentially no rooted plants and the 2008 survey found relatively little plant coverage. Water chemistry data indicated an increase in total phosphorus (TP) and chlorophyll *a* from 1999 to 2001, largely attributable to an abandoned feedlot operation in the nearshore area of the lake. Subsequent efforts by the Yellow Medicine Watershed District, Lincoln County officials, and the local sportsman's group sought to address the problem, but TP and chlorophyll *a* remained above the trophic status thresholds for the NGP ecoregion (Engel et al. 2009).

In 2008, Shaokotan Lake was included in the Sentinel Lakes Program and Phase 1 sampling was initiated to monitor physical, chemical and biological characteristics of 25 sentinel lakes across Minnesota. The Sentinel Lakes Program focusses on understanding the impacts of stressors on lake's physical and ecological features. Climate change, invasive species, changing land use, lakeshore development and angling pressure are among some of the stressors being evaluated.

Ecological Changes Observed During Phase 2

Phase 2 of the Sentinel Lakes Program began in 2013 and data collection was again a collaborative effort between Minnesota Department of Natural Resources (MN DNR) and MPCA. Data was gathered for a large suite of surface and ground water chemical parameters, temperature and dissolved oxygen levels were regularly monitored, as were lake water levels, water clarity, and local weather conditions. Biological data collection included fish surveys, submerged and emergent vegetation surveys, and composition and abundance of phytoplankton and zooplankton.

In 2013 and 2014, summer Secchi transparency (a measure of water clarity) averaged 1.6 (5.2 feet) and 1.8 meters (5.9 feet) respectively, which were approximately twice as high as measurements taken in the 1990's and 2000's (Figure 1). These data along with reports of plants growing in the deepest parts of the lake prompted a point-intercept vegetation survey by the MN DNR Division of Ecological and Water Resources in July 2015. A grid of evenly-spaced points was used to systematically sample submerged aquatic vegetation over the entire surface of the lake, with extra points added in shallow areas and around the shoreline, as outlined by the Minnesota Lake Plant Survey Manual (Perleberg et al. 2015). In the past, the deeper portions of the lake were excluded from such surveys, because poor water clarity did not allow plant growth. In 2015, submerged vegetation was detected at nearly 100% of sampling points including the deepest portions of the lake, which is in stark contrast to previous surveys where vegetation was sparse and limited to shallow areas around the shoreline and west end of the lake (Figure 2).

It is well documented that shallow lakes of intermediate fertility (i.e., P concentrations) can exist in either a clear water or a turbid water state and can alternate between these two stable states (Scheffer and Carpenter 2003, Bayley et al. 2007, Zimmer et al. 2009, Bronmark et al. 2010). Alternate states are reinforced by feedback mechanisms that follow from macrophytes and phytoplankton competing for light. Once one primary producer becomes abundant, it tends to dominate (e.g., phytoplankton shades out macrophytes, wind can better interact with and continue to stir nutrients and sediments, etc.; oppositely, when macrophytes dominate they reduce wind-driven sediment suspension, release alleolopatheic chemicals, and provide refuge for zooplankton that graze phytoplankton, among other feedbacks). Specific abundance thresholds where one group dominates over the other are difficult to identify because each lake is affected differently by factors such as depth, wind, emergent/floating leaf vegetation, and fish (Zimmer et al. 2009). While it's the exception for most Minnesota lakes, at Shaokotan and the other sentinel lakes, the annual, monthly, and even hourly data needed to understand thresholds and the mechanisms driving such lake changes are becoming available. Over time, these data may provide researchers with a better understand when and why shallow lakes flip clear to turbid or turbid to clear.

Both mean summer chlorophyll *a* and TP concentrations have exhibited a decline in Shaokotan Lake since 1989 (Figures 3 and 4). In 2014 and 2015, chlorophyll *a* concentrations measured 7.2 µg/L and 8.4 µg/L, respectively, which fell below the threshold identified by Zimmer et al. (2009) (22 µg/L), where 95% of their shallow study lakes remained in a clear water state (Figure 3). These data indicate an overall reduction in the abundance of phytoplankton, and companion data on phytoplankton composition document a shift away from blue-green algae (Cyanophyta) dominance (Figure 5). Nuisance algal blooms are typically observed at chlorophyll *a* concentrations >20 µg/L and in 2014 concentrations at Shaokotan Lake remained below this threshold for the entire summer. In 2015, however, a reading of 26.7 µg/L was recorded in August, suggesting Shaokotan was near the level where nuisance blooms are possible. Mean summer TP concentrations in 2014 and 2015 measured 30.7 ppb and 37.4 ppb respectively, which also fell below the threshold identified by Zimmer et al. (2009) (62 ppb) where lakes typically exist in a stable, clear water state (Figure 4). It is likely that improved land use practices within the watershed have led to reductions in phosphorus loading and are at least partially responsible for the shift to a clear water state by limiting nutrients available to phytoplankton, with resulting clear water allowing rooted plants to grow throughout the lake. In 2015, mean summer TP concentration increased some compared to 2014 with a maximum observed value of 62 ppb, which again indicates Shaokotan remains vulnerable to a turbid water state shift. Future monitoring should be useful to help identify mechanisms behind the regime shifts should that lake revert back to a turbid state. Over time, it may be possible identify lake-specific abundance thresholds for both aquatic macrophytes and phytoplankton preceding state shifts, and future predictions may be possible.

Fish Population Responses Follow a Major Lake Shift?

In Phase 2, fish sampling continued on Shaokotan Lake from 2013-2015. This included both standard MN DNR fish surveys using gill nets and trap nets as well as nearshore fish surveys using seines and backpack electrofishing. Prior to 2008, standard surveys were conducted approximately every 4 years, and since 2008, when the Sentinel Lakes Program was initiated, standard surveys have been conducted every year except 2012. From 2008 to 2013, gill net catches of Walleye remained relatively consistent, around 20 fish per gill net, but dropped significantly in 2014 and 2015 to around 10 fish per gill net (Table 1). In 2014 and 2015, there was a noticeable increase in water clarity and submerged aquatic vegetation abundance. Though it is unclear if the reduced gill net catch corresponds to a decrease in Walleye abundance or if gill net catchability of Walleye decreased due to clearer water and abundant vegetation, it will be important to continue to monitor gill net catch to see if this trend continues.

Northern Pike *Esox lucius* gill net catch has historically been low in Shaokotan Lake, however 2014 and 2015 produced the two highest gill net catches of Northern Pike since 1996 (Table 1). Though more years of sampling will be required to identify a trend, Northern Pike should thrive in an environment with clear water and abundant vegetation. Yellow Perch gill net catches have historically been variable and continue to be (Table 1), but it would also be expected that Yellow Perch abundance would increase with abundant aquatic vegetation, by providing plenty of rearing habitat for young fish. Yellow Perch growth appears to be fast, with fish reaching 8-10 inches by age 2. Black Bullhead gill net catches have historically been high, but Black Bullhead were nearly absent from the gill net sample from 2008 to 2011. 2013 produced another high catch of Black Bullhead which steadily declined through the 2014 and 2015 surveys (Table 1).

Common Carp remain absent in the lake. Because Common Carp often have negative impacts on aquatic vegetation abundance and water quality by uprooting aquatic vegetation and re-suspending nutrients from the lake bed into the water column (King and Hunt 1967, Crivelli 1983, Zambrano et al. 2001), monitoring efforts should remain as frequent as possible. Shaokotan Lake sits near the P threshold where clear water and turbid water states are each possible. Common Carp re-establishment would make Shaokotan Lake vulnerable to flipping into the turbid water state.

Nearshore fish sampling captured 10 species including Black Bullhead, Fathead Minnow, Brook Stickleback *Culea inconstans*, Green Sunfish, hybrid sunfish, Orange-spotted Sunfish *Lepomis humilis*, Johnny Darter *Etheostoma nigrum*, Walleye, Northern Pike and Yellow Perch. If the trend of improved water quality continues, it is possible that species less tolerant of poor water quality could be found in the nearshore fish surveys. However, because Common Carp, an exceptionally strong swimmer, have not re-established, connections to other water bodies might be weak, therefore chances of new species immigrating into Shaokotan Lake are low.

Social Considerations, Management Implications, and a Case for Long-Term Monitoring

As Shaokotan Lake has become clearer and aquatic vegetation has started to grow in deeper parts of the lake, natural resource managers have been faced with a new set of challenges. The most evident is that the abundant vegetation is making it difficult for people to operate boats. The vegetation wraps around propellers and clogs water intakes making both recreational boating and fishing difficult. A second challenge will be managing game fish populations with a new set of environmental conditions. While Shaokotan Lake has a reputation of producing abundant Walleye and Yellow Perch, the shift to a clear water state may provide habitat niches preferred (or needed) for species such as Northern Pike, Largemouth Bass and Bluegill *Lepomis macrochirus*. While it is difficult to predict exactly how the fish community will respond to new lake conditions, changes seem likely and may require new fish management actions. Frequent monitoring efforts will ensure managers can quickly identify these changes and respond accordingly.

While members of the natural resource community view this shift from turbid to clear water as a great success, some people in the community around Shaokotan Lake find it to be a nuisance. After just one year of aquatic vegetation being problematic for lake users, the community is asking for permission to manage (remove) aquatic vegetation. The citizens around Shaokotan Lake and MN DNR have begun drafting a Lake Vegetation Management Plan (LVMP). This plan will allow some vegetation removal across the surface of the lake. The treatment would provide lanes through the vegetation for lakeshore owners to operate their boats out to a larger common area where recreational boating can occur.

As organizations throughout the state of Minnesota strive to improve the state's water quality, it is important to recognize that a situation similar to what has taken place at Lake Shaokotan is a possibility on other shallow lakes. Because lakes in southern Minnesota are typically shallow, vegetation could grow throughout the lake if water clarity improves and present the same challenges observed at Shaokotan Lake. Shaokotan is one of the first large recreational (fishing) lakes in the southern part of the state to experience a dramatic shift from turbid to clear and much can be learned as we monitor how these changes influence the ecosystem and the community around the lake.

The usefulness of the monitoring efforts of the Sentinel Lakes Program has been demonstrated by measuring key physical, chemical, and biological metrics during a dynamic time period at Shaokotan Lake. By examining historical data and data that will be collected into the future, we can start to understand the changes we are seeing in Minnesota's Lakes and what is driving those changes. This will become increasingly important as we continue to manage aquatic ecosystems in a time when our world is rapidly changing.

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Table 1. Mean gill net catch per net lift of Black Bullhead, Northern Pike, Walleye and Yellow Perch from Shaokotan Lake from 1996-2015.

	1996	2000	2004	2008	2009	2010	2011	2013	2014	2015
Black Bullhead	115.0	225.0	287.0	1.7	0.0	0.0	0.3	168.0	64.3	45.7
Northern Pike	0.0	0.7	0.7	0.3	0.0	0.7	1.3	0.3	1.7	4.0
Walleye	13.8	10.3	6.0	19.7	22.7	18.7	18.3	23.3	10.7	10.3
Yellow Perch	24.0	147.7	0.7	38.7	19.3	73.3	21.7	15.7	43.3	80.0

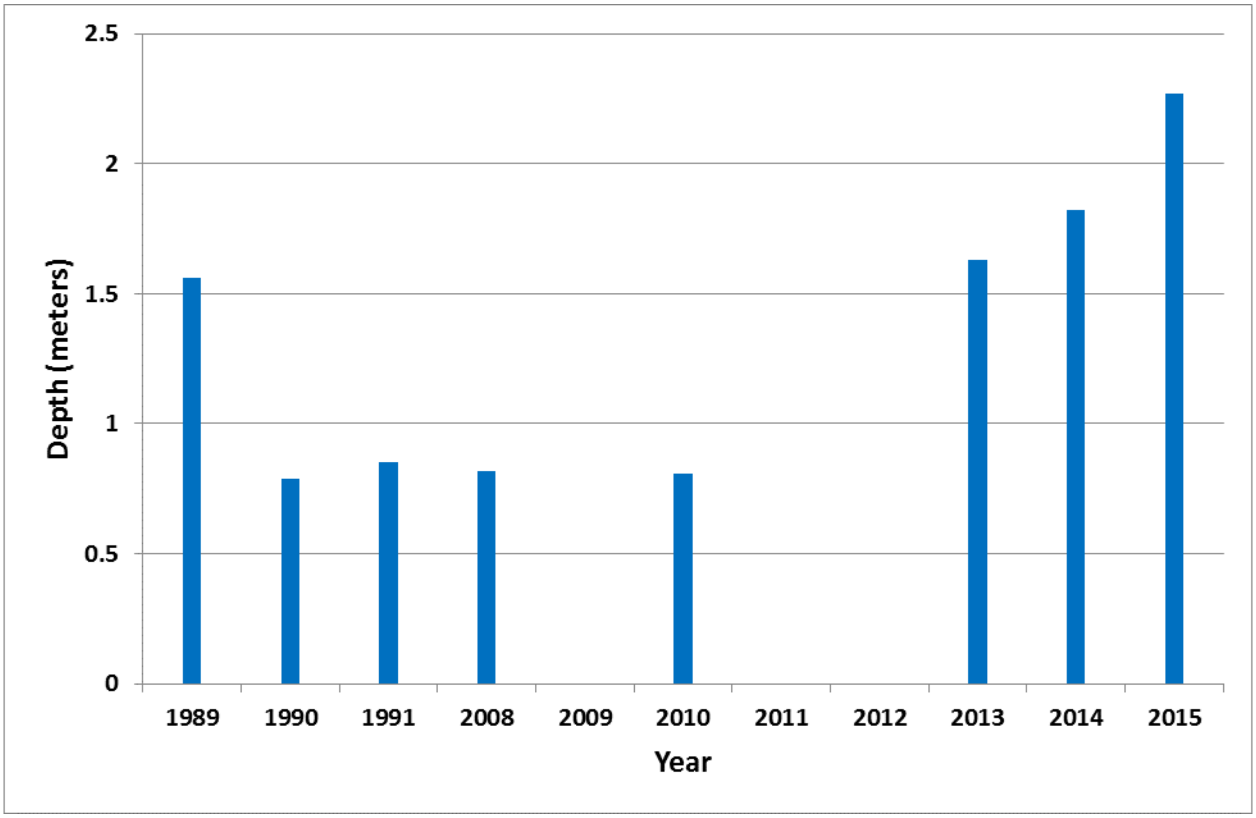

Figure 1. Mean summer Secchi transparency at Shaokotan Lake from 1989 to 2015.

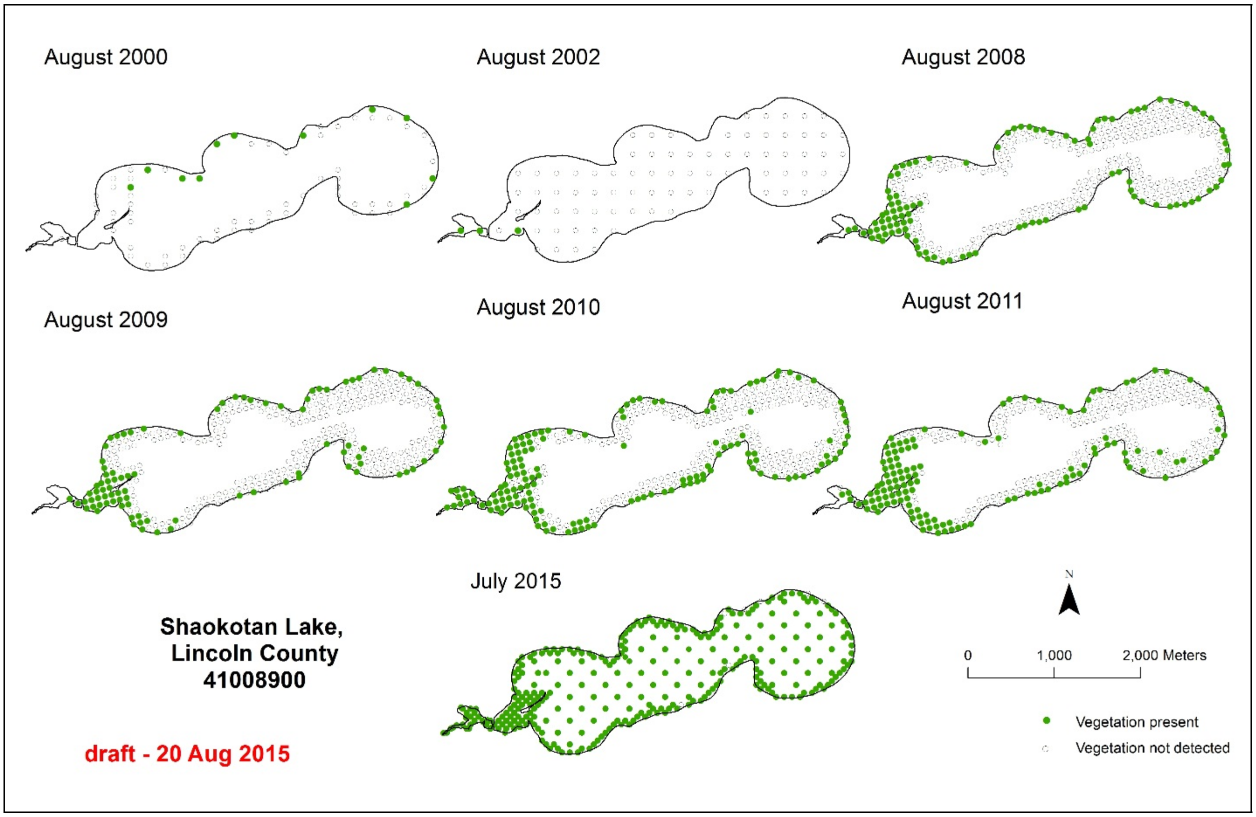

Figure 2. Maps of Shaokotan Lake from 2000 to 2015 showing results of point-intercept vegetation surveys. Green dots indicate vegetation was detected and white dots indicate no vegetation was detected.

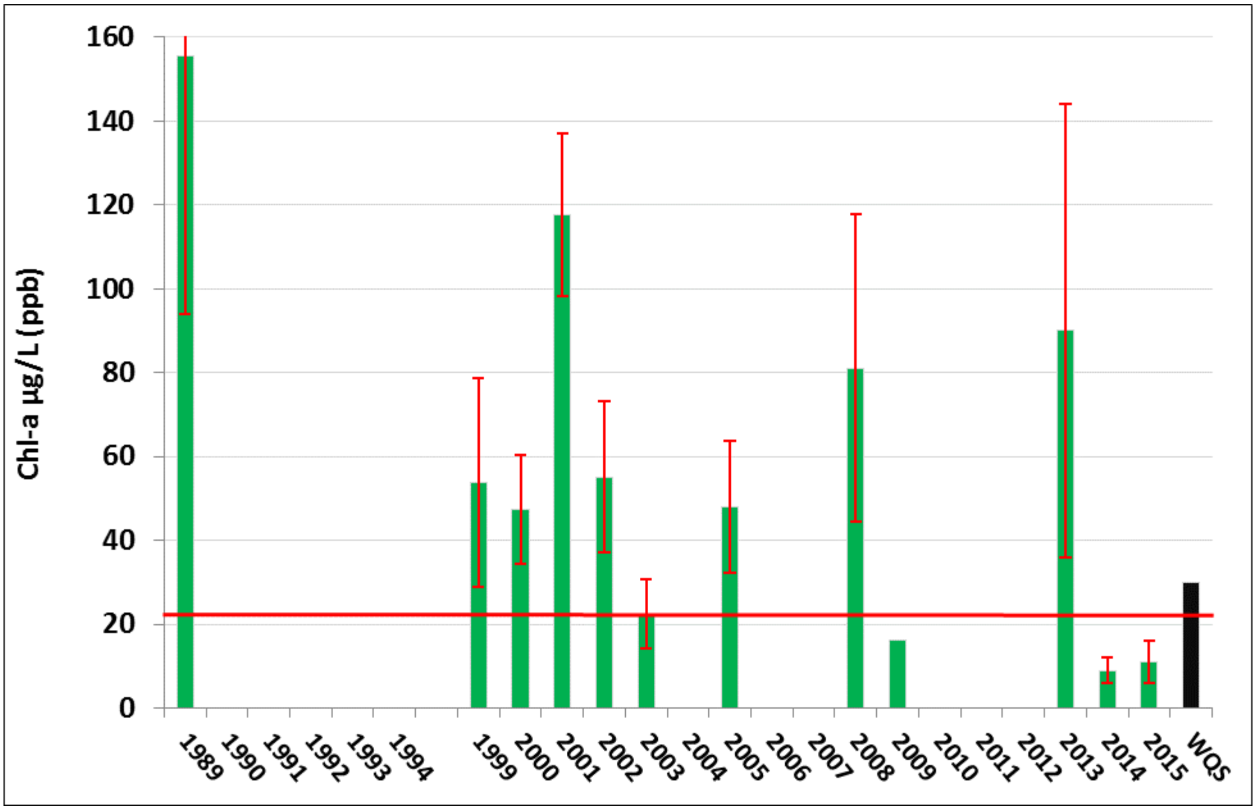

Figure 3. Mean summer chlorophyll-*a* concentration in Shaokotan Lake from 1989 to 2015. The black bar indicates the water quality standard (WQS) for the Northern Glaciated Plains ecoregion. Error bars represent the standard error of the mean. Zimmer et al. (2009) identified a chlorophyll-a threshold of 22 µg/L (red line), where 95% of study lakes existed in a clear water state when chlorophyll-a measurements were below this level.

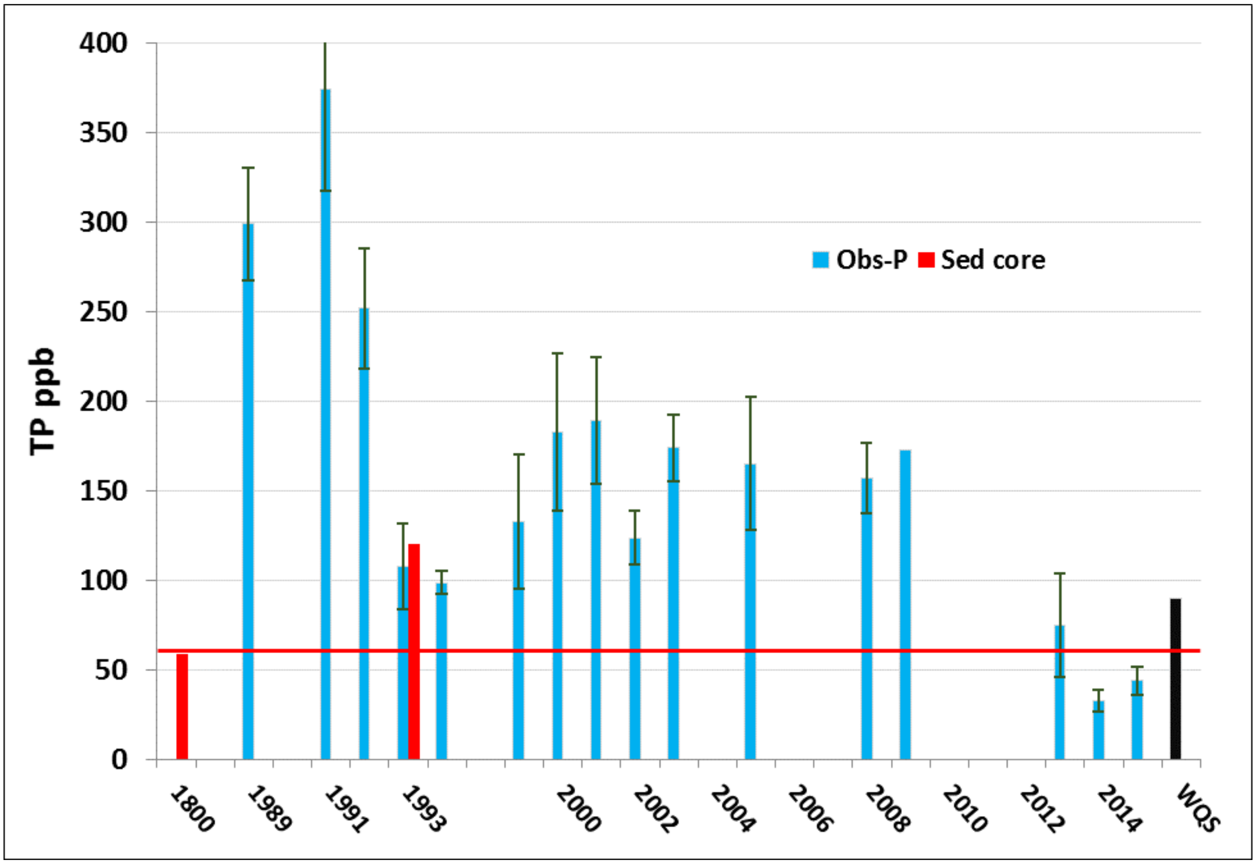

Figure 4. Observed mean summer total phosphorus concentration (blue bars) in Shaokotan Lake and sediment core reconstructed total phosphorus (red bars). The black bar indicates the water quality standard (WQS) for Northern Glaciated Plains ecoregion. Error bars represent the standard error of the mean. Zimmer et al. (2009) identified a TP threshold of 62 ppb (red line), where lakes typically exist in a clear water state when TP measurements were below this level.

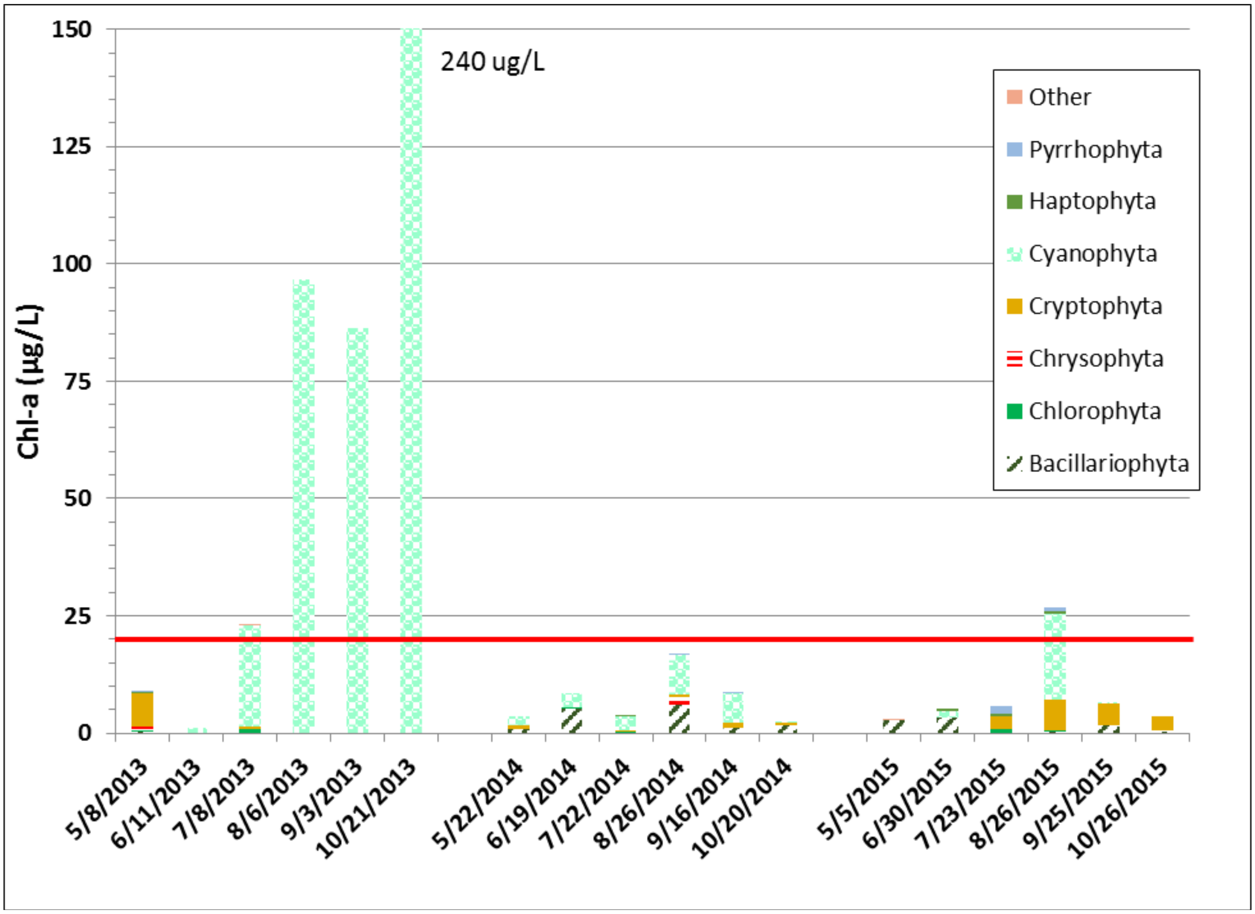

Figure 5. Phytoplankton community composition data collected monthly in 2013 and 2014 at Shaokotan Lake. Nuisance algal blooms typically occur at chlorophyll-a concentrations above 20 µg/L (indicated by the red line).

Outcome 2 – Work related to inventorying watershed land cover and land uses, and drainage features has been completed and final reports for these activities have been provided in the progress summaries above.

Outcome 3 - Large amounts of data were collected in this area, most of which will be of value to researchers and managers over a long time period. However, two short summaries included in the final report demonstrate the short-term value of such data. Complete data sets are continually archived by DNR and are available through Jeff Reed (jeffrey.reed@state.mn.us). The summaries included in the final report were compiled by Eric Katzenmeyer (water levels) and Jeff Reed and Andrew Carlson (temperatures and dissolved oxygen).

Water Level Monitoring

Water level within a lake can be quite dynamic and has the potential to influence a lake ecosystem by influencing variables such as nutrient inflow and outflow and available fish spawning and nursery habitat (Kallemyn 1987). For these and other reasons, it is important to monitor how lake level changes over time. Submerged pressure transducers were installed in Elk, Carlos, Trout, Pearl, Madison and Shaokotan lakes following ice out and retrieved just before ice up in 2014 and 2015. These sensors collect water level and temperature data every 15 minutes and store them internally until they are downloaded at the end of the open water season. This equipment was initially deployed to provide data to the United States Geological Survey (USGS) for use in tier 1 sentinel lake bio-physical models. Because this data may also be useful in explaining changes in the biological communities of sentinel lakes, monitoring equipment will continue to be deployed in lakes where water level data would otherwise be sparse.

It is evident by looking at the data for Pearl and Madison Lakes that water level fluctuates both within years and between years, which could have profound impacts on how the lake ecosystem functions (Figures 1, 2). Continued operation of this monitoring equipment will help to better understand the effects of water level on in-lake processes.

References

Kallemyn, L.W. 1987. Correlations of regulated lake levels and climatic factors with abundance of young-of-the-year Walleye and Yellow Perch in four lakes in Voyageurs National Park. North American Journal of Fisheries Management. 7(4): 513-521.

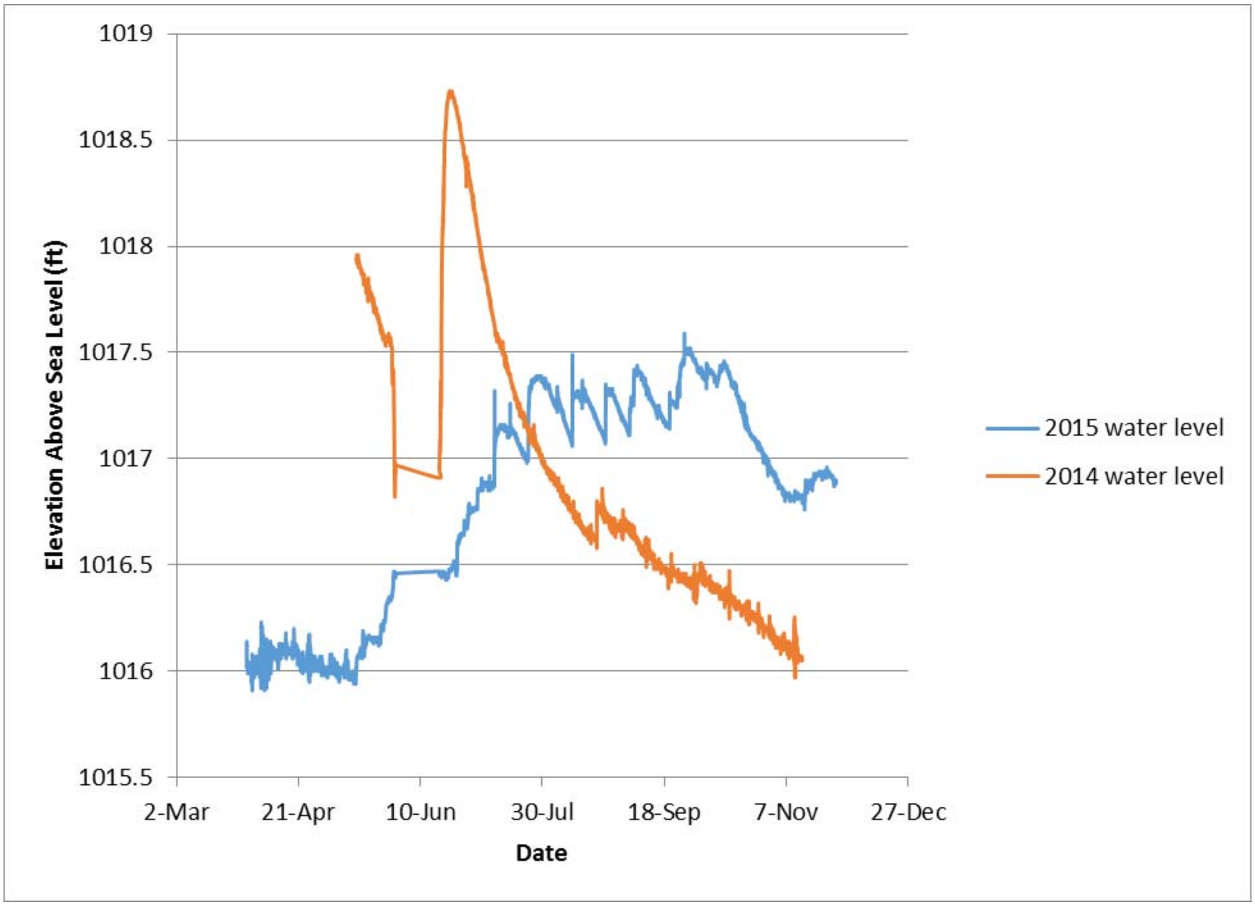

Figure 1. Water level collected at 15 minute intervals with a pressure transducer at the outlet of Madison Lake in 2014 and 2015.

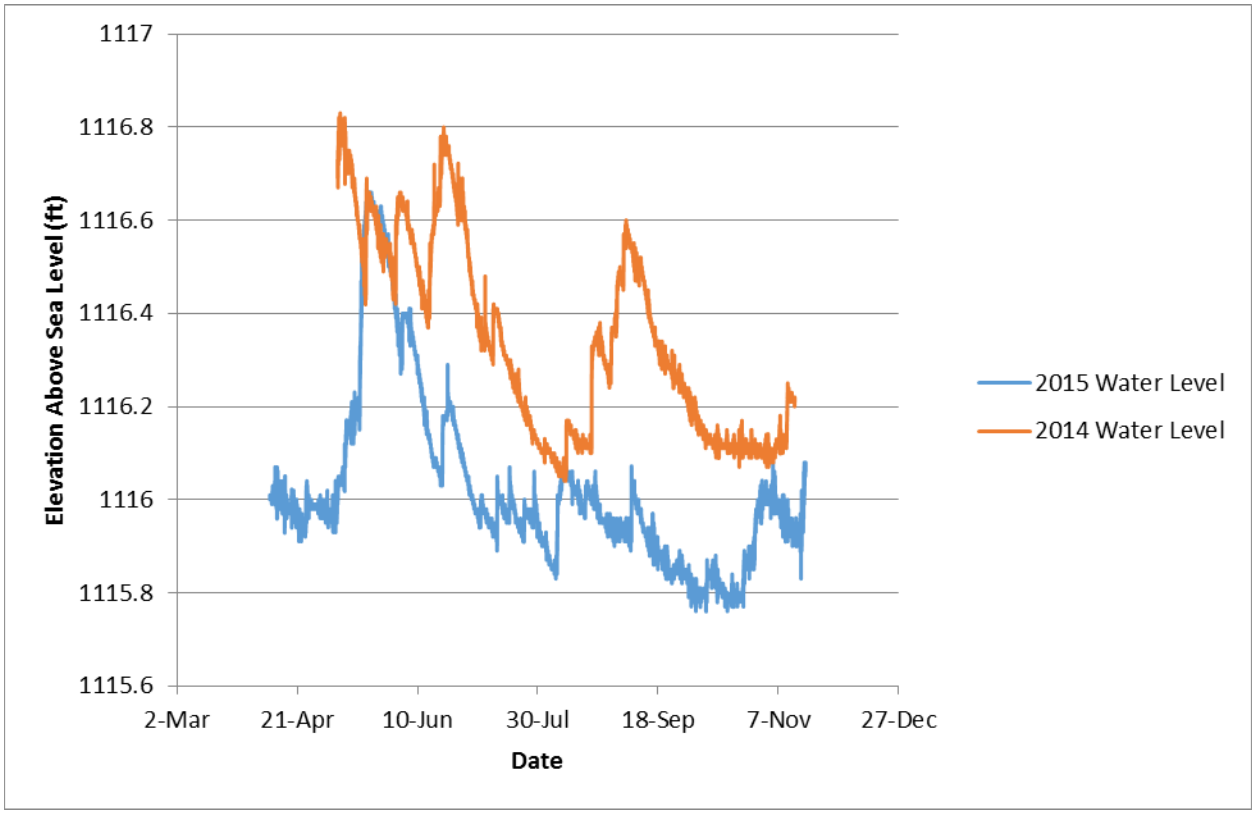

Figure 2. Water level collected at 15 minute intervals with a pressure transducer at the outlet of Pearl Lake in 2014 and 2015.

Dissolved Oxygen and Temperature Monitoring on Sentinel Lakes

Lake temperature monitoring has been a foundational part of Sentinel Lakes monitoring since the program's inception in 2007. These monitoring efforts have enabled researchers to develop a much better understanding of how lakes behave thermally and as a result have been able to develop and refine predictive models of how future climate change scenarios may affect fish populations. For example, USGS has used these data to develop sophisticated in-lake models for Carlos and Elk lakes (Smith and Kiesling 2013). In Phase II, temperature monitoring efforts on Elk Lake were supplemented with monitoring of dissolved oxygen (DO) with the deployment of several continuously recording dissolved oxygen sensors. The combination of continual temperature and dissolved oxygen monitoring on Elk Lake provided researchers with greater insight into how these physical and chemical changes affect fish movement in the lake. The combination of data collected through the Sentinel Lakes Program with those collected by DNR Fisheries Researchers demonstrates the benefits of collaborative efforts on Sentinel Lakes (Carlson and Holbrook 2016). Furthermore, in addition to updating in-lake models developed in Phase 1, USGS researchers have completed model development for warm-water lakes in central and southern Minnesota and as with the previous models, temperature and dissolved oxygen measurements are key metrics for proving biophysical lake models are properly calibrated (see Activity 2 for these results).

<u>ELK LAKE EXAMPLE</u>

Methods:

Lake temperature was continuously monitored in 15-minute intervals using a custom thermistor chain (HOBO temp loggers (Model: U22-001) at 0.5m – 1m intervals along a buoy line). Lake dissolved oxygen was continuously monitored in 15-minute intervals using a custom DO chain (HOBO dissolved oxygen logger Model: U26-001) with loggers spaced at 2.8, 5.8, 7.8, 9.8, 11.8, and 14.8m below the surface. Temperature and dissolved oxygen (Hach LDO meter, model HQ40d) profiles were conducted at the standard water quality monitoring station in 1-10 week intervals. Daily depths for each logger were calculated from the water level data and temperature or DO was linearly interpolated into standardized 0.5m bins (Figures 1 and 2, respectively).

Summary of Findings:

- Dissolved oxygen is depleted in the hypolimnion during summer stratification and depletion? continues until fall turnover
- Hypoxia (<2mg/L) extends into the metalimnion by mid-summer and limits fish distribution at these depths
- Dissolved oxygen is depleted in the hypolimnion during winter stratification, though not to the same magnitude as during the summer
- Thermocline degradation began in September and continued through early November, when the lake was almost completely turned over (i.e., when the lake's water temperature is the same from the top to the bottom, ~4^{o}C).

RECOMMENDATIONS:

- Continue temperature monitoring on all 25 Sentinel Lakes.
- Expand monitoring to include a wider range of depths.
- Encourage continued temperature and dissolved oxygen profile monitoring by agency staff and citizen-science volunteers.

References

Carlson, A.J., and B. Holbrook. 2016. Seasonal Variability in Standard Assessment Techniques. Study 617 Progress Report.

Smith, E.A., and R.L. Kiesling. 2013. Assessing the Water Quality and Habitat Dynamics of Deepwater Lakes with Coldwater Fish Populations. Final Report to LCCMR ML 2009.

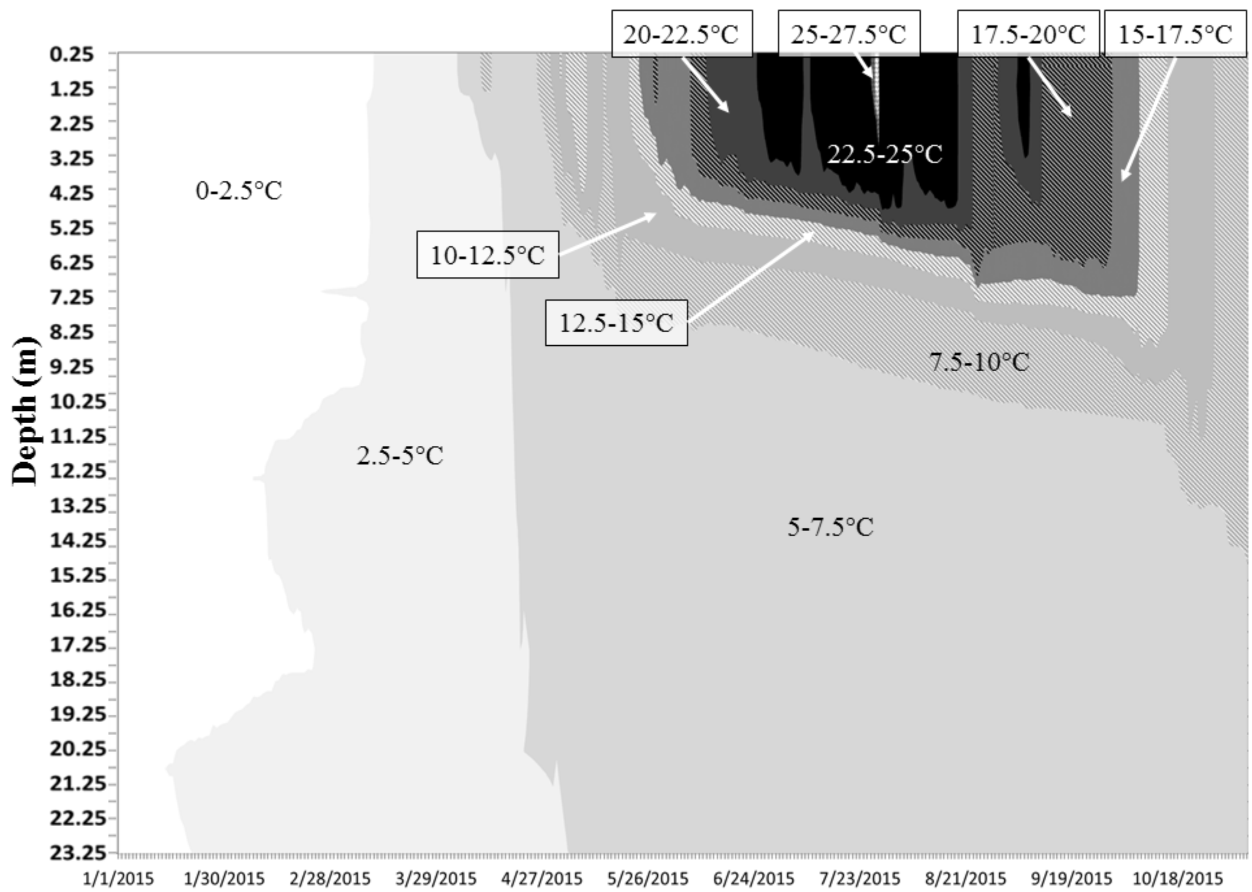

Figure 1. 2015 Seasonal (1 January – 6 November) Temperatures at Depths in Elk Lake.

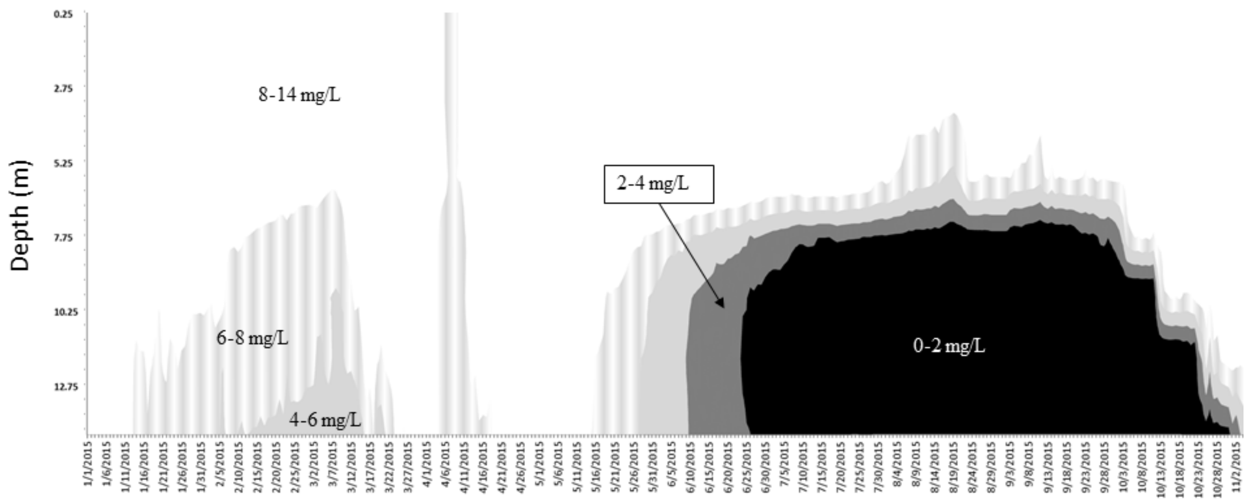

Figure 2. 2015 Seasonal (1 January – 6 November) Dissolved Oxygen Levels at Depths in Elk Lake.

Outcome 4

Assessment of Stable Isotope Analysis as a Tool to Investigate Current and Changing Lake Food Webs to Stressors

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Introduction

Stable isotope analysis using compositions of organic carbon (^{13}C:^{12}C), nitrogen (^{15}N:^{14}N), hydrogen (^{2}H:^{1}H), and sulfur (^{34}S:^{32}S) has become an important tool for understanding linkages in aquatic food webs (Peterson and Fry 1987; Vander Zanden et al. 1999; Croisetière et al. 2009), yet regional and individual lake characteristics sometimes limits the utility of the stable isotope approach for describing aquatic food web structure (Cabana and Rasmussen 1994). When there is sufficient separation between stable isotope source signatures, combined measurements of $\delta^{13}C$, $\delta^{34}S$, $\delta^{2}H$, and $\delta^{15}N$ can be used to determine how basal carbon within an ecosystem or linked ecosystems is successively transferred to higher trophic levels within the food web (Fry 2006). This works because the stable carbon, hydrogen, and sulfur isotope signatures of a consumer reflects its source of dietary carbon, hydrogen, and sulfur (Peterson and Fry 1987; Post 2002, Cole et al. 2011), while consumer $\delta^{15}N$ values can be used to describe trophic position, because animals become enriched in a predictable 2.2-3.4 ‰ relative to their diet (Mingawa and Wada 1984; Post 2002; McCutchan et al. 2003).

Ideally, if sufficient separation in stable isotope signatures exist among primary producers, then combined $\delta^{13}C$, $\delta^{34}S$, $\delta^{2}H$, and $\delta^{15}N$ signatures should be a sensitive tool to understand present day food webs (e.g., understand niche separation among native and stocked predatory fish) as well as to detect disruptions to food webs. Potential food web disruptions include species introductions (e.g., zebra mussels – Mitchell et al. 1996; Maguire and Grey 2006, smallmouth bass – Vander Zanden et al. 1999) and altered nitrogen cycling and climate change (Fung et al. 1997; Bousquet et al. 2000; Holtgrieve et al. 2011). Taking it a step further, isotopes could possibly be used to help us measure and understand lake responses to routine management activities (e.g., fish stocking), ecosystem restoration efforts, or possibly even climate adaptation practices likely to be implemented in the future. For these reasons, we think stable isotope analysis could be a valuable component within a long-term monitoring program, which we have begun to evaluate in this study, and further detail in this report.

Long-term monitoring programs are perhaps most useful to management agencies (e.g., MN DNR) when they include a proactive management component intentioned to identify and then react to change, such as is envisioned in the MN DNR/MN PCA Sentinel Lakes Program (http://www.dnr.state.mn.us/fisheries/slice/index.html). A key element in this cycle is to be able to first establish baselines of historical and current conditions, from which key departures from normal conditions can be detected, and then addressed. Isotopes may be ideally suited for this application as they provide a standardized way to characterize an organism's ecological niche (both resource- and habitat-related aspects) because isotope systems (i.e., different combinations of isotopes such $\delta^{13}C/\delta^{15}N$, $\delta^{13}C/\delta^{34}S$, $d\delta^{2}H/\delta^{18}O$) vary in predictable ways along key environmental and resource gradients (Newsome et al. 2007). Thus, stable isotope signatures of organisms faithfully record both anthropogenic and natural changes because they vary predictably along key environmental and resource gradients, including across trophic levels, across types and according to distributions of plants and animals, with levels and types of eutrophication, according to relative importance of benthic/pelagic pathways, and across oxic vs. anoxic conditions (Newsome et al. 2007). Because humans and their activities can influence environmental and resource gradients, stable isotope compositions should track anthropogenic stressors in predictable ways (sensu Cabana and Rasmussen 1994).

Our study included two lake sets with different research questions. Carlos and Ida lakes are part of system of interconnected lakes along the Long Prairie River north of Alexandria, MN. Lakes here are in the midst of zebra mussel invasion and colonization, which is the focus of our research questions for these lakes. A second lake group included Elk Lake, which is a sentinel lake located in Itasca SP in north central Minnesota, north of Park Rapids. Work here focused on establishing baseline isotope signatures for the food web and included an applied aspect where we describe the ecological niches of the major large piscivores in the system (walleye, northern pike, and muskellunge). Our study had three main goals. Our first goal was to establish baseline data to describe energy flow, isotope signature patterns of predator and prey fish, and overall food web structure in 3 Minnesota lakes faced with anthropogenic stressors, such as climate change, eutrophication, and invasive species. Our second goal was to establish data to quantify the impacts of zebra mussels on the food webs of Carlos and Ida lakes. For goal three, we used Elk Lake as a case study to establish baseline food web linkages, energy flow, and ecological niches of piscivore species. Data collected for goal two in Lake Ida, when combined with future data collected after the zebra mussel population increases, will allow us to test for several different outcomes in response to zebra mussel infestation. First, zebra mussels may have no discernible impact on sources of energy for primary game and prey fish, nor any impact on fish population sizes. This may be due to zebra mussels remaining at low enough densities so that energy inputs from littoral (and benthic), pelagic, and allochthonous sources remain unchanged. This outcome would be indicated by isotopic signatures staying constant through time. Second, zebra mussel density may get high enough to reduce phytoplankton abundance, but energy sources for food webs and fish may remain constant due to increased water clarity resulting in a deeper photic zone that keeps pelagic production similar. This outcome would be indicated by higher values of ^{2}H (Cole et al. 2011) and lower values of ^{13}C (Matthews and Mazumder 2006) in zooplankton and fish. Third, zebra mussels may decrease pelagic production and stimulate benthic production due to higher water transparency. This outcome would be indicated by ^{13}C and ^{34}S concentrations changing in fish tissue to indicate a benthic origin of C (Hecky and Hesslein 1995) or S (Croisetière et al. 2009). Finally, if zebra mussels reduce pelagic production, and benthic production does not compensate, then the lake food web may become more reliant on allochthonous production. This would be indicated by changes in ^{13}C and especially in ^{2}H, as terrestrial vegetation is enriched in ^{2}H relative to autochthonous production (Cole et al. 2011). As allochthonous inputs should be the same before and after zebra colonization, this outcome would indicate less overall energy input into the lake food web, and reduced abundance of fish would be expected.

This project will produce results immediately valuable to researchers and lake managers in the state, as it will quantify energy flow and trophic relationships among most of the important game fish and their prey. More importantly, it will establish a baseline data set for assessing how lake food webs change in the future in response to anthropogenic-induced change.

Methods

Our study approach involved collection of natural abundance stable isotope samples of resources and consumers to characterize the food web of two lakes included in the Sentinel Lakes Long-Term Monitoring Program, Elk and Carlos lakes, and one non-Sentinel lake (Lake Ida) for "baseline" data sets to be used for future assessments of how each lake responds to eutrophication, climate change, and invasive species introductions. Zebra mussels have been present previously in Lake Carlos, and sampling in July of 2013 indicated heavy infestation. Due to the advancement of this infestation, available water quality data suggests the food web has already been highly altered by the mussels, thus making it hard to assess how zebra mussels have impacted the lake after the impacts of zebra mussels have already been strongly manifested. As a comparison we also conducted an isotopic analysis of the food web of Lake Ida, a lake directly downstream from Carlos, where zebra mussels were first detected in 2014. Given Ida and Carlos are adjacent to each other in the same watershed, have similar land use in the watershed, and similar fish communities, these two lakes provided an excellent pair for assessing impacts of zebra mussels, with Ida and Carlos serving as the pre and post infestation sites, respectively.

Elk, Carlos and Ida lakes were sampled during July 2013 and in mid-June and late-August during 2014. This sampling scheme gave us the ability to assess both inter- and intra-annual variability. Intra-annual variable was not a concern for fish, thus were sampled just once per year due to slower tissue turnover time.

Dissolved oxygen and temperature profiles were determined on each sampling date using a YSI ProODO optical dissolved oxygen meter at the deepest location of the lake, and we also estimated the light extinction coefficient at the same time using a LI-COR LI-250A submersible light meter. Dissolved oxygen and temperature data were used to identify the hypolimnetic habitat in each lake, and light data was used to determine if sufficient light reaches the hypolimnion to drive photosynthesis (which could influence ^{13}C values of seston in the hypolimnion). Quantifying the current photic zone for Ida and Carlos lakes is especially important for assessing how benthic and pelagic production change as sources of energy for the lake food webs.

Samples for obtaining isotopic signatures of fish were collected for each of the target species by DNR biologists using trap nets, vertical and horizontal gill nets, beach seines, and electrofishing. All major game fish were sampled, as well as the representative forage fish for each lake. A tissue sample was removed from the lateral muscle from each fish, placed on ice, and then frozen until analysis in the lab. We collected samples from up to 5 fish in each of three size classes for each fish species so that we can ultimately assess ontogenetic changes in diets and food web position. In the lab, samples were freeze dried and ground with a hand pestle. Recent work has shown that lipids have lower ^{13}C values than other types of tissues, causing problems in food web studies as differences in ^{13}C for a given species could be due to differences in lipid content instead of differences in C source (Smyntek et al. 2007). Surveys have shown that this problem is prevalent in both fish and aquatic invertebrates (Logan et al. 2008), and that ^{13}C cannot be analyzed on tissue with lipids extracted because the extraction procedure changes ^{15}N values (Murry et al. 2006). Thus, we corrected our ^{13}C samples for lipid content using the C:N ratio of tissue with lipids extracted to the C:N ratio of non-extracted tissue (Smyntek et al. 2007, Logan et al. 2008). This approach is based on the fact that most of the C in a sample of animal tissue is in lipid form (largely in cell membranes), and so the reduction in the C:N value after lipid extraction can be used to measure the overall lipid content, and be used to adjust the ^{13}C estimate based on amount of lipid present. Lipids were extracted by placing a 1 ml quantity of dried tissue in a microcentrifuge tube, adding 3 mls of 2:1 chloroform: methanol mixture, shaking and centrifuging, and then decanting the solvent. This process was repeated three times, and the tissue then dried at 60°C. We then determined the C:N ratio of the extracted tissue for each species on a CN Elemental Analyzer at the University of St Thomas. A separate subsample of tissue was analyzed for ^{13}C, ^{15}N, and C:N by the stable isotope lab at UC Davis. A third sample was analyzed for ^{34}S by the UC Davis Stable Isotope Lab, and a fourth sample was analyzed for ^{2}H by the University of Wyoming Stable Isotope Lab after benchtop equilibration for exchangeable H.

Zooplankton were collected from both the epilimnion and the hypolimnion using a Birge closing plankton net (80 µm mesh). The epilimnion sample was towed from the top of the metalimnion to the surface, while the hypolimnion sample was collected from 1m off the sediment to the bottom of the metalimnion. Collected animals were rinsed with lake water into plastic sample jars and placed on ice until analyzed in the lab. Repeated tows were taken until sufficient material was collected for analysis. We also collected seston samples from both the epilimnion (1 m below the surface) and hypolimnion (half way between the bottom of the metalimnion and the sediment) using a Van Dorn sampler, and kept the samples on ice until processing in the lab. We collected samples from three different locations in each lake. In the lab zooplankton were placed in lake water filtered with a GF/F filter for two hours to allow organisms time to clear their digestive tracks. The zooplankton were then condensed onto 80µm mesh, non-herbivorous zooplankton material was removed by hand (including predatory zooplankton like *Chaoborus* which were placed in a separate vial for analysis), and resulting samples were then rinsed with nanopure water into 1.5 ml microcentrifuge tubes. The nanopure water was then decanted and the sample frozen until further analysis. We then estimated lipid content, C:N of extracted and non-extracted tissue, and ^{13}C, ^{15}N, ^{2}H, and ^{34}S using the same techniques as used for fish. For seston, hypolimnetic and epilimnetic samples were prefiltered through a 80 µm mesh to remove zooplankton, then filtered unto precombusted GF/F filters. The filter was then wetted with 1% HCL to remove precipitated carbonates, rinsed with nanopure water, and then frozen. Stable isotopes of ^{13}C, ^{15}N, ^{2}H, and ^{34}S were analyzed by the same labs as used for fish.

We sampled both profundal and littoral habitats for at least one representative group of invertebrates typical of each habitat. Benthic macroinvertebrates were collected from profundal habitat with a ponar grab and placed in lake water on ice until analysis in the lab. In the lab the animals were placed in filtered lake water for 2 hours to allow them to clear their guts, and then samples were handpicked and individuals sorted into taxonomic categories and functional feeding groups (e.g. predatory versus non predatory chironomids). The animals in each taxonomic group were rinsed with nanopure water, then frozen and analyzed for lipids, C:N, ^{13}C, ^{15}N, ^{2}H, and ^{34}S in the same way as for fish and zooplankton samples. Littoral macroinvertebrates were collected using dip nets, placed in filtered lake water, hand sorted into taxonomic and feeding groups not requiring magnification, rinsed with nanopure water, and frozen. Frozen samples were analyzed for lipids, C:N, ^{13}C, ^{15}N, ^{2}H and ^{34}S in the same manner as profundal invertebrates.

Macroalgae (e.g. *Chara*), and emergent, floating leaf, and submerged macrophytes were collected using a weighted plant rake and dip nets. Periphyton was removed by scrubbing with a tooth brush, and calcium carbonate deposits removed by wetting in 1% HCL. The plants were frozen, then dried and ground using a hand pestle. Periphyton removed from macrophytes was poured onto a precombusted GF/C filter, wetted with 1% HCL to remove carbonates, frozen, and then dried. We also collected leaf tissue of several species of terrestrial plants in the watershed of each lake, and samples were processed similarly to aquatic plants. Macroalgae, macrophytes, periphyton, and terrestrial plants were all analyzed for ^{13}C, ^{15}N, ^{2}H, and ^{34}S by the same labs as used for fish and aquatic invertebrates.

Our focus is on understanding trophic relationships and patterns of energy flow in our three study lakes, but we were also interested in testing whether patterns of energy flow differed between Lake Carlos and Lake Ida due to high densities of zebra mussels in the former. Our overall working hypothesis is zebra mussels reduce the abundance of phytoplankton, and may cause the lake food web to become more dependent on littoral and sediment energy sources in Lake Carlos relative to Lake Ida. If this is true, $\delta^{13}C$ values should be higher in fish and aquatic invertebrates in Lake Carlos, reflecting the increased importance of littoral primary production. Moreover, $\delta^{34}S$ in aquatic invertebrates and fish should also be lower in Lake Carlos if lake sediment (which is depleted in $\delta^{34}S$) is more important for the food web relative to Lake Ida. One problem for between-lake comparisons is stable isotope values naturally vary among lakes. Thus, we used a baseline-correction where the $\delta^{34}S$ and $\delta^{13}C$ values of fish and aquatic invertebrates are subtracted from the $\delta^{34}S$ and $\delta^{13}C$ values of mussels in each lake (Vander Zanden et al. 1999). Mussels are preferred for baseline corrections as they are long-lived and don't show the within-lake temporal variability often observed with zooplankton or other small-bodied primary consumers feeding on phytoplankton (reviewed by Cabana and Rasmussen 1994). We then used t-tests to test whether baseline corrected $\delta^{34}S$ and $\delta^{13}C$ values differed between Lake Carlos and Lake Ida. Mussels were collected under a DNR collector's permit for each lake by snorkeling.

Our analyses in this report have two focal areas. First we assessed how well isotopes of C, N, S, and H describe food webs and patterns of energy flow in our study lakes. For this focal point we tested whether $\delta^{2}H$ differed between terrestrial and aquatic primary producers in each study site, and we also tested whether $\delta^{13}C$ differed between littoral versus pelagic primary producers. We also used biplots of $\delta^{15}N$ and $\delta^{13}C$ and biplots of $\delta^{34}S$ and $\delta^{13}C$ of major groups of primary producers, aquatic invertebrates, and fish to assess the ability of the isotopes to distinguish patterns of energy flow in each lake. We grouped fish species into the common trophic groups of benthivores, planktivores, and piscivores for this analysis, but kept Cisco as a distinct group as they are thought to occupy a different niche and be heavily dependent on pelagic energy sources. We also created a "mixed-diet" group of fish (e.g. Black Crappie, Rock Bass) that was comprised of fish species thought to fall somewhere between piscivores and planktivores in their diet.

Our second focal area was focused on demonstrating the application of isotope studies to address questions on lake food web structure. For this focal area we first did a species-level analysis of fish in Elk Lake to assess niche overlap and potential competition among major groups of gamefish. Second, we use baseline-corrected values of $\delta^{34}S$ and $\delta^{13}C$ to test whether the importance of littoral and benthic energy sources differed between Lake Carlos and Lake Ida. As described above, the hypothesis here is zebra mussels reduce phytoplankton abundance, potentially forcing the lake food web to be more reliant on littoral and sediment energy sources. If this is the case, baseline-corrected fish and aquatic invertebrates should have lower $\delta^{34}S$ and higher $\delta^{13}C$ values in Lake Carlos compared to Lake Ida.

We are still assessing the impact of lipids on $\delta^{13}C$ values in our samples, but preliminary results indicate effects are small. Thus, results presented here are not corrected for lipid content.

Results

Values of $\delta^{13}C$ successfully distinguished energy originated from pelagic versus littoral primary producers, as $\delta^{13}C$ was significantly higher in littoral primary producers in all three lakes (Fig. 1 top panel). In contrast, $\delta^{2}H$ did not consistently distinguish between terrestrial and aquatic primary producers, as $\delta^{2}H$ was significantly higher in terrestrial plants in Lake Carlos only (Fig. 1 bottom panel). Given the inability of $\delta^{2}H$ to distinguish between terrestrial and aquatic primary producers (its main application in this study), we conducted no further testing with $\delta^{2}H$ and focus the rest of our analyses on isotopes of C, S, and N.

Biplots also showed that $\delta^{13}C$ is useful for distinguishing patterns of energy flow originating from pelagic versus littoral primary producers in all three lakes, while $\delta^{15}N$ was able to distinguish trophic position in food webs (Fig. 2). In all three lakes $\delta^{13}C$ values indicated profundal chironomids, zooplankton, and cisco were more reliant on phytoplankton production, constituting a "pelagic" component of the food webs. Littoral primary producers (submerged aquatic plants, filamentous algae, and periphyton) had larger values of $\delta^{13}C$, and littoral invertebrates had $\delta^{13}C$ values between those of phytoplankton and littoral plants. While Cisco appeared to acquire most of their energy from phytoplankton, all other groups of fish had $\delta^{13}C$ values between phytoplankton and littoral plants, indicating more balanced use of energy from pelagic and littoral habitats. Worth noting is $\delta^{13}C$ values for fish in Lake Carlos were much closer to $\delta^{13}C$ values of littoral primary producers (approximately 3 $\delta^{13}C$ units lower) compared to nearby Lake Ida (where fish were approximately 11 $\delta^{13}C$ units lower than littoral primary producers). This suggests fish communities in Lake Carlos have a greater reliance on littoral production compared to Lake Ida, one of our hypothesized impacts of zebra mussels. $\delta^{15}N$ was also effective in discerning trophic position in all three lakes. Primary producers had the lowest values, herbivorous invertebrates were higher, followed by benthivores, planktivores, and mixed-diet fish, with piscivores having the highest values overall. It is also worth noting that phytoplankton had higher $\delta^{15}N$ values relative to littoral primary producers in all three lakes, suggesting $\delta^{15}N$ may also be useful for distinguishing pelagic sources of energy production.

Biplots of $\delta^{13}C$ and $\delta^{34}S$ were effective for distinguishing energy sources from detritus in lake sediments versus the lake water column ($\delta^{34}S$) and pelagic versus littoral energy sources ($\delta^{13}C$) (Fig. 3). Results for $\delta^{13}C$ are discussed above, so here we focus on $\delta^{34}S$. Across all three lakes $\delta^{34}S$ was much lower in sediment-feeding chironomids in the profundal zone, indicating this isotope works well for assessing the importance of deep-water midges in lake food webs. The results indicate that fish in Lake Ida may be more reliant on sediment-feeding prey than fish in Lake Carlos and Elk Lake, as all groups of fish in Lake Ida had $\delta^{34}S$ values closer to profundal chironomids than did fish in Elk Lake and Lake Carlos. It is possible the low importance of sediment as an energy sources in Lake Carlos is due to zebra mussels reducing phytoplankton abundance, which would reduce the amount of energy entering the profundal zone, which may in turn reduce chironomid abundance. Supporting this notion is the fact that chironomid abundance was extremely low on Lake Carlos compared to both Elk Lake and Lake Ida, as several sampling trips were needed to collect sufficient mass for isotope analysis. In contrast, sufficient chironomid mass was easily obtained for both Elk Lake and Lake Ida on a single sampling trip.

Our second focal point was applying stable isotope analysis to address specific questions about structure of lake food webs. Our first analysis was a species-level assessment of trophic position of fish in Elk Lake to assess species niche space, thus potential competition among gamefish. $\delta^{13}C$ results indicated a gradient of habitat use by planktivores, with Cisco the most planktonic, White Sucker and Blackchin Shiner intermediate, and Bluegill, Pumpkinseed, Bluntnose Minnow, and Banded Killifish the most littoral (Fig. 4). Classifying Yellow Perch and Rock Bass as "mixed-diet" fish was confirmed by isotope value obtained for these species, as both fish had $\delta^{15}N$ values intermediate between planktivores and piscivores. For piscivorous gamefish, Northern Pike and Walleye were more reliant on littoral energy than Muskellunge. In fact, adult Muskellunge appeared to feed nearly exclusively on pelagic Cisco in Elk Lake.

Higher $\delta^{15}N$ values in Muskellunge, Walleye, and Northern Pike indicate these fish are strongly piscivorous, but Northern Pike appear to feed at a different trophic position. Thus, these three species of gamefish appear to have partitioned niche space in Elk Lake, which may reduce competition. Muskellunge and Walleye feed at similar trophic levels but reduce competition by feeding in different habitats, while Walleye and Northern Pike feed in similar habitats but reduce competition by feeding at slightly different trophic positions.

Our last analysis of lake food webs was focused on using $\delta^{34}S$ and $\delta^{13}C$ to test whether zebra mussels cause the food web in Lake Carlos to be more reliant on sediment and littoral energy sources compared to Lake Ida. Results showed no differences in baseline-corrected $\delta^{34}S$ values for three species of fish between lakes (Fig. 5), and only one group of invertebrates (profundal chironomids) was significantly lower in Lake Carlos than Ida (Fig. 6). This suggests the importance of energy from sediment sources may be similar between the two lakes, in contrast to results shown in Figure 3. Further analyses will be required to clarify the degree to which energy from sediment differs between Lake Carlos and Ida. Results for $\delta^{13}C$ indicated that the Lake Carlos food web was much more reliant on littoral energy relative to Lake Ida. All three species of fish showed higher baseline-corrected $\delta^{13}C$ values in Lake Carlos (Fig. 7), as did five out of six invertebrate groups (Fig. 8).

Discussion

Our results indicate that stable isotopes of C, N, and S are powerful tools for assessing patterns of energy flow in lakes, and for testing how lake food webs respond to exotic species and other anthropogenic stressors. $\delta^{13}C$ was useful for identifying pelagic versus littoral sources of energy, $\delta^{34}S$ for distinguishing between sediment and water-column energy sources, and $\delta^{15}N$ for assessing trophic position. Taken together, these isotopes provide a comprehensive assessment of energy flow and trophic position in lakes, an assessment not possible with other techniques such as diet analysis. Thus, we recommend isotope analysis as one of many tools that can be used to understand and manage lake food webs in Minnesota.

In contrast, our analysis of $\delta^{2}H$ indicates this isotope has limited use for studying lake food webs at the present time. Previous research has used this isotope to successfully distinguish aquatic from terrestrial production (Doucett et al. 2007, Cole et al. 2011), but we failed to achieve similar results here. We suspect the problem lies with the methods used by stable isotope labs to analyze $\delta^{2}H$. The lab that processed our $\delta^{2}H$ samples used the same "benchtop equilibrium" approach researchers have used in previous published papers. However, the lab that did the analyses for these papers no longer uses the benchtop equilibrium method because they feel it doesn't work well enough, and they are in the process of developing a new technique. Our results support the notion that using $\delta^{2}H$ to distinguish terrestrial from aquatic production is difficult and we cannot recommend its use at the current time. Perhaps improvements or developments of new analytical techniques will make it more reliable in the future.

The utility of isotope analyses was demonstrated in the two case studies presented here. Our species-level analysis of fish in Elk Lake highlighted the niche partitioning of major game fish, and also indicated Muskellunge are likely to be most sensitive to changes in abundance of Cisco. Results from Lake Carlos and Lake Ida indicate that zebra mussels may increase the reliance of littoral energy sources in lake food webs. Fish species balanced on pelagic and littoral energy sources may be forced to greater reliance on littoral energy sources, and our data indicates this is already the case. Whether this ultimately results in lower population abundance of certain fish species or whether fish communities begin to shift to species focused more heavily on the littoral habitat both remain to be detected.

We emphasize the results for Elk Lake and Lake Carlos presented here are preliminary and may change with subsequent analyses. Our next steps are to further assess lipid corrections for ^{13}C, perform additional species-level analyses in all three lakes, and use mixing models to assess the proportion of energy coming from different prey and different types of habitat in lakes. Our ultimate goal is to publish this work in peer-reviewed journals.

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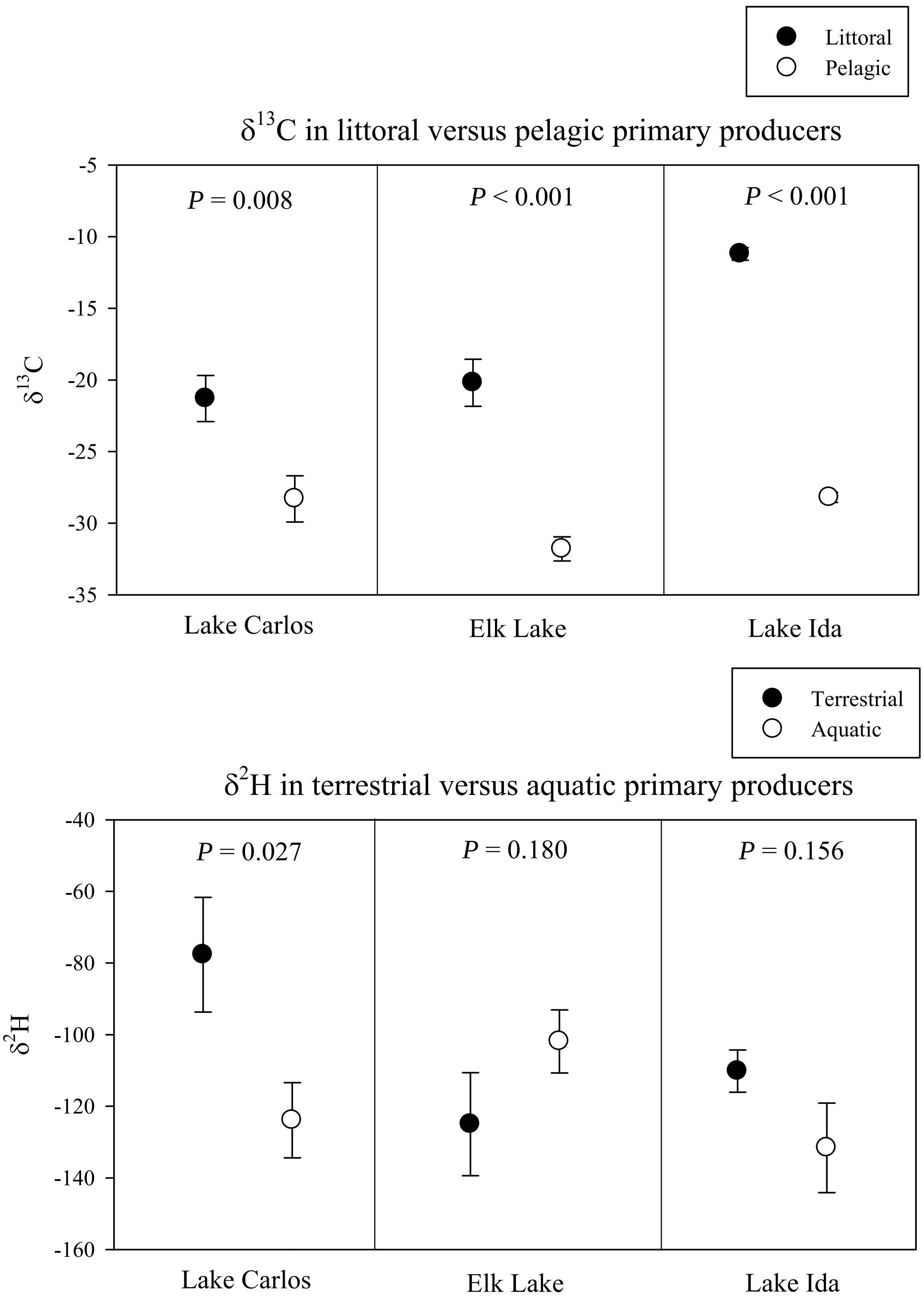

Figure 1. Results for testing for significant differences in $\delta^{13}C$ between littoral versus pelagic primary producers (top panel) and for significant differences in $\delta^{2}H$ between terrestrial versus aquatic primary producers (bottom panel) in the three study lakes.

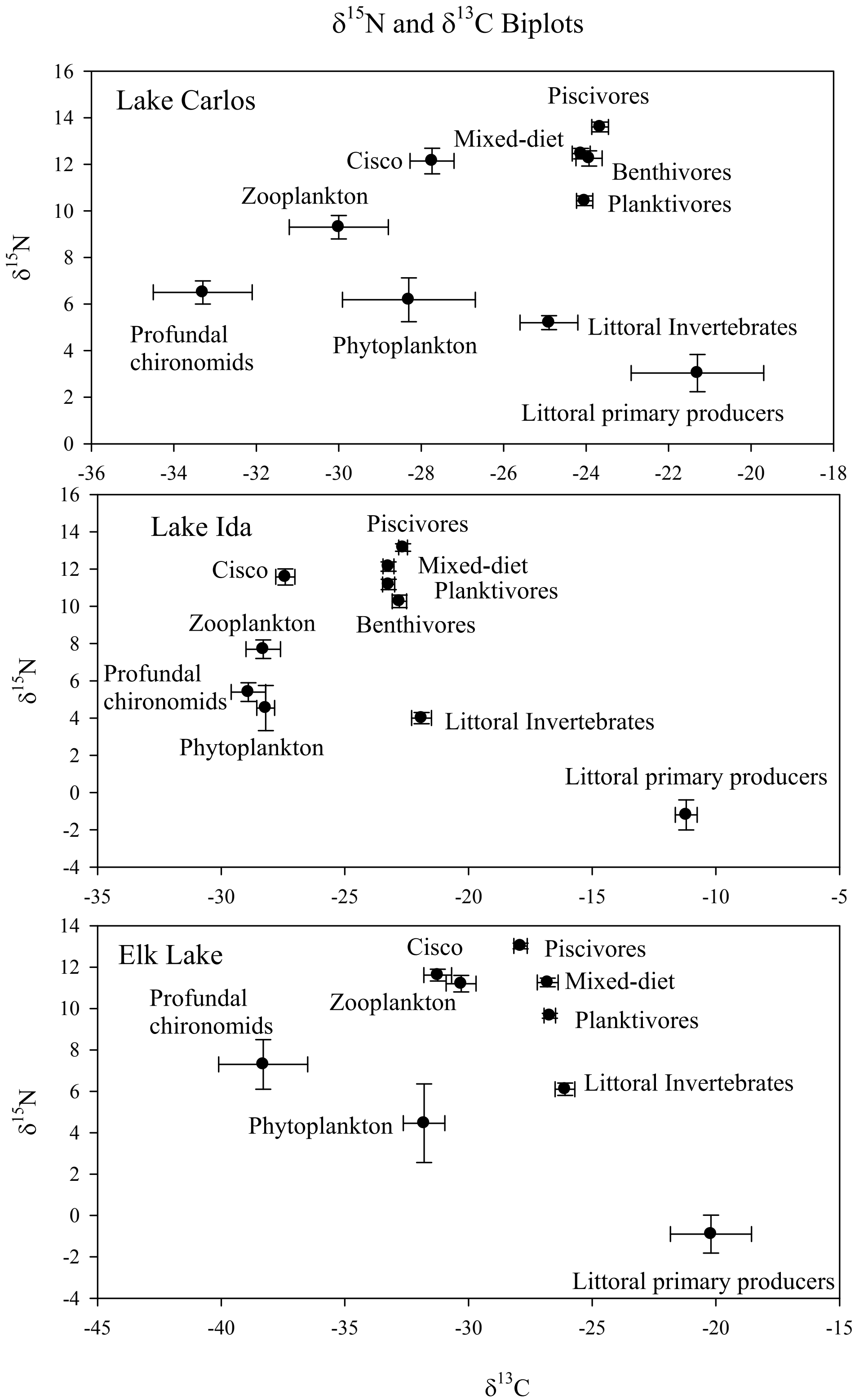

Figure 2. Biplots of $\delta^{15}N$ and $\delta^{13}C$ in major groups of primary producers, aquatic invertebrates, and fish in the three study lakes.

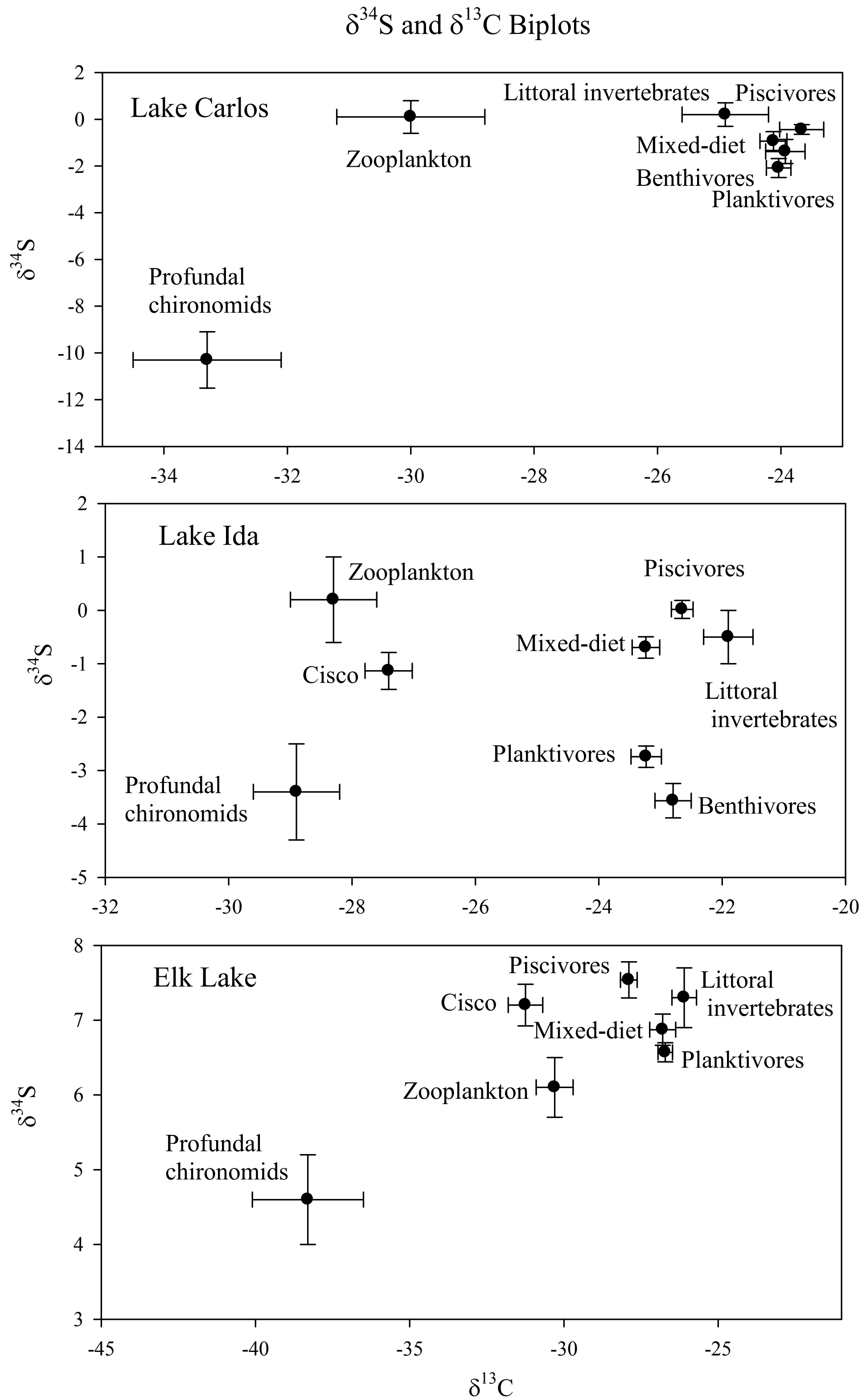

Figure 3. Biplots of $\delta^{34}S$ and $\delta^{13}C$ in major groups of aquatic invertebrates and fish in the three study lakes.

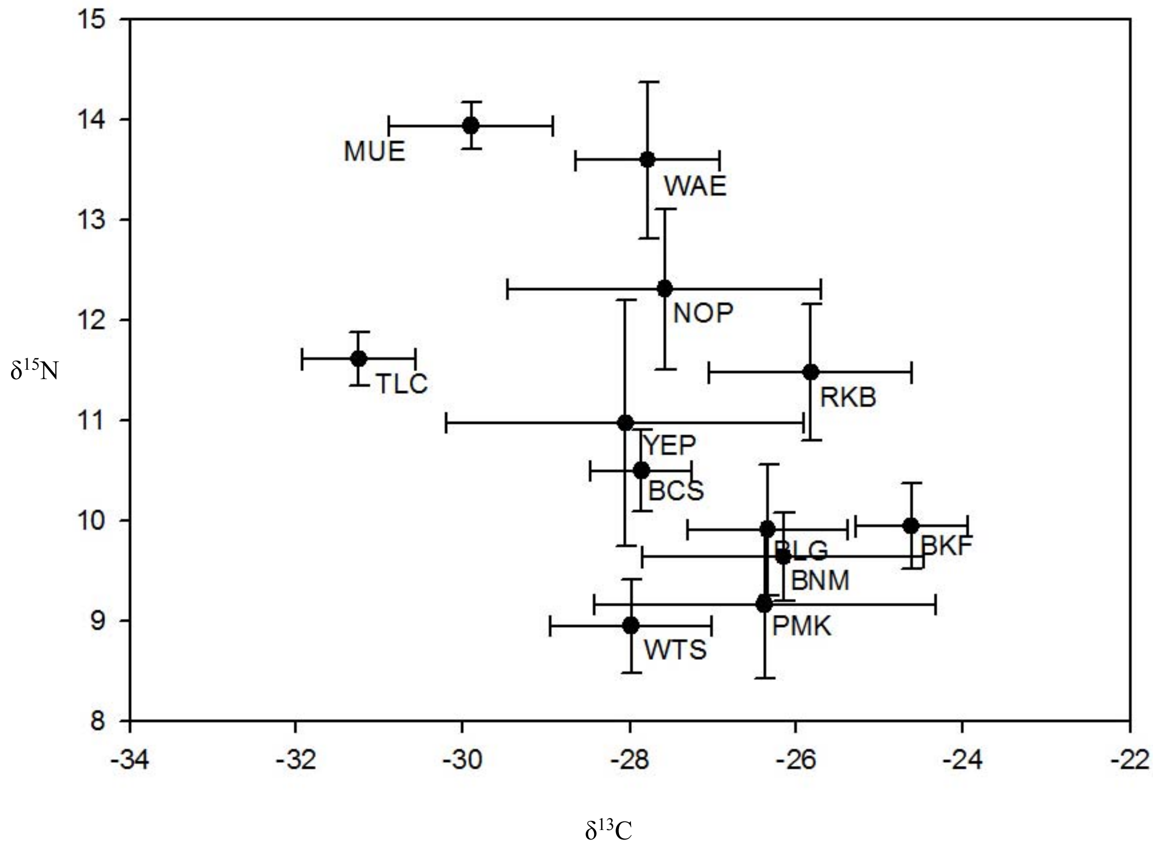

Figure 4. Biplots of $\delta^{15}N$ and $\delta^{13}C$ in 12 species of fish in Elk Lake. Fish acronyms are MUE= Muskellunge, WAE= Walleye, NOP= Northern Pike, TLC= Cisco, RKB= Rock Bass, YEP= Yellow Perch, BCS= Blackchin shiner, BLG= Bluegill, BKF= Banded Killifish, BNM= Bluntnose Minnow, WTS= White Sucker, PMK= Pumpkinseed.

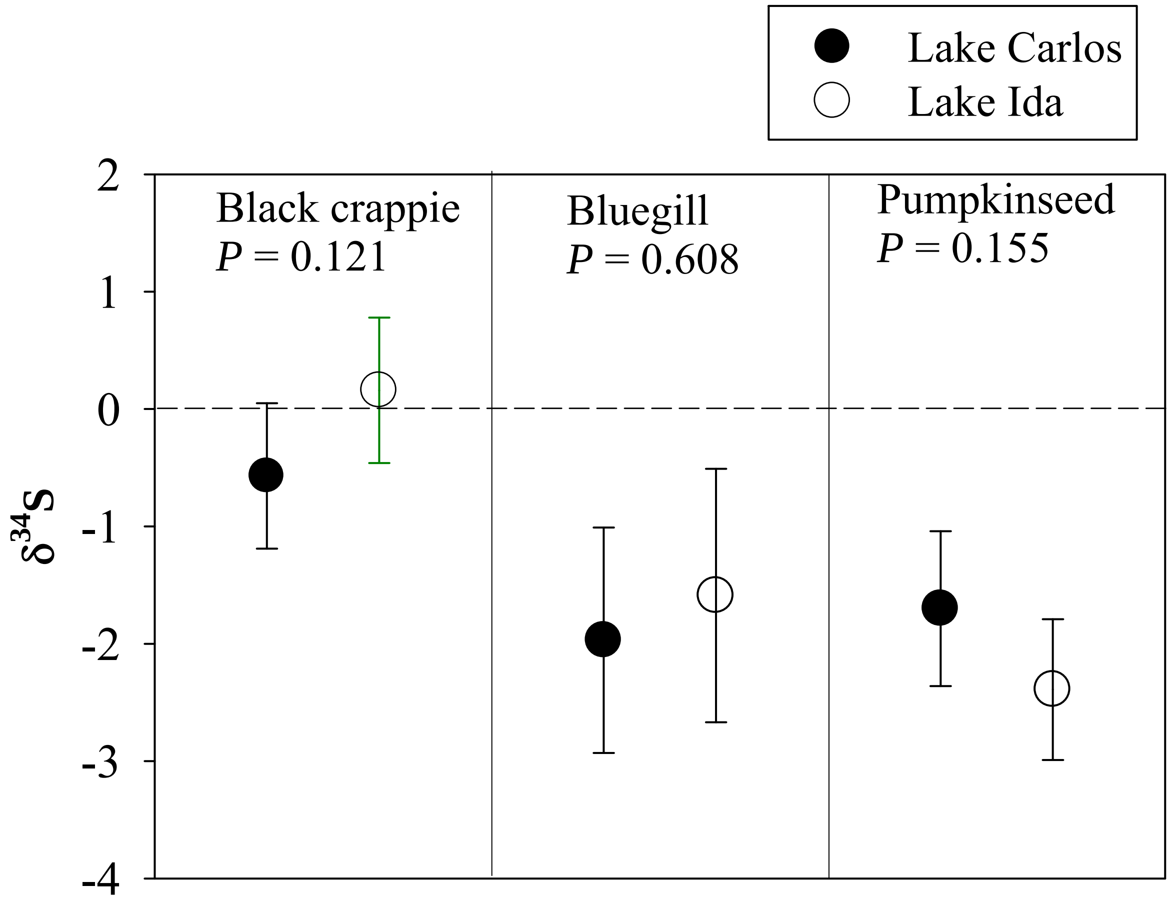

Figure 5. Results of testing for significant differences between Lake Ida and Lake Carlos in baseline-corrected $\delta^{34}S$ values of three fish species. Lower $\delta^{34}S$ values indicate higher importance of sediment energy sources, but no differences were detected between the two lakes. The dashed horizontal line at zero represents the $\delta^{34}S$ value of mussels in each lake that were used for baseline corrections.

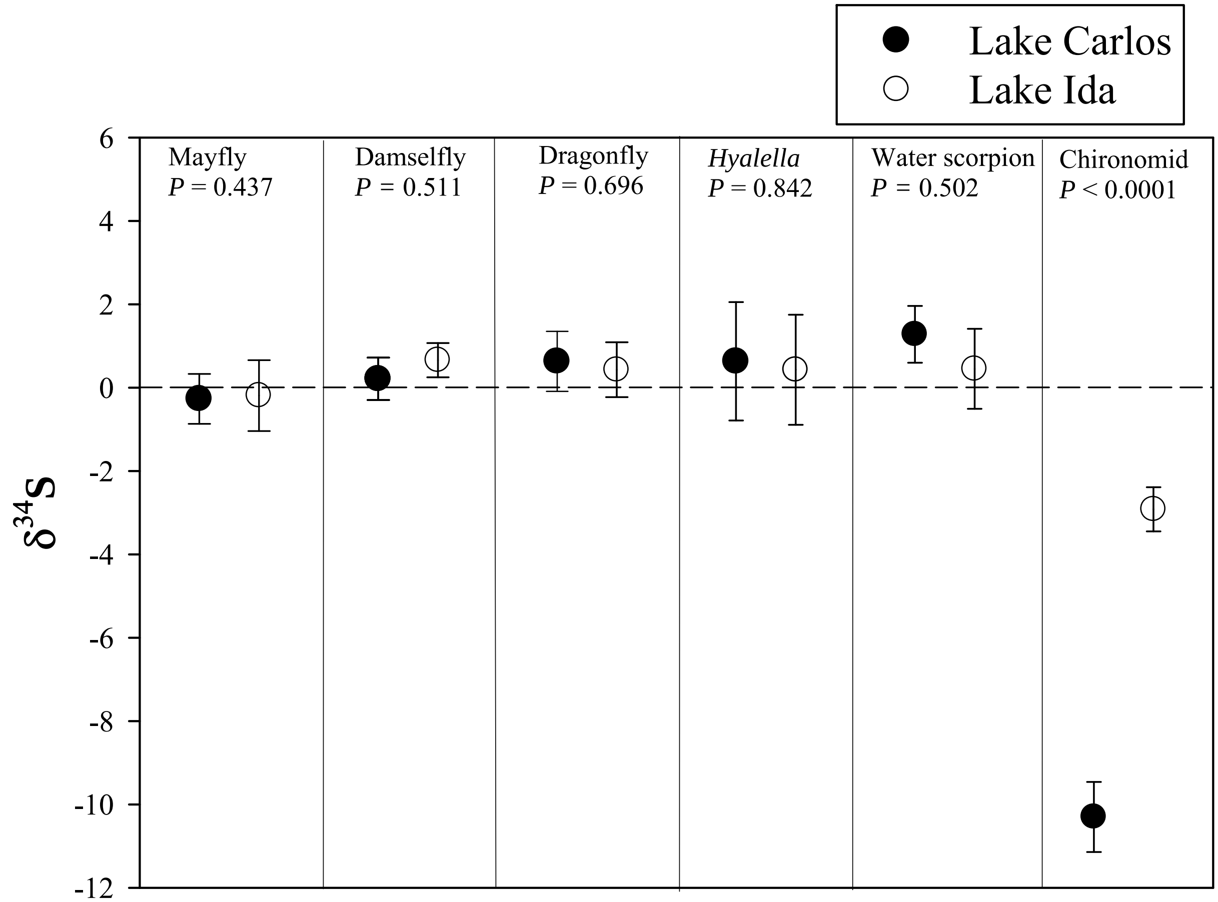

Figure 6. Results of testing for significant differences between Lake Ida and Lake Carlos in baseline-corrected $\delta^{34}S$ values of six groups of aquatic invertebrates. Lower $\delta^{34}S$ values indicate higher importance of benthic energy sources, but chironomids were the only group that differed between the two lakes. The dashed horizontal line at zero represents the $\delta^{34}S$ value of mussels in each lake that were used for baseline corrections.

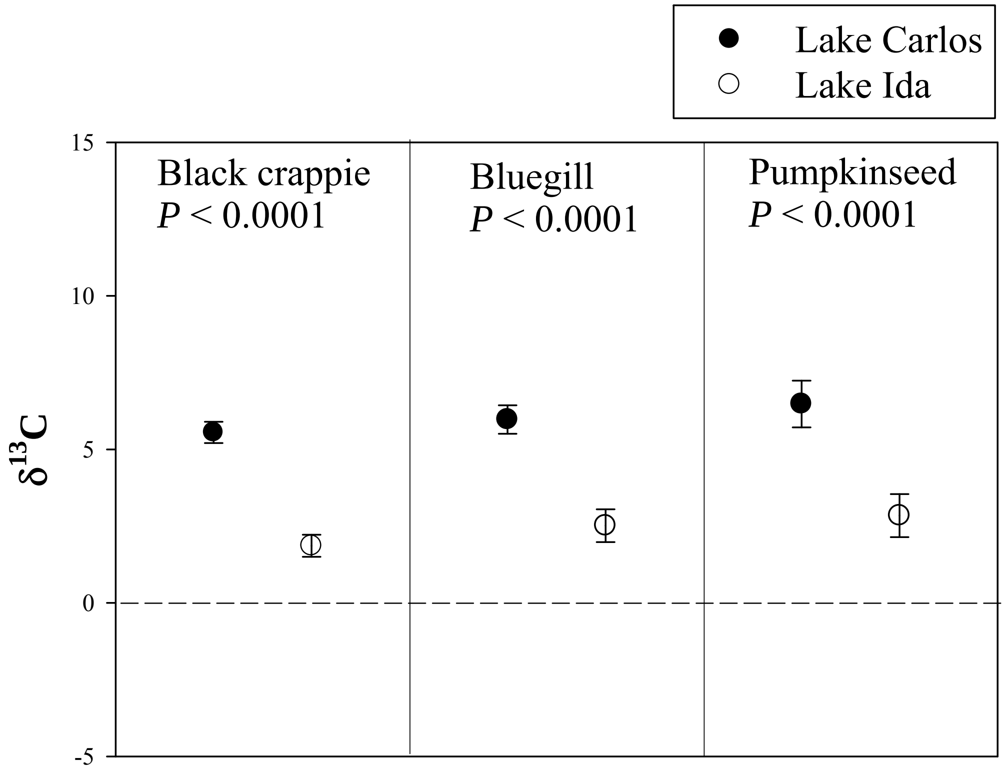

Figure 7. Results of testing for significant differences between Lake Ida and Lake Carlos in baseline-corrected $\delta^{13}C$ values of three fish species. Higher $\delta^{13}C$ values indicate higher importance of littoral energy sources, and $\delta^{13}C$ values were significantly higher in Lake Carlos in all three species. The dashed horizontal line at zero represents the $\delta^{13}C$ value of mussels in each lake that were used for baseline corrections.

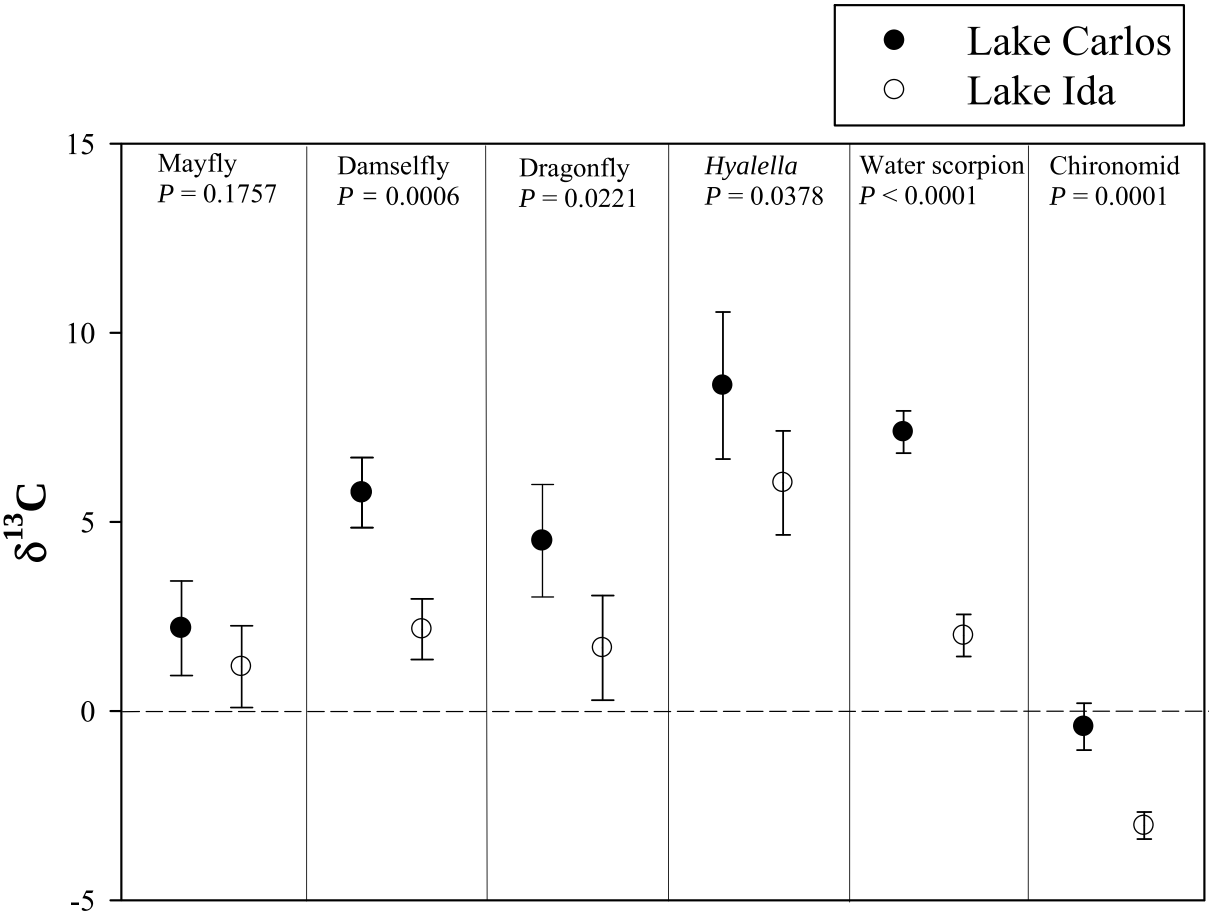

Figure 8. Results of testing for significant differences between Lake Ida and Lake Carlos in baseline-corrected $\delta^{13}C$ values of six groups of aquatic invertebrates. Higher $\delta^{13}C$ values indicate higher importance of littoral energy sources, and $\delta^{13}C$ values were significantly higher in Lake Carlos in five of the six groups. The dashed horizontal line at zero represents the $\delta^{13}C$ value of mussels in each lake that were used for baseline corrections.

Outcome 5

Assessment of Biodiversity of Chironomidae in Tier 1 Sentinel Lakes

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Introduction: The Section of Fisheries of the Minnesota Department of Natural Resources is leading a collaborative effort to monitor and record biological and chemical changes that may occur over long periods in a subset of lakes that are representative of the state's most common aquatic environments. As part of this monitoring effort, we have been tasked with determining the taxonomic composition of Chironomidae in 12 of the Tier 1 Sentinel Lakes and assessing their significance as relates to lake trophic states.

Scientific Context for the Biodiversity of Chironomidae Portion of the Study: Chironomidae are a family of aquatic flies (Diptera) that are exceptionally species rich, and collections of larvae have been shown to be useful in documenting water and habitat quality in freshwater systems (Ferrington, 2008; Ferrington *et al*., 2008). In Minnesota lakes, these flies serve as the primary food sources for a number of sport and forage fish species, and they are essential links in nutrient cycling and energy flows in freshwater systems. In Minnesota lakes they are usually the most biologically diverse family of aquatic insects. In some parts of the world they have been integrated into biological monitoring programs and are considered to be reliable indicators of lake water quality, habitat structure and climate conditions.

Chironomidae can be very species rich and numerically abundant in healthy aquatic habitats, with more than 100 species known to occur in some streams in Minnesota (Bouchard, 2007), and at least 80 species have been documented in some urban lakes of the Twin Cities (Rufer & Ferrington, 2008). At the beginning of this study, we anticipated encountering more than 100 species across the wide range of trophic states represented among the 12 Tier 1 Sentinel Lakes.

There are several practical limitations when using collections of larvae as part of routine biological monitoring programs. Larval populations can often be very dense, requiring substantial time commitment to sort and process larvae. It is generally not possible to identify larvae consistently to species-level, consequently many programs use genus, tribe or even subfamily as their target level for taxonomic resolution, which can mask water quality relationships that are only evident at genus or species level. In addition, sampling for larvae in lakes typically requires boats to deploy grab- or dredge-type samplers, and may even require winches for retrieval of the grab or dredge samplers, which adds to work loads and costs. Consequently we have utilized collections of surface-floating pupal exuviae (SFPE) as our preferred method to assess the biodiversity of Chironomidae in the Tier 1 Sentinel Lakes selected for this project. This method is being used in England, Portugal, Spain, Germany and several Scandinavian countries, and is explained in more detail in a subsequent section of this report.

No previous comprehensive studies of Chironomidae using field sampling designs and collection protocols similar to ours has been completed in all 12 Tier 1 Sentinel Lakes. Prior to initiating this study it was anticipated that some undescribed species and/or life stages would be encountered.

In this report our project scope, sampling method and scientific research findings are summarized (SECTION I), along with a status overview of other deliverables that were part of the subcontract (SECTIONS II.1- II.5).

SECTION I

Scope of Research: For this project we used collections of SFPE, an innovation field collection technique that has been shown to be efficient at detecting species and effective in determining phenological patterns. Review of literature shows that the SFPE method has been used for two contrasting types of study designs: (1) randomized sampling during emergence seasons, versus (2) repeated sampling at regular intervals during ice-free periods.

The relative effectiveness of these two approaches for assessing biodiversity of Chironomidae has not been previously evaluated in lakes in Minnesota, although Rufer and Ferrington (2008) have shown that three repeated samples at approximately monthly intervals can detect up to 80% of species in lakes known to have in excess of 75 species of Chironomidae. Consequently, we used randomized sampling during the 2014 ice-free period and repeated sampling at approximately one-month intervals during 2015, to evaluate the relative merits of these two designs for assessing species composition versus lake trophic states. We selected a lake trophic state model developed by Sæther (1979) to use as our basis of comparison.

Species-level identifications of Chironomidae are primarily made or confirmed using the adult male stages, which require substantial amounts of time and specialized sampling gear to collect quantitatively. However, adults collected qualitatively with aerial nets can be used to assist or guide species-level identifications. Although this type of collection is not quantitative, and may oversample, undersample, or even miss species altogether, males are not typically strong fliers and do not travel in large numbers to great distances from natal habitats. Consequently, adults collected in moderate or high numbers around our sample sites are typically considered to have emerged from that specific lake, and the collections with aerial nets can be valuable. On most sample dates adults were collected and have been partially examined or preserved and stored for future processing. It is anticipated that these extra collections will continue to provide a valuable reserve for confirming generic-level and species-level identifications into the future.

Sampling Method: The innovative field collection technique employed during this research is highly cost-effective (Ferrington 1987; Ferrington *et al*. 1991; Bouchard & Ferrington 2011). When used in lake monitoring, the method relies on collections of surface-floating pupal exuviae (SFPE) that are concentrated on the water surface by wind and wave action. Both wind direction and intensity of wave action can change over time, consequently we collect at pre-determined points in each lake (depending on bay structure, prevailing winds, habitats and lake access) to ensure that collections are representative of the species emerging on each sample date.

The project consisted of two years of fieldwork. Collections of SFPE representing species occurring in a broad array of habitats at each of the pre-determined sample sites in each SLICE Lake were taken during ice-free months. During the first year of the project (2014) samples were collected at random time intervals. Therefore, these collections of SFPE can be interpreted as providing spatially and temporally integrated samples of chironomid emergence, allowing us to efficiently determine species composition at various points in time, and begin to assess community emergence phenology for each lake.

By contrast, during the second year of the project (2015) samples of SFPE were collected approximately monthly (as weather conditions permitted) from early spring (soon after ice-out) through early fall and provided better evaluation of phenology.

GPS readings were made for each of the collection sites, and habitat descriptors (e.g., substrate and vegetative cover, water temperature pH conductivity and dissolved oxygen) were recorded. These abiotic habitat descriptors are linked to individual species and site records, in construction of our electronic databases. Similarly, data for collections made during the study have been and will continue to be linked to GIS shells of each respective lake, and maps of sample sites were constructed.

Figure 1: Collection pan, sieve and sample jars.

Our sampling devices consisted of a moderate-sized white plastic pan to scoop SFPE, and a 125 micron aperture sieved to retain the scooped SFPE but allow excess water to drain off (see Figure 1). We used a ten-15 minute timed sampling approach as recommended by Ferrington *et al.* (1991), except in instances where the numbers of exuviae accumulated through wind or wave action were excessively dense. In these types of conditions, sampling time was shortened, but often more than 1,000 SFPE were quickly collected.

Samples were sorted in their entirety if less than 300 specimens were collected. Otherwise, only 300 randomly selected SFPE were subsampled and identified. Selected specimens were permanently slide mounted and identified. Voucher specimens were designed from slide mounted specimens. All other sorted exuviae were stored in labeled one-dram vials with 80% ethanol as preservative. Samples have all been curated for long-term storage according to pre-approved lab protocols.

We have recently published a video of our lab processing procedures (Kranzfelder et al. 2015) that can be viewed for details related to sample sorting, specimen preparation, labeling and curation. This video resource also provides discussions of some relative merits and cost-effectiveness compared to other types of collections for inventorying Chironomidae faunas. It is also useful to click on the link below to access a video that demonstrates in more detail the methodology for collecting SFPE: https://www.facebook.com/The-Chironomidae-Research-Group-Page-114778358598205/

All twelve of the pre-selected 12 Tier I lakes were sampled in both 2014 and 2015. These lakes are the first twelve lakes listed in Table 1. In 2015 three additional lakes (White Iron Lake, Elephant Lake and Echo Lake) were sampled as time and resources permitted. White Iron Lake was sampled on several dates at two locations. The remaining lakes (Elephant Lake and Echo Lake) were each only sampled once in 2015. Two additional lakes (Hill and Red Sand) were sampled once each in early 2016.

Identification to genus was accomplished using the taxonomic keys in Ferrington *et al.* (2008). Primary literature was used for delineation of species. List of literature used is available on request, but consisted largely of the same publications used by Egan and Ferrington (2015). All slide-mounted specimens are affixed with two labels. The larger label provides information regarding County, locality, lake, date of collection and person taking the sample. The second label lists the genus (and species, when possible) and name of person preforming the identification. SFPE that were not slide mounted are stored in vials with locality and identification labels inside the vial.

Data generated in the project during 2015 were used to test the applicability of a Lake Trophic State Model published by Sæther (1979) based on studies in northern lakes primarily in Canada and across Scandinavia. The model use the ratio of total phosphorus to mean lake depth (TP/MLD) to categories lakes into trophic categories, and maps the occurrences and abundances of chironomid species to differing lake trophic states. The publication also includes a text key, based on abundances of particular "indicator species," to assign lakes to trophic categories based on occurrence and abundance (or absence) of particular species. The data used as input to develop this model were based on collections of larvae. However, it can logically be assumed that data from collections of SFPE will also be useful for mapping taxa to lake trophic state, and thereby allow for assignment of individual lakes to the different trophic categories proposed in the model. To begin the process it is necessary to determine TP and MLD for each of the 12 Tier 1 Sentinel Lakes. These data were obtained from existing lake reports made available on-line by MN DNR, and some of the historical data are presented in Table 1.

Table 1. Background information for lakes sampled for this project. Lakes are listed by the ratio of total phosphorus (TP) to mean lake depth (MLD). This ratio was proposed by Sæther (1979) as a useful metric for predicting the taxonomic composition of Chironomidae as a function of lake trophic state. (Data obtained from MN DNR web site at http://www.dnr.state.mn.us/fisheries/slice/sentinel.html)

Lake	DNR ID#	Watershed Acres	Surface Acres	Max Depth (ft)	Mean Depth (ft) (=MLD)	TP	Ratio TP to MLD	Mixing Class
Trout	16004900	1,148	259	77	35	6	0.17	Stratified
Ten Mile	11041300	25,510	5,069	208	53	12	0.23	Stratified
Carlos	21005700	156,569	2,606	160	46	16	0.35	Stratified
Cedar	49014000	1,603	236	88	37	13	0.35	Stratified
Elk	15001000	8,759	300	97	22	21	0.96	Stratified
Bear Head	69025400	2,723	663	46	12	14	1.16	Stratified
South Twin	44001400	6,745	1,128	29	12	18	1.50	Mixed
Pearl	73003700	16,311	753	18	10	50	5.00	Mixed
Madison	07004400	11,167	1,442	58	11	81	7.36	Stratified
Portage	29025000	2,996	429	15	7	60	8.57	Mixed
St. James	83004300	2,340	203	14	4	54	13.50	Mixed
Shaokotan	41008900	8,817	995	11	8	132	16.50	Mixed
White Iron	69000400	595,864	3,241	43	16	21	1.31	Mixed
Elephant	69081000	4,420	725	30	15	21	1.40	Stratified
Hill	01014200	25,736	795	48	9	24	2.67	Stratified
Red Sand	18038600	4,555	510	15	4	24	6.00	Mixed
Echo	69061500	32,069	1,140	11	6	43	7.17	Mixed

RESULTS

Taxa Collected

A comprehensive summary of the genera and species documented emerging from all 12 Tier 1 SLICE lakes is provided in Table 2. The genera are arranged by subfamily, then tribe, and listed alphabetically within the subfamily or tribe. Seventy-one genera were documented, representing at least 114 species.

The subfamily Chironominae (includes Tribes Chironomini, Pseudochironomini and Tanytarsini) was represented by the largest number of genera (26) and species (53). Twenty-one genera, consisting of 35 species, of Orthocladiinae were documented. The subfamilies Tanypodinae (includes Tribes Coelotanypodini, Macropelopiini, Pentaneurini, Procladiini and Tanypodini) consisted of 15 genera and 26 species. Diamesinae was represented by one species.

Table 2. Chironomidae species detected in Sentinel Lakes during 2014/2015.

Orthocladiinae		**Chironominae**	
Acricotopus	(one species)	**Tribe Chironomini**	
Brillia	(one species)	*Chironomus*	(three species)
Chaetocladius	(one species)	*Cladopelma*	(two species)
Corynoneura	(three species)	*Cryptochironomus*	(two species)
Cricotopus	(five species)	*Cryptotendipes*	(two species)
Epoicocladius	(one species)	*Demeijeria*	(one species)
Eukiefferiella	(one species)	*Dicrotendipes*	(three species)
Heterotrissocladius	(one species)	*Einfeldia*	(one species)
Hydrobaenus	(one species)	*Endochironomus*	(two species)
Hydrosmittia	(one species)	*Goeldichironomus*	(one species)
Lapposmittia (?)	(one species)	*Glyptotendipes*	(three species)
Limnophyes	(one species)	*Harnischia*	(one species)
Nanocladius	(two species)	*Lauterborniella*	(one species)
Orthocladius	(two species)	*Microtendipes*	(one species)
Parakiefferiella	(three species)	*Omisus*	(one species)
Parametriocnemus	(one species)	*Parachironomus*	(three species)
Psectrocladius	(three species)	*Paracladopelma*	(one species)
Pseudosmittia	(one species)	*Paratendipes*	(one species)
Synorthocladius	(one species)	*Polypedilum*	(five species)
Thienemanniella	(three species)	*Saetheria*	(one species)
Zalutschia	(one species)	**Tribe Pseudochironomini**	
		Pseudochironomus	(three species)
Tanypodinae		**Tribe Tanytarsini**	
Tribe Coelotanypodini		*Cladotanytarsus*	(three species)
Clinotanypus	(one species)	*Micropsectra*	(two species)
Coelotanypus	(one species)	*Paratanytarsus*	(three species)
Tribe Macropelopiini		*Rheotanytarsus*	(one species)
Psectrotanypus	(one species)	*Stempellina*	(one species)
Tribe Pentaneurini		*Stempellinella*	(one species)
Ablabesmyia	(five species)	*Tanytarsus*	(five species)
Arctopelopia	(one species)		
Conchapelopia	(two species)		
Guttipelopia	(one species)	**Tribe Procladiini**	
Helopelopia	(one species)	*Procladius*	(four species)
Labrundinia	(two species)	**Tribe Tanypodini**	
Larsia	(one species)	*Tanypus*	(two species)
Nilotanypus	(one species)		
Paramerina	(one species)	**Subfamily Diamesinae**	
Thienemannimyia	(one species)	Potthastia	(one species)

A comprehensive summary of the genera documented emerging from all 12 Tier 1 Sentinel Lakes in 2014 is provided in Table 3. The genera are listed alphabetically. Fifty-nine genera were documented.

Table 3. Chironomidae species detected in individual Tier 1 Sentinel Lakes during 2014.

Lake	Trout	Ten Mile	Cedar	Bear Head	Carlos	South Twin	Elk	Pearl	Saint James	Portage	Madison	Shaokotan	Total
Total Phosphorus (ug/L)	5.6	12	13	14.5	17	18	21	52	54	60	81	132	**Collected in**
Mean Lake Depth (feet)	35	52	37	12	45.7	12	21.8	9.8	4	6.8	11	7.9	**2014**
Genus													
Ablabesmyia	6	16	9	31	51	11	7	16	26	29	0	65	267
Arctopelopia	1	0	0	0	0	0	0	0	0	0	0	0	1
Brillia	1	0	0	0	0	0	0	0	0	0	0	0	1
Chironomus	0	1	16	6	28	12	9	24	15	6	15	23	155
Cladopelma	1	0	0	4	3	1	2	0	7	0	31	0	49
Cladotanytarsus	15	16	0	0	4	4	1	4	61	19	25	76	225
Clinotanypus	0	2	1	0	0	0	0	0	0	0	0	0	3
Coelotanypus	0	0	0	0	0	0	0	0	0	0	0	2	2
Conchapelopia	0	0	1	0	1	0	6	0	0	0	0	0	8
Corynoneura	16	13	13	23	8	4	0	23	0	8	0	21	129
Cricotopus	6	1	7	4	10	26	15	7	1	6	47	46	176
Cryptochironomus	0	2	0	2	6	5	1	4	1	6	10	10	47
Cryptotendipes	0	0	0	1	0	0	0	0	1	1	0	0	3
Dicrotendipes	1	0	58	12	81	6	42	37	32	4	20	1	294
Einfeldia	0	0	0	4	0	0	0	2	0	0	0	0	6
Endochironomus	0	1	0	0	0	5	24	3	0	2	52	1	88
Glyptotendipes	1	0	0	2	0	10	10	4	5	5	55	17	109
Guttipelopia	0	0	0	0	0	0	2	0	4	7	0	0	13
Harnischia	0	0	0	0	0	1	0	14	0	0	0	10	25
Helopelopia	2	0	0	3	0	0	0	0	0	0	0	0	5
Hydrobaenus	7	0	0	0	0	0	0	0	0	0	0	0	7
Labrundinia	0	3	16	6	5	16	12	13	3	13	0	0	87
Lapposmittia	0	2	0	0	0	2	0	0	0	0	0	0	4
Larsia	0	0	0	9	0	1	0	1	1	2	0	0	14
Lauterborniella	0	0	0	0	0	4	0	0	0	0	0	0	4
Limnophyes	1	0	0	0	0	0	0	0	0	0	0	1	2
Microtendipes	0	0	0	13	8	0	2	0	0	0	0	0	23
Nanocladius	2	1	6	2	4	42	0	15	0	6	3	0	81
Orthocladius	0	0	0	0	0	4	1	0	0	0	0	0	5
Parachironomus	0	2	0	2	0	0	1	8	25	4	16	31	89
Paracladopelma	0	1	0	0	0	0	0	0	0	0	0	0	1
Parakiefferiella	19	0	1	3	0	0	1	0	0	0	0	0	24
Paramerina	0	0	0	1	0	0	0	0	0	0	0	0	1
Parametriocnemus	0	0	0	1	0	0	0	0	0	0	0	0	1
Paratanytarsus	0	1	16	3	6	8	2	19	0	6	1	0	62
Paratendipes (?)	0	3	4	5	6	0	0	0	0	0	0	0	18
Phaenopsectra	0	0	0	0	1	0	1	0	0	0	0	0	2
Polypedilum	21	1	6	9	22	9	9	9	20	8	72	1	187
Procladius	16	42	4	18	14	13	8	3	34	4	50	50	256
Psectrocladius (Psect.)	4	1	15	11	61	9	2	5	0	0	1	1	110
Psectrocladius (Allops.)	0	0	0	0	0	0	0	0	0	1	0	0	1
Pseudochironomus	9	0	9	6	7	5	7	0	0	1	0	0	44
Pseudorthocladius	0	0	0	0	0	1	0	0	0	0	0	0	1
Rheotanytarsus	0	0	0	0	0	0	1	0	0	0	0	0	1
Saetheria	0	2	0	0	0	0	0	0	0	0	0	0	2
Stempellinella	23	1	17	3	2	4	0	0	0	0	0	0	50
Stenochironomus	1	0	0	2	0	1	0	0	0	0	0	0	4
Stictochironomus	0	1	1	4	0	0	0	0	0	0	0	0	6
Synorthocladius	0	0	0	59	0	0	0	0	0	0	0	0	59
Tanypus	0	0	0	0	0	0	0	0	2	0	4	3	9
Tanytarsus	16	40	55	35	69	15	22	15	68	23	19	2	379
Thienemanniella	9	8	4	0	0	6	0	2	0	0	0	0	29
Thienemannimyia	8	1	1	1	0	2	0	0	0	0	0	0	13
Xenochironomus	0	0	0	2	0	0	0	1	0	0	0	0	3
Zavreliella	0	0	1	0	0	6	16	0	0	0	0	0	23
Unknown Chironomini	1	1	32	8	21	3	27	0	3	1	0	1	98
Unknown Tanytarsini	0	0	0	2	0	0	0	12	0	0	0	0	14
Unknown (not sure of tribe)	1	0	0	0	0	0	0	0	0	0	0	0	1
Unknown (Chironomid?)	0	0	0	1	0	0	0	0	0	0	0	0	1
Total Taxa	**25**	**25**	**23**	**35**	**22**	**30**	**26**	**23**	**18**	**22**	**16**	**19**	**59**
Total Specimens	**188**	**163**	**293**	**298**	**418**	**236**	**231**	**241**	**309**	**162**	**421**	**362**	**3322**

A comprehensive summary of the genera and documented emerging from all 12 Tier 1 Sentinel Lakes in 2015 is provided in Table 4. The genera are listed alphabetically. Sixty-three genera were detected in 2015.

Table 4. Chironomidae species detected in individual Tier 1 Sentinel Lakes during 2015.

Lake	Trout	Ten Mile	Cedar	Bear Head	Carlos	South Twin	Elk	Pearl	Saint James	Portage	Madison	Shaokotan	Total
Total Phosphorus (ug/L)	5.6	12	13	14.5	17	18	21	52	54	60	81	132	Collected in
Mean Lake Depth (feet)	35	52	37	12	45.7	12	21.8	9.8	4	6.8	11	7.9	2015
Genus													
Ablabesmyia	14	9	8	22	42	125	22	18	35	39	5	283	622
Ablabesmyia annulata	0	0	0	0	1	0	0	0	0	0	0	0	1
Ablabesmyia peleensis	0	0	1	0	0	1	0	0	0	0	0	0	2
Acricotopus	0	0	0	0	0	0	0	0	0	0	0	0	0
Arctopelopia	0	0	0	0	0	0	0	0	0	0	0	0	0
Brillia	22	0	0	11	0	0	0	0	0	0	0	0	33
Chaetocladius	4	0	0	0	0	0	0	0	0	0	0	0	4
Chironomus	6	2	26	9	29	2	158	292	262	91	29	249	1155
Cladopelma	0	0	0	11	0	0	4	0	247	1	15	0	278
Cladotanytarsus	5	17	3	8	2	8	12	12	3	49	50	75	244
Clinotanypus	0	0	0	1	1	1	2	1	0	0	1	0	7
Coelotanypus	0	0	0	0	0	0	0	0	0	0	0	1	1
Conchapelopia	0	0	0	0	1	0	3	0	0	1	0	0	5
Corynoneura	99	176	4	32	45	182	3	32	1	6	0	129	709
Cricotopus	22	25	4	13	78	28	5	35	47	5	0	37	299
Cryptochironomus	3	0	1	11	6	0	0	0	1	1	27	2	52
Cryptotendipes	0	0	0	2	1	0	0	1	17	1	14	1	37
Demeijeria	0	0	0	0	0	0	0	1	0	0	11	0	12
Dicrotendipes	5	0	34	31	89	5	18	115	103	7	75	14	496
Einfeldia	0	0	38	0	1	9	20	22	1	0	0	0	91
Endochironomus	1	0	1	3	0	0	15	4	10	2	23	24	83
Eukiefferiella	0	2	0	0	0	0	0	0	0	0	0	0	2
Glyptotendipes	13	0	2	15	0	52	43	36	25	1	61	0	248
Goeldichironomus	0	0	0	0	0	0	0	0	0	0	1	0	1
Guttipelopia	0	0	0	0	0	3	3	1	4	14	0	0	25
Harnischia	0	0	0	4	0	0	1	5	0	0	4	0	14
Helopelopia	0	0	0	0	0	0	0	0	0	0	0	0	0
Heterotrissocladius	66	0	0	0	0	0	0	0	0	0	0	0	66
Hydrobaenus	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydrosmittia	0	0	0	0	1	0	0	0	0	0	0	0	1
Labrundinia	6	0	2	8	1	40	5	10	1	31	0	22	126
Lapposmittia	23	0	0	0	0	0	0	0	0	0	0	0	23
Larsia	0	0	1	0	0	0	1	0	0	1	0	0	3
Lauterborniella	0	0	0	0	0	7	8	0	0	6	0	0	21
Limnophyes	0	0	0	0	1	0	0	0	0	0	0	1	2
Micropsectra	89	269	0	0	0	3	4	0	0	3	0	31	399
Microtendipes	0	0	1	43	174	0	4	1	0	78	0	0	301
Nanocladius	4	71	2	0	6	121	1	13	3	12	2	118	353
Orthocladius	32	0	0	0	1	0	0	0	0	0	0	0	33
Parachironomus	0	10	1	0	1	10	1	40	119	4	13	31	230
Paracladopelma	0	0	0	0	0	0	0	0	0	0	0	0	0
Parakiefferiella	57	25	3	29	2	20	1	1	0	64	0	0	202
Paramerina	4	0	0	0	0	0	0	0	0	0	0	0	4
Parametriocnemus	0	0	0	0	0	0	0	0	0	0	0	0	0
Paratanytarsus	32	0	82	43	16	21	2	11	9	2	3	12	233
Paratendipes (?)	19	6	13	28	14	0	8	2	0	1	0	0	91
Phaenopsectra	27	0	0	0	1	0	0	0	0	0	0	0	28
Polypedilum	16	0	2	7	5	8	12	21	1	24	16	3	115
Polypedilum simulans	0	0	0	0	0	1	0	0	14	2	3	0	20
Potthastia (?)	3	0	0	0	0	0	0	0	0	0	0	0	3
Procladius	23	3	54	24	57	7	9	44	0	56	224	9	510
Psectrocladius (Psect.)	6	33	9	18	55	31	11	3	8	35	1	112	322
Psectrocladius (Allops.)	0	0	0	0	0	0	0	0	0	0	0	13	13
Pseudochironomus	4	0	2	30	20	8	0	3	0	5	1	0	73
Pseudorthocladius	0	0	0	0	0	0	0	0	0	0	0	0	0
Pseudosmittia	0	0	0	0	0	0	0	0	1	0	0	0	1
Rheotanytarsus	0	0	0	0	0	0	0	0	0	0	0	0	0
Saetheria	0	1	0	0	0	0	0	0	0	0	0	0	1
Stempellina	35	0	0	0	0	0	0	0	0	0	0	0	35
Stempellinella	29	7	2	0	0	8	0	0	0	0	0	0	46
Stenochironomus	0	0	2	0	0	0	0	0	0	0	0	0	2
Stictochironomus	0	0	42	0	4	2	0	0	0	0	0	0	48
Synorthocladius	22	0	0	64	0	0	0	0	0	1	0	0	87
Tanypus	0	0	0	0	5	0	0	0	2	1	61	2	71
Tanytarsus	23	11	20	41	7	28	63	22	28	53	19	8	323
Thienemanniella	0	13	0	0	8	1	1	0	0	0	0	0	23
Thienemannimyia	8	0	0	15	0	0	0	0	0	0	0	0	23
Xenochironomus	0	0	0	0	0	0	0	2	0	0	0	0	2
Zalutschia	72	89	0	0	0	19	6	0	0	0	0	0	186
Zavreliella	0	0	2	0	1	0	0	0	0	1	0	0	4
Zavrelimyia	0	0	0	3	0	0	0	0	0	0	0	0	3
Unknown Chironomini	2	0	0	0	1	1	0	0	0	6	2	0	12
Unknown Chironomini Type 2	2	0	0	0	0	0	0	0	0	0	0	0	2
Unknown Tanytarsini	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown (not sure of tribe)	0	0	0	0	0	0	0	0	0	0	0	0	0
Unknown (Chironomid?)	0	0	0	0	0	0	0	0	0	0	0	0	0
Total Taxa	**35**	**18**	**28**	**27**	**33**	**29**	**30**	**27**	**23**	**33**	**24**	**22**	**65**
Total Specimens	**798**	**769**	**362**	**526**	**677**	**752**	**446**	**748**	**942**	**604**	**661**	**1177**	**8462**

Similarity of emergence in 2014 versus 2015 was calculated using both Jaccard's Coefficient, which uses presence/absence data, and Whittaker's Percentage Similarity Coefficient. Based on presence absence data, emergence across all lakes for the two years was 68.4% similar, which is relatively high compared to some of the year-to-year similarities we have obtained for emergence of Chironomidae in smaller streams in Minnesota. Thirteen taxa that were only collected in either 2014 or 2015 were rare (defined as represented by 5 or fewer specimens) and may have been emerging during the other year but undetected because of their rarity. Rare taxa have traditionally been either deleted from calculations or it has been assumed that all rare taxa were present in both years but not detected in both years. When deleted from the similarity calculation, the similarity across increases to 72.3% and when added into the yearly totals the similarity value increases to 86.3%. Five of the remaining taxa were abundant in the second year in the more oligotrophic to slightly mesotrophic lakes primarily in samples collected soon after ice-out in April. Most of these lakes were not sampled as early in the season in 2014 because of the random sampling approach used in the first year of the project.

Whittaker's Percentage Similarity for emergence in 2014 versus 2015 was also relatively high at 64.3%. This measure of similarity is not strongly influenced by rare species, so calculations dropping or adding the rare species only detected during one of the years does not change the value appreciably.

Comparison To Lake Trophic State Model Developed By Sæther (1979): The lake trophic state model developed by Sæther (1979) was based on prior studies in Canada and elsewhere across North America and Scandinavia, and consisted of historical data from literature, which were combined with more recently generated data from several studies completed in Lake Winnipeg and elsewhere across Canada, some of which were not published at the time the model was published. The 1979 model was calibrated by depth (one calibration for chironomids occurring in profundal zones and a second calibration for taxa occurring in littoral and sub-littoral zones) and was based primarily on collections of larvae using dredge or grab samplers. Our method collects SFPE of adults that have emerged from all depth zones, so some caution must be taken when attempting to reconcile our results with the model calibrations. However, the publication also included a text key that can be used to test chironomid community patterns for lake trophic state concordance. This model has not be tested for lakes at lower latitudes in North America, such as the 12 Tier 1 Sentinel Lakes in this study, so our results are the first attempt to validate the applicability of the model to another geographic region.

The model uses ratios of total phosphorus/mean lake depth (TP/MLD) to classify oligohumic lakes into Oligotrophic, Mesotrophic and Eutrophic categories, and also has some reference to mesohumic and polyhumic situations. The three main trophic categories for oligohumic lakes are further sub-divided and referenced by Greek symbols of increasing trophic state as follows:

Oligotrophic: α, β, γ, δ, ε, ζ
Mesotrophic: η, Θ,ι
Eutrophic: κ, λ, μ, ν, ξ, ο

Table 5 shows the twelve Tier 1 Sentinel Lakes used in this study, arranged from top to bottom in the left column according to the ratio of TP/MLD (provided in column 2). Column 3 lists the MN DNR lake trophic state classification based on individual lake reports. Column 4 provides the lake trophic state(s) best predicted by the text key in Sæther (1979). The designation in Column 4 represents the first attempt to use the model for predicting lake trophic states for lakes in Minnesota, and may be subject to slight modification into the future. We presently are receiving feedback from peers regarding our justifications for this classification scheme, and will submit a manuscript for publication in a peer-reviewed journal after evaluating the feedback that we receive of our draft manuscript.

Table 5. Preliminary classification of 12 Tier 1 Sentinel Lakes based on data generated using SFPE collections in 2015.

SLICE Lake	TP/MLD	MN DNR CLASSIFICATION	SÆTHER (1979)
Trout	0.16	Oligotrophic	**δ oligothropic**
Ten Mile	0.23	Mesotrophic	**ε- ζ oligotrophic**
Cedar	0.35	Mesotrophic	**η-Θ mesotrophic**
Carlos	0.37	Mesotrophic	**ζ oligotrophic**
Elk	0.96	Mesotrophic	**Θ mesotrophic**
Bear Head	1.21	Mesotrophic	**η mesotrophic**
South Twin	1.5	Mesotrophic	**η mesotrophic**
Pearl	5.31	Eutrophic	**μ eutrophic**
Madison	7.36	Eutrophic	**ξ eutrophic**
Portage	8.82	Eutrophic	**ν eutrophic**
Saint James	13.5	Eutrophic	**ν eutrophic**
Shaokotan	16.71	Hypereutrophic	**ξ-o eutrophic**

DISCUSSION

Collections of SFPE from 12 Tier 1 Sentinel Lakes revealed a highly diverse assemblage of Chironomidae emerging from the lakes. This type of sampling approach appears to be highly effective in documenting biodiversity by lake. Identifications to genus-level were easy to achieve, except for a small number of taxa that should be reared to the adult stage in order to determine the genus. Several taxa collected during this study can be differentiated to species, but do not match species-level descriptions published in the primary literature for the respective genera. More targeted collections of mature larvae or whole pupae of these species should be made, and specimens should be reared to the adult. After identification of the adult, the respective pupal exuviae should be formally described and published in the primary taxonomic literature. In order to facilitate use of the SFPE method on a routine basis for monitoring into the future, a key with illustrations and digital photographs should be assembled for all taxa collected in this project. The illustrations and photographs should be made of voucher specimens, and distributional and seasonal data regarding where and when the specimens were collected should be integrated into a discussion section of the key for each of the 117 species. A manuscript is also currently in preparation that focuses on the biological diversity of Chironomidae in the 12 Tier 1 Sentinel Lakes. The format is patterned after the format used by Egan and Ferrington (2015), and the manuscript will be submitted to the same journal for publication.

Both random sampling and regular sampling at defined intervals of approximately one month generated very similar, but not identical, patterns of biodiversity in 2014 and 2015, respectively. However, repeating sampling at fixed intervals (2015) yielded slightly more taxa, and this approach was better suited for documenting species that emerge very soon after ice-out. Consequently, more targeted sampling, focused on early season periods (April to mid-May in southern lakes and late April through May in more northerly lakes) is necessary to better resolve the biodiversity of Chironomidae. One month intervals between June and September appear to be sufficient for resolving summer-emerging species. This outcome of the research project is very significant for planning large-scale monitoring projects focusing on chironomids in lakes of differing trophic states and a manuscript will be prepared and submitted for publication based on the outcome of the two contrasting approaches used for the two years of this study.

With results from this project we were able to provide a test of the Lake Trophic State model published by Sæther (1979). Our initial attempt to classify the 12 lakes appears to be broadly consistent with MN DNR classifications, but the model provides more sensitivity than the classification scheme presently in use by MN DNR. In the future, additional Sentinel Lakes should be sampled in a similar manner and attempts should be made to classify them using the 15-category scheme of Sæther (1979). A manuscript related to this outcome is currently being circulated to colleagues for friendly review and comment.

To the extent possible, collections of SFPE and species-level identifications should be the goal for assessing climate change influences on Chironomidae emerging from lakes with different trophic states. Sentinel Lake-specific identification keys will help achieve this goal, along with focused efforts to collect, rear, and describe species that do not currently conform to published descriptions. Both reference and voucher collections should be designated, and some should be permanently deposited into Entomological Museums. Teaching collections would also be a valuable resource, as would periodic workshops focusing on species-level identifications.

SECTION II

Products and Deliverables

Section II.1 Data Base: A database of material from this research project has been established and will continue to be maintained at the University of Minnesota. The database will be at the specimen level and our longer-term intention is to include associated images (photographs and line drawings of identifying characters). Individual specimens have been selected as vouchers. For these vouchers, identification labels and barcode labels will be added to the slide-mounted specimens. Each adult specimen will then be uniquely identified by its bar code, and all information associated with that specimen will be retrievable by code. For alcohol preserved material, one vial of multiple individuals will be functionally equivalent to a specimen in a series. To minimize confusion managing the database, a trained Graduate Research Assistant or citizen volunteer will always do all data entry.

Section II.2 Continued Development and Enhancement of Web Site: During the first year of the project we initiated development of a project World Wide Web site, which is dedicated for this project. We constructed links for each Tier 1 lake, and our data bases that are linked will enable users to generate comprehensive taxon lists, by locality, and by subfamily, tribe, genus and species for Chironomidae. Once image files are established, the database of species and images will form the basis of an interactive identification system. As faunas of Sentinel Lakes and other lakes in Minnesota become better known, we will develop taxonomic keys with links to images and specimen/collection data for each taxon. Our web site and associated database are potentially of use to systematists, ecologists, conservationists, resource managers, and policy makers that need to access information about known and newly discovered aquatic biota of the Sentinel Lakes. Web pages will continue to reside on a server maintained by the CFANS at the University of Minnesota. The home page has links to the home page of the Chironomidae Research Group, the UMSP, and the home page of the DNR Sentinel Lakes Program. Other links to various aquatic and environmental forums on the Internet are being considered and evaluated for development. The home page will continue to be updated into the future.

Section II.3 Deposition of Specimens: Voucher collections of each species encountered in each Sentinel Lake have been assembled and prepared for long-term deposition and curation in the Insect Museum of the Department of Entomology at the University of Minnesota. Undescribed species encountered in this project will be prepared for description, and holotype material will be incorporated into the Insect

Museum of the Department of Entomology at the University of Minnesota. When possible, paratype or additional voucher material of Chironomidae will be sent to the United States National Museum and to the University of Bergen for incorporation into their comprehensive Chironomidae Collection. Material will also be made available to interested colleagues as formal loans.

Section II.4 Value Added Components of This Study: Several value-added components developed from this research. The first is that collections of SFPE often inadvertently yield specimens of other aquatic insects either as adults or as specimens in the process of emergence. These other insects can include Odonata, Trichoptera, Ephemeroptera, Plecoptera, Megaloptera, other Diptera families, and aquatic Heteroptera in addition to terrestrial insects of many orders. Selected voucher specimens of these orders collected along with SFPE are sorted to family, preserved, curated, and incorporated into research collections. Specimens incorporated into the research collection will be made available upon request to other entomologists conducting taxonomic, systematic or faunal studies.

The second value added component is related to the teaching mission of my unit, the Department of Entomology, at the University of Minnesota. Some specimens of extremely abundant taxa have been incorporated into teaching collections and will be used during laboratory sections for teaching insect identification skills. Typically, student teaching material can be broken or otherwise suffer wear and tear when used by many inexperienced students trying to learn how to identify insects during lab periods. Consequently, the teaching collections require periodic replenishment. Some of the most common taxa encountered in this project have been removed from samples that were sub-sampled, and the specimens were added to the teaching collection.

The third value added component of this project was that I was able to incorporate an international student from Brazil (Ms. Hanna Lisa Leffever) into our field and lab activities from June through August 2015. This student selected our lab because of the project and wanted to learn our field techniques and laboratory protocols as part of a programmatic internship. She was funded by the Brazilian Exchange Program and joined our research effort with no added expense to the grant. In addition, the student had a great time working with us and was extremely satisfied with her experience. Consequently, we now have a strong connection to this Brazilian program and expect that we will receive requests from students wanting to join our lab for internship experience in the future. Her home institution is Federal Rural University of Rio de Janeiro.

Section II.5 Training: The final value added component has been training of an undergraduate student (Alec Brown) and a Master's-level graduate student (Kara Fitzpatrick) in field methods, laboratory processing, identification, quantification and curation of Chironomidae. Results from this project will form at least part of the conceptual basis for developing and testing hypotheses for the students' research topics. In addition, two graduate students from my lab (Ms. Petra Kranzfelder and Mr. Alexander Egan) assisted with some field and lab work, including sample collection, processing, identification, development of web site and coordination of supplies purchases and scheduling of fieldwork.

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Outcome 6

The following two reports were completed for Outcome 6. The first, summary of algal and zooplankton interactions in 13 of the Sentinel Lakes was completed by Steve Heiskary (MPCA), and Jodie Hirsch and Heidi Rantala (MNDNR). The executive summary follows and the report can be found in its entirety at www.dnr.state.mn....

Patterns in Phytoplankton and Zooplankton in Minnesota Lakes
Executive Summary

The purpose of this report was to examine patterns in phytoplankton and zooplankton based on a subset of the Sentinel Lakes and to serve as a baseline for future monitoring and analysis. While zooplankton sampling has been a routine part of Sentinel Lake sample collection and was addressed in individual lake reports, phytoplankton has been sampled sporadically. This analysis provides basic descriptions of phytoplankton and zooplankton composition, seasonal cycling, and interactions in 13 Sentinel Lakes. Observations included herein can be used in future assessments as a basis for comparison and evaluating change over time. Though this analysis was limited to 13 lakes, their geographic, morphometric, and trophic ranges allow for some generalizations that may be applicable to Minnesota lakes in general. A summary of some important findings follows.

Numerous factors influence the cycling of algal forms in Minnesota lakes. Examples of these factors include: climatic drivers - sunlight, temperature, and wind; nutrients - phosphorus and nitrogen; minerals - carbonate and silica; and biological: zooplankton. These factors vary in their significance in various lakes and interactions among variables may be complex. While the literature offers numerous examples of seasonal cycling of algal forms, this analysis allows for descriptions based on Minnesota lake data. Analyzing the Sentinel Lakes across a trophic gradient provided a reasonable approach for characterizing phytoplankton and zooplankton composition and seasonal succession for Minnesota lakes. Moving from oligotrophic to hypereutrophic we observed the following:

- In oligotrophic and mesotrophic lakes, diatoms and chrysophytes are the dominant forms. While diatoms are prominent across the entire trophic gradient, chrysophytes are uncommon in the eutrophic lakes. This is likely the result of competition from algal forms that prosper in more nutrient-enriched water and the warmer temperatures of the lakes of central and southern Minnesota. Algal biomass is often highest at spring turnover in the oligotrophic and mesotrophic lakes and usually declines thereafter, until fall turnover. Blue-greens, while present, are never dominant and biomass, based on Chl-*a*, is far below nuisance bloom levels. Nutrient limitation, silica depletion, zooplankton grazing, and water temperature are important drivers in the observed seasonal succession.

- In eutrophic and hypereutrophic lakes, spring turnover is marked by blooms of diatoms, cryptophytes, and green algae. By June, these forms decline in prominence and are replaced by blue-greens, which remain prominent until October when the water cools and Cryptophytes become prominent again. In general, there is a steady increase in algal biomass from May through September/October and much of that is in the form of blue-greens. Since nutrients are abundant in these lakes, warm temperatures and zooplankton grazing of other algal forms (reduce competition) are among the two most important drivers allowing for blue-green dominance.

This study also provided an opportunity to identify some phytoplankton and zooplankton indicators that merit tracking in future monitoring efforts.

a. Phytoplankton indicators are summarized below:

- *Cylindrospermopsis* is a known toxin producer and was relatively uncommon in the 2013 and 2014 collections in this study, with the exception of Madison and South Center lakes. It was previously believed to be tropical, preferring warm temperatures. Its presence and relative abundance in other Sentinel Lakes should be tracked in future collections. A significant expansion in its presence and abundance could be an indication of significant environmental changes in the affected lakes. Northward expansion would be of particular concern.
- Blue-green algae were present in all Sentinel Lakes and are in all Minnesota lakes. Blue-green forms, shift in dominant genera, and potential increases in toxin-producers could be useful indicators of change. In the oligotrophic and mesotrophic lakes a variety of blue-green genera are found; whereas in the more eutrophic lakes potential toxin-producing *Anabaena, Microcystis*, and *Aphanizomenon* were dominant.
- The blue-green alga *Gloeotrichia* may be worth watching over time in these lakes. It is not considered an indicator of poor water quality and is more common in lakes of high clarity. However, literature reports suggest it draws most of its nutrients from the sediment of the lake and that it is not a good food source for zooplankton. Its increase in the future could be indicative of changes in a lake system.
- Increased water temperature favors blue-green algae over other forms. In particular, as summer temperatures increase above 20 °C and remain at 25 °C or more for extended periods, blue-greens prosper. As lakes warm, chrysophytes may be among the first forms to decrease, given their preference for cool temperatures. Chrysophytes are important in mesotrophic lakes and a good food source; should they be reduced via increased temperature or nutrient enrichment, blue-greens may fill the niche they occupy.

b. Zooplankton indicators

- *Daphnia longiremis*- This deep-water daphnid may be an indicator species, as it is found in the hypolimnion of deep, well-oxygenated lakes. This is reinforced by the findings from zooplankton analysis of 150 random lakes in the 2012 National Lakes Assessment (Hirsch 2014) where this species was found only in the more north central MN lakes.
- *Leptodiaptomus minutus*- a cool-water Diaptomidae which is restricted to deep lakes at the southern end of its range. In Minnesota, it is only found in oligotrophic and mesotrophic lakes, and its loss could be an indication of eutrophication and/or climate-induced warming.
- *Leptodiaptomus siciloides* and *Aglaodiaptomus clavipes*- both these calanoid species tend to favor or adapt better to more eutrophic conditions.

This study also provided some initial insights on the impact of invasive species on phytoplankton, zooplankton, and lake water quality.

- The impact of zebra mussels in Lake Carlos is already apparent based on trends in trophic status; whereby Secchi depth is increasing, and algal biomass appears to be decreasing. There are not adequate data to determine if there are any significant shifts in algal composition. Based on 2013 and 2014 data there was not a significant shift to blue-green forms, which sometimes occurs with increased zebra mussel infestation. However, the low TP in Lake Carlos may minimize the likelihood of this occurring.
- Zebra mussels appear to be impacting the zooplankton community in Lake Carlos, as densities of grazers have declined in post-zebra mussel infestation years, suggesting competition effects, although more post-infestation data are necessary to test this statistically.
- Spiny waterfleas in Trout Lake appear to be impacting small cladocerans by direct predation and possibly cyclopoid copepods by indirect food competition interactions, although more data are necessary to confirm this as well.

The full report can be found on the MNDNR website:
http://www.dnr.state.mn.us/publications/fisheries/special_reports.html

A lake assessment report was completed for Greenwood Lake by Steve Heiskary (MPCA) and Steve Persons and Jodie Hirsch (MNDNR). Similar to all other lake assessment reports for the Sentinel Lakes, the final report can be found on the Minnesota Pollution Control website (https://www.pca.state.mn.us/water/sentinel-lakes). The report specific to Greenwood Lake can be accessed via the PCA website as well (https://www.pca.state.mn.us/sites/default/files/wq-2slice16-0077.pdf).

ACTIVITY 2: Build or adapt biophysical lake ecosystem models for 6 Tier 1 lakes capable of forecasting future lake conditions based on scenarios that cause change but vary across lakes.

Description:
To understand complex systems like natural lakes, we need both high quality and comprehensive monitoring data, but also the capacity to synthesize those data in mechanistic system models. For example, we have a generalized idea what zebra mussels do to lake food webs, and those effects likely vary across lakes. Consequently, we can develop sound mechanistic models to explain these changes, but these models need to be fit and calibrated to our specific lakes. To further elaborate on this example,

biologists have learned a lot about how zebra mussels have changed Oneida Lake in New York state, but it would be very wrong to use the Oneida data and model to predict changes in Lake Carlos. This is because the two lakes are so different, Oneida is very fertile and shallow while Carlos is deep and of moderate fertility. In Activity 2, we propose to continue building mechanistic lake models for distinct classes of Sentinel Lakes and to refine the three existing models developed for deep, coldwater lakes. The planned 6 lake models will continually be used and recalibrated with new monitoring data to forecast lake conditions given certain scenarios that will cause some type of change.

Summary Budget Information for Activity 2: **ENRTF Budget: $ 207,544**
Amount Spent: $ 207,544
Balance: $ 0

Activity Completion Date: June 30, 2016

Outcome	Completion Date	Budget
1. *Build biophysical lake system models for 6 Tier 1 lakes & establish change scenarios*	*June 30, 2015*	$88,948
2. *Calibrate and validate models*	*December 31, 2015*	$59,298
3. *Develop and run forecast scenarios for evaluating various lake management tools & BMPs*	*June 30, 2016*	$59,298

Activity Status as of 10/30/2013:
The USGS assisted with the set-up and installation of water level transducers at three of the Sentinel Lakes (Carlos, Pearl, and Elk). The three existing Super Sentinel lake models were upgraded to Release 3.7 of the CE-QUAL W2 software. All model calibrations and validation runs were repeated with the new version of the software and assessed for continuity with previous results. Model performance was similar to simulations performed with version 3.6 and was well within acceptable limits.

Using the output from the new version of the CE-QUAL W2 software, model runs were evaluated for two historic climate years for each of the Phase 1 Super Sentinel Lakes. The predicted oxy-thermal fish habitat in each lake was evaluated for optimal, good, and lethal thermal habitat as defined by thermal ranges and the dissolved oxygen minimum concentration for cold water fish communities. Habitat volumes for each of the simulations were summarized in graphs of percent lake volume in each habitat class over the ice-free season in each lake.

Activity Status as of 4/15/2014:
The USGS recovered data from the three water level transducers at Lake Carlos and from the weather station on the southeast shore of Lake Carlos. Water-level data records were processed and review for quality assurance and finalized in the USGS online database as of March 31, 2014. Weather station data were downloaded and archived for use with future lake model runs. Lake Carlos inflow and outflow discharge estimates were calculated from water level data using existing rating curves. Download and review of available water quality data from the three new super-sentinel lakes was started in March.

A USGS Scientific Investigation Report detailing the Phase 1 model calibration and validation received Bureau approval. The report is undergoing final formatting for publication during the next reporting period.

Activity Status as of 10/30/2014:
The USGS conducted water level transducer installation training and Flowtracker ® stream flow measurement training for MN DNR Fisheries staff at Pearl Lake in May 2014. USGS installed and

maintained three water level transducers at Lake Carlos and a weather station on the lake's southeast shore. Water-level and weather data were downloaded and reviewed for quality assurance in preparation for working the station records for publication and for use with future lake model runs. USGS downloaded 2013 water quality data for the three new super-sentinel lakes. USGS staff provided quality assurance review of MN DNR inflow and outflow discharge measurement. USGS staff began to compile watershed data for model development support.

Results from lake models evaluating two historic climate years for each of the Phase 1 Super Sentinel Lakes were presented at the 2014 Minnesota Water Resources Conference. The presentation focused on the predicted oxy-thermal fish habitat in each lake as a function of multiple cold-water fisheries habitat classes.

Activity Status as of 4/15/2015: The USGS provided the following data review and analysis during the reporting period –

1. **2014 Water level data from six Phase 1 and Phase 2 Sentinel Lakes:**
   Available continuous water level data for all six lakes have been entered into the USGS records processing database and processors for applying stage corrections have been added to the individual records. Gage height corrections from MNDNR field data have been used to process the data and to produce a provisional record for publication.

2. **2014 Stage – Discharge Relationships from instantaneous discharge measurements:**
   USGS has reviewed the discharge rating developed by MNDNR Eco-waters for the outlets of Pearl, Madison, and Shaokotan Lakes. USGS has compared instantaneous discharge measurements made by MN DNR Fisheries staff to the MNDNR Eco-waters ratings. Data were analyzed to determine how best to proceed with development of continuous discharge estimates. The following alternatives were evaluated for each site:
   a) use the original rating as developed by MNDNR Eco-waters
   b) improve the rating using the MNDNR Fisheries discharge measurements; or
   c) develop statistically significant regression equations between stage and discharge to provide a better fit to the data.

As of 30 April 2015, ratings and/or regression equations have been developed for every site.

Activity Status as of 10/30/2015:

1. **2014 Water level data from Phase 1 and Phase 2 Sentinel Lakes:**

Progress through 31 October 2015: Continuous water level records have been checked and reviewed in preparation for use in model development and calibration.

2. **2014 Stage – Discharge Relationships from instantaneous discharge measurements:**

Progress through 31 October 2015: MN DNR Fisheries stage data have been used by USGS to validate or develop and evaluate discharge rating curves developed for the outlets of Pearl, Madison, and Shaokotan Lakes. Water levels and discharge estimates have been used to develop water balance models for the three lakes.

3. **Overall Project Progress:**

Progress through 31 October 2015: MN DNR completed data collection for Phase 2 in October of 2014. Data delivery to USGS was completed in August of 2015. Bathymetry and segmentation schema were developed for the new lake models. Discharge measurements and stream cross-section elevation data were used to establish initial stage-discharge relationships for inflow data. Inflow data, bathymetry and lake level records were used to develop water balance and heat budget base models. A preliminary model was completed for and is being tested for Pearl Lake.

Activity Status as of 4/15/2016:

1. **Water level data from Phase 1 and Phase 2 Sentinel Lakes:**
   Progress through 15 April 2016: Continuous water level records have been incorporated into CE-QUAL-W2 model calibration for Pearl Lake and Madison Lake. Water budget calibrations are complete.

2. **Stage – Discharge Relationships from instantaneous discharge measurements:**
   Progress through 15 April 2016: To date, ratings and/or regression equations have been developed for every site and used to develop a CE-QUAL-W2 model calibration for water balance of Pearl Lake and Madison Lake.

3. **Overall Project Progress:**
   Progress through 15 April 2016: Load estimates and water quality data were used to develop calibrated CE-QUAL-W2 models for Pearl and Madison Lakes. The calibrated models are ready for use to evaluate oxy-thermal habitat for target species. Calibration efforts included temperature profiles, dissolved oxygen profiles, nutrients, and algal dynamics.

Global climate change consensus model predictions for 2050 were used to select climate record analogs for Elk and Trout Lake. Meteorological records were obtained for these analog climate stations for the Elk Lake and Trout Lake model calibration years. Data were used to run future response scenarios for the two lakes. Elk Lake seasonal cisco oxy-thermal habitat and bluegreen algae growth were altered under predicted changes for 2050. Trout Lake response was more muted, and the lake appeared to be more resilient to the predicted changes.

Final Report Summary:
A draft U.S. Geological Survey (USGS) Scientific Investigations Report (SIR) for the Pearl and Madison Lake CE-QUAL-W2 models has been submitted to the LCCMR, and is currently in final report preparation with U.S. Geological Survey editing staff. The final online USGS SIR will replace the interim draft report, once available. The Lake Shaokotan CE-QUAL-W2 model and SIR has been delayed, mainly because of a low frequency of available discharge measurements, and low overall flow in/out of the lake that makes Lake Shaokotan difficult to hydrodynamically model. However, the U.S. Geological Survey team is continuing to work towards a final CE-QUAL-W2 modeling solution.

The title of the draft USGS SIR mentioned above is “Two-dimensional lake water-quality model simulations of algal community dynamics for two agricultural land-use dominated lakes in Minnesota”, by Erik A. Smith, Richard L. Kiesling, and Jeffrey R. Ziegeweid.

Abstract: Degradation of fish habitat occurs in many lakes due to summer blue-green algal blooms—a threat that may be exacerbated with land-use change and climate change. To better manage and mitigate against loss of fish habitat due to these changes, predictive models are needed. The U.S. Geological Survey (USGS), in cooperation with the Minnesota Department of Natural Resources, developed predictive water quality models for two agricultural land-use dominated lakes in Minnesota—Madison Lake and Pearl Lake, which are part of Minnesota’s Sentinel Lakes monitoring program. The interaction of watershed processes to these two lakes, via the delivery of nutrient loads, were simulated using CE-QUAL-W2, a carbon-based, laterally averaged, two-dimensional water quality model that predicts

distributions of temperature and oxygen from interactions between nutrient cycling, primary production, and trophic dynamics.

The CE-QUAL-W2 models successfully predicted water temperature and dissolved oxygen, on the basis of the two metrics of absolute mean error and root mean square error. Both temperature and dissolved oxygen are key metrics for proving these models are calibrated. These calibrated lake models also simulated water quality and algal community dynamics. The model simulations presented potential explanations for persistently high total phosphorus concentrations in Madison Lake, key differences in nutrient concentrations between these lakes, and summer blue-green algal bloom persistence. Fish habitat suitability simulations illustrated that in general both lakes contained a large proportion of good-growth habitat and sustained period of optimal growth habitat in the summer, without any periods of lethal oxythermal habitat. Sensitivity analyses were completed to understand lake response effects through the usage of controlled departures on certain calibrated model parameters and input nutrient loads. These sensitivity analyses also operated as land-use change scenarios because alterations in agricultural practices, for example, could potentially increase or decrease nutrient loads.

Climate change scenario modeling for Elk and Trout lakes

Executive summary for the climate change scenarios: Phase 2 of Sentinel Lakes monitoring and research includes applying lake models to predict ecosystem impacts of major environmental and ecological stressors (e.g., changing land use and climate change). Of the completed Tier 1 sentinel lakes developed by the U.S. Geological Survey, climate change scenarios were developed for Elk Lake and Trout Lake. Elk Lake and Trout Lake were determined to be the most susceptible to climate change of the completed Tier 1 sentinel lakes with published CE-QUAL-W2 ecosystem models. Elk Lake is close to the ecoregion boundary between Northern Lakes and Forests and North Central Hardwood Forests (Figure 1) and this boundary has been shown to shift in the past (Whitlock et al. 1993). Additionally, projected warming trends could cause the Elk Lake's thermocline to deepen, affect the lake's dissolved oxygen distribution, and the lake habitat volume suitable for coldwater fish such as Northern Cisco to survive. For Trout Lake, warming trends could also cause the thermocline to deepen, which would affect the lake habitat volume suitable for coldwater fish.

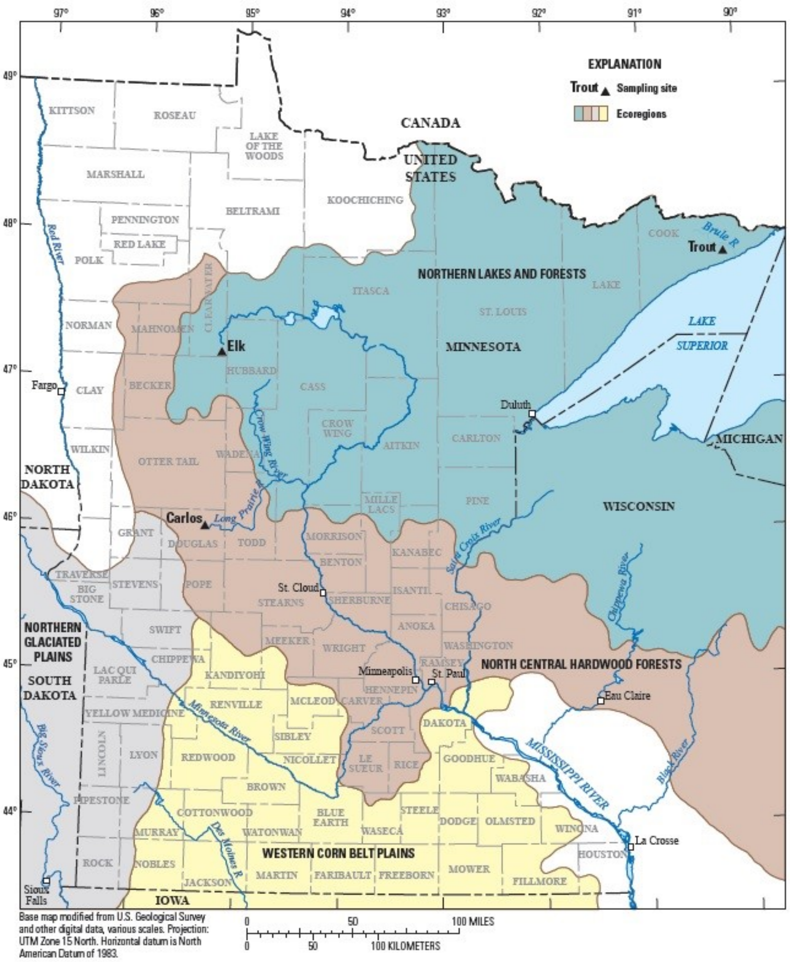


Figure 1: Major ecoregions of Minnesota, including locations of Elk and Trout Lakes.

Methods for climate change scenarios: A total of 12 different climate scenarios were developed for Elk Lake and three climate scenarios were developed for Trout Lake. All of the climate scenarios used the previously developed CE-QUAL-W2 models as a baseline condition (Smith et al. 2014). For Elk Lake, nine of the 12 climate scenarios were based upon the Special Report on Emissions Scenarios (SRES) used in two different Intergovernmental Panel on Climate Change (IPCC) reports (IPCC, 2001; IPCC, 2007). The SRES were used in both the IPCC Third Assessment Report (IPCC, 2001) and the IPCC Fourth Assessment Report (IPCC, 2007). These SRES consist of several greenhouse gas emissions scenarios that were included to make future climate change projections. Three of the 12 scenarios were based upon the change in guidance for IPCC Fifth Assessment Report (IPCC, 2014), which supersedes the SRES. The new method, known as Representative Concentration Pathways (RCPs), focuses on different increases in radiative forcing values by the year 2100 relative to pre-industrial values (IPCC, 2014). Three RCP scenarios were adopted for both Elk Lake and Trout Lake.

For the SRES, six families of individual scenarios exist, of which three were chosen for Elk Lake: A1B, A2, and B1. These selected families are defined as the following:

- A1B: aggressive economic and population growth, with a peak global population of 9 billion in 2050 with gradual declines afterwards; world economy and standard of living begins to converge, with new and efficient technologies spreading across the world (IPCC, 2001);
- A2: global population continues to grow, with a world made up of independent and self-reliant

nations without major international cohesion of technologies (IPCC, 2001);

- B1: similar to A1B, with the difference of the introduction of clean and resource efficient technologies and economic, social and environmental stability (IPCC, 2001).

Of the three different selected scenarios (A1B, A2, and B1), A2 leads to the largest increase in greenhouse gases followed by A1B and B1 having the lowest increase in greenhouse gases (IPCC, 2001; IPCC, 2007). In particular, both A1B and B1 show declines in CO_2 and N_2O emissions, both major greenhouse gases, by 2100 after peaking around 2050 whereas A2 continues to increase to 2100 and beyond (Figure 2).

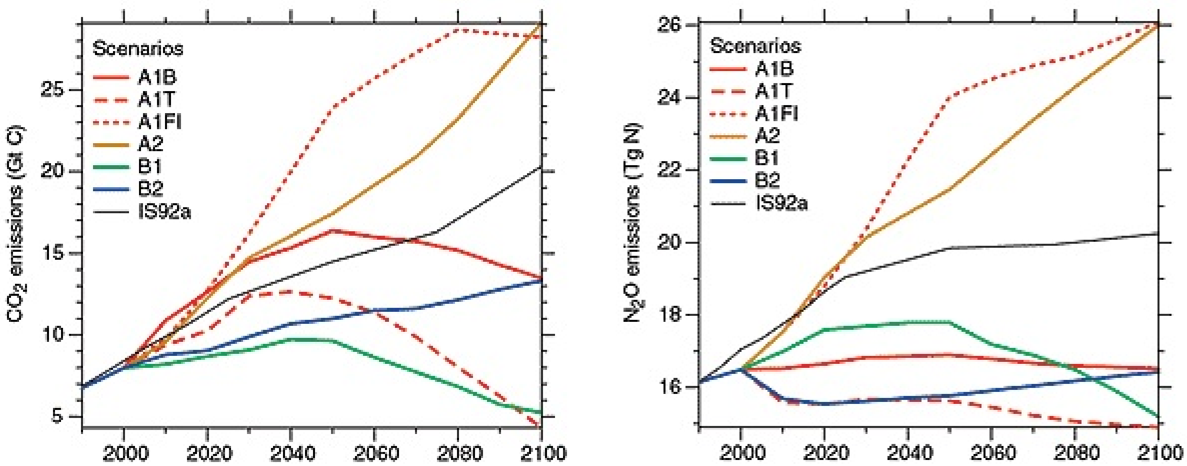


Figure 2: Anthropogenic emissions of CO_2 and N_2O for the six SRES scenarios, including A1B, A2, and B1 (figure copied from https://www.ipcc.ch/ipccreports/tar/wg1/figts-17.htm).

The following time periods were included for Elk Lake for the nine SRES climate scenarios: 2030, 2050, and 2070. With the three different scenarios (A1B, A2, and B1) combined with the three time periods, a total of nine scenarios were created. The meteorological data for all nine of the SRES came from Meteonorm, a comprehensive climatological database for every location on the globe (Remund et al. 2013). For each IPCC SRES scenario, the year and particular scenario (for example, A1B in 2030) were selected, and based upon the interpolated long-term monthly means of the climatological data, a stochastically generated dataset of hourly meteorological data was generated. This data included all of the hourly data necessary for running a CE-QUAL-W2 model, including: air temperature, dewpoint temperature, wind speed and direction, incoming solar radiation, and cloud cover.

For the final three scenarios, the RCPs (IPCC, 2014) were used in lieu of the SRES. As mentioned earlier, RCPs focus on different increases in radiative forcing values rather than emissions scenarios (IPCC, 2014). There are four different RCP families, known as RCP2.6, RCP4.5, RCP6, and RCP8.5. The number behind RCP is the value in watts per square meter (W/m^2), so RCP8.5 is a +8.5 W/m^2 increase in the radiative forcing values.

Since the Meteonorm software package has not been updated to include RCPs, the method known as "space-for-time" was used to generate the necessary meteorological data to run the CE-QUAL-W2 models (Blois et al. 2013) for the RCP scenarios. The "space-for-time" concept is borrowed from biodiversity modeling, which uses this method to infer past or future trajectories of ecological systems. Basically, the substitution takes contemporary spatial phenomena, such as plant succession, and infers that if another record (alternate) in the past (or future) has the same pattern, than the ecological patterns from the contemporary location can be transferred to the alternate location.

The way "space-for-time" is employed for generating the necessary climatological data works in a similar manner. Since it is often difficult to get the high-resolution (sub-daily) meteorological data, particularly wind speed and direction, future climate change projections from a location are used to locate the present-day analog. This alternate location that has the same basic climatological characteristics currently as the location of concern's projected future climatological characteristics is substituted into the calibrated CE-QUAL-W2 model.

In order to assist with determining the analogous geographical locations for both Elk and Trout Lakes, the National Climate Change Viewer (NCCV) was utilized to determine the optimal substitute locations (U.S. Geological Survey, 2016). On NCCV, any area within the conterminous United States can be selected by either individual HUC-12 watershed or county. For Elk Lake, Hubbard County was selected; for Trout Lake, Cook County was selected. Once the area was selected, the time period can be selected as either an annual mean or by month; for this exercise, the annual mean was selected. Two of the four different RCP families can be selected: RCP4.5 and RCP8.5; for the three RCP scenarios, only RCP8.5 was selected.

For NCCV, results from any of 30 different climate models can be viewed, or a synthesis mean model of all 30 climate models can be selected; for this modeling exercise, the mean model was selected. Once the combination of location, time period, and climate model was selected, the following parameters were summarized: maximum 2-meter air temperature, minimum 2-meter air temperature, average mean precipitation, average mean runoff, annual mean snow, annual mean soil storage, and annual evaporation deficit. Figure 3 shows a screenshot of the basic NCCV layout, with bottom panels that summarize the maximum temperature by month and the annual deviations from the present to the future time period.

Three different summary time periods were summarized on NCCV: 2025-2049, 2050-2074, and 2075-2099. For the final three climate scenarios, the following combinations were used: RCP8.5 for 2025-2049, RCP8.5 2050-2074, and RCP8.5 for 2075-2099. Based on the basic increases in greenhouse gas emissions for RCP8.5, this RCP scenario is most analogous to the A2 climate scenario of the SRES. As mentioned earlier, several climatological parameters were summarized on NCCV of which emphasis was placed on finding another location with similar mean values in the future. However, only the following parameters were considered: minimum and maximum air temperature, annual mean precipitation, and annual evaporation deficit.

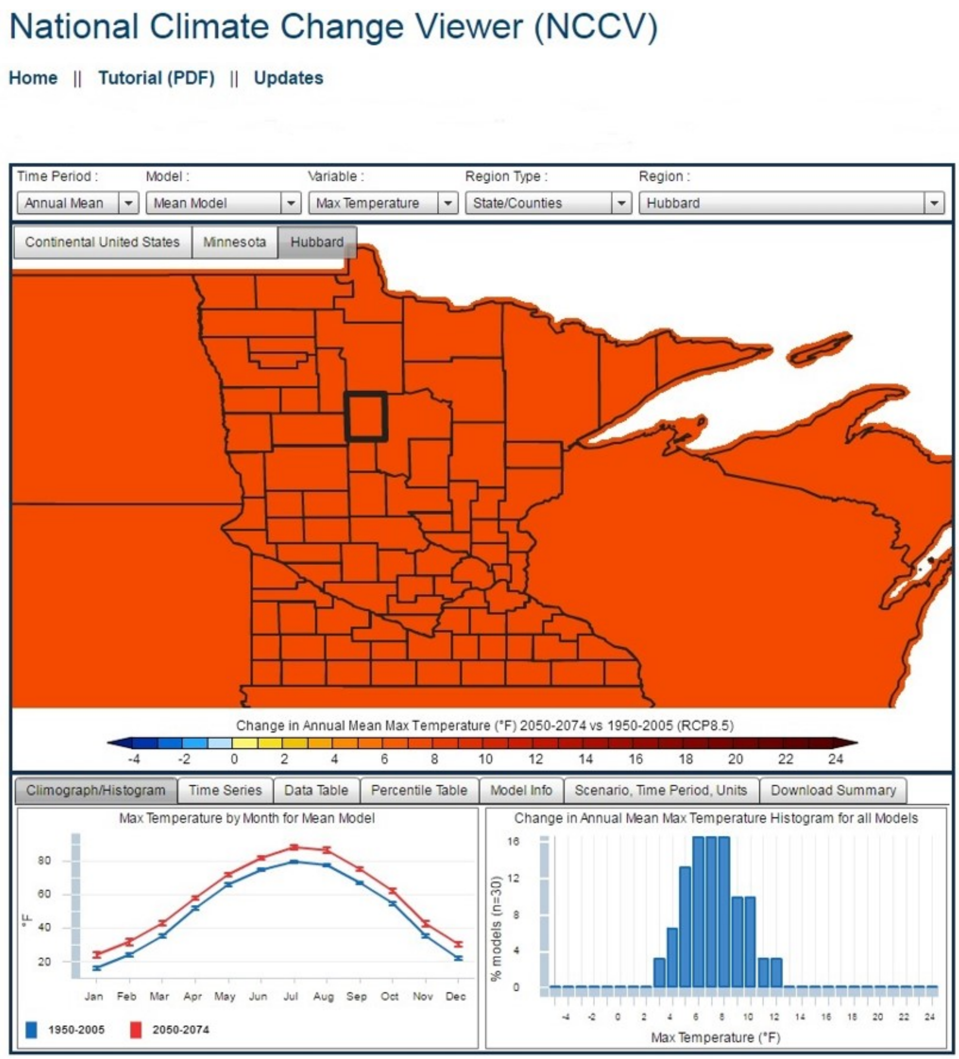


Figure 3: National Climate Change Viewer (NCCV) layout, with map view shown (with Hubbard County, Minnesota selected). The bottom left panel shows the maximum 2-meter air temperature (by month) for the mean model for two different time periods: 1950-2005, and the projected 2050-2074 time period. In the bottom right panel, the change in annual mean maximum temperature histogram is shown for the mean models.

Based on the above approach using NCCV, the following locations were determined to have the most analogous climatological annual means to Elk Lake for RCP8.5 (Figure 4):

- RCP8.5, 2025-2049: McLeod County, Minn. (Hutchinson Municipal Airport, Hutchinson, Minn.);
- RCP8.5, 2050-2074: Clay County, Iowa (Northwest Iowa Regional Airport, Spencer, IA);
- RCP8.5, 2075-2099: Union County, Iowa (Creston Municipal Airport, Creston, IA).

For each of these three scenarios, climate data was downloaded from the National Climatic Data Center for 2011, the calibrated model year for Elk Lake (Smith et al. 2014), from the nearest continuous climate station (NCDC, 2016). As with the nine SRES scenarios, the following data was required at hourly time steps: air temperature, dewpoint temperature, wind speed and direction, incoming solar radiation, and cloud cover.

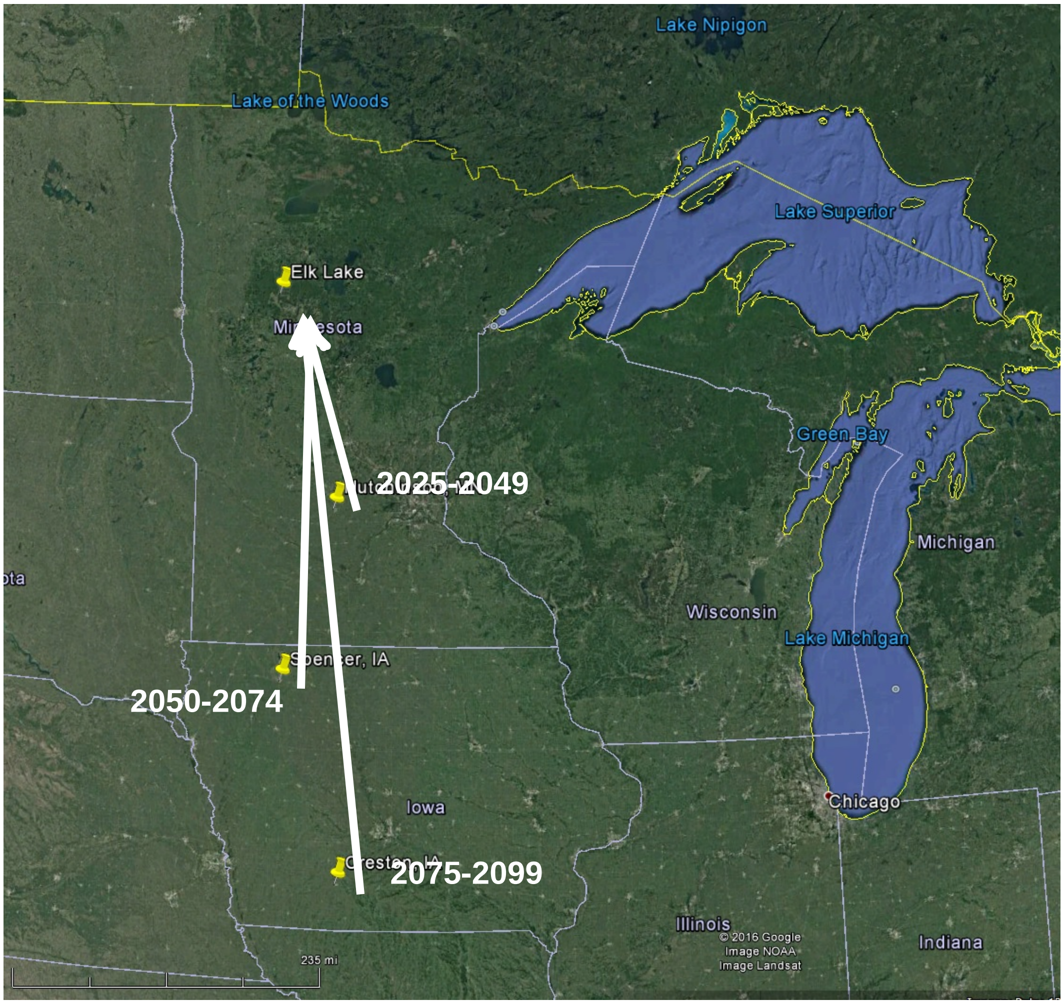


Figure 4: Similar climatological locations in "space-for-time" substitution for Elk Lake (Hubbard County, Minn.): Hutchinson, Minn. (McLeod County, Minn.) for RCP8.5, 2025-2049; Spencer, Iowa (Clay County, Iowa) for RCP8.5, 2050-2074; and Creston, Iowa (Union County, Iowa) for RCP8.5, 2075-2099.

For Trout Lake, the following locations were determined to have the most analogous climatological annual means for the RCP8.5 scenarios (Figure 5):

- RCP8.5, 2025-2049: Ashland County, Wisc. (John F. Kennedy Memorial Airport, Ashland, Wisc.);
- RCP8.5, 2050-2074: Brown County, Wisc. (Austin Straubel International Airport, Green Bay, Wisc.);
- RCP8.5, 2075-2099: Lake County, Ill. (Waukegan National Airport, Waukegan, Ill.).

As with Elk Lake, climate data was downloaded from the NCDC for 2010, the calibrated model year for Trout Lake (Smith et al. 2014), from the nearest continuous climate station (NCDC, 2016).

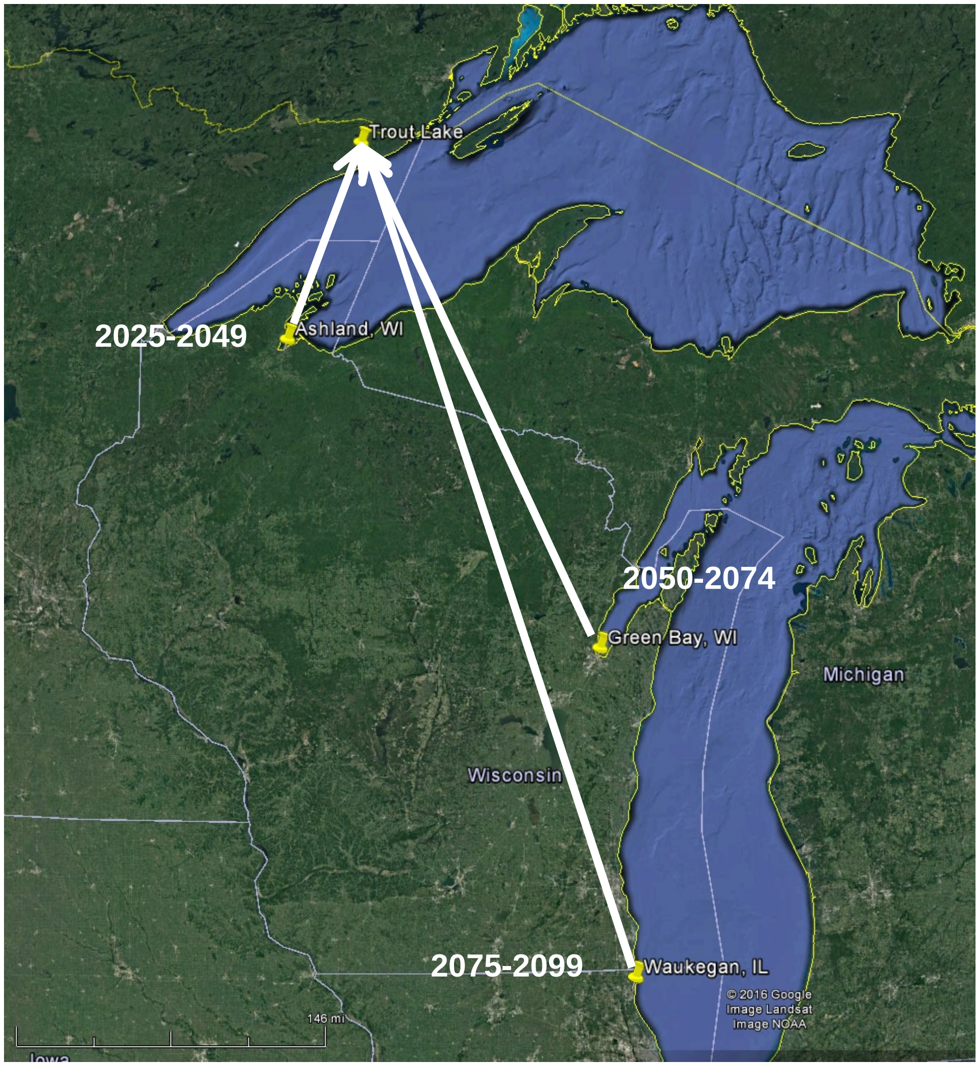


Figure 5: Similar climatological locations in "space-for-time" substitution for Trout Lake (Cook County, Minn.): Ashland, Wisc. (Ashland County, Wisc.) for RCP8.5, 2025-2049; Green Bay, Wisc. (Brown County, Wisc.) for RCP8.5, 2050-2074; and Waukegan, Ill. (Lake County, Ill.) for RCP8.5, 2075-2099.

With the approach developed for procuring the meteorological data for all of the different climate scenarios, the next step was to run the calibrated CE-QUAL-W2 models (Smith et al. 2014) to set the baseline condition for Elk Lake and Trout Lake. Since the publication of the original Elk Lake and Trout Lake models (Smith et al. 2014), minor revisions to the hydraulic and water-quality parameters were made to improve the original performance of both models. This was highlighted in an earlier update to the LCCMR, as the new model version required updating some parameters. These newly calibrated CE-QUAL-W2 models were used for the three RPC8.5 climate scenarios on both lakes, including the original meteorological file used for the Smith et al. (2014) models. For the nine SRES climate scenarios on Elk Lake (e.g., A1B in 2030, A2 in 2070), the wind sheltering coefficient and a few other hydraulic parameters of the newly calibrated Elk Lake model had to be updated as the wind speed from the Meteonorm software package overestimated wind speed compared to actual Elk Lake meteorological station measurements. The Elk Lake SRES baseline model was calibrated to an independent 2011 Elk Lake meteorological data set, as provided by the Meteonorm software package.

Results for climate change scenarios: All climate scenarios for Elk Lake and Trout Lake were run using CE-QUAL-W2 version 4, compiled on July 14, 2015. *These results are preliminary and are subject to revision. It is being provided to meet the need for timely best science. The information is provided on the condition that neither the U.S. Geological Survey nor the U.S. Government shall be held liable for any damages resulting from the authorized or unauthorized use of this information.*

For each climate scenario, temperature, dissolved oxygen, and coldwater fish lake habitat volume were used as the primary methods for determining relative changes in the lake. For these different metrics, several of the figures in the following sections (temperature, dissolved oxygen, lake volume habitat) show the measured values, the baseline calibration, and the revised climate scenario. Aside from the differences explained above for setting the baseline model runs, the only difference from run to run was the new meteorological dataset. Generally, a single recalibration of the water balance was necessary after each run, as the new climatological set would slightly unbalance the distributed tributary flow. For more details on water balance adjustments, refer to Smith et al. (2014). For all climate scenarios (Elk Lake, Trout Lake), comparison figures have been generated but only particular examples will be shown for this progress report. Summary table and graphs will also follow that summarize the annual mean for the different climate scenarios and the percent change between the baseline and climate change scenarios.

Elk Lake had a single monitoring location, located in deepest portion of the south basin, with continuously monitored temperature and dissolved oxygen (Smith et al. 2014). This monitoring data was summarized into daily values, and existed for the following depths: 2 meters, 7 meters, 20 meters, and 28 meters (temperature only). These locations monitored the epilimnion (at 2 meters), the transitional zone (7 meters), and the hypolimnion (20 meters, 28 meters). The 7-meter profile was particularly important, as the thermocline and chemocline (for dissolved oxygen) was often located near this boundary for the 2011 calibrated baseline model; however, this shifted substantially in the climate change scenarios, which had implications for lake volume habitat suitable for coldwater fish.

Trout Lake also had a single monitoring location, located in the deepest portion north basin, with only continuously monitored temperature for 2010 (Smith et al. 2014). This monitoring data was summarized into daily values, and existed for two depths: 1 meter (epilimnion) and 18 meters (hypolimnion). Without continuous dissolved oxygen for Trout Lake, the periodic dissolved oxygen profiles will be shown, but the difference and percent change were not calculated for these depth profiles.

Temperature results (Elk Lake): Figure 6 shows the percent change for annual mean water temperature from the baseline (2011) Elk Lake model for all 12 scenarios. The figure is divided into four sub-graphs, each showing the percent change with time (simulation year) for the four different depths.

At 2 meters, all climate scenarios caused the epilimnion water temperatures to warm as the century progressed. The simple addition of more heat over time, caused by increased air temperature, caused these temperatures to rise. At 28 meters, these changes were muted and the different scenarios were barely distinguishable, although warmer groundwater temperatures were not modeled, in which case deep hypolimnion temperatures could have warmed by another 1 to 2 degrees Celsius if this had been factored into the model. The results were mixed at both 7 meters and 20 meters, with a grouping of the SRES scenarios (A1B, A2, and B1) in the 5 to 15 percent change range; alternatively, the RCP8.5 scenarios had large percent changes, up to 43 percent (Table 1).

For the A1B scenarios (2030, 2050, and 2070), the temperature generally increased throughout the lake as the 21st century progresses. Although the atmospheric greenhouse gas concentrations has been projected to decline in the second half of the 21st century (IPCC, 2007; IPCC, 2014), the lag effect on meteorological data does not seem to translate to the meteorological record and therefore the climatological forcing on Elk Lake. Similar to A1B, the A2 scenarios also increased for two of the three different time periods selected (2030, 2050, and 2070); however, the A2 scenarios dipped in 2050 at all depths except the epilimnion (2 meters). This was likely due to either a cancellation effect from higher temperatures for part of the year with lower temperatures in a different portion of the year, or a subtle effect of different wind mixing regimes that could have led to the 2050 A2 results deviating from the temperature trends for the A1B and B1 climate scenarios.

For the RCP8.5 scenarios, the percent change was higher for all of the depths except at 2 meters. The RCP8.5 is an aggressive climate change scenario compared to the SRES, and also is highly dependent on the climatological data for 2011 at the alternate locations. However, the basic idea behind "space-for-time" should hold although more robust testing of this method would need to occur before further publication.

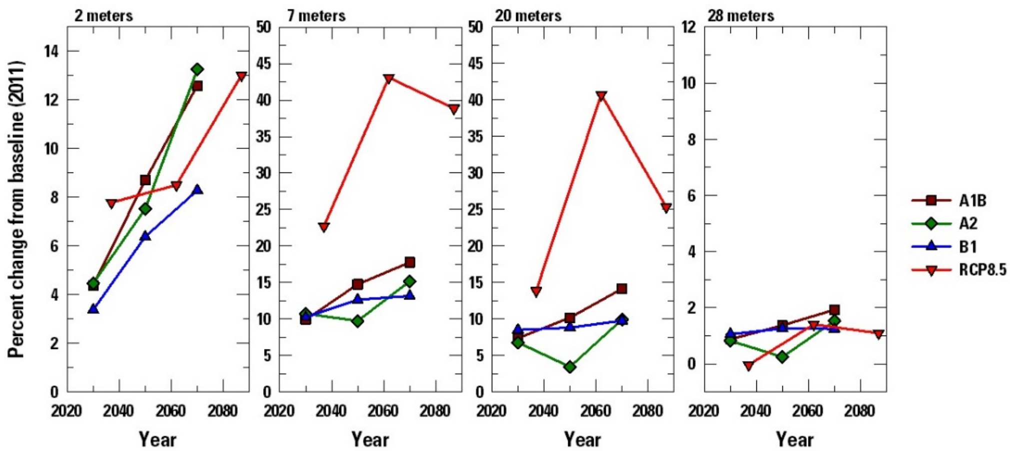


Figure 6: Percent change for annual mean water temperature from the baseline (2011) Elk Lake model for the 3 SRES climate scenarios (2030, 2050, 2070) and the RCP8.5 scenarios (plotted at midpoint of 2025-2049, 2050-2074, 2075-2099). Each panel represents a different continuous profile depth: 2 meters, 7 meters, 20 meters, 28 meters.

Table 1: Summary of annual water temperature means for four continuous profiles (2 meters, 7 meters, 20 meters, and 28 meters) for the 2011 Elk Lake baseline (SRES, RCP8.5) and the 12 climate scenarios. The percent change (departure) from the 2011 Elk Lake baseline is shown for the 12 climate scenarios.

Climate Scenario	Water temperature (degrees Celsius)			
	2 meters	7 meters	20 meters	28 meters
Annual mean				
Baseline (2011)	17.62	10.32	6.66	5.35
Baseline (2011), RCP8.5	17.61	11.12	6.27	5.80
A1B, 2030	18.40	11.34	7.16	5.40
A1B, 2050	19.16	11.84	7.34	5.42
A1B, 2070	19.84	12.15	7.61	5.45
A2, 2030	18.41	11.43	7.12	5.39
A2, 2050	18.95	11.33	6.89	5.36
A2, 2070	19.96	11.89	7.33	5.43
B1, 2030	18.22	11.38	7.23	5.41
B1, 2050	18.75	11.63	7.25	5.42
B1, 2070	19.08	11.68	7.32	5.42
RCP8.5, 2025-2049	18.98	13.64	7.14	5.80
RCP8.5, 2050-2074	19.11	15.91	8.83	5.88
RCP8.5, 2075-2099	19.90	15.44	7.87	5.86
Percent change (from 2011 baseline)				
A1B, 2030	4.39	9.92	7.37	0.88
A1B, 2050	8.71	14.77	10.19	1.37
A1B, 2070	12.60	17.78	14.14	1.93
A2, 2030	4.47	10.74	6.79	0.82
A2, 2050	7.53	9.76	3.44	0.25
A2, 2070	13.26	15.19	9.93	1.53
B1, 2030	3.38	10.26	8.50	1.06
B1, 2050	6.38	12.66	8.83	1.26
B1, 2070	8.29	13.17	9.78	1.24
RCP8.5, 2025-2049	7.79	22.72	13.87	-0.04
RCP8.5, 2050-2074	8.51	43.10	40.74	1.41
RCP8.5, 2075-2099	13.02	38.87	25.35	1.09

Three examples are shown in Figures 7-9 to demonstrate the progressive changes in water temperature for all four continuous profile depths, in this case for the A1B scenario in 2030 (Figure 7), the A1B scenario in 2070 (Figure 8), and the A2 scenario in 2070 (Figure 9). For the deeper parts of the lake (20 meters, 28 meters), the differences between the calibrated (2011) model and the climate scenarios were difficult to discern. However, the epilimnion (2 meters) shows higher August and September water temperatures for both the 2030 A1B (Figure 7) and the 2070 A1B (Figure 8), but is up to 4 degrees Celsius warmer than the 2011 baseline for the 2070 A1B and 2070 A2. At the transitional zone (7 meters), the 2030 A1B results are similar to the measured values but by the 2070 A1B and A2 climate scenarios the simulated values deviate particularly towards the end of the simulation period in late September and October.

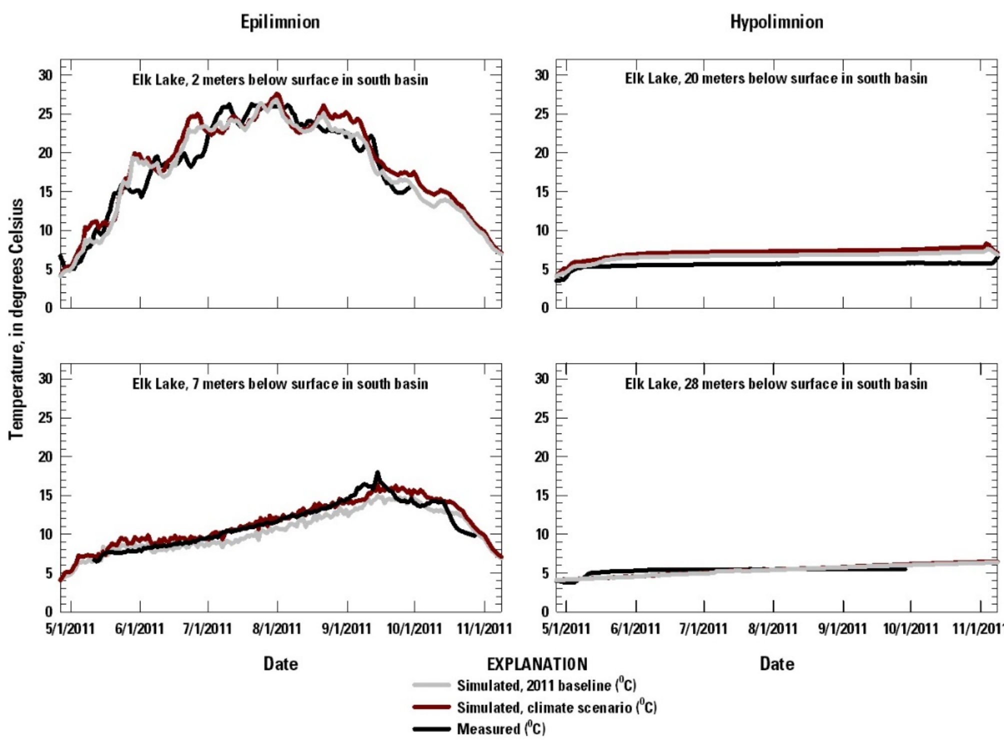


Figure 7: Simulated and measured water temperature for the epilimnion, transitional zone (7 meters), and the hypolimnion at the south basin hole in Elk Lake. Also shown is the A1B climate scenario results for 2030 (highlighted in red).

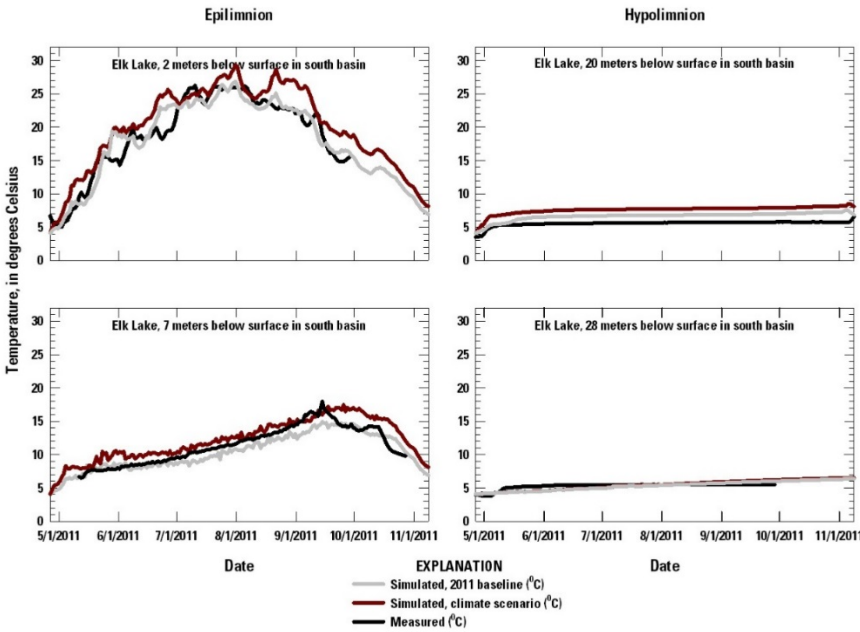


Figure 8: Simulated and measured water temperature for the epilimnion, transitional zone (7 meters), and the hypolimnion at the south basin hole in Elk Lake. Also shown is the A1B climate scenario results for 2070 (highlighted in red).

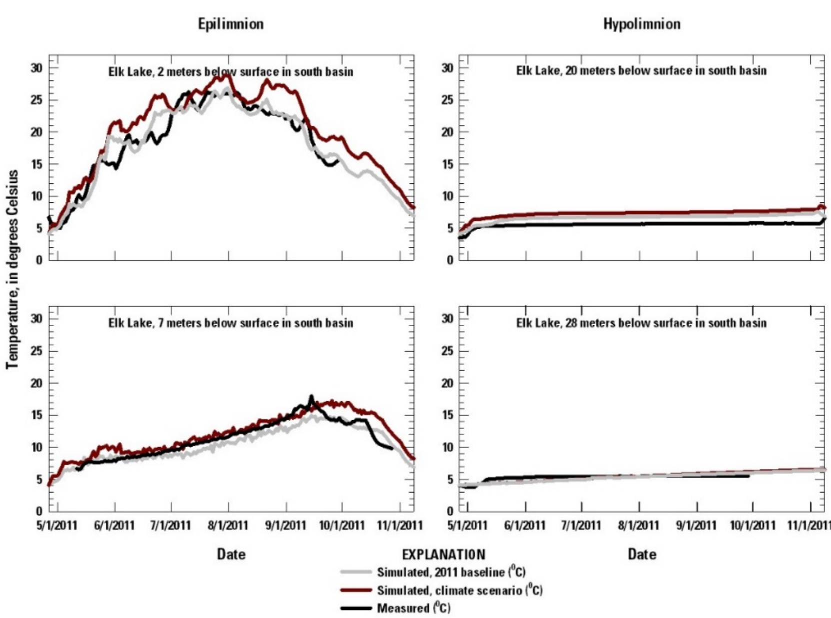


Figure 9: Simulated and measured water temperature for the epilimnion, transitional zone (7 meters), and the hypolimnion at the south basin hole in Elk Lake. Also shown is the A2 climate scenario results for 2070.

Two more examples are shown in Figures 10 and 11 for the more dramatic differences for the RCP8.5 scenarios, in this case the RCP8.5 2025-2049 (Figure 10) and the RCP8.5 2075-2099 (Figure 11).

Particularly in the transitional zone at 7 meters, warmer water is pushed deeper into the lake. For the earlier time period, 2025-2049 (Figure 9), the warmer water is not pushed deeper into the lake until July. However, with the 2075-2099 RCP8.5 scenario (Figure 10), the warm water is pushed into the lake from the beginning of the simulation period in late April. With both scenarios, the water temperatures begin to equilibrate as the lake mixes and approaches the measured data in mid- to late October.

It should be noted for the RCP8.5 scenarios, the “space-for-time” could introduce to much more energy into the lake. While the usage of comparing NCCV results seemed fair for generating the meteorological data, this method is highly dependent on meteorological data for year the data has been transferred. A better method would like be to average several years together to account for this variability, and is the preferred method being considered for peer-review literature.

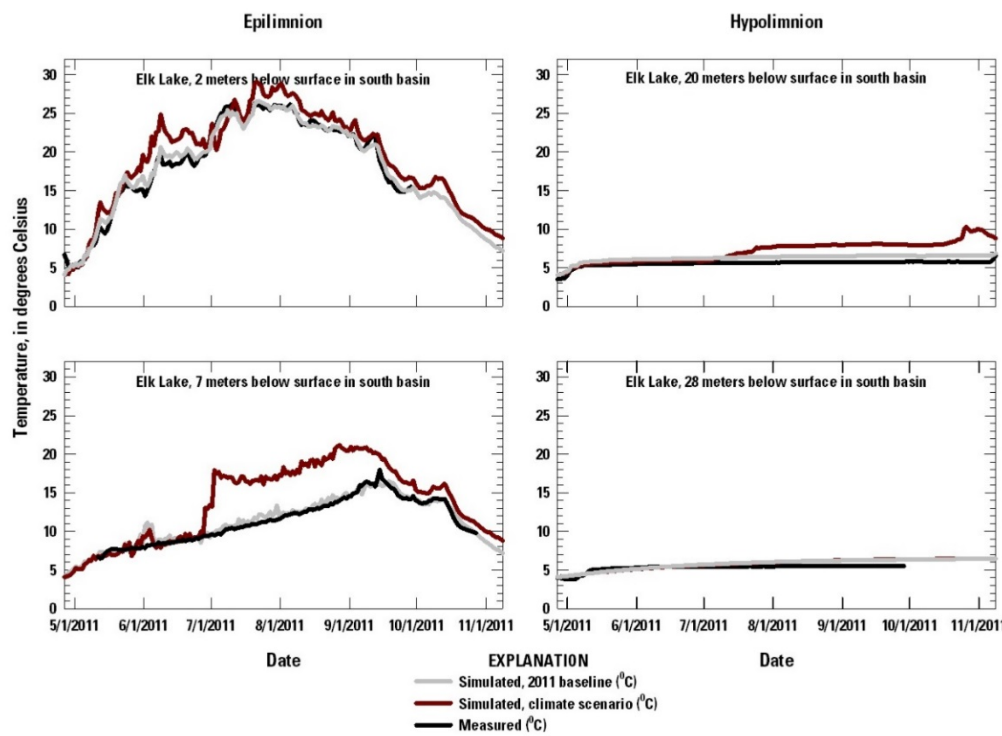


Figure 10: Simulated and measured water temperature (in degrees Celsius) for the epilimnion, transitional zone (7 meters), and the hypolimnion at the south basin hole in Elk Lake. Also shown is the RCP8.5 climate scenario results for 2025-2049 (McLeod County, Hutchinson, Minn.), highlighted in red.

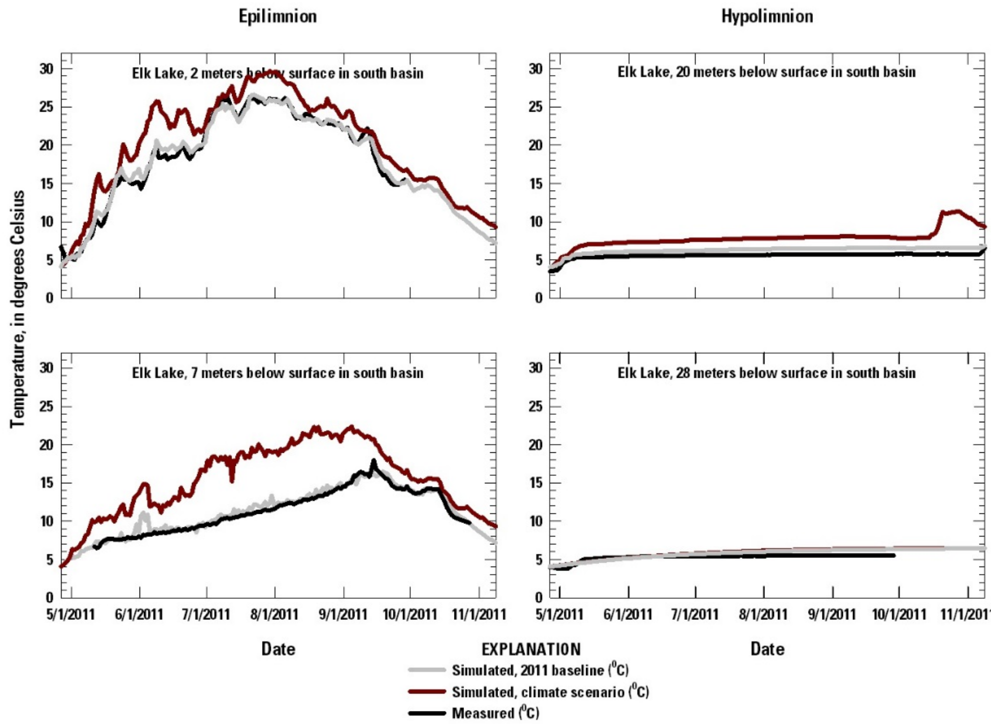


Figure 11: Simulated and measured water temperature (in degrees Celsius) for the epilimnion, transitional zone (7 meters), and the hypolimnion at the south basin hole in Elk Lake. Also shown is the RCP8.5 climate scenario results for 2075-2099 (Union County, Creston, Iowa), highlighted in red.

<u>*Temperature results (Trout Lake):*</u> Trout Lake only had the three RCP8.5 scenarios and did not include the nine SRES scenarios. Overall, the RCP8.5 scenarios for Trout Lake caused a more uniform change in temperature as compared to Elk Lake. The most likely reason for the difference is that Trout Lake is a simpler model.

Figure 12 shows the percent change for annual water temperature from the baseline (2010) Trout Lake model for the three RCP8.5 scenarios with time (simulation year) at 1 meter and 18 meters. Each point corresponds to the mid-point of the RCP8.5 scenarios; for example, the RCP8.5 2025-2049 scenario (Ashland, Wisc.) is plotted at year 2037. Over the course of the 21st century, the epilimnion (at 1 meter) becomes progressively warmer as air temperatures become warmer; the hypolimnion (at 18 meters) shows little variation with the new climate scenarios. Also highlighted in Table 2, annual mean water temperature at 1 meter increases from 17.4 degrees Celsius from the baseline (2010) Trout Lake model to 21.8 degrees Celsius for RCP8.5 (2075-2099), an increase of 4.4 degrees Celsius. This change happens at a fairly uniform rate of 0.6 degrees Celsius per decade.

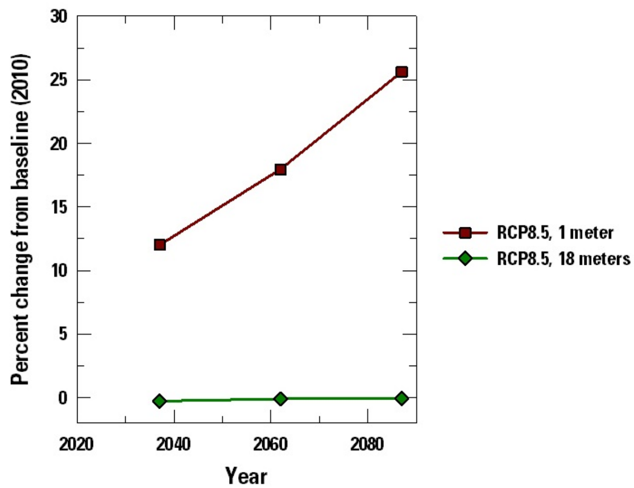


Figure 12: Percent change for annual mean water temperature from the baseline (2010) Trout Lake model for the RCP8.5 scenarios (plotted at midpoint of 2025-2049, 2050-2074, 2075-2099).

Table 2: Summary of annual water temperature means for two continuous profiles (1 meter and 18 meters) for the 2010 Trout Lake baseline (RCP8.5) and the three RCP8.5 climate scenarios. The percent change (departure) from the 2010 Trout Lake baseline is shown for the RCP8.5 climate scenarios.

Climate Scenario	**Water temperature (degrees Celsius)**	
	1 meter	**18 meters**
Annual mean		
Baseline (2010)	17.38	7.57
RCP8.5, 2025-2049	19.47	7.55
RCP8.5, 2050-2074	20.50	7.56
RCP8.5, 2075-2099	21.83	7.56
Percent change (from 2011 baseline)		
RCP8.5, 2025-2049	12.02	-0.26
RCP8.5, 2050-2074	17.97	-0.10
RCP8.5, 2075-2099	25.63	-0.09

In Figures 13-15, the three RCP8.5 scenarios are plotted. Figure 13 shows the continuous water temperature profiles at 1 meter and 18 meters for the RCP8.5 2025-2049 scenario (Ashland, Wisc.). Unlike the Elk Lake scenarios, the epilimnion shows a fairly uniform warming of the epilmnion beginning in early June through the remainder of the simulation period. For the following two scenarios, the epilimnion begins to warm in early May and warms up further starting in early June. The primary difference between RCP8.5 2050-2074 (Green Bay, Wisc.; figure 14) and RCP8.5 2075-2099 (Waukegan, Ill.; figure 15) is the extent of the mid- to late summer warming. By the end of the 21st century in the RCP8.5 2075-2099 scenario (Figure 15), daily mean water temperatures reach close to 27 to 28 degrees Celsius. This is in contrast to the 2010 baseline high temperatures of 22 to 23 degrees Celsius.

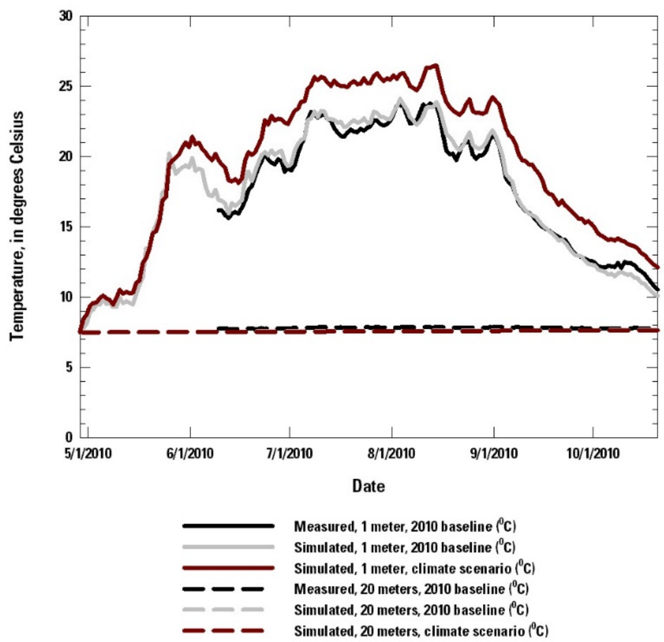


Figure 13: Simulated and measured water temperature (in degrees Celsius) for the epilimnion (1 meter) and the hypolimnion (18 meters) at the north basin hole in Trout Lake. Also shown is the RCP8.5 climate scenario results for 2025-2049 (Ashland County, Ashland, Wisc.), highlighted in red.

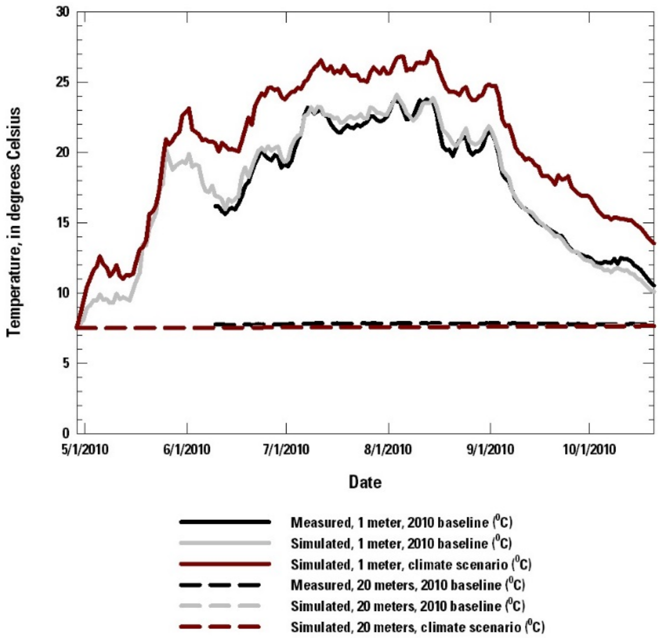


Figure 14: Simulated and measured water temperature (in degrees Celsius) for the epilimnion (1 meter) and the hypolimnion (18 meters) at the north basin hole in Trout Lake. Also shown is the RCP8.5 climate scenario results for 2050-2074 (Brown County, Green Bay, Wisc.).

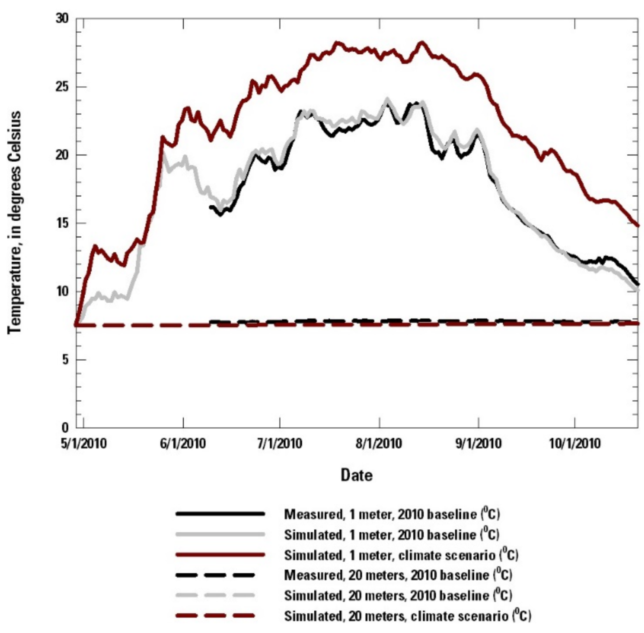


Figure 15: Simulated and measured water temperature (in degrees Celsius) for the epilimnion (1 meter) and the hypolimnion (18 meters) at the north basin hole in Trout Lake. Also shown is the RCP8.5 climate scenario results for 2075-2099 (Lake County, Waukegan, Ill.).

Dissolved oxygen results (Elk Lake): Figure 16 shows the percent change for annual dissolved oxygen from the baseline (2011) Elk Lake model for all 12 scenarios. The figure is divided into three sub-graphs, each showing the percent change with time (simulation year) for the three different depths available for dissolved oxygen.

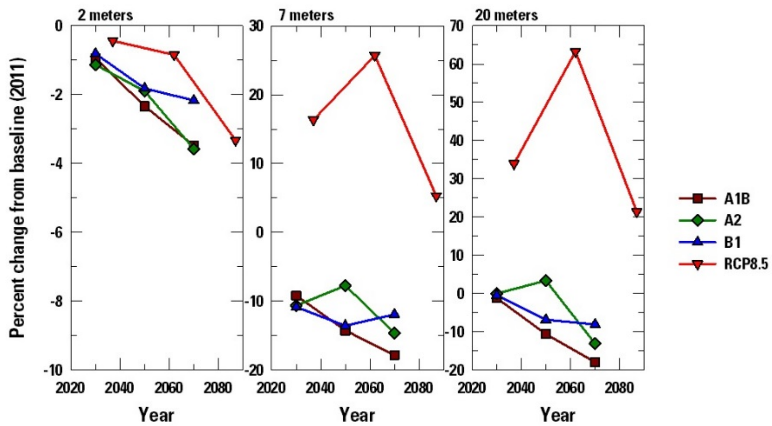


Figure 16. Percent change for annual mean dissolved oxygen from the baseline (2011) Elk Lake model for the 3 SRES climate scenarios (2030, 2050, 2070) and the RCP8.5 scenarios (plotted at midpoint of 2025-2049, 2050-2074, 2075-2099). Each panel represents a different continuous profile depth: 2 meters, 7 meters, and 20 meters.

Table 3. Summary of annual dissolved oxygen (in milligrams per liter) means of three continuous profiles (2m, 7m, and 20m) for the 2011 Elk Lake baseline, the nine Special Reports on Emissions Scenarios (SRES) and the three RCP8.5 climate scenarios. The percent change (departure) from the 2011 Elk Lake baseline is shown for the nine SRES and three RCP8.5 climate scenarios.

Climate Scenario	Dissolved oxygen (in milligrams per liter)		
	2 meters	7 meters	20 meters
Annual mean			
Baseline (2011), SRES	8.96	5.65	2.60
Baseline (2011), RCP8.5	9.07	4.95	1.73
A1B, 2030	8.88	5.13	2.57
A1B, 2050	8.75	4.84	2.33
A1B, 2070	8.65	4.64	2.14
A2, 2030	8.86	5.05	2.60
A2, 2050	8.79	5.21	2.69
A2, 2070	8.64	4.82	2.26
B1, 2030	8.89	5.04	2.59
B1, 2050	8.80	4.88	2.43
B1, 2070	8.77	4.98	2.39
RFP8.5, 2025-2049	9.03	5.75	2.31
RFP8.5, 2050-2074	8.99	6.22	2.82
RFP8.5, 2075-2099	8.77	5.21	2.10
Percent change (from 2011 baseline)			
A1B, 2030	-0.93	-9.16	-1.22
A1B, 2050	-2.34	-14.23	-10.60
A1B, 2070	-3.48	-17.80	-17.95
A2, 2030	-1.13	-10.62	-0.02
A2, 2050	-1.89	-7.72	3.38
A2, 2070	-3.58	-14.61	-13.07
B1, 2030	-0.81	-10.82	-0.48
B1, 2050	-1.82	-13.54	-6.86
B1, 2070	-2.17	-11.90	-8.06
RFP8.5, 2025-2049	-0.43	16.36	33.96
RFP8.5, 2050-2074	-0.84	25.71	63.27
RFP8.5, 2075-2099	-3.32	5.31	21.39

The three RCP8.5 scenarios, on the other hand, showed positive percent change at both 7 meters and 20 meters. Unlike the nine SRES scenarios (A1B, A2, and B1), the enhanced wind profiles from the substituted meteorological records in combination with warmer temperatures led to enhanced algal production, which thereby led to elevated primary production of dissolved oxygen. Since the warmer temperatures were not as increased for the nine SRES scenarios, the algal production was not as affected.

Two examples are shown in Figures 17 and 18 to demonstrate the progressive changes in dissolved oxygen temperature for the three continuous profile depths, in this case for the RCP8.5 2025-2049 scenario (Figure 17) and the RCP8.5 2075-2099 scenario (Figure 18). At 7 meters for both figures, enhanced mixing events of elevated dissolved oxygen occurred in early July and late August to early September. These events were likely because of elevated primary production, sinking deeper into the metalimnion, related back to the elevated water temperatures which enhance algal growth. At 20 meters, late season lake mixing pushed dissolved oxygen to the bottom of the lake (Figures 17 and 18).

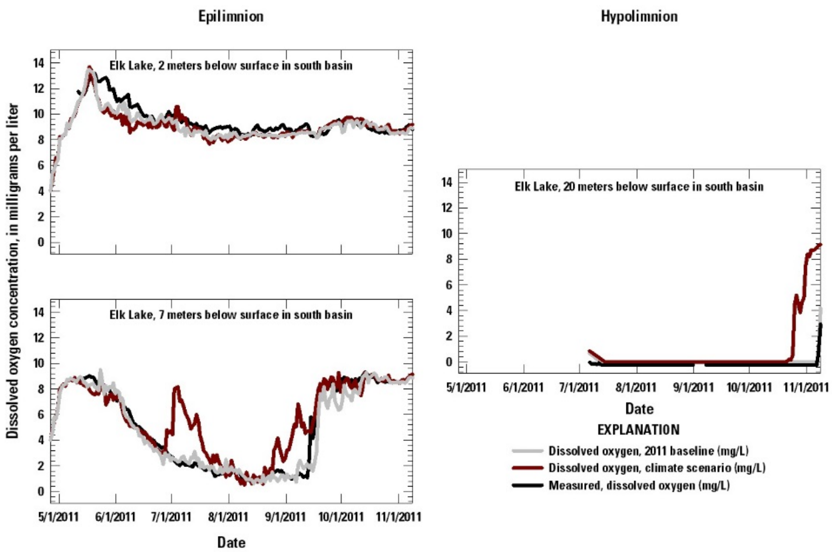


Figure 17: Simulated and measured dissolved oxygen (in milligrams per liter) for the epilimnion (2 meters), transitional zone (7 meters), and the hypolimnion (20 meters) at the south basin hole in Elk Lake. Also shown is the RCP8.5 climate scenario results for 2025-2049 (McLeod County, Hutchinson, Minn.), highlighted in red.

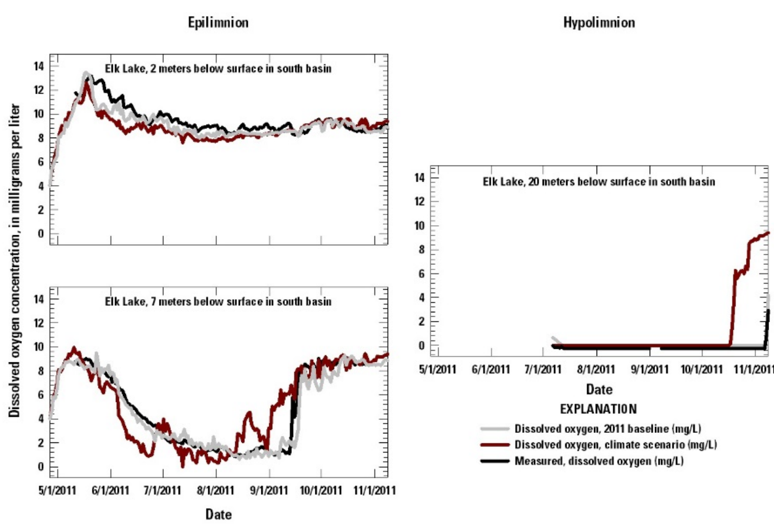


Figure 18: Simulated and measured dissolved oxygen (in milligrams per liter) for the epilimnion (2 meters), transitional zone (7 meters), and the hypolimnion (20 meters) at the south basin hole in Elk Lake. Also shown is the RCP8.5 climate scenario results for 2075-2099 (Union County, Creston, Iowa), highlighted in red.

Dissolved oxygen results (Trout Lake): As opposed to Elk Lake, Trout Lake does not exhibit a large range of variability for dissolved oxygen as Trout Lake has limited primary production compared to Elk Lake, even under warmer temperatures. Figures 19 and 20 show changes in dissolved oxygen across 15 depth profiles of dissolved oxygen measured across the simulation year. The measured profiles for 2010 are shown in addition to the simulated 2010 profiles of dissolved oxygen and the RCP8.5 scenario profiles. In Figure 19 which shows the RCP8.5 2025-2049 scenario (Ashland County, Ashland, Wisc.), the metalimnetic oxygen maximum is exhibited in both the measured profile and calibrated 2010 results; however, this feature erodes as the year progresses. Slightly lower primary production combined with warmer temperatures caused the lower dissolved oxygen concentrations in the lake. Figure 20, which shows the RCP8.5 2075-2099 climate scenario, differs little to Figure 19, except the metalimnetic oxygen maximum is further eroded for this later scenario.

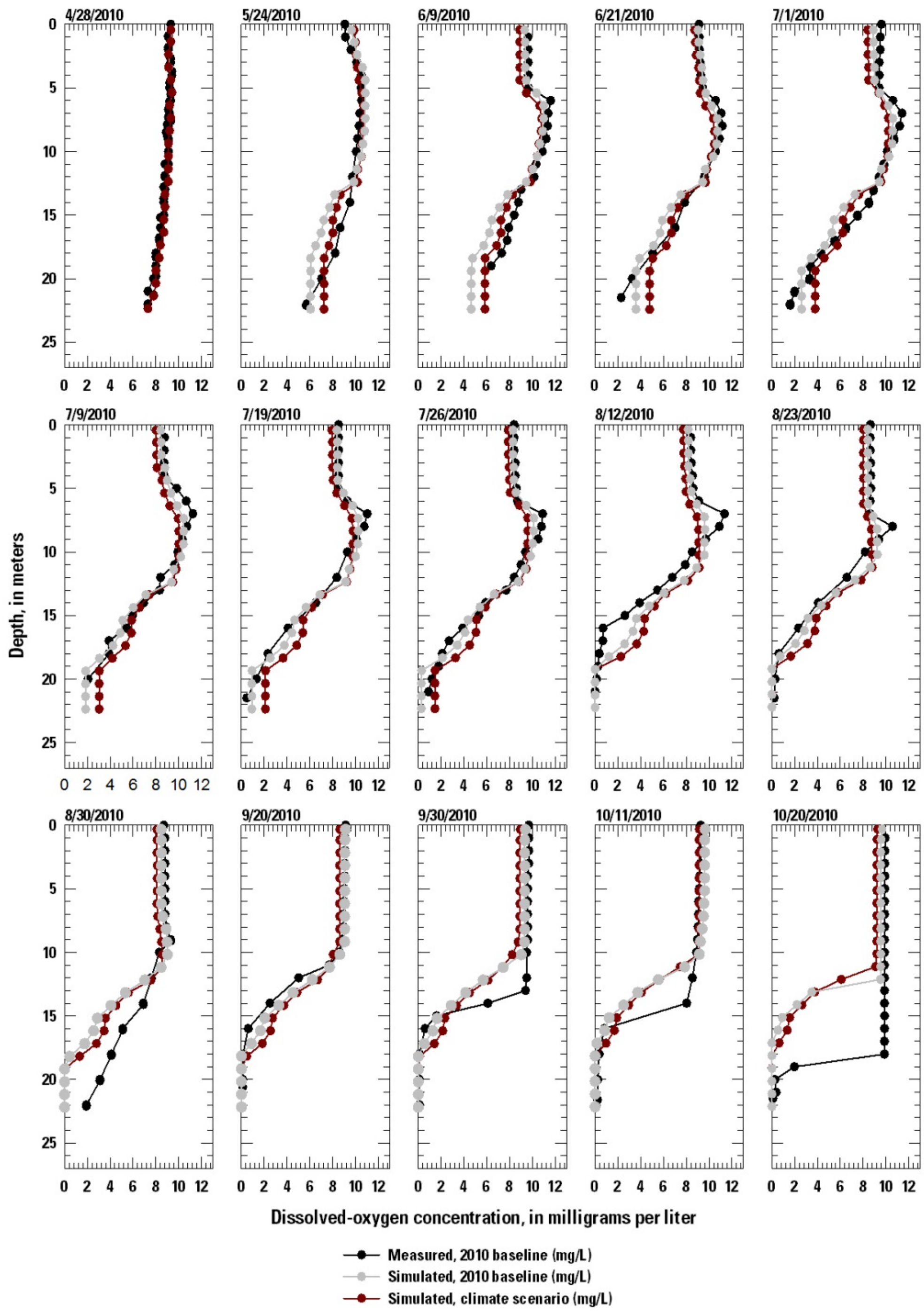


Figure 19: Simulated and measured dissolved oxygen concentrations at the north basin hole in Trout Lake for 15 dates. Also shown in red is the RCP8.5 2025-2049 scenario (Ashland County, Ashland, Wisc.).

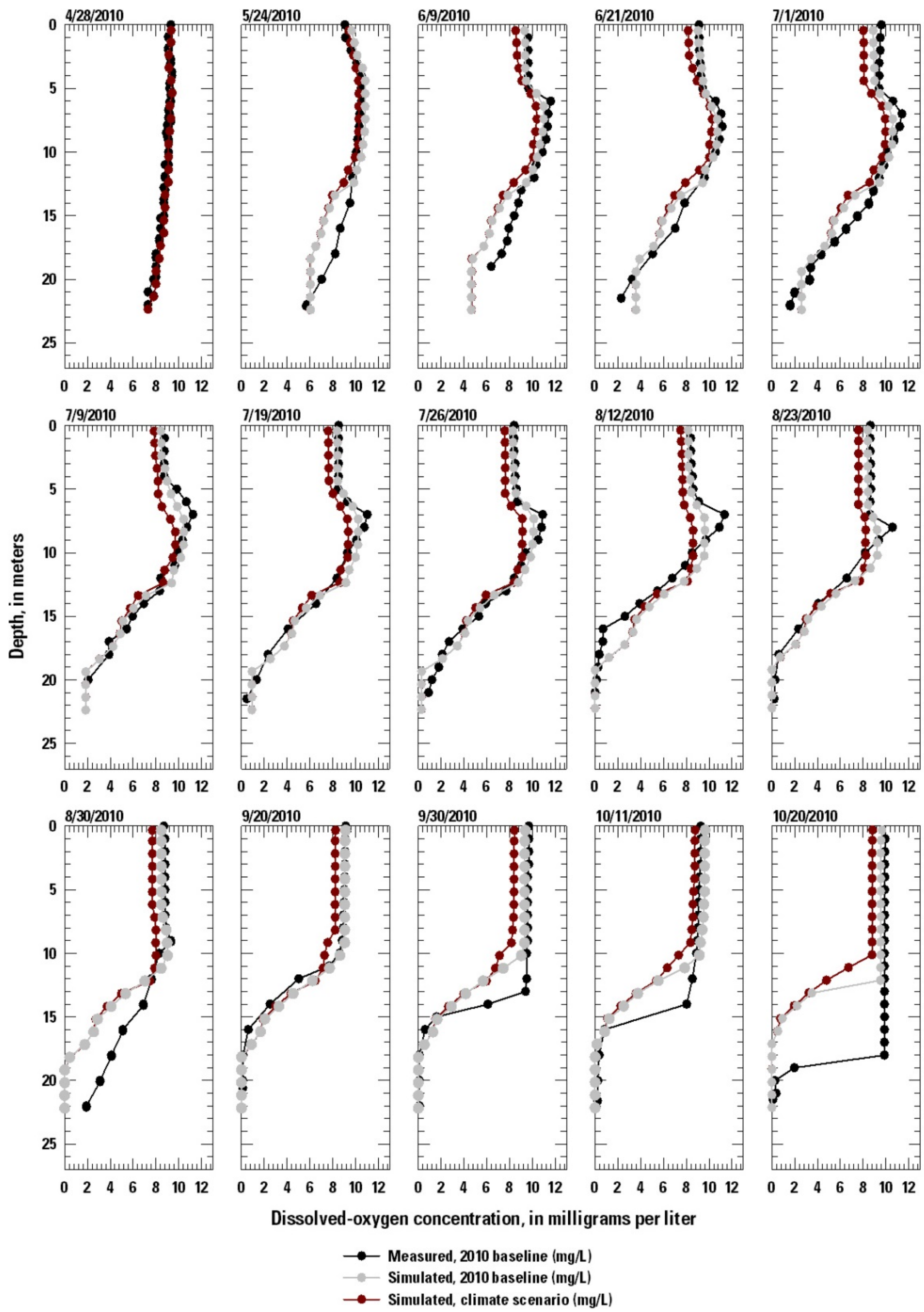


Figure 20: Simulated and measured dissolved oxygen concentrations at the north basin hole in Trout Lake for 15 dates. Also shown in red is the RCP8.5 2075-2099 scenario (Lake County, Waukegan, Ill.).

<u>*Lake habitat volume for coldwater fish (Elk Lake and Trout Lake):*</u> In order to evaluate fish lake habitat suitability for coldwater fish species, with emphasis on Northern Cisco, the CE-QUAL-W2 model was evaluated for total volume of good-growth habitat and lethal oxythermal habitat. The good-growth habitat temperature ranges for Northern Cisco *Coregonus artedi* were developed using models developed by Fang et al. (1999); for all of the following figures, good-growth habitat was modeled as

water temperatures between 9 and 18.5 degrees Celsius and a minimum dissolved oxygen concentration of 3 milligrams per liter (mg/L). Lethal habitat volume was approximately modeled as temperatures above 22 degrees Celsius; the minimum dissolved oxygen concentration threshold was lower for cooler temperatures, and as the water temperatures rise above 22-25 degrees Celsius, the minimum lethal dissolved oxygen concentration would rise. For further guidance on the development of the lethal temperature-dissolved oxygen curve, Fang et al. (2014) includes the detailed development of the function and relationship.

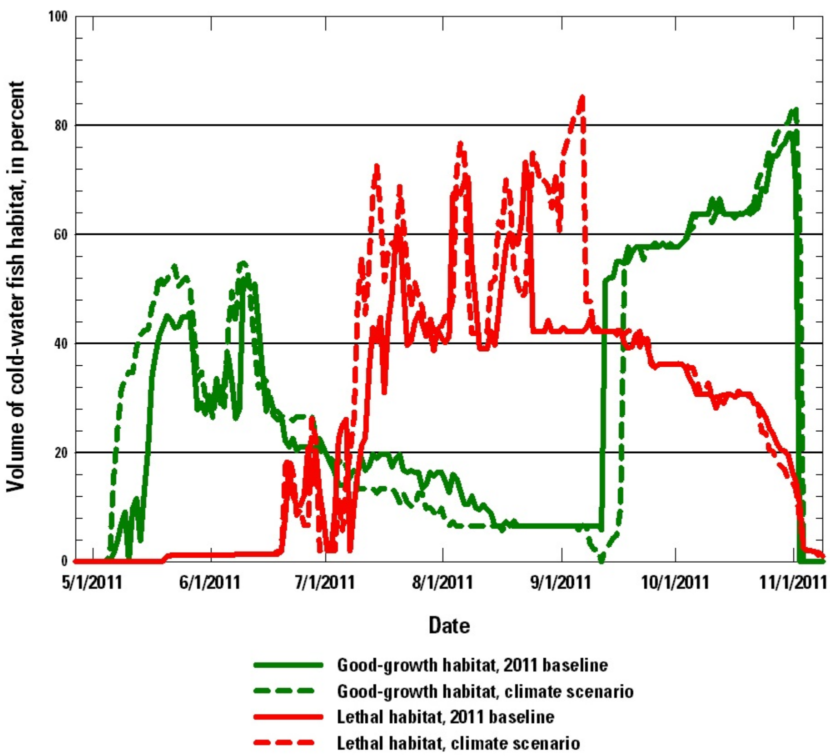


Figure 21: Good-growth and lethal habitat for Elk Lake, as shown for the 2011 Elk Lake baseline model (solid lines) and the A1B scenario for 2030 (dashed lines).

For the Elk Lake baseline model (2011), the summer lethal habitat will spike as high as 70 percent of the lake volume. The primary cause of these periods of high lethal habitat was due to an expansion of low dissolved oxygen volume particularly as the metalimnion oxygen maximum eroded. Good-growth habitat mainly existed, for both the baseline model (2011) and the A1B scenario (Figure 21) in May/June and again in the fall period during October as the lake mixes. The availability of good-growth habitat also slightly eroded in Figure 21.

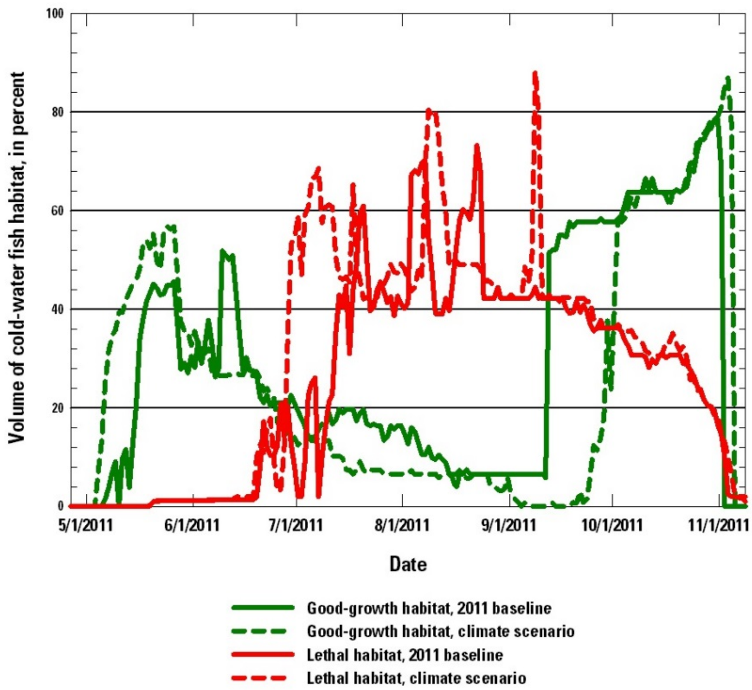


Figure 22: Good-growth and lethal habitat for Elk Lake, as shown for the 2011 Elk Lake baseline model (solid lines) and the A1B scenario for 2070 (dashed lines).

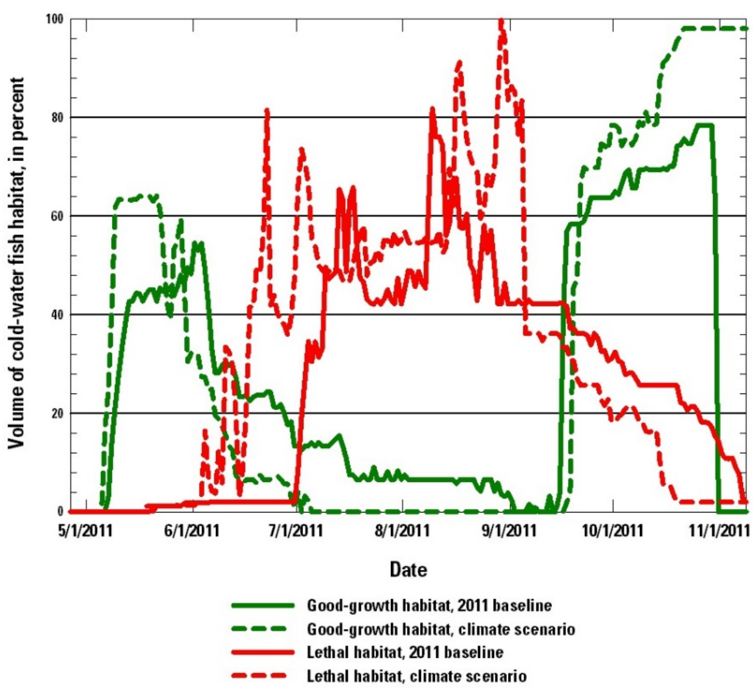


Figure 23: Good-growth and lethal habitat for Elk Lake, as shown for the 2011 Elk Lake baseline model (solid lines) and the RCP8.5, 2075-2099 scenario (Union County, Creston, Iowa; dashed lines).

Figures 22 and 23 show the good-growth and lethal habitat for Elk Lake for the A1B scenario for 2070 and the RCP8.5, 2075-2099 scenario (Union County, Creston, Iowa), respectively. As the century progresses, lethal habitat for Northern Cisco expands for longer periods of time, at higher frequencies, and a higher overall percentage of the lake volume. Furthermore, good-growth habitat approaches zero during the summer months. The combination of lower good-growth habitat and a large expansion of lethal habitat would likely cause the existence of Northern Cisco to become threatened under these climate scenarios.

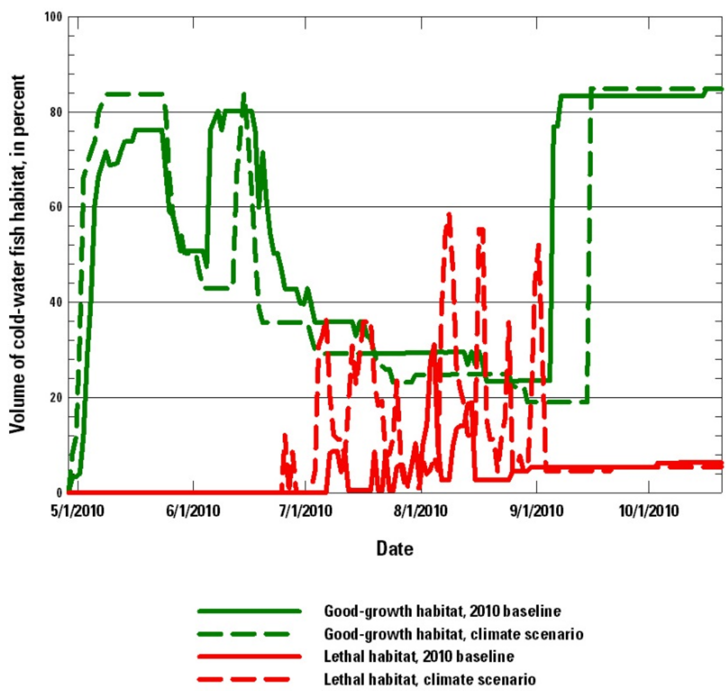


Figure 24: Good-growth and lethal habitat for Trout Lake, as shown for the 2010 Trout Lake baseline model (solid lines) and the RCP8.5, 2025-2049 scenario (Ashland County, Ashland, Wisc.; dashed lines).

Figures 24 and 25 show the Trout Lake good-growth and lethal habitat volumes for the RCP8.5, 2025-2049 scenario (Ashland County, Ashland, Wisc.) and the RCP8.5, 2075-2099 scenario (Lake County, Waukegan, Ill.), respectively. Unlike Elk Lake, lethal habitat lake volume was less frequent and would last for shorter periods in Trout Lake. While the lethal habitat volume could spike as high as 60 percent, such as the mid-August events in Figure 24, these events would last 4-5 days and would quickly return to a lake volume below 10 percent. In Figure 25, lethal habitat events actually occurred with less frequency and at an overall lower lake volume; however, the events started earlier in the year, by up to a month. Also, the largest change for the climate change scenarios for Trout Lake was the erosion over the century in good-growth habitat. In Figure 24, the change in good-growth habitat had only begun; however, in Figure 25 (RCP8.5, 2075-2099 scenario) up to 60 percent of the good-growth habitat is lost for Northern Cisco.

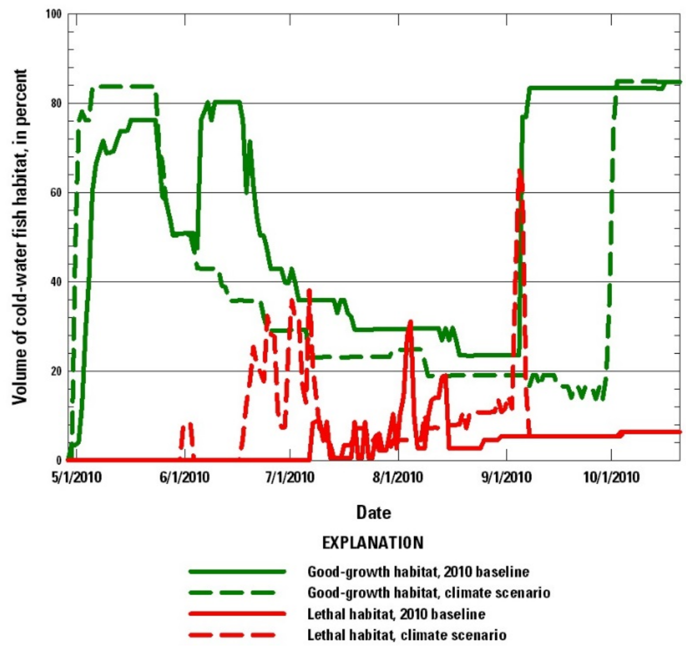


Figure 25: Good-growth and lethal habitat for Trout Lake, as shown for the 2010 Trout Lake baseline model (solid lines) and the RCP8.5, 2075-2099 scenario (Lake County, Waukegan, Ill.; dashed lines).

For both Elk and Trout Lakes, lethal habitat expands by the end of the 21st century although the expansion seemed to be a much larger problem for Elk Lake. Trout Lake, instead of showing more lethal habitat volume, had an erosion of good-growth habitat. Since Trout Lake is further north, the water temperatures do not reach as high of temperatures as Elk Lake so Trout Lake is less vulnerable to Northern Cisco loss.

References for climate change scenarios:

Blois, J.L., Williams, J.W., Fitzpatrick, M.C., Jackson, S.T., and Ferrier, S., 2013, Space can substitute for time in predicting climate-change effects on biodiversity: Proceedings of the National Academy of Sciences, v. 110, p. 9374–9379.

Intergovernmental Panel on Climate Change (IPCC), 2001, Climate Change 2001: Synthesis Report, A Contribution of Working Groups I, II, and III to the Third Assessment Report of the Intergovernmental Panel on Climate Change, Watson, R.T. and the Core Writing Team, eds., Cambridge University Press, Cambridge, United Kingdom, and New York, NY, USA, 398 p.

Intergovernmental Panel on Climate Change (IPCC), 2007, Climate Change 2007: Synthesis Report, A Contribution of Working Groups I, II, and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Pachauri, R.K., Reisinger, A., and the Core Writing Team, eds., IPCC, Geneva, Switzerland, 104 p.

Intergovernmental Panel on Climate Change (IPCC), 2014, Climate Change 2014: Synthesis Report, A Contribution of Working Groups I, II, and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Pachauri, R.K., Meyer, L.A., and the Core Writing Team, eds., IPCC, Geneva, Switzerland, 151 p.

National Climatic Data Center, 2016, Climate data online: accessed May 5, 2016, at https://www.ncdc.noaa.gov/cdo-web/.

Remund, J., Müller, S., Kunz, S., Huguenin-Landl, B., Studer, C., Klauser, D., Schilter, C., and Lehnherr, R., 2012, Meteonorm , global meteorological database, version 7: Meteotest, Bern, Switzerland, 79 p

Smith, E.A., Kiesling, R.L., Galloway, J.M., and Ziegeweid, J.R., 2014, Water quality and algal community dynamics of three Sentinel deepwater lakes in Minnesota utilizing CE-QUAL-W2 models: U.S. Geological Survey Scientific Investigations Report, no. 2014-5066, 73 p.

United States Geological Survey (USGS), 2016, National Climate Change Viewer: accessed June 22, 2016, at https://www2.usgs.gov/climate_landuse/clu_rd/nccv.asp.

Whitlock, Cathy, Bartlein, P.J., and Watts, W.A., 1993, Vegetation history of Elk Lake, in Bradbury, J.P., and Dean, W.E., eds., Elk Lake, Minnesota—Evidence for rapid climate change in the north-central United States: Boulder, Colorado, Geological Society of America Special Paper 276, p. 251–274.

V. DISSEMINATION:

Description:

In addition to the scheduled status updates, and Phase 1 final report due to LCCMR, we currently provide or envision:

1. An updated description of the overall long-term lake monitoring program will be available on MN DNR's public website at (http://www.dnr.state.mn.us/fisheries/slice/index.html). Basic "fact-sheets and retrospective lake assessment reports on all 24 sentinel lakes are available on MN PCA's public website at (http://www.pca.state.mn.us/water/sentinel-lakes.html).
2. Data visualization and data download tools for variables measured as part of the long-term lake monitoring program will be available on the MN DNR's public website (forthcoming, and to be updated yearly once tools are developed). Currently, status and trends graphs for several indicators measured 2008-2011 are available at (http://www.dnr.state.mn.us/slice/indicator-graphs.html). Graphs and data accessible for download will be updated as data become available, typically within a few months after the field season.

3. Several manuscripts submitted to peer-reviewed journals by project coordinators, long-term monitoring biologists, Fisheries Research Unit staff, and other project partners (PCA, USGS, and university partners to name just three). Due to length of time required to process and analyze samples, and the time frames required for true long-term patterns to emerge and be detected, some manuscripts for peer-review journals will necessarily need to be developed after June 2016.
4. Several technical presentations given at state, regional, national, and potentially international symposia. Local outlets include MN chapter of the American Fisheries Society and organized lake groups.
5. Water quality data will be housed in EPA's national water quality database STORET (http://www.epa.gov/storet/dbtop.html). GIS data is available at (http://deli.dnr.state.mn.us/). Fish, zooplankton, and aquatic plant data will be housed in central databases and be made available upon request.
6. Multiple publications and online database portals will provide open access to all data collected by the USGS for the Sentinel Lakes Program (organized by topic below):
    a. Discrete water quality data in manuscript and database format will be accessible through an interactive web portal at: http://wdr.water.usgs.gov/adrgmap/
    b. Water quality data, water level data, and discharge data will be in the National Water Information System, or NWIS. NWIS data will be available for download at: http://waterdata.usgs.gov/mn/nwis/qw
    c. Interactive, web-based data portal for all continuous temperature data will be available at: http://mn.water.usgs.gov/projects/sentinel_lakes/map.html
7. Sentinel lakes data will be housed on a shared network drive that will be available to all internal DNR staff throughout the state.
8. A "file transfer protocol" (ftp) site will also be maintained by the project coordinators which will house all GIS layers, reports, analyses, and raw data relevant to the project. This information will be available to any interested parties.
9. A data sharing philosophy that encourages free access to comprehensive high quality data by outside researchers. The program and the state benefit greatly from analyses performed by outside researchers on raw datasets. These partnerships may bring in additional matching grants from outside funding sources.

Status as of 10/30/2013:

- Will be updating the MN DNR's SLICE website this winter – Project Coordinator, Jeff Reed, and Long-term Monitoring Biologist, Matt Hennen will be trained on the DNR process and guidelines for updating the website.
- SLICE was featured as a case study on the Climate Adaptation Knowledge Exchange website on July 24, 2013. Citation: Hitt, J. L. (2012). *Sustaining Lakes in a Changing Environment (SLICE): A Long-term Monitoring and Evaluation Program* [Case study on a project of the Minnesota Department of Natural Resources]. Product of EcoAdapt's State of Adaptation Program. Retrieved from CAKE: http://www.cakex.org/case-studies/sustaining-lakes-changing-environment-slice-long-term-monitoring-and-evaluation-program (accessed Oct. 23, 2013)
- We were contacted by Ray Newman, University of Minnesota, about obtaining SLICE plant data for a meta-analysis in support of research proposal and funding for a project focusing on curly-leaf pondweed and Eurasian watermilfoil control thru the Minnesota Aquatic Invasive Species Research Center.
- 2013 water quality data is being entered in STORET. Fish data collected in 2013 is being entered into DNR's lake survey database. Zooplankton that were collected this summer are being identified to taxonomic groups, counted, and entered into the EWR zooplankton database. We will be assembling these data sets and all other 2013 data into excel spreadsheets for subsequent statistical summarization.
- Project coordinators have given updates on the program through presentations to MDNR Fisheries research staff, MDNR Area Supervisors, and Regional Fisheries Staff. Long-term monitoring biologists are

conducting outreach activities with lake associations as well as nurturing cooperative sampling protocols with fisheries field staff.

Status as of 4/15/2014:

- Project coordinators gave updates at the Section of Fisheries Academy in February.
- LTM biologists presented a poster on Yellow Perch and Bluegill aging at the Minnesota Chapter of the American Fisheries Society meeting in Mankato in March.
- Jeff Reed presented an overview of long-term monitoring at Alexandria Technical and Community College's "Senior College" in March.
- Website improvements are still ongoing as are discussions with DNR's Parks and Trails to develop informational kiosks at access points on sentinel lakes located within Minnesota State Parks.

Status as of 10/30/2014:

- Website improvements are pending internal coordination with internal DNR media consultants.
- Richard Kiesling, USGS partner investigator, presented research results from Phase 1 at the Minnesota Water Resources Conference.
- Jeff Reed gave a program update to MDNR Area Supervisors.
- Brian Herwig and Jeff Reed were involved with conversations regarding long-term monitoring and the Sentinel Lakes Program with natural resource professionals from the Red Lake Nation.
- A USGS Scientific Investigation Report detailing the Phase 1 model calibration and validation was released to the public. The report citations and web-based digital object identifier (doi) code are:

Smith, E.A., Kiesling, R.L., Galloway, J.M., and Ziegeweid, J.R., 2014, Water quality and algal community dynamics of three deepwater lakes in Minnesota utilizing CE-QUAL-W2 models: U.S. Geological Survey Scientific Investigations Report 2014–5066, 73 p., http://dx.doi.org/10.3133/sir20145066.

Status as of 4/15/2015:

- Jeff Reed and Brian Herwig coordinated a two-day gathering in early February of past and present 'SLICE' collaborators (Sentinel Lakes Summit) in Brainerd, MN. Over 70 people attended.
- Matt Hennen and Eric Katzenmeyer presented findings of fisheries investigations related to Sentinel Lakes at the Sentinel Lakes Summit.
- Richard Kiesling, USGS partner investigator, presented research results from Phase 1 at the Sentinel Lakes Summit.
- Jacqueline Amor, GIS Analyst funded by LCCMR, presented results of her work with LiDAR in the watersheds of Sentinel Lakes at the Sentinel Lakes Summit, MN TWS Annual Meeting, and MN AFS Annual Meeting.
- Dr. Kyle Zimmer, University of St. Thomas, presented a summary of stable isotope work at the Sentinel Lakes Summit.
- Jeff Reed presented an overview of long-term monitoring to the Forada Lions Club and the Alexandria Chain of Lakes Association.
- Sentinel Lakes was highlighted in the DNR's 2015 Conservation Agenda.
- Jeff Reed and Brian Herwig met with DNR Parks and Trails naturalists to develop outreach materials that will highlight ongoing work in Elk Lake which is located within Itasca State Park.

Status as of 10/30/2015:

- Jeff Reed presented an overview of the Sentinel Lakes program and long-term monitoring efforts to Section of Fisheries leadership in Grand Rapids, MN in June.

- Brian Herwig and Jeff Reed co-authored the presentation "Sentinel Lakes: A Review of the Past and Look Toward the Future" with Steve Heiskary (MPCA Partner) at the Minnesota Water Conference, St. Paul, MN on October 13th.
- Biologists Eric Katzenmeyer and Matt Hennen were featured on a StarTribune video story on climate change and White Suckers in the May 2nd edition.

Status as of 4/15/2016:

- Collaborators Steve Heiskary (MPCA), Richard Keisling (USGS), and Eric Smith (USGS) presented lectures featuring Sentinel Lakes research to the Macalester College's Limnogeology course in February.
- Biologist Matt Hennen presented the results of his research on Bluegill catchability and growth assessment at the American Fisheries Society Meeting (MN Chapter) in Duluth, February 2016.
- Jeff Reed presented research directions on Yellow Perch at the Section of Fisheries Academy (February 2016), highlighting some of the efforts of Sentinel Lakes Program staff regarding this species. Additionally, Brian Herwig, Matt Hennen, and Eric Katzenmeyer presented brief overviews of their work on Sentinel Lakes to Section staff.
- Eric Katzenmeyer and Brian Herwig have been working with SW Regional Staff, providing data regarding the current status of Lake Shaokotan, for use in public meetings.
- Two poster presentations from University of St. Thomas were included at the American Fisheries Society Meeting (MN Chapter) in Duluth, February 2016:
  - Margaret Thompson, Ryan Trapp, Zach George, Kyle Zimmer, and Brian Herwig, *Stable isotopes indicate zebra mussels (Dresseina polymorpha) increase dependence of lake food webs on littoral energy sources*
  - Ryan Grow, Angela Tipp, Nicole Graziano, Kyle Zimmer, Brian Herwig, Pete Jacobson, *Stable isotopes indicate cisco diets shift from shallow to deep-water prey as they increase in body size*
- Data collected by Sentinel Lakes staff were provided to several university students for class projects including the following:
  - *Bioenergetics of cisco in Elk Lake under predicted future climate change and its subsequent effects on the ecosystem* by William Reed, Andrew Day, and Praseuth Yang for the University of Minnesota course on Ecology.
  - *Effects of temperature and dissolved oxygen on cisco populations* by Maya Grantier, Eric Kittilsby, and Michael Murphy for the Macalester College course on Limnogeology.
- Drs Richard Kiesling and Eric Smith, USGS, were keynote speakers at the International Rainy-Lake of the Woods Watershed Forum and presented information regarding the effects of climate change on the water quality and algal communities of Minnesota's Sentinel Lakes.
- Updates to the Sentinel Lakes website (www.dnr.state.mn.us/fisheries/slice/index.htm) were made in January.

Final Report Summary:

In addition to the activities already completed we anticipate continued outreach via presentations to the scientific community and interested citizen groups. We also expect several peer-reviewed publications from major aspects of the monitoring project and partner research projects (isotopes, climate scenario lake modeling, and pupal chironomid exuvia work). We are also planning to develop a second Sentinel Lakes Summit, similar to the gathering convened in 2015, to once again bring together past and present Sentinel Lakes collaborators and interested agency staff for 1-2 days of research, monitoring and program updates (probably March 2017).

The following research was presented at the Ecological Society of American Annual Meeting in Fort Lauderdale, FL in August 2016 as posters:

- Zachary George, Meg Thompson, Ryan Trapp, Brian Herwig, and Kyle Zimmer, *Stable isotopes reveal niche separation among three piscivores in a northern mesotrophic lake.*
- Margaret Thompson, Ryan Trapp, Zach George, Kyle Zimmer, and Brian Herwig, *Stable isotopes indicate zebra mussels (Dreissena polymorpha) increase dependence of lake food webs on littoral energy sources.*

The following peer-reviewed papers were recently published by scientists with National Science Foundation funding to work on Sentinel Lakes. While LCCMR funds were not used for this research, they were instrumental in developing the network of lakes and monitoring efforts that lead to this type of collaborative research.

- Frisch, D., Morton, P.K., Culver, B.W., Edlund, M.B., Jeyasingh, P.D., Weider, L.J. 2016. Palaeogenetic records of *Daphnia pulicaria* in two North American lakes reveal the impact of cultural eutrophication. *Global Change Biology* doi: 10.1111/gcb.13445

- Muñoz, J., A. Chaturvedi, L. De Meester, and L. J. Weider. 2016. Characterization of genome- wide SNPs for the water flea *Daphnia pulicaria* generated by genotyping-by-sequencing (GBS). Scientific Reports **6**:28569 | DOI: 10.1038/srep28569

VI. PROJECT BUDGET SUMMARY:

A. ENRTF Budget:

Budget Category	$ Amount	Explanation
Personnel:	$539,983	3 Long-Term Monitoring Fisheries Specialists (100% FTE) to coordinate project surveys, train and lead field crews in data collection efforts, perform database management and data QA/QC, and report on status and trends for sentinel lakes located in 4 different landtypes. 7 Student Interns (100% FTE), field data collection activities (6students) and GIS work (1 student) in support of project objectives.
Professional/Technical/Service Contracts:	$355,489	USGS Water Sciences Center to build or adapt biophysical lake ecosystem models for 6 Tier 1 lakes capable of forecasting future lake conditions based on scenarios that cause change but vary across lakes. University of St. Thomas to use carbon and nitrogen stable isotopes to track food web responses to expanding zebra mussel populations in Lake Carlos, and to characterize food web linkages in Elk Lake. University of Minnesota to develop pupal chironomid indicators of lake trophic and thermal conditions.
Direct and Necessary Services:*	$61,614	Direct and Necessary Services for the Appropriation
Equipment/Tools/Supplies:	$126,754	Water level gauges, remote continuously-recording temperature and dissolved oxygen

		sensors, fish and invertebrate sampling equipment, field wear and safety equipment, other miscellaneous survey equipment, and analytical services in support of long-term monitoring objectives outlined in the proposal.
Travel Expenses in MN:	$116,160	In support of project objectives, with approximate breakdown as follows: 40% for fleet for travel to study lakes to install sensors and conduct survey work, and to attend coordination meetings; 40% for hotels for overnight stays associated with lake survey work and project coordination, 20% for meal reimbursement in accordance to DNR travel guidelines, and meal reimbursement limits.
TOTAL ENRTF BUDGET:	**$1,200,000**	

* Direct and Necessary expenses include both Department Support Services (Human Resources, IT, Financial Management, Communications, Procurement, and Facilities) and Division Support Services. Department Support Services are described in agency Service Level Agreements, and billed internally to divisions based on indices that have been developed for each area of service. Department leadership (Commissioner's Office and Regional Directors) are not assessed. Division Support Services include costs associated with Division, business office and clerical support. Those elements of individual projects that put little or no demand on support services such as large single-source contracts, large land acquisitions, and funds that are passed-thru to other entities are not assessed Direct and Necessary costs for those activities. For this work plan, Professional/Technical/Service Contracts to USGS Water Sciences Center, University of St. Thomas, and University of Minnesota with an associated cost of $355,489 has not been assessed Direct and Necessary costs.

Explanation of Use of Classified Staff: N/A

Explanation of Capital Expenditures Greater Than $3,500: N/A

Number of Full-time Equivalent (FTE) funded with this ENRTF appropriation: 13.5

Number of Full-time Equivalent (FTE) estimated to be funded through contracts with this ENRTF appropriation: 4.0

B. Other Funds:

Source of Funds	**$ Amount Proposed**	**$ Amount Spent**	**Use of Other Funds**
Non-state			
USGS Water Science Center	$140,000	$	In-kind matching funds in support of proposal objectives
University of St. Thomas, Department of Biology	$6,900	$	In-kind match funds in support of stable isotope study of L. Carlos and Elk L. food webs (undergraduate assistant support, travel to study sites, and lab supplies)
State			
DNR Div. of Fish and Wildlife	$171,000	$	In-kind match funding for approximately 11 DNR staff ($130,000) involved in field

			data collection and data analysis efforts.
DNR Div. of EWR	$83,750	$	In-kind match funding to support bi-monthly zooplankton and monthly (in some cases annual only) benthic invertebrate sampling and sample processing costs. In-kind matching funds to support stream gauging and the subsequent watershed modeling efforts of Jim Solstad and others.
MPCA – Env. Anlys. & Outcomes	$74,200	$	In-kind matching funds to support water quality sampling and analytical costs. In-kind labor provided by citizen lake monitoring volunteers (not quantified).
TOTAL OTHER FUNDS:	**$475,850**	**$**	

VII. PROJECT STRATEGY:

A. Project Partners:

1. DNR Division of Fish and Wildlife Section of Fisheries – Program administration, Fisheries technical and field support, data management, ($492,456 ENRTF + in-kind; Dr. Don Pereira Project Manager).
2. USGS Water Science Center – Hydrologic and limnologic technical and field support ($207,544 ENRTF + in-kind; Dr. Richard Kiesling PI).
3. University of St. Thomas – Stable isotope analysis of Carlos and Elk lake food webs: technical, lab, and field support ($55,312 ENRTF + in-kind, Dr. Kyle Zimmer PI)
4. University of Minnesota – Analysis of chironomid pupal exuvia as indicators of lake trophic and thermal conditions: technical, lab, and field support ($82,633 ENRTF, Dr. Leonard Ferrington Jr. PI)

Partners providing support but not receiving funds from the ENRTF:

5. DNR Divisions of Parks and Trails, Ecological and Water Resources – Survey support on Bear Head Lake (Parks), lake level gauging, watershed modeling, invertebrate data (DNR Div. EWR, in-kind).
6. MPCA – Environmental Analysis and Outcomes Division – Water quality assessments, ground-water monitoring, volunteer coordination (in-kind).

B. Project Impact and Long-term Strategy:

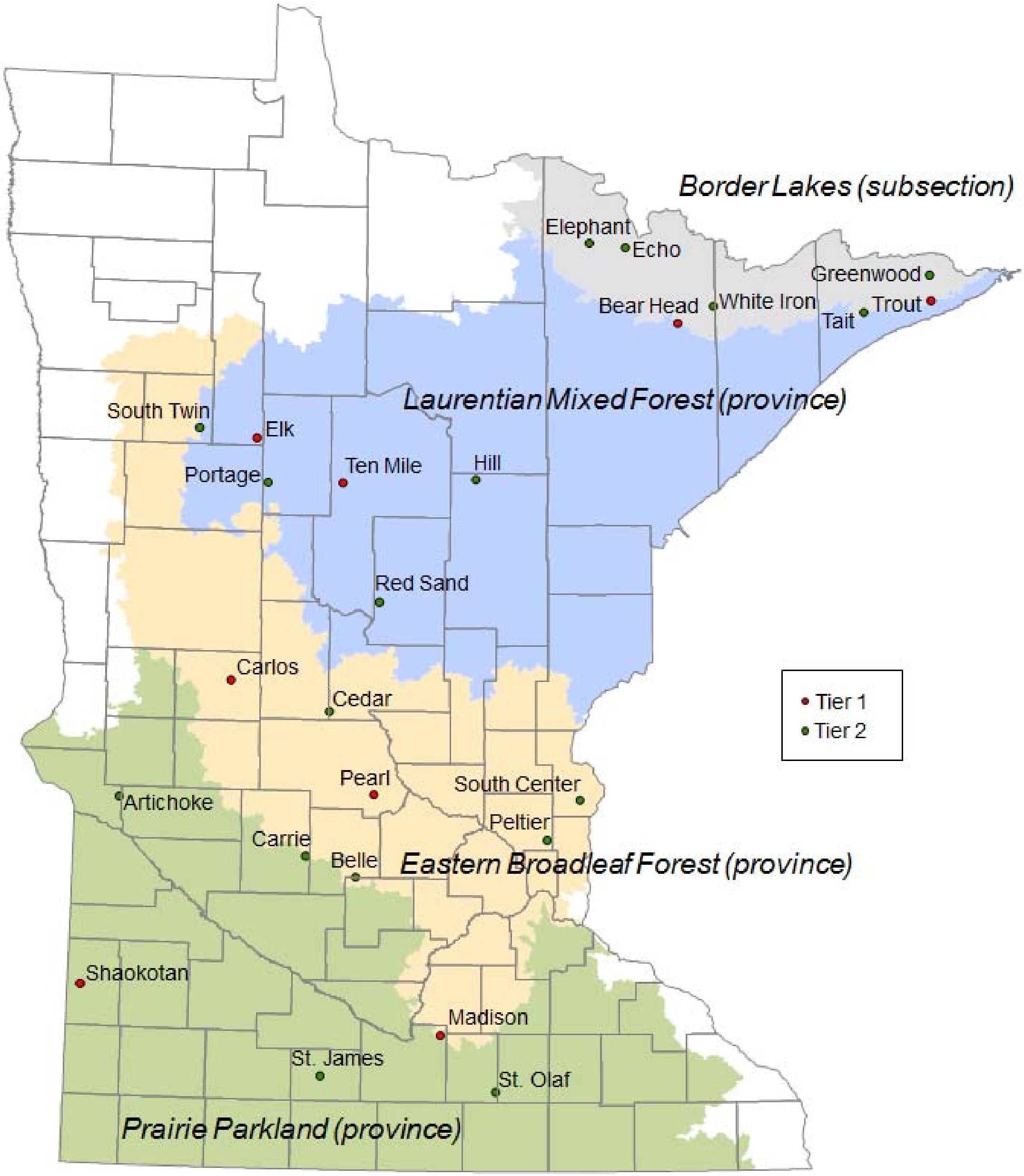
Healthy lakes are an important component of our Minnesota identity. While losses to lake health have already undoubtedly occurred in many areas, numerous high-quality lakes still exist throughout the state, yet all lakes remain vulnerable to a myriad of threats (excess nutrients from land use and human populations, climate changes, and invasive species, to name a few). Lakes are especially vulnerable as not only are lakes the collectors of waters moving across our landscape (thus strongly reflect human modifications within watersheds), but are also sensitive integrators of climatic conditions. For these reasons, timing is urgent for effective monitoring and protection tools not only to prevent further, possibly irreversible damage, but also to document lake improvements to our actions. Foremost, we hope to offer lake managers, conservation planners, lakeshore residents, fishers, other lake users, and the Minnesota public a better understanding of historical, present-day, and future factors influencing lake conditions. Our monitoring and modeling efforts will not only help reveal the cause-effect mechanisms affecting lake status, but will also help lead us to the most appropriate indicators to track the status of our state's lakes. Detailed assessment and modeling of lake conditions will inform revisions to lake monitoring programs, provide an empirical foundation for understanding impacts of state's varied land uses and watershed restoration programs, and inform climate change adaptation policies related to lake management. Understanding the myriad of factors driving changes to lake habitats is one our main project goals, and critical to the societal, economic, and ecological well-being of our state.

Our long-term strategy is to complete development of a fully integrated lake monitoring program that combines and focuses the activities of key, collaborative management agencies (e.g. DNR and MPCA). Such a system will greatly increase our understanding of how these lakes change, and what management actions are most likely to provide cleaner water and healthier fish populations. After funding Phase 2, our hope will be to fully incorporate this program in to the regular activities of both agencies (thus move from concept development to operational), and thus cover future work with regular agency funding sources.

C. Spending History:

Funding Source	**M.L. 2007 or FY08**	**M.L. 2008 or FY09**	**M.L. 2009 or FY10**	**M.L. 2010 or FY11**	**M.L. 2011 or FY12-13**
ENRTF			$825,000		
In-kind support originating from the Game and Fish Fund, USGS cooperative funds, US Forest Service operating budgets, PCA operating budgets and Clean Water Legacy. FY12-13 includes 50% salary for project coordinators, Jeff Reed and Brian Herwig, to finish the SLICE Phase 1 project and to design and coordinate the implementation of the SLICE Phase 2 project.			$169,000	$169,000	$243,000
DJ Study 605 - Designing a long-term monitoring program to track the status of fish communities and their habitats in Minnesota lakes, identify efficient indicators, and evaluate mechanisms.		$157,775	$174,585	$196,855	$66,658

VIII. ACQUISITION/RESTORATION LIST: N/A

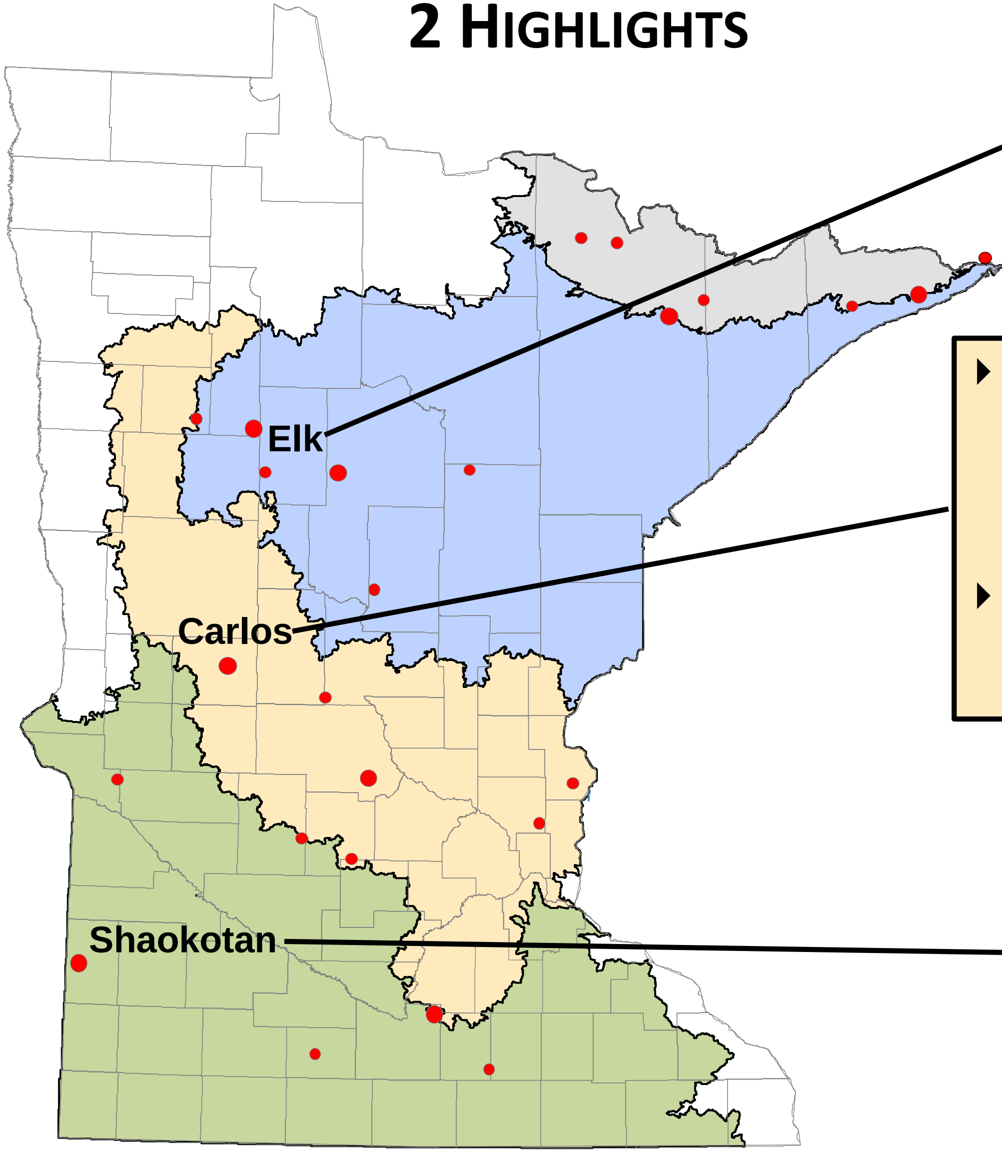


IX. MAP(S): Map showing the location of all 25 sentinel lakes in Minnesota, and the four major land types that they cover. Proposed tiered classification of lakes is also indicated. Tier 1 lakes (red) are the focus of work presented in this proposal and will receive annual sampling in 2013-16 (and beyond). Tier 2 lakes (green) will be sampled at a reduced frequency of approximately one in every 4-5 years.

X. RESEARCH ADDENDUM: N/A

XI. REPORTING REQUIREMENTS: Periodic work plan status update reports will be submitted not later than 10/30/2013, 4/15/2014, 10/30/2014, 4/15/2015, 10/30/2015, and 4/15/2016. A final report and associated products will be submitted between June 30 and August 15, 2016 as requested by the LCCMR.

Sentinel Lakes – 25 Laboratories for Understanding Lake Changes: Phase 2 Highlights



- Evaluation of new biological indicators of lake health (pupal skins of aquatic flies)
- Physical lake models to understand nutrient, temperature and oxygen dynamics were used to simulate impacts of future climate conditions

- Food web research conducted to understand impacts of zebra mussels (ZM) finds that lake biota (insects and fish) shifted to alternative food sources
- Long-term water quality data collection reveals water clarity increases and zooplankton changes due to ZM

- Long-term and baseline water quality and aquatic plant surveys detect a major lake shift from a turbid, algae-dominated state to a clear water, aquatic plant-dominated condition due to watershed restoration and BMP implementation
- Predictive lake models were developed for Pearl and Madison lakes (the Shaokotan Lake model is in development)

Attachment A: Budget Detail for M.L. 2013 Environment and Natural Resources Trust Fund Projects

Project Title: Sustaining Lakes in a Changing Environment (SLICE): Phase 2

Legal Citation: M.L. 2013, Chp. 52, Sec. 2, Subd. 05a

Project Manager: Jeffrey R. Reed

M.L. 2013 ENRTF Appropriation: $1,200,000

Project Length and Completion Date: June 30, 2016

Date of Update: 06/30/2016

ENVIRONMENT AND NATURAL RESOURCES TRUST FUND BUDGET	Activity 1 Budget	Revised Activity 1 Budget 04/25/2016	Amount Spent 06/30/2016	Balance 06/30/2016	Activity 2 Budget	Amount Spent 06/30/2016	Balance 06/30/2016	TOTAL BUDGET	TOTAL BALANCE
BUDGET ITEM	*Monitoring sentinel lakes to gage status and trends of lake health*				*Build or adapt biophysical lake ecosystem models and forecast future lake conditions*				
Personnel (Wages and Benefits) - Overall	565,743	565,743	565,743	0				565,743	0
3 Long-Term Monitoring Fisheries Specialists (8L): 77% salary, 23% benefits; 100% FTE (30 mos: Jul 1, 2013 - Dec 30, 2015) (Est. $410,623)	463,133	463,133	463,133	-					
7 Student Interns: 100% salary; 100% FTE (14 weeks in FY14, 14 weeks in FY15, 14 weeks in FY16) (Est. $129,360)	102,610	102,610	102,610	-					
Professional/Technical/Service Contracts									
USGS Water Sciences Center - Dr. Richard Kiesling (PI) • Lake model constuction (three 0.25 FTE Hydrologists for three years; $183,644) • Supplies ($7,800) • Travel ($3,000) • Water sample shipping ($600) • Analytical lab costs ($13,100)					207,544	207,544	0	207,544	0
Univ of St. Thomas, Dept. of Biology - Dr. Kyle Zimmer (PI) • Collect, process, and analyze stable isotope samples to examine food web reponses to expanding zebra mussel populations in Lake Carlos and to obtain baseline characterization of the Elk Lake food web - UST intern salary (Est. $11,299) - Equipment and stable isotopes analytical costs (Est. $41,003) -1.5 mos summer salary for K. Zimmer (Est. $13,010) -Travel Expenses (Est. $3,675)	68,987	68,987	67,562	1,425				68,987	1,425

Iowa State Univ, Ag & Biosystems Engr. Dept. – Dr. Brian Gelder (PI) • Apply automated process for hydro-modification of the LiDAR-derived DEM for the Shaokotan, Madison and Pearl lake watersheds – resulting DEM will be used in future watershed modeling efforts (estimating sediment & nutrient loads, hydrological modeling) - Salary and benefits ($3,000)	3,000	3,000	3,000	0				3,000	0
UMN, Department of Entomology - Dr. Len Ferrington (PI) • Sample chironomid pupal exuvia communities from 8 Tier 1 sentinel lakes, explore relationships within and among lakes, and develop indicators that reflect lake trophic and thermal conditions - Graduate Research Assistant (M.S.) - 2 yrs (Est. $76,137) - Travel Expenses (Est. $6,496)	82,633	82,633	82,633	0				82,633	0
Direct and Necessary Services for the Appropriation	61,614	61,614	61,614	0				61,614	0
Equipment/Tools/Supplies (estimates, actual may vary slightly)									
Transducers for water level gauges	7,170	7,170	7,170	0				10,054	0
Temperature thermistors and buoys	16,365	16,365	16,365	0				16,400	0
Dissolved oxygen sensors (8 @ $1250/ea) and buoys ($1000)	526	526	0	526				4,000	526
Lowrance HDS7 depth finder with card and sidescan (1 @ $1,700/ea)	1,700	1,700	1,700	0				1,700	0
Contour Innovations BioBase online subscription to aquatic bathymetric and vegetation mapping software for 8 Tier 1 lakes for 3 yrs	2,200	2,200	2,200	0				4,000	0
Factory calibration and new calibration sphere for Biosonics DTx (or Biosonics DTE) and 420 kHz transducer	0	0		0				2,000	0
2 sets of vertical gill nets ($3,500/set)	7,000	7,000	7,000	0				7,000	0
Supplies for invertebrate sampling (e.g., sample jars and preservative ethanol)	4,000	4,000	4,000	0				4,000	0
Waders, rain gear, PPE, etc.	2,400	2,400	2,400	0				2,400	0
Miscellaneous survey equipment and repairs	20,000	20,000	20,000	0				20,000	0
Phytoplankton taxonomy (MN Rapid Assessment Method)	23,040	23,040	22,776	264				23,040	264
Analytical services (water chemistry $29,160)	29,160	29,160	29,160	0				29,160	0
Travel expenses in Minnesota For DNR field staff to conduct regular bi-monthly sampling to all study lakes, and specialized seasonal sampling at study lakes, and to attend coordination meetings (hotels, fleet costs, meals)	96,918	96,918	96,586	332				112,485	332
COLUMN TOTAL	**$992,456**	**$992,456**	**$989,909**	**$ 2,547**	**$207,544**	**$207,544**	**$0**	**$1,200,000**	**$2,547**

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Water-Quality Models to Assess Algal Community Dynamics, Water Quality, and Fish Habitat Suitability for Two Agricultural Land-Use Dominated Lakes in Minnesota, 2014

Scientific Investigations Report 2017–5056

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

Front cover. Pearl Lake north shoreline at the public boat access, looking south across Pearl Lake, Minnesota, July 2015. Photograph by Les Warren, St. Cloud State University.

Back cover. Pearl Lake, Minnesota, May 2016. Photograph by Les Warren, St. Cloud State University.

Water-Quality Models to Assess Algal Community Dynamics, Water Quality, and Fish Habitat Suitability for Two Agricultural Land-Use Dominated Lakes in Minnesota, 2014

By Erik A. Smith, Richard L. Kiesling, and Jeffrey R. Ziegeweid

Prepared in cooperation with the Minnesota Department of Natural Resources

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

U.S. Department of the Interior
RYAN K. ZINKE, Secretary

U.S. Geological Survey
William H. Werkheiser, Acting Director

U.S. Geological Survey, Reston, Virginia: 2017

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Acknowledgments

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Laura Hubbard and Norman Buccola of the U.S. Geological Survey are greatly acknowledged for technical reviews of the report.

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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

$$°F = (1.8 \times °C) + 32.$$

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
GIS	geographic information system
HAB	harmful algal bloom
ID	identification number
lidar	light detection and ranging
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
SLICE	Sustaining Lakes in a Changing Environment
SOD	sediment oxygen demand
USACE	U.S. Army Corps of Engineers
USAF	U.S. Air Force
USGS	U.S. Geological Survey
V4.0	version 4.0
WSC	wind sheltering coefficient

Water-Quality Models to Assess Algal Community Dynamics, Water Quality, and Fish Habitat Suitability for Two Agricultural Land-Use Dominated Lakes in Minnesota, 2014

By Erik A. Smith, Richard L. Kiesling, and Jeffrey R. Ziegeweid

Abstract

Fish habitat can degrade in many lakes due to summer blue-green algal blooms. Predictive models are needed to better manage and mitigate loss of fish habitat due to these changes. The U.S. Geological Survey (USGS), in cooperation with the Minnesota Department of Natural Resources, developed predictive water-quality models for two agricultural land-use dominated lakes in Minnesota—Madison Lake and Pearl Lake, which are part of Minnesota's sentinel lakes monitoring program—to assess algal community dynamics, water quality, and fish habitat suitability of these two lakes under recent (2014) meteorological conditions. The interaction of basin processes to these two lakes, through the delivery of nutrient loads, were simulated using CE-QUAL-W2, a carbon-based, laterally averaged, two-dimensional water-quality model that predicts distribution of temperature and oxygen from interactions between nutrient cycling, primary production, and trophic dynamics.

The CE-QUAL-W2 models successfully predicted water temperature and dissolved oxygen on the basis of the two metrics of mean absolute error and root mean square error. For Madison Lake, the mean absolute error and root mean square error were 0.53 and 0.68 degree Celsius, respectively, for the vertical temperature profile comparisons; for Pearl Lake, the mean absolute error and root mean square error were 0.71 and 0.95 degree Celsius, respectively, for the vertical temperature profile comparisons. Temperature and dissolved oxygen were key metrics for calibration targets. These calibrated lake models also simulated algal community dynamics and water quality. The model simulations presented potential explanations for persistently large total phosphorus concentrations in Madison Lake, key differences in nutrient concentrations between these lakes, and summer blue-green algal bloom persistence.

Fish habitat suitability simulations for cool-water and warm-water fish indicated that, in general, both lakes contained a large proportion of good-growth habitat and a sustained period of optimal growth habitat in the summer, without any periods of lethal oxythermal habitat. For Madison and Pearl Lakes, examples of important cool-water fish, particularly game fish, include northern pike (*Esox lucius*), walleye (*Sander vitreus*), and black crappie (*Pomoxis nigromaculatus*); examples of important warm-water fish include bluegill (*Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), and smallmouth bass (*Micropterus dolomieu*). Sensitivity analyses were completed to understand lake response effects through the use of controlled departures on certain calibrated model parameters and input nutrient loads. These sensitivity analyses also operated as land-use change scenarios because alterations in agricultural practices, for example, could potentially increase or decrease nutrient loads.

Introduction

The ecology and water quality of small lakes with a large proportion of agricultural land use in the drainage area are often controlled by nutrient dynamics (Sharpley and others, 1987; Daniel and others, 1998; Bennett and others, 2001). The large input of nitrogen and phosphorus into the receiving water body dominates the growth dynamics of phytoplankton and macrophyte communities (Xu and others, 2010; Paerl and Paul, 2012). Lakes with long lake residence times can become established as eutrophic, or even hypereutrophic, as a stable state because of the internal recycling of nutrients from sediments (Lerman, 1974). Small, shallow lakes in particular are vulnerable because of the large sediment contact area relative to lake volume. These shallow lakes, and even deeper lakes with large nutrient loading, are susceptible to habitat degradation from enhanced growth of algal blooms (Schindler, 2006).

In Minnesota, lakes are facing substantial risks from land-use change and climate change (Tong and Chen, 2002). Although improved management practices are being implemented on agricultural land, increased economic pressures towards high intensity row-crop agriculture challenges the paradigm of improving water quality (Harding and others, 1999; Dumanski and others, 2006). Lakes from across the State are threatened by current (2017) and legacy nutrients, so it is imperative to gain an understanding of lake responses to increased and decreased fluxes of nutrients. In recent years, water-resource scientists have been making the case for focused assessments and monitoring of “sentinel” systems (Jassby, 1998; Magner and Brooks, 2008, Williamson and others, 2008), which are more closely monitored and studied to assess how these stressors affect lakes long term. Lakes and their contributing drainage basins are complex, and development of a mechanistic understanding of the linkage between basin-based stressors and lake metabolism is best accomplished by taking a long-term, adaptive approach towards water-resource management (Magnuson and others, 1990). Intensive, detailed study of representative systems is critical to understanding cause and effect mechanisms, but an equally important need is to compare this detailed information to a broader set of similar systems. For the Minnesota Department of Natural Resources (MNDNR) Sustaining Lakes in a Changing Environment (SLICE) research program, these study design requirements are being met by coupling intensive, predictive modeling of a subset of “super sentinel” lakes with 24 Minnesota sentinel lakes distributed in a split-panel design of environmental monitoring that includes basic basin, water-quality, habitat, and fish indicators across a gradient of ecoregions, depths, and nutrient levels (McDonald, 2003).

The ability to simulate the effects of large-scale stressors (for example, basin land-use alterations or decadal climate changes) on lake ecosystems is a critical component of a proactive management plan for Minnesota lakes. Several regional and statewide lake modeling studies have illustrated the potential linkages between climate change, lake morphology, and reductions in fish habitat in the form of temperature and dissolved oxygen (DO) distributions for Minnesota and the north-central United States (for example, see summaries in Stefan and others, 1995, 1996; De Stasio and others, 1996; Fang and others, 1999, 2010; Jacobson and others, 2008; Jiang and others, 2012). These models have documented the relative importance of lake-basin geometry, ice-free season, thermal stratification, DO stratification, and wind-driven mixing to the development of sustainable fish habitat in deepwater lakes of the region; however, the potential trophic-dynamic response to simultaneous changes in land use and climate is less understood, as is the response of specific lakes to these historical and hypothetical changes. Questions also remain as to how the complex food webs that support fish guilds within these modeled systems will respond to the predicted physical changes in fish habitat (De Stasio and others, 1996).

The U.S. Geological Survey (USGS), in cooperation with the Minnesota Department of Natural Resources, developed predictive water-quality models for two agricultural land-use dominated lakes in Minnesota—Madison Lake and Pearl Lake, which are part of Minnesota’s sentinel lakes monitoring program—to assess algal community dynamics, water quality, and fish habitat suitability of these two lakes (fig. 1; table 1) under recent (2014) meteorological conditions. The two selected lakes, Madison Lake and Pearl Lake, have abundant cool-water and warm-water fish communities but are located within active agricultural drainage basins. Both lakes have frequent summer blue-green algal blooms, as supported by the algal count data (PhycoTech, 2017). The chosen modeling framework for this study, CE-QUAL-W2 (Cole and Wells, 2015), is a two-dimensional, laterally averaged, hydrodynamic and water-quality model originally developed by the U.S. Army Corps of Engineers (USACE) and currently supported by Portland State University (Cole and Wells, 2015). The CE-QUAL-W2 model addresses the interaction between nutrient cycling, primary production, and trophic dynamics to predict responses in the distribution of temperature and oxygen in lakes, which was a primary goal of this study.

Purpose and Scope

The purpose of this report is to document the development of predictive models to assess algal community dynamics, water quality, and fish habitat suitability of two selected lakes (Madison Lake and Pearl Lake) in Minnesota under recent (2014) meteorological conditions. Both lakes are classified as supporting large cool-water and warm-water fish communities. The water-quality models were calibrated using data collected from April 2014 through November 2014. A sensitivity analysis was done to better understand model response to some of its most important parameters, including the wind sheltering coefficient (WSC), sediment release rates of phosphorus, sediment oxygen demand (SOD), and the extinction coefficients. The sensitivity analysis was also used as a surrogate for potential basin modifications, potentially due to changes in management practices, for constituents such as inflowing phosphorus, nitrogen, and organic matter.

Study Area

Two Minnesota lakes dominated by agricultural land use, classified as super sentinel lakes (Minnesota Department of Natural Resources, 2010), are the focus of this study: Madison Lake in Blue Earth County and Pearl Lake in Stearns County. Previous extensive characterization of both lakes and their drainage basins was done during the initial phase of SLICE (Lindon and others, 2010; Anderson and others, 2012) and was summarized into separate lake reports.

Madison Lake

Madison Lake (fig. 2) 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 located on the boundary of the Western Corn Belt Plains and the North Central Hardwood Forests ecoregions (Soulard and others, 2014). Madison Lake was carved out during the last glaciation, and the area surrounding Madison Lake is made up of thick deposits of poorly drained loam soils derived from Des Moines lobe glacial tills (Lindon and others, 2010). Madison Lake is weakly dimictic, generally starting off as

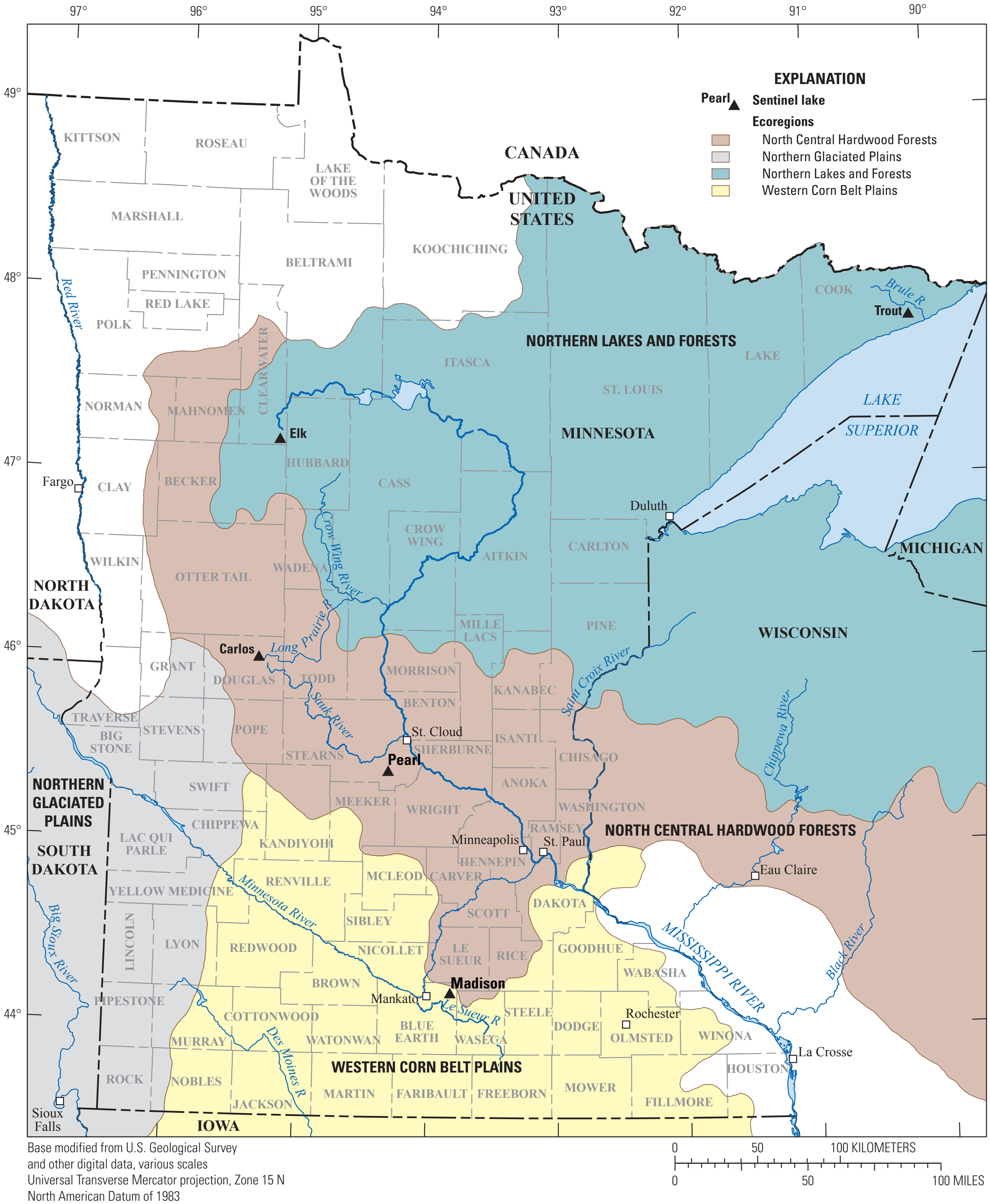


Figure 1. Major ecoregions and locations of two lakes of this study (Madison Lake and Pearl Lake) and two other sentinel lakes (Elk Lake and Trout Lake) from Smith and others (2014), Minnesota.

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 of water temperature, dissolved oxygen, and water-quality constituents in the Madison Lake and Pearl Lake studies, Minnesota.

[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
Madison Lake								
Unnamed stream to Madison Lake at CR-48 near Madison Lake, Minn.	Northeast inlet	05320130	--	44° 11′ 53.4″ -93° 46′ 39.9″	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′ 07.9″ -93° 47′ 11.2″	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″ -93° 48′ 57.1″	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″ -93° 48′ 39.7″	DO, T	DO, pH, SC, T, MI, L-L N, TC/TN, Alk, Alg	WQ (I), T (C), DO (C)	7
Madison Lake northeast deep point near Madison Lake, Minn.	Northeast deep point	--	07-0044-00-202	44° 11′ 48.8″ -93° 48′ 23.3″	T	--	T (C)	5
Pearl Lake								
Unnamed tributary to Pearl Lake, southwest corner, near Marty, Minn.	Southwest corner inlet	05270447	--	45° 23′ 33.9″ -94° 19′ 05.6″	S, S→Q, T	DO, pH, SC, T, MI, L-L N, TC/TN, Alk	I	2 (I)
Mill Creek at inlet to Pearl Lake near Marty, Minn.	Mill Creek inlet	05270448	--	45° 24′ 13.2″ -94° 19′ 07.9″	S, S→Q, T	DO, pH, SC, T, MI, L-L N, TC/TN, Alk	I	5 (I)
Mill Creek at Pearl Lake Outlet near Marty, Minn.	Pearl Lake outlet	05270449	--	45° 24′ 22.7″ -94° 18′ 36.4″	S, S→Q, T	--	S (C), I (S→Q), T (C)	6 (O)
Pearl Lake Deep Point near Marty, Minn.	Pearl Lake deep point	--	73-0037-00-202	45° 24′ 02.3″ -94° 18′ 35.1″	DO, T	DO, pH, SC, T, MI, L-L N, TC/TN, Alk, Alg	WQ (I), T (C), DO (C)	4

[1]U.S. Geological Survey, 2016a.

[2]Minnesota Department of Natural Resources, 2016e.

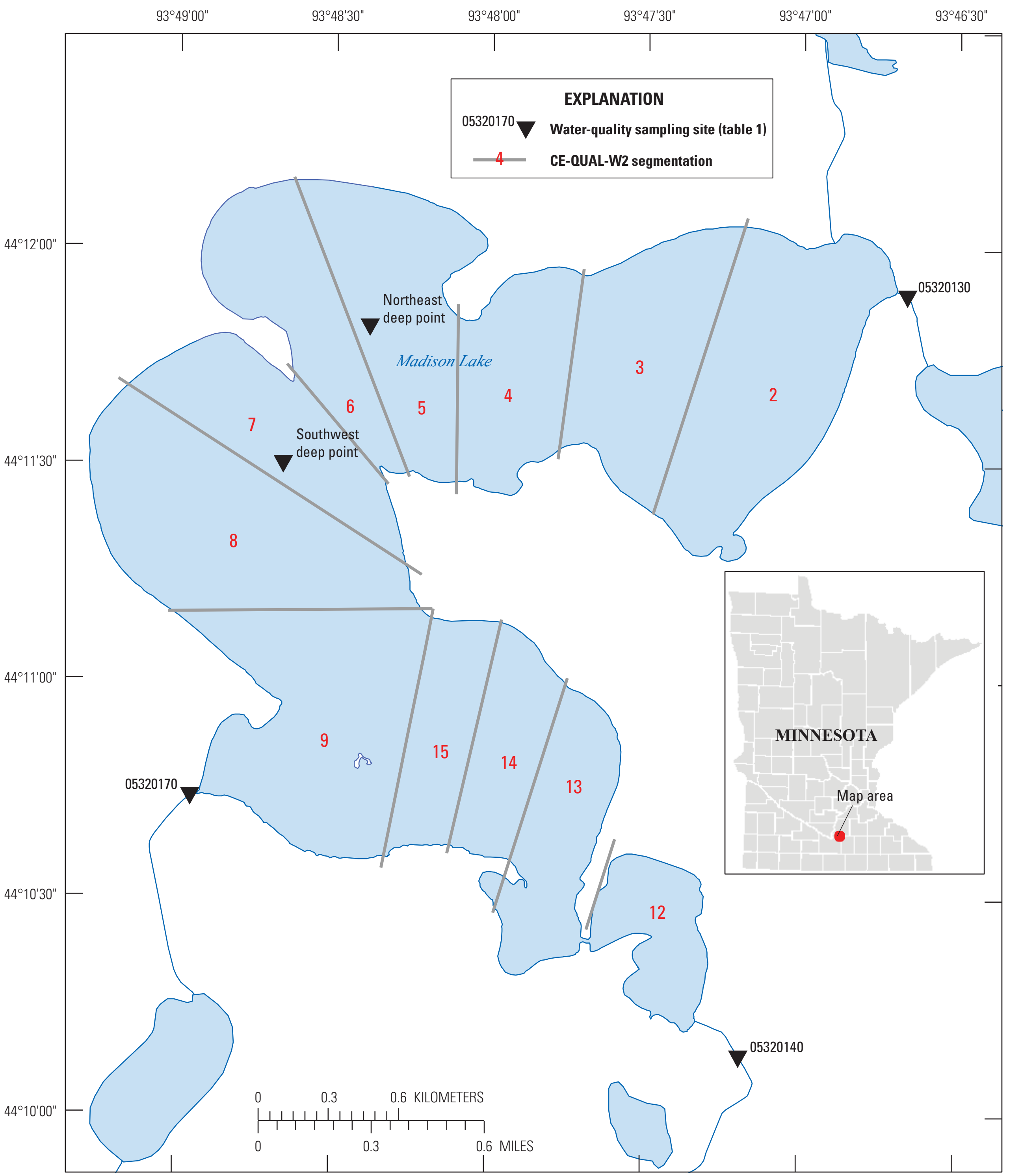
well-mixed before early summer, with a weak thermocline that develops in the summer months; the lake mixes again in the late fall (Lindon and others, 2010). Dissolved oxygen is well-mixed in the early spring (April to May) and late fall (mid-October), with a substantial portion of the hypolimnion becoming 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. Lake residence time is on the order of 3–4 years, with small surface-water inflow and outflow (Lindon and others, 2010). The mean precipitation in the region for 1981–2010 is 0.82 meter per year (m/yr) (National Centers for Environmental Information, 2016a). Climate in the region has been known to consist of sustained drought periods, which can have a substantial effect on lake level. Lake levels decreased 3 meters (m) from 1939 to 1944 after a period of sustained drought and have been stable since that time. The lowest lake level recorded on Madison Lake was 305.71 m above the National Geodetic Vertical Datum of 1929 in May 1939 (Minnesota Department of Natural Resources, 2016a); the general water-level changes during the last 50 years have been within a narrow range of less than (<) 1 m.

Primary inflows to Madison Lake are located in the northeast and southeast parts of the lake, both of which were primary sampling locations during this study for nutrient and major inorganic constituents, continuous 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, 2016a]; hereafter referred to as the "northeast inlet") flows into the relatively large and shallow northeast bay of Madison Lake; this is considered the primary inflow into the main water body for purposes of the CE-QUAL-W2 modeling. 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; this is considered the primary inflow into the secondary water body for purposes of the CE-QUAL-W2 modeling. 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, 2016a]; hereafter referred to as the "Madison Lake outlet"), located along the southwest part of the lake. Madison Lake has three other documented inlets, but these have intermittent flow and were not gaged for purposes of this study.

The lake has an area of 5.4 square kilometers (km^2) and a volume of 22.7 million cubic meters (m^3), with a maximum depth of 18 m (Lindon and others, 2010). The drainage basin of Madison Lake is 22.4 km^2, for a small ratio of basin to lake area of 4:1. The lake has three distinct bays, with two of the three bays containing deep areas; the large and shallow bay is located in the northeast part of the lake (fig. 2). The deepest parts of the lake are close to one another and are near the shallow narrows between the northeast and southwest bays of the lake. 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. The other deep location, located in the northeast bay close to the shallow narrows, was sampled at site Madison Lake northeast deep point near Madison Lake, Minn. (hereafter referred to as "northeast deep point") with a depth of approximately 18 m. This location was used primarily for continuous monitoring of water temperature at various depths.

Pearl Lake

Pearl Lake (fig. 3) in Stearns County, Minn., is in the Sauk River Basin, which is part of the greater Mississippi River Basin (Anderson and others, 2012). Pearl Lake is located within the North Central Hardwood Forests ecoregion (Soulard and others, 2014) in southeast Stearns County within a broad outwash plain that was deposited from the Des Moines lobe during the late Wisconsinan glaciation (Meyer and others, 1995). Based on the Stearns County quaternary stratigraphy map, Pearl Lake lies within more than 15 m of sand and gravel over two or more till beds (Meyer and others, 1995). Pearl Lake is an intermittently stratified polymictic lake, having a slight decline in temperatures earlier in the year, but is generally well-mixed before early summer through late fall (Anderson and others, 2012). Only a portion of the deeper mixed layer is anoxic earlier in the year, and by July, DO is well-mixed throughout the water column. The water balance of the basin for Pearl Lake is typically controlled by a spring snowmelt in late March or early April, followed by periodic large rain events in the summer. Lake residence time is on the order of 1–2 years, with small surface-water inflow and outflow (Anderson and others, 2012). The mean precipitation in the region for 1981–2010 is 0.70 m/yr (National Centers for Environmental Information, 2016b). Climate in the region has been known to consist of sustained drought periods, although Pearl Lake has not shown the historical fluctuations that Madison Lake has shown. Intermittent lake-level measurements date back to 1946, with fluctuations of approximately 1.1 m. The lowest lake level recorded on Pearl Lake was 339.94 m above the National Geodetic Vertical Datum of 1929 in September 1988 (Minnesota Department of Natural Resources, 2016b).



Base modified from U.S. Geological Survey
and other digital data, various scales
Universal Transverse Mercator projection, Zone 15 N
North American Datum of 1983

Figure 2. Location of water-quality sampling sites for Madison Lake, Minnesota.

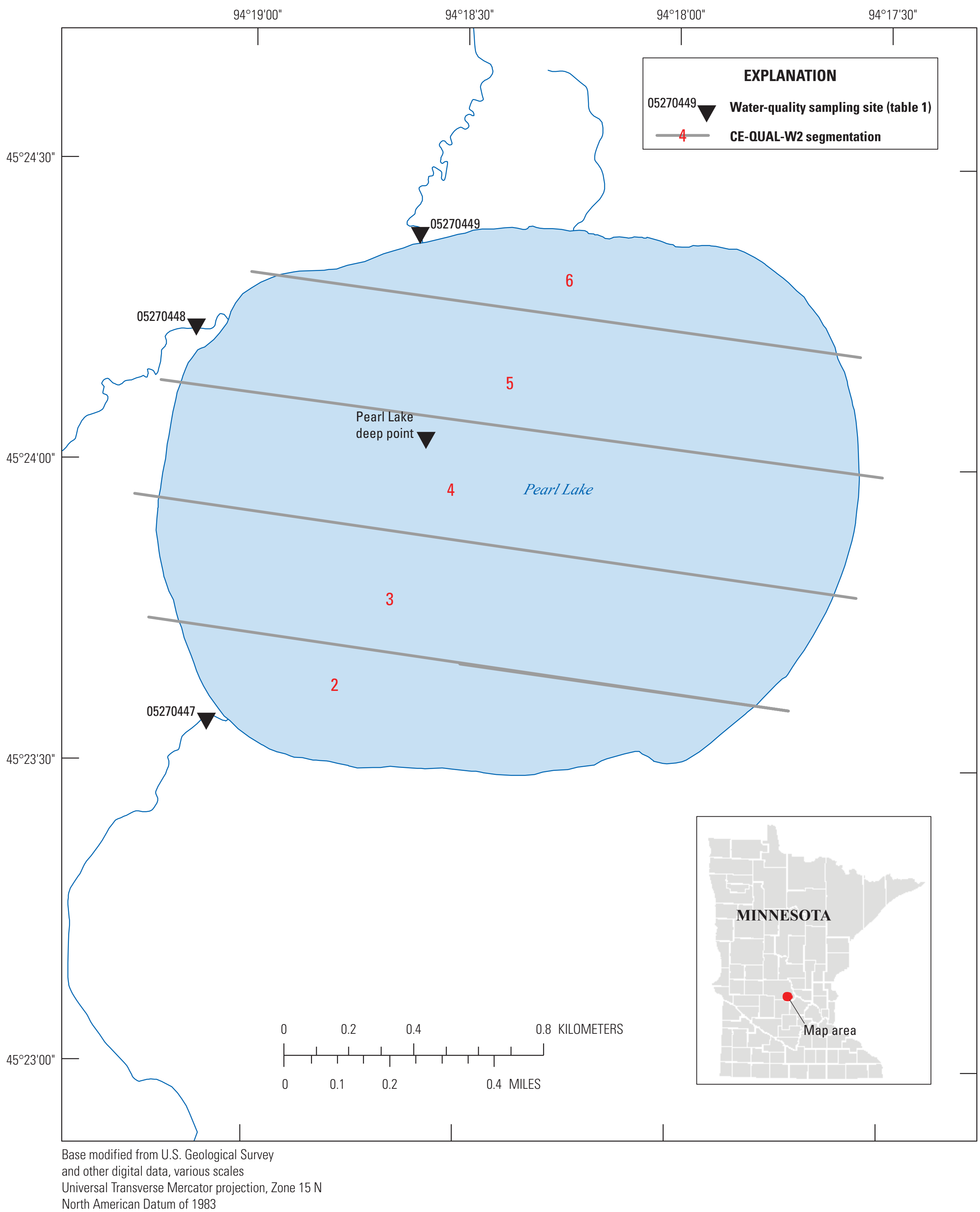


Figure 3. Location of water-quality sampling sites for Pearl Lake, Minnesota.

Primary inflows to Pearl Lake are through the northwest and southwest parts of the lake, both of which were primary sampling locations during this study for nutrient and major inorganic constituents, continuous water temperature, and streamflow. The site unnamed tributary to Pearl Lake, southwest corner, near Marty, Minn. (USGS station number 05270447 [U.S. Geological Survey, 2016a]; hereafter referred to as the "southwest corner inlet") flows into the southwest corner of Pearl Lake; this is considered the primary inflow into the water body for purposes of the CE-QUAL-W2 modeling. The Mill Creek at inlet to Pearl Lake near Marty, Minn. (USGS station number 05270448 [U.S. Geological Survey, 2016a]; hereafter referred to as the "Mill Creek inlet") flows into the northwest corner of Pearl Lake; this is considered tributary inflow into the water body for purposes of the CE-QUAL-W2 modeling. The main primary outflow for Pearl Lake is site Mill Creek at Pearl Lake outlet near Marty, Minn. (USGS station number 05270449 [U.S. Geological Survey, 2016a]; hereafter referred to as the "Pearl Lake outlet"), is along the north shoreline of the lake. No other inlets were documented for Pearl Lake.

Pearl Lake has an area of 3.0 km^2 and a volume of 9.1 million m^3, with a maximum depth of 5.5 m (Anderson and others, 2012). The Pearl Lake drainage basin is 40.0 km^2, with a large ratio of basin to lake area of 24:1. The lake has a roughly oval shape, with the long axis oriented east to west (fig. 3). The lake has an extensive littoral area of 68 percent with deep areas in the middle of the lake. The deep area in the west central part of the lake was sampled at site Pearl Lake deep point near Marty, Minn. (hereafter referred to as "Pearl Lake deep point"), with a depth of approximately 5.5 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.

Previous Studies

Both Madison and Pearl Lakes have been extensively sampled as part of the sentinel lake studies. Water-quality (including nutrients and major ion chemistry), phytoplankton biomass, zooplankton biomass, and macrophyte surveys were all completed during portions of the intensive sampling years between 2006 and 2009. Data collection summaries for Madison Lake were documented in the Sentinel Lake Assessment Report for Madison Lake (Lindon and others, 2010). Long-term fish community surveys have been recorded and stored in the MNDNR long-term fisheries survey database species assessments, available on the LakeFinder website by lake name and county (Minnesota Department of Natural Resources, 2016c, 2016d). Madison Lake was included as part of a study of microcystin levels in eutrophic south-central Minnesota lakes by the Minnesota Pollution Control Agency (Lindon and Heiskary, 2007). Fish community integrity surveys were completed for both lakes following the methods of Drake and Pereira (2002). Pearl Lake was included in the potential future climate scenarios of cisco (*Coregonus artedii*) refuge lakes to assess potential cisco refugia, using the MINLAKE 2010 water-quality model (Fang and others, 2012).

Development of Water-Quality Models to Assess Algal Community Dynamics and Water Quality

Two lake models (one for Madison Lake and one for Pearl Lake) were constructed using CE-QUAL-W2, version 4.0 (V4.0) (Cole and Wells, 2015), which is 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). Because the model is laterally averaged, the model is best suited for water bodies with a fairly homogenous cross section. The CE-QUAL-W2 V4.0 model calculates the hydrodynamic properties of water-surface elevation, velocities, and temperature and can simulate 28 water-quality variables in addition to temperature. An advantage of the CE-QUAL-W2 model over other hydrodynamic and water-quality models is that the hydrodynamic and water-quality modules are coupled together through an equation of state for density, which is dependent on temperature, suspended solids, and dissolved solids. This enables the water-quality model to feed back into the hydrodynamic part of the model. Although the lateral averaging of the CE-QUAL-W2 model is better suited for long, narrow water bodies, such as reservoirs, rivers, and estuaries, the CE-QUAL-W2 model has been successfully applied in lake settings (Sullivan and Rounds, 2004; Sullivan and others, 2007; Smith and others, 2014). Although Madison Lake and Pearl Lake did not meet the same criterion of a long and narrow body, homogeneity in water-quality and water temperature data are indicated for both lakes such that laterally averaging did not seem to compromise the integrity of the model. Vertical variations captured with the CE-QUAL-W2 model are important for distinguishing temporal variations in the lake epilimnion, hypolimnion, and mixed layers. Initial calibration included a water balance based on water-surface elevation and continuous water temperature for each lake. Additional calibration targets included water temperature and DO depth profiles, in addition to discrete measurements of algae (chlorophyll *a*) and nutrients (ammonia, nitrate plus nitrite, total Kjeldahl nitrogen, total phosphorus, orthophosphate).

The individual lake models were developed in several phases. First, data were collected to determine the hydrological, thermal, and water-quality boundary conditions. A summary of the discrete and continuous constituents collected for both lakes, further split by sampling locations, is shown in table 1. Selection of the calibration year for both lakes was based on the most extensive datasets available, specifically for streamflow, water-surface elevation, and water temperature data, because these datasets were critical for driving the model hydrodynamics. All other data were aggregated to best define the initial boundary conditions. These data were also used later in the calibration process. Next, the model grid was constructed based on available lake bathymetry data (Minnesota Geospatial Information Office, 2016a). Datasets necessary to run the CE-QUAL-W2 model were formatted to fit the input data structure. Prior to initial water-balance calibration, input parameters were selected, mainly based on default values either prepopulated within the CE-QUAL-W2 model (Cole and Wells, 2015) or previous USGS CE-QUAL-W2 modeling efforts (Galloway and Green, 2006; Galloway and others, 2008; Smith and others, 2014).

Water Balance

The following subsections provide details of the water-balance approach used for each of the lakes. The water-balance approach for the lakes included an initial calibration followed by refined calibrations.

Madison Lake

The water balance of Madison Lake was calibrated for May 15–November 1, 2014, by comparing measured water levels to simulated water levels at the Madison Lake outlet (USGS station number 05320170), which is the main surface-water outflow for Madison Lake (fig. 2; table 1). Two gaged inflow tributaries, the northeast inlet (USGS station number 05320130) and the southeast inlet (USGS station number 05320140) (fig. 2; table 1), provided the continuous streamflow measurements (U.S. Geological Survey, 2016a) for the entire calibration period. Adjustments were made to the gains and losses in the distributed tributary flow, which lumps all ungaged inflow and groundwater interactions, until the mean absolute error (MAE) and root mean square error (RMSE) values were <0.02 m.

Pearl Lake

The water balance of Pearl Lake was calibrated for May 14–November 13, 2014, by comparing measured water levels to simulated water levels at the Pearl Lake outlet, which is the main surface-water outflow located at the north end of the lake. Two gaged inflow tributaries, the southwest corner inlet (USGS station number 05270447) and the Mill Creek inlet (USGS station number 05270448) (fig. 3; table 1), provided the continuous streamflow measurements (U.S. Geological Survey, 2016a) for the entire calibration period. Adjustments were made to the gains and losses in the distributed tributary flow, which lumps all ungaged inflow and groundwater interactions, until a reasonable water balance was attained. Similar to methods for Madison Lake, adjustments were made to the gains and losses in the distributed tributary flow until a reasonable water balance, as well as low MAE and RMSE values for lake-level elevation, could be achieved.

Bathymetric Data and Computational Grid

Information from a digital elevation model (DEM) (U.S. Geological Survey, 2016b) and available bathymetric data from 2014 (Minnesota Geospatial Information Office, 2016a) were used to generate bathymetric cross sections for the CE-QUAL-W2 model. Accurate model reconstruction is important given that this reconstruction is the finite-difference representation of the lake itself. This accuracy can be verified by comparisons between the measured bathymetry and model grid for the curves relating water-surface elevation and lake volume and curves relating water-surface elevation and lake-surface area for each of the lakes (figs. 4–5).

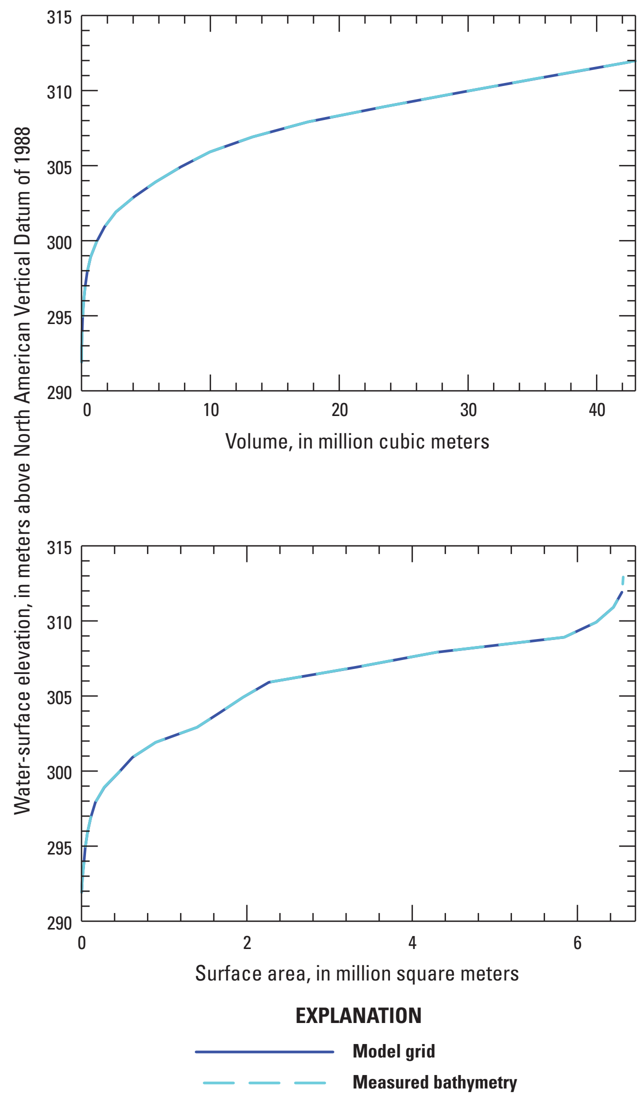


Figure 4. Lake volume and lake-surface area compared to water-surface elevation for Madison Lake using the measured bathymetry (Minnesota Geospatial Information Office, 2016a) and as represented by the model grid.

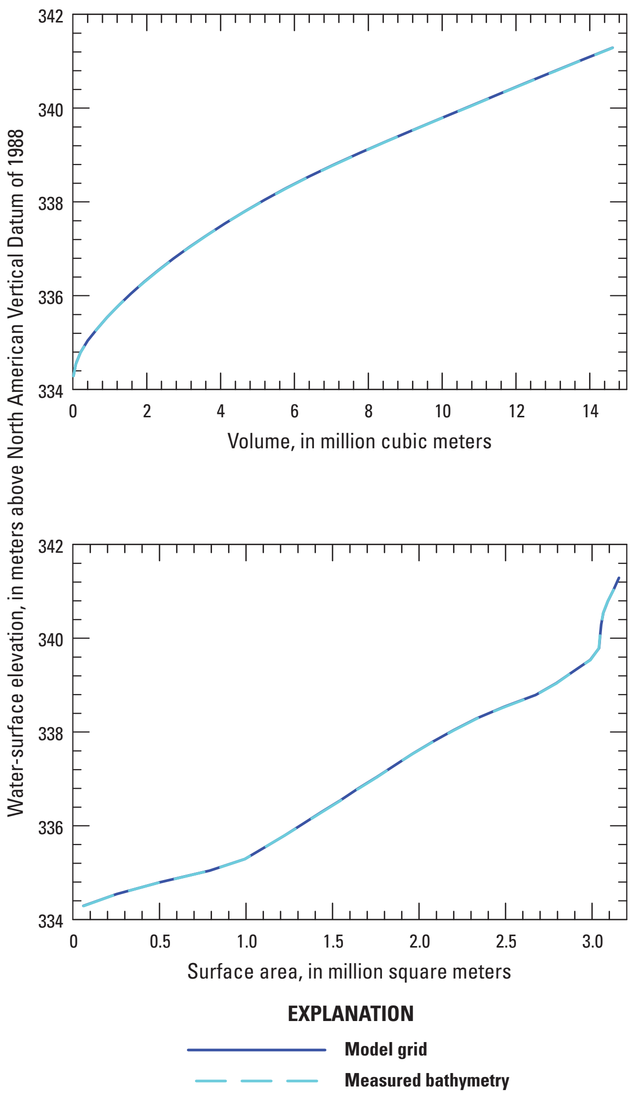


Figure 5. Lake volume and lake-surface area compared to water-surface elevation for Pearl Lake using the measured bathymetry (Minnesota Geospatial Information Office, 2016a) and as represented by the model grid.

The best available elevation data, 1-m DEMs based on light detection and ranging (lidar), were used to develop the area of potential inundation around the perimeter of both lakes (Minnesota Geospatial Information Office, 2016b) and merged with the bathymetric datasets. Bathymetric surveys of both lakes were available from the MNDNR as geographical information system (GIS) layers (Minnesota Geospatial Information Office, 2016a). The basic process was to combine the land elevation layer (1-m lidar DEM) with the bathymetric data to produce a gridded, three-dimensional model of the surface area and depth of each lake. The next step was to identify the deepest elevation value of the lake and then divide the lake model into 1-m slices starting at the bottom of the lake and ending approximately 2 m above the lake's base elevation (static water-level elevation); for Pearl Lake, the lake model was divided into 0.25-m slices rather than 1-m slices for numerical stability of the model due to its overall shallower depths. The base-lake elevation was obtained from lake-level data on the MNDNR Lake Finder website (Minnesota Department of Natural Resources, 2016a, 2016b). All model grid cells represented in each vertical section were identified and converted to a GIS polygon dataset. All slice polygons were then compiled into a single polygon GIS dataset, and the area of each polygon was calculated.

After completion of the GIS polygon dataset, each lake was divided into lateral segments (figs. 2–3). Within each lateral segment, vertical layers were drawn from the bottom of the lake up to 2 m above the static lake-level elevation. Distance along the longitudinal axis for individual CE-QUAL-W2 lateral segments varied considerably. Considerations for the number of segments selected included a balance between full-scale representation of the real structure of the lake and a segment structure that avoids numerical instability. Segments were grouped together into branches, with all of the branches grouped together representing the computational grid of the water body. The approximate segment boundaries for the CE-QUAL-W2 two-dimensional computational grids are shown in figures 2 and 3. With the ability to use different branches to represent separate bays or embayments, the southeast part of Madison Lake (where the southeast inlet flows into the lake) was separated out as a second distinct branch that flows into segment 9 (fig. 2); branch 1 includes segments 2 through 9, and branch 2 includes segments 12 through 15. Pearl Lake was modeled as a single water body with 5 computational segments (fig. 3).

Boundary and Initial Conditions

The success of the model largely depended on a high data density of biological, chemical, and physical lake characteristics from which lake parameters could be calculated and the model could be calibrated. Several continuous flow and water-quality monitoring systems were installed to calculate the initial and boundary conditions for the models and to provide a robust calibration dataset. Streamflow was measured on a regular schedule according to methods described in Buchanan and Somers (1969) and Mueller and Wagner (2008). All streamflow measurements were done by the MNDNR, with initial training provided by USGS personnel; streamflow data are available from U.S. Geological Survey (2016a). Continuous water-surface level (stage or gage height) and water temperature were collected for selected inflows and all outflows. Continuous streamflows were estimated for inflows and outflows using relations between corresponding water-surface levels and streamflows measured during site visits. Additional discussion of the continuous water levels is given in the "Hydraulic and Thermal Boundary Conditions" section.

Hydraulic and Thermal Boundary Conditions

The following subsections describe the collection of water-level (stage) measurements, streamflow measurements, temperature data, and meteorological data at each of the lakes. These data were used as hydraulic and thermal boundary conditions in the model.

Madison Lake

Lake inflow and water temperature data used in the CE-QUAL-W2 model for Madison Lake were obtained from two separate channels that flow into Madison Lake. The northeast inlet streamflow (fig. 2; table 1) was measured in the channel connecting several small lakes and wetlands to Madison Lake (USGS station number 05320130). The southeast inlet streamflow (fig. 2; table 1) was measured in the channel connecting Schoolhouse and Goolsby Lake to Madison Lake (USGS station number 05320140). Submersible pressure transducers were installed at ice off and removed just before ice on for the northeast inlet and southeast inlet. While in operation (May–November 2014), transducers collected continuous measurements of water temperature and water-surface level (stage or gage height) 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, 2016a).

Corresponding measurements of streamflow and water-surface level were used to develop rating curves that estimate continuous streamflow values based on continuously recorded water-surface levels. The continuous 15-minute interval water-surface levels were made at Madison Lake outlet (USGS station 05320170), from which daily mean water-surface elevations were calculated for May 15–November 1, 2014, available online through National Water Information System (NWIS) (U.S. Geological Survey, 2016a). Rating curves were developed using graphical plotting methods similar to those described in Rantz and others (1982a, 1982b) (appendix table 1–1). Linear extrapolations were added to the upper and lower end of the rating curves to estimate streamflows outside of the range of measured streamflows. For the purposes of the model, the northeast inlet streamflow is considered the main inflow into the main water body (appendix table 1–2), flowing into segment 2, and the southeast inlet streamflow is considered the main inflow into the secondary water body (appendix table 1–3), flowing into segment 12 (fig. 2; table 1). Additional water inflows to Madison Lake also were assumed from ungaged locations in the lake and from groundwater flow, known as distributed flow. This distributed flow was input into the model in daily time steps and distributed evenly across all of the model segments; more detail of the distributed flow is provided in the “Water Balance” section of the model calibration.

The main outflow streamflow from Madison Lake (fig. 2; table 1) is through the Madison Lake outlet, located along the southwest part of the lake out of the largest bay (USGS station number 05320170). The Madison Lake outlet streamflow is considered the sole outflow for Madison Lake (appendix table 1–4). Four corresponding measurements of streamflow and water level were made in 2014. Similar to methods used at both inflow sites, a rating curve was developed using graphical plotting methods similar to those described in Rantz and others (1982a, 1982b). Linear extrapolations were added to the upper and lower end of the rating curves to estimate streamflows outside of the range of measured streamflows, and the final rating curve was used to estimate continuous streamflow using continuously measured water levels.

The temperature data collected at 15-minute intervals at the two inlet sites were used to calculate daily mean temperature for those sites. The daily mean temperatures are available online through NWIS (U.S. Geological Survey, 2016a) for the northeast inlet (USGS station number 05320130) and for the southeast inlet (USGS station number 05320140).

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 [USAF] station identification number [ID] 726585) from the Climate Data Online portal (National Climatic Data Center, 2016), 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.

Pearl Lake

Lake inflow and water temperature data used in the CE-QUAL-W2 model for Pearl Lake were obtained from two separate channels that flow into Pearl Lake. The southwest corner inlet streamflow was measured in the channel located at the southwest corner of Pearl Lake (fig. 3; table 1; USGS station number 05270447). The Mill Creek inlet streamflow was measured in the channel located at the northeast corner of Pearl Lake (fig. 3; table 1; USGS station number 05270448). Submersible pressure transducers were installed at ice off and removed just before ice on (May–November 2014) for the southwest corner inlet and Mill Creek inlet. While in operation, these transducers collected continuous water temperature and water-level (stage) measurements every 15 minutes. Streamflow was measured once per month from May through October at these two inflow sites (U.S. Geological Survey, 2016a).

Methods for developing relations between streamflow and water-surface elevation for inflow and outflow sites at Pearl Lake differed from those used at Madison Lake. Graphical plotting techniques were used at Madison Lake sites because fewer streamflow measurements were collected and because the relations between streamflow and water-surface elevation were nonlinear. In contrast, linear regression analyses were used to develop relations between streamflow and water-surface elevation for inflow and outflow sites at Pearl Lake because more streamflow measurements were collected at Pearl Lake sites. In addition, relations between streamflow and water-surface elevation were linear for Pearl Lake sites. Developed linear relations were applied to water-surface elevation data collected with submersible pressure transducers at 15-minute intervals to develop continuous streamflow records for inflow and outflow sites at Pearl Lake.

Linear relations between streamflow and water-surface elevation for the southwest corner inlet and Mill Creek inlet are presented in equations 1 and 2, respectively. Equation 1 for the southwest corner inlet has a coefficient of determination (R^2) of 0.9855 and is as follows:

$$Q_{SWCornerInlet} = \left(2.1672 * StE_{SWCornerInlet}\right) - 738.26 \tag{1}$$

where

$Q_{SWCornerInlet}$ is the streamflow, in cubic meters per second; and

$StE_{SWCornerInlet}$ is the water-surface elevation, in meters.

Daily mean streamflow was calculated based on the continuous (15-minute) streamflow records generated using equation 1, available online (USGS station number 05270447) through NWIS (U.S. Geological Survey, 2016a). Equation 2 for the Mill Creek inlet has an R^2 of 0.9715 and is as follows:

$$Q_{MillCreekInlet} = \left(1.7206 * StE_{MillCreekInlet}\right) - 585.39 \tag{2}$$

where

$Q_{MillCreekInlet}$ is the streamflow, in cubic meters per second; and

$StE_{MillCreekInlet}$ is the water-surface elevation, in meters.

Daily mean streamflows were calculated based on the continuous (15-minute) streamflow records generated using equation 2, available online (USGS station number 05270448) through NWIS (U.S. Geological Survey, 2016a). For the purposes of the model, the southwest corner inlet is considered the main inflow into the water body, flowing into segment 2, and the Mill Creek inlet is considered a tributary inflow, flowing into segment 5 (fig. 3; table 1). Additional water inflows to Pearl Lake also were assumed from ungaged locations in the lake and from groundwater flow, known as distributed flow. This distributed flow was input into the model in daily time steps and distributed evenly across all the model segments; more detail of the distributed flow is provided in the "Water Balance" section of the model calibration.

The main outflow from Pearl Lake is through the Pearl Lake outlet, located along the north part of the lake (fig. 3; table 1). Equation 3 for the Pearl Lake outlet has an R^2 of 0.9586 and is as follows:

$$Q_{PearlLakeOutlet} = \left(4.6483 * StE_{PearlLakeOutlet}\right) - 1581.1 \tag{3}$$

where

$Q_{PearlLakeOutlet}$ is the streamflow, in cubic meters per second; and

$StE_{PearlLakeOutlet}$ is the water-surface elevation, in meters.

Daily mean streamflows were calculated based on the continuous (15-minute) streamflow records generated using equation 3 for Pearl Lake outlet, available online (USGS station number 05270449) through NWIS (U.S. Geological Survey, 2016a). Water-surface elevations for Pearl Lake were based on the transducer record collected at the Pearl Lake outlet site and are also available online (USGS station number 05270449) through NWIS (U.S. Geological Survey, 2016a).

The temperature data collected at 15-minute intervals at the two inlet sites were used to calculate daily mean temperature for those sites. The daily mean temperatures are available online through NWIS (U.S. Geological Survey, 2016a) for the southwest corner inlet (USGS station number 05270447) and the Mill Creek inlet (USGS station number 05270448).

All of the required meteorological data were available at hourly intervals for the Saint Cloud Regional Airport (USAF station ID 726550) from the Climate Data Online portal (National Climatic Data Center, 2016), approximately 25 km northeast of Pearl 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. The meteorological data required for the Pearl Lake CE-QUAL-W2 model were the same as those described in the previous section for Madison Lake.

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

Limnological characteristics, including properties that could affect trophic state, were examined at one site for each lake (table 1): southwest deep point at Madison Lake and Pearl Lake deep point at Pearl Lake. The Madison Lake site was sampled five times and the Pearl Lake site was sampled six times from May through November 2014 by MNDNR staff. Samples were collected near the surface and at depth, respectively (2 m and 16.5 m in Madison Lake; 2 m and 4.5 m in Pearl Lake), 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, 1993d). 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.

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 lakes. For Madison Lake, inflow sites were sampled 4–5 times in 2014. For Pearl Lake, inflow sites were sampled 6 times in 2014.

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., with the exception of the algae data. All of the samples analyzed by the Minnesota DHEL have been previously reviewed and published and are available online at the MNDNR Lake Finder (Minnesota Department of Natural Resources, 2016e) by searching by lake number (Madison Lake is 07004400; Pearl Lake is 73003700) or by lake name and county (Minnesota Department of Natural Resources, 2016e). The algae data were produced by a phytoplankton enumeration technique performed by PhycoTech, Inc. (PhycoTech, 2017); all of the raw algal data are presented in appendix table 2–1, 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 0.10 and multiplying by the chlorophyll *a* concentration collected on the same day.

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.

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

[EPA, U.S. Environmental Protection Agency; mg/L, milligram per liter; SM, standard method; --, not analyzed]

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 nitrate plus nitrite as 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 as nitrogen	EPA 351.2 (U.S. Environmental Protection Agency, 1993c)	0.20 mg/L
Total phosphorus as phosphorus	SM 4500–P (American Water Works Association and others, 1997a)	0.01 mg/L
Dissolved phosphorus as phosphorus	EPA 365.1 (U.S. Environmental Protection Agency, 1993d)	0.01 mg/L
Dissolved orthophosphate as phosphorus	EPA 365.1 (U.S. Environmental Protection Agency, 1993d)	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 (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 zero (U.S. Environmental Protection Agency, 2002).

Initial Conditions

Water-quality modeling was incorporated into a hydrodynamic model for each lake. 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 for Madison and Pearl Lakes are presented in table 3; 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. Initial water-surface elevation and water temperature were set to the measured value at the simulation start for both lakes.

Table 3. Initial constituent concentrations for Madison and Pearl Lakes for the 2014 calibration runs.

[m NAVD 88; meters above North American Vertical Datum of 1988; mg/L, milligram per liter; DOM, dissolved organic matter; POM, particulate organic matter; °C, degrees Celsius]

Constituent	Madison Lake	Pearl Lake
Initial water-surface elevation, m NAVD 88	310.57	340.38
Total dissolved solids, mg/L	177.7	199.1
Dissolved orthophosphate as phosphorus, mg/L	0.005	0.0025
Dissolved ammonia as nitrogen, mg/L	0.05	0.025
Dissolved nitrate plus nitrite as nitrogen, mg/L	0.38	0.32
Dissolved silica, mg/L	3.95	9.84
Particulate silica, mg/L	1	1
Total iron, mg/L	0.014	0.014
Labile DOM, mg/L	4.9510	1.6598
Refractory DOM, mg/L	11.5522	3.8730
Labile POM, mg/L	0.1490	0.5902
Refractory POM, mg/L	0.3478	1.3770
Bacillariophyta/crysophyta, mg/L	2.7	0.025
Chlorophyta (green algae), mg/L	0.0003	0.025
Cyanophyta (blue-green algae), mg/L	0.002	0.025
Haptophyta/cryptophyta, mg/L	0.0007	0.025
Dissolved oxygen, mg/L	[1]0.75–10.25	13
Inorganic carbon, mg/L	170.4	207
Alkalinity, mg/L	140	170
Initial temperature, °C	9.9	9.0
Sediment temperature, °C	14.2	6.1

[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.

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 Madison Lake 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. The Pearl Lake distributed tributary inflow constituents were based on the mean concentrations for the southwest corner inlet site.

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 V4.0 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, 200 parameters control the constituent kinetics (table 4). 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. Only the parameters required for the lake applications were included in table 4. As with the hydraulic and heat exchange parameters that control the hydrodynamics, all of 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.

Table 4. Model parameters used for the water-quality algorithms for Madison Lake and Pearl Lake.

[**Bold** text indicates parameters adjusted from default value. m^{-1}, per meter; $m^{-1}/(g \cdot m^{-3})$, per meter per grams per cubic meter; $g \cdot m^{-3}$, grams per cubic meter; day^{-1}, per day; $m \cdot day^{-1}$, meters per day; $W/(m^2 \cdot °C)$, watts per square meter per degree Celsius; $W \cdot m^{-2}$, watts per square meter; °C, degrees Celsius; $m^2 \cdot s^{-1}$; square meters per second; $m \cdot s^{-1}$; meters per second; POM, particulate organic matter; DOM, dissolved organic matter; SOD, sediment oxygen demand; SED, sediment]

Parameter	Description	Parameter value	
		Madison Lake	Pearl Lake
AX	Horizontal eddy coefficient, $m^2 \cdot s^{-1}$	1.0	1.0
DX	Vertical eddy coefficient, $m^2 \cdot s^{-1}$	1.0	1.0
CBHE	Sediment heat exchange coefficient, $W/(m^2 \cdot °C)$	**1.5**	**1.3**
FI	Interfacial friction factor	0.015	0.015
TSEDF	Heat lost to sediments that is added back to water column	**0.75**	**0.10**
AZC	Vertical turbulence closure algorithm	**W2**	TKE
EXH2O	Light extinction for pure water, m^{-1}	0.25	0.25
EXSS	Light extinction due to inorganic suspended solids, m^{-1}	0.10	**0.20**
EXOM	Light extinction due to organic suspended solids, m^{-1}	0.10	**0.20**
BETA	Fraction of incident solar radiation absorbed at water surface	**0.55**	0.45
EXA1	Light extinction due to algae (bacillariophyta/crysophyta), $m^{-1}/(g \cdot m^{-3})$	**0.1**	0.2
EXA2	Light extinction due to algae (green), $m^{-1}/(g \cdot m^{-3})$	**0.1**	0.2
EXA3	Light extinction due to algae (blue-green), $m^{-1}/(g \cdot m^{-3})$	**0.1**	0.2
EXA4	Light extinction due to algae (haptophyta/cryptophyta), $m^{-1}/(g \cdot m^{-3})$	**0.1**	0.2
EXM1	Light extinction due to macrophytes, $m^{-1}/(g \cdot m^{-3})$	**0.05**	0.01
AG	Maximum algal growth rate (bacillariophyta/crysophyta), day^{-1}	**3.82**	**1.38**
AG	Maximum algal growth rate (green), day^{-1}	**2.10**	**1.21**
AG	Maximum algal growth rate (blue-green), day^{-1}	**1.62**	**1.38**
AG	Maximum algal growth rate (haptophyta/cryptophyta), day^{-1}	**2.12**	**1.52**
AR	Maximum algal respiration rate (bacillariophyta/crysophyta), day^{-1}	**0.02**	0.04
AR	Maximum algal respiration rate (green), day^{-1}	0.04	0.04
AR	Maximum algal respiration rate (blue-green), day^{-1}	0.04	0.04
AR	Maximum algal respiration rate (haptophyta/cryptophyta), day^{-1}	0.04	0.04
AE	Maximum algal excretion rate (bacillariophyta/crysophyta), day^{-1}	**0.02**	**0.025**
AE	Maximum algal excretion rate (green), day^{-1}	**0.05**	**0.025**
AE	Maximum algal excretion rate (blue-green), day^{-1}	**0.07**	**0.025**
AE	Maximum algal excretion rate (haptophyta/cryptophyta), day^{-1}	0.04	**0.025**
AM	Maximum algal mortality rate (bacillariophyta/crysophyta), day^{-1}	**0.06**	**0.07**
AM	Maximum algal mortality rate (green), day^{-1}	**0.09**	**0.07**
AM	Maximum algal mortality rate (blue-green), day^{-1}	**0.08**	**0.12**
AM	Maximum algal mortality rate (haptophyta/cryptophyta), day^{-1}	**0.09**	**0.07**
AS	Algal settling rate (bacillariophyta/crysophyta), $m \cdot day^{-1}$	**0.20**	**0.15**
AS	Algal settling rate (green), $m \cdot day^{-1}$	**0.12**	0.10
AS	Algal settling rate (blue-green), $m \cdot day^{-1}$	**0.08**	0.10
AS	Algal settling rate (haptophyta/cryptophyta), $m \cdot day^{-1}$	0.10	0.10
AHSP	Algal half-saturation for phosphorus-limited growth (bacillariophyta/crysophyta), $g \cdot m^{-3}$	**0.0065**	**0.0040**
AHSP	Algal half-saturation for phosphorus-limited growth (green), $g \cdot m^{-3}$	**0.0065**	**0.0040**
AHSP	Algal half-saturation for phosphorus-limited growth (blue-green), $g \cdot m^{-3}$	**0.0062**	**0.0045**
AHSP	Algal half-saturation for phosphorus-limited growth (haptophyta/cryptophyta), $g \cdot m^{-3}$	**0.0065**	**0.0040**
AHSN	Algal half-saturation for nitrogen-limited growth (bacillariophyta/crysophyta), $g \cdot m^{-3}$	**0.025**	**0.010**
AHSN	Algal half-saturation for nitrogen-limited growth (green), $g \cdot m^{-3}$	**0.030**	0.014
AHSN	Algal half-saturation for nitrogen-limited growth (blue-green), $g \cdot m^{-3}$	**0.002**	**0.002**
AHSN	Algal half-saturation for nitrogen-limited growth (haptophyta/cryptophyta), $g \cdot m^{-3}$	**0.030**	**0.010**

Table 4. Model parameters used for the water-quality algorithms for Madison Lake and Pearl Lake.—Continued

[**Bold** text indicates parameters adjusted from default value. m^{-1}, per meter; $m^{-1}/(g \cdot m^{-3})$, per meter per grams per cubic meter; $g \cdot m^{-3}$, grams per cubic meter; day^{-1}, per day; $m \cdot day^{-1}$, meters per day; $W/(m^2 \cdot °C)$, watts per square meter per degree Celsius; $W \cdot m^{-2}$, watts per square meter; °C, degrees Celsius; $m^2 \cdot s^{-1}$; square meters per second; $m \cdot s^{-1}$; meters per second; POM, particulate organic matter; DOM, dissolved organic matter; SOD, sediment oxygen demand; SED, sediment]

Parameter	Description	Parameter value	
		Madison Lake	Pearl Lake
AHSSI	Algal half-saturation for silica-limited growth (bacillariophyta/crysophyta), $g \cdot m^{-3}$	0	0
AHSSI	Algal half-saturation for silica-limited growth (green), $g \cdot m^{-3}$	0	0
AHSSI	Algal half-saturation for silica-limited growth (blue-green), $g \cdot m^{-3}$	0	0
AHSSI	Algal half-saturation for silica-limited growth (haptophyta/cryptophyta), $g \cdot m^{-3}$	0	0
ASAT	Light saturation intensity at maximum photosynthetic rate (bacillariophyta/crysophyta), $W \cdot m^{-2}$	**150**	**30**
ASAT	Light saturation intensity at maximum photosynthetic rate (green), $W \cdot m^{-2}$	**50**	**45**
ASAT	Light saturation intensity at maximum photosynthetic rate (blue-green), $W \cdot m^{-2}$	**60**	**120**
ASAT	Light saturation intensity at maximum photosynthetic rate (haptophyta/cryptophyta), $W \cdot m^{-2}$	**20**	**30**
AT1	Lower temperature for algal growth (bacillariophyta/crysophyta), °C	5	5
AT1	Lower temperature for algal growth (green), °C	**10**	**10**
AT1	Lower temperature for algal growth (blue-green), °C	**13**	**11**
AT1	Lower temperature for algal growth (haptophyta/cryptophyta), °C	**14**	**6**
AT2	Lower temperature for maximum algal growth (bacillariophyta/crysophyta), °C	**13**	**14**
AT2	Lower temperature for maximum algal growth (green), °C	**22**	**18**
AT2	Lower temperature for maximum algal growth (blue-green), °C	**20**	**17**
AT2	Lower temperature for maximum algal growth (haptophyta/cryptophyta), °C	**24**	**16**
AT3	Upper temperature for maximum algal growth (bacillariophyta/crysophyta), °C	**18**	**23**
AT3	Upper temperature for maximum algal growth (green), °C	**28**	**25**
AT3	Upper temperature for maximum algal growth (blue-green), °C	**32**	**28**
AT3	Upper temperature for maximum algal growth (haptophyta/cryptophyta), °C	**28**	**24**
AT4	Upper temperature for algal growth (bacillariophyta/crysophyta), °C	**21**	**27**
AT4	Upper temperature for algal growth (green), °C	**32**	**30**
AT4	Upper temperature for algal growth (blue-green), °C	**35**	**35**
AT4	Upper temperature for algal growth (haptophyta/cryptophyta), °C	**30**	**29**
AK1	Fraction of algal growth rate at AT1 (bacillariophyta/crysophyta)	0.1	0.1
AK1	Fraction of algal growth rate at AT1 (green)	**0.005**	0.1
AK1	Fraction of algal growth rate at AT1 (blue-green)	**0.01**	**0.3**
AK1	Fraction of algal growth rate at AT1 (haptophyta/cryptophyta)	**0.005**	0.1
AK2	Fraction of maximum algal growth rate at AT2 (bacillariophyta/crysophyta)	0.99	0.99
AK2	Fraction of maximum algal growth rate at AT2 (green)	**0.65**	0.99
AK2	Fraction of maximum algal growth rate at AT2 (blue-green)	0.99	**0.95**
AK2	Fraction of maximum algal growth rate at AT2 (haptophyta/cryptophyta)	**0.88**	**0.94**
AK3	Fraction of maximum algal growth rate at AT3 (bacillariophyta/crysophyta)	0.99	**0.90**
AK3	Fraction of maximum algal growth rate at AT3 (green)	0.99	0.99
AK3	Fraction of maximum algal growth rate at AT3 (blue-green)	0.99	**0.85**
AK3	Fraction of maximum algal growth rate at AT3 (haptophyta/cryptophyta)	0.99	**0.64**
AK4	Fraction of algal growth rate at AT4 (bacillariophyta/crysophyta)	0.1	0.1
AK4	Fraction of algal growth rate at AT4 (green)	0.1	0.1
AK4	Fraction of algal growth rate at AT4 (blue-green)	0.1	0.1
AK4	Fraction of algal growth rate at AT4 (haptophyta/cryptophyta)	0.1	0.1
ALGP	Stoichiometric equivalent between algal biomass and phosphorus (bacillariophyta/crysophyta)	**0.0045**	**0.0041**
ALGP	Stoichiometric equivalent between algal biomass and phosphorus (green)	**0.0085**	**0.0036**
ALGP	Stoichiometric equivalent between algal biomass and phosphorus (blue-green)	**0.0075**	**0.0049**
ALGP	Stoichiometric equivalent between algal biomass and phosphorus (haptophyta/cryptophyta)	**0.0060**	**0.0037**

Table 4. Model parameters used for the water-quality algorithms for Madison Lake and Pearl Lake.—Continued

[**Bold** text indicates parameters adjusted from default value. m^{-1}, per meter; $m^{-1}/(g \cdot m^{-3})$, per meter per grams per cubic meter; $g \cdot m^{-3}$, grams per cubic meter; day^{-1}, per day; $m \cdot day^{-1}$, meters per day; $W/(m^2 \cdot °C)$, watts per square meter per degree Celsius; $W \cdot m^{-2}$, watts per square meter; °C, degrees Celsius; $m^2 \cdot s^{-1}$; square meters per second; $m \cdot s^{-1}$; meters per second; POM, particulate organic matter; DOM, dissolved organic matter; SOD, sediment oxygen demand; SED, sediment]

Parameter	Description	Parameter value	
		Madison Lake	Pearl Lake
ALGN	Stoichiometric equivalent between algal biomass and nitrogen (bacillariophyta/crysophyta)	**0.085**	**0.0795**
ALGN	Stoichiometric equivalent between algal biomass and nitrogen (green)	**0.085**	**0.0825**
ALGN	Stoichiometric equivalent between algal biomass and nitrogen (blue-green)	0.080	**0.0655**
ALGN	Stoichiometric equivalent between algal biomass and nitrogen (haptophyta/cryptophyta)	**0.085**	**0.0850**
ALGC	Stoichiometric equivalent between algal biomass and carbon (bacillariophyta/crysophyta)	0.45	0.45
ALGC	Stoichiometric equivalent between algal biomass and carbon (green)	0.45	0.45
ALGC	Stoichiometric equivalent between algal biomass and carbon (blue-green)	0.45	0.45
ALGC	Stoichiometric equivalent between algal biomass and carbon (haptophyta/cryptophyta)	0.45	0.45
ALGSI	Stoichiometric equivalent between algal biomass and silica (bacillariophyta/crysophyta)	0.18	0.18
ALGSI	Stoichiometric equivalent between algal biomass and silica (green)	0.18	0.18
ALGSI	Stoichiometric equivalent between algal biomass and silica (blue-green)	0.18	0.18
ALGSI	Stoichiometric equivalent between algal biomass and silica (haptophyta/cryptophyta)	0.18	0.18
ACHLA	Ratio between algal biomass and chlorophyll *a* in terms of milligrams of algae to micrograms of chlorophyll *a* (bacillariophyta/crysophyta)	**0.09**	**0.10**
ACHLA	Ratio between algal biomass and chlorophyll *a* in terms of milligrams of algae to micrograms of chlorophyll *a* (green)	**0.10**	**0.15**
ACHLA	Ratio between algal biomass and chlorophyll *a* in terms of milligrams of algae to micrograms of chlorophyll *a* (blue-green)	**0.06**	**0.15**
ACHLA	Ratio between algal biomass and chlorophyll *a* in terms of milligrams of algae to micrograms of chlorophyll *a* (haptophyta/cryptophyta)	**0.10**	**0.10**
ALPOM	Fraction of algal biomass that is converted to particulate organic matter when algae die (bacillariophyta/crysophyta)	**0.85**	**0.75**
ALPOM	Fraction of algal biomass that is converted to particulate organic matter when algae die (green)	**0.70**	**0.75**
ALPOM	Fraction of algal biomass that is converted to particulate organic matter when algae die (blue-green)	**0.70**	**0.75**
ALPOM	Fraction of algal biomass that is converted to particulate organic matter when algae die (haptophyta/cryptophyta)	**0.70**	**0.75**
ANEQN	Equation number for algal ammonium preference (bacillariophyta/crysophyta)	**1**	2
ANEQN	Equation number for algal ammonium preference (green)	**1**	2
ANEQN	Equation number for algal ammonium preference (blue-green)	**1**	2
ANEQN	Equation number for algal ammonium preference (haptophyta/cryptophyta)	**1**	2
ANPR	Algal half-saturation constant for ammonium preference (bacillariophyta/crysophyta)	**0.02**	**0.02**
ANPR	Algal half-saturation constant for ammonium preference (green)	**0.02**	**0.02**
ANPR	Algal half-saturation constant for ammonium preference (blue-green)	**0.02**	**0.02**
ANPR	Algal half-saturation constant for ammonium preference (haptophyta/cryptophyta)	**0.02**	**0.02**
O2AR	Oxygen stoichiometry for algal respiration (bacillariophyta/crysophyta)	1.10	1.10
O2AR	Oxygen stoichiometry for algal respiration (green)	1.10	1.10
O2AR	Oxygen stoichiometry for algal respiration (blue-green)	**0.95**	1.10
O2AR	Oxygen stoichiometry for algal respiration (haptophyta/cryptophyta)	**1.25**	1.10
O2AG	Oxygen stoichiometry for algal primary production (bacillariophyta/crysophyta)	**1.65**	1.40
O2AG	Oxygen stoichiometry for algal primary production (green)	1.40	1.40
O2AG	Oxygen stoichiometry for algal primary production (blue/green)	**1.10**	**2.40**
O2AG	Oxygen stoichiometry for algal primary production (haptophyta/cryptophyta)	1.40	1.40
MG	Maximum macrophyte growth rate, day^{-1}	**0.47**	**0.57**
MR	Maximum macrophyte respiration rate, day^{-1}	0.05	0.05
MM	Maximum macrophyte mortality rate, day^{-1}	0.05	0.05

Table 4. Model parameters used for the water-quality algorithms for Madison Lake and Pearl Lake.—Continued

[**Bold** text indicates parameters adjusted from default value. m^{-1}, per meter; $m^{-1}/(g \cdot m^{-3})$, per meter per grams per cubic meter; $g \cdot m^{-3}$, grams per cubic meter; day^{-1}, per day; $m \cdot day^{-1}$, meters per day; $W/(m^2 \cdot °C)$, watts per square meter per degree Celsius; $W \cdot m^{-2}$, watts per square meter; °C, degrees Celsius; $m^2 \cdot s^{-1}$; square meters per second; $m \cdot s^{-1}$; meters per second; POM, particulate organic matter; DOM, dissolved organic matter; SOD, sediment oxygen demand; SED, sediment]

Parameter	Description	Parameter value	
		Madison Lake	Pearl Lake
MSAT	Light saturation intensity at maximum photosynthetic rate, $W \cdot m^{-2}$	**75**	**20**
MHSP	Macrophyte half-saturation for phosphorus-limited growth, $g \cdot m^{-3}$	0	0
MHSN	Macrophyte half-saturation for nitrogen-limited growth, $g \cdot m^{-3}$	0	0
MHSC	Macrophyte half-saturation for carbon-limited growth, $g \cdot m^{-3}$	0	0
MPOM	Fraction of macrophyte biomass that is converted to particulate organic matter when macrophytes die	**0.85**	**0.75**
LRPMAC	Fraction of POM that originates as dead macrophytes becoming labile POM	0.20	0.20
PSED	Fraction of phosphorus uptake by macrophytes obtained from sediments	1	1
NSED	Fraction of nitrogen uptake by macrophytes obtained from sediments	1	1
MBMP	Threshold macrophyte concentration for which growth is moved to the above layer, $g \cdot m^{-3}$	40	40
MMAX	Maximum macrophyte concentration, $g \cdot m^{-3}$	500	500
CDDRAG	Macrophyte drag coefficient	2	2
DMV	Macrophyte dry weight to wet volume ratio, $g \cdot m^{-3}$	70,000	70,000
DWSA	Macrophyte dry weight to surface area ratio, $g \cdot m^{-3}$	8	8
ANORM	Fraction of macrophyte surface area normal to direction of flow	0.3	0.3
MT1	Lower temperature for macrophyte, °C	**7**	**7**
MT2	Lower temperature for maximum macrophyte, °C	**10**	**10**
MT3	Lower temperature for maximum macrophyte, °C	**24**	**18**
MT4	Upper temperature for macrophyte, °C	**34**	**24**
MK1	Fraction of macrophyte growth rate at MT1	0.10	0.10
MK2	Fraction of maximum macrophyte growth rate at MT2	0.99	0.99
MK3	Fraction of maximum macrophyte growth rate at MT3	0.99	0.99
MK4	Fraction of macrophyte growth rate at MT4	0.10	0.10
MP	Stoichiometric equivalent between macrophyte biomass and phosphorus	0.005	0.005
MN	Stoichiometric equivalent between macrophyte biomass and nitrogen	0.08	0.08
MC	Stoichiometric equivalent between macrophyte biomass and carbon	0.45	0.45
O2MR	Oxygen stoichiometry for macrophyte respiration	1.10	1.10
O2MG	Oxygen stoichiometry for macrophyte primary production	1.40	1.40
LDOMDK	Labile DOM decay rate, day^{-1}	**0.0725**	0.10
RDOMDK	Refractory DOM decay rate, day^{-1}	**0.002**	0.001
LRDDK	Labile-to-refractory DOM decay rate, day^{-1}	**0.02**	0.01
LPOMDK	Labile POM decay rate, day^{-1}	**0.05**	0.08
RPOMDK	Refractory POM decay rate, day^{-1}	**0.002**	0.001
LRPDK	Labile-to-refractory POM decay rate, day^{-1}	**0.02**	0.01
POMS	POM settling rate, $m \cdot day^{-1}$	**0.125**	**0.125**
ORGP	Stoichiometric equivalent between organic matter and phosphorus	**0.0065**	**0.0025**
ORGN	Stoichiometric equivalent between organic matter and nitrogen	**0.0950**	**0.1050**
ORGC	Stoichiometric equivalent between organic matter and carbon	0.45	0.45
ORGSI	Stoichiometric equivalent between organic matter and silica	0.18	0.18
OMT1	Lower temperature for organic matter decay, °C	**5**	4
OMT2	Upper temperature for organic matter decay, °C	25	25
OMK1	Fraction of organic matter decay at OMT1	0.1	0.1
OMK2	Fraction of organic matter decay at OMT2	0.99	0.99
PO4R	Sediment release rate of phosphorus, fraction of SOD	**0.012**	**0.0025**

Table 4. Model parameters used for the water-quality algorithms for Madison Lake and Pearl Lake.—Continued

[**Bold** text indicates parameters adjusted from default value. m^{-1}, per meter; $m^{-1}/(g \cdot m^{-3})$, per meter per grams per cubic meter; $g \cdot m^{-3}$, grams per cubic meter; day^{-1}, per day; $m \cdot day^{-1}$, meters per day; $W/(m^2 \cdot °C)$, watts per square meter per degree Celsius; $W \cdot m^{-2}$, watts per square meter; °C, degrees Celsius; $m^2 \cdot s^{-1}$; square meters per second; $m \cdot s^{-1}$; meters per second; POM, particulate organic matter; DOM, dissolved organic matter; SOD, sediment oxygen demand; SED, sediment]

Parameter	Description	Parameter value	
		Madison Lake	Pearl Lake
PARTP	Phosphorus partitioning coefficient for suspended solids	0	0
NH4R	Sediment release rate of ammonium, fraction of SOD	**0.004**	**0.0**
NH4DK	Ammonium decay rate, day^{-1}	**0.095**	**0.2**
NH4T1	Lower temperature for ammonia decay, °C	**7**	5
NH4T2	Lower temperature for maximum ammonia decay, °C	**30**	25
NH4K1	Fraction of nitrification rate at NH4T1	0.1	0.1
NH4K2	Fraction of nitrification rate at NH4T2	0.99	0.99
NO3DK	Nitrate decay rate, day^{-1}	**0.0575**	**0.16**
NO3S	Denitrification rate from sediments, $m \cdot day^{-1}$	**0.003**	**0.15**
FNO3SED	Fraction of nitrate-nitrogen diffused into the sediments that become part of organic nitrogen in the sediments	0	0
NO3T1	Lower temperature for nitrate decay, °C	**7**	5
NO3T2	Lower temperature for maximum nitrate decay, °C	**30**	**30**
NO3K1	Fraction of denitrification rate at NO3T1	0.1	**0.3**
NO3K2	Fraction of denitrification rate at NO3T2	0.99	0.99
DSIR	Dissolved silica sediment release rate, fraction of SOD	0.1	0.1
PSIS	Particulate biogenic settling rate, $m \cdot s^{-1}$	1.0	1.0
PSIDK	Particulate biogenic silica decay rate, day^{-1}	0.3	0.3
PARTSI	Dissolved silica partitioning coefficient	0.0	0.0
SEDS	Sediment settling or focusing velocity, $m \cdot day^{-1}$	0.1	0.1
SEDK	Sediment decay rate, $1 \cdot day^{-1}$	**0.04**	0.1
FSOD	Fraction of SOD	1.0	1.0
FSED	Fraction of SED	1.0	1.0
SOD	Zero-order SOD	2.5	5.5
O2LIM	Dissolved oxygen half-saturation constant or concentration at which aerobic processes are at 50 percent of their maximum, $g \cdot m^{-3}$	**0.1**	0.7
FER	Iron sediment release rate, fraction of SOD	0.5	0.5
FES	Iron settling velocity, $m \cdot day^{-1}$	2	2
CO2R	Sediment carbon dioxide release rate, fraction of SOD	1.2	1.2
O2NH4	Oxygen stoichiometry for nitrification	4.57	4.57
O2OM	Oxygen stoichiometry for organic matter decay	1.4	1.4
TYPE	Type of waterbody	LAKE	LAKE
EQN#	Equation number used for determining reaeration	**9**	**1**

Many of the parameters in table 4 were left as the default values (88 of 200 parameters for Madison Lake; 112 of 200 parameters for Pearl Lake), whereas the remaining parameters (112 of 200 parameters for Madison Lake; 88 of 200 parameters for Pearl Lake) 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

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 4 (for example, see usage in Smith and others, 2014), is a measure of the mean difference between the simulated (model) value and the measured value:

$$\text{MAE} = \frac{1}{n}\sum_{i=1}^{n}\left|simulated\ value - measured\ value\right| \quad (4)$$

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 5 (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:

$$\text{RMSE} = \sqrt{\frac{1}{n}\sum_{i=1}^{n}\left(simulated\ value - measured\ value\right)^2} \quad (5)$$

where

n is the number of observations.

The degree of fit between the simulated and measured outlet water-surface elevation and between the simulated and measured water temperature was only considered during the initial calibration for each of the lake models. By calibrating to water-surface elevation and water temperature first, the subsequent water-quality calibration was easier given that the effects such as wind stress, inflow water temperature, meteorological effects, and the amount of flow in and out of the lake had already been taken into account. The water-quality calibration for DO, algae, and nutrients followed, using the MAE and RMSE metrics. In a few cases, the measured lake water-quality value was outside the simulation time period by as much as 1 day; in such cases, the water-quality measurement was compared to the closest simulated value.

Refined calibration focused on the vertical profiles of temperature and DO for both lakes (figs. 2 and 3; 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, after several hundred iterations, 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 location. 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. Most model runs included one adjustment with a subsequent model run to characterize the parameter sensitivity.

Water Balance

The first step in the calibration process for both lake models was the water balance. Before the water temperature and water-quality calibration could proceed, the differences between the simulated and measured water-surface elevations were rectified. A water balance was considered complete when the MAE and RMSE values were <0.02 m for the simulated water-surface elevation.

Madison Lake

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 May 15–November 1, 2014; 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. To include the unaccounted inflow, two different distributed tributary flows were added iteratively for each of the two water bodies of Madison Lake. These distributed tributary flows were added to all segments equally until the simulated lake water-surface elevation matched the measured lake water-surface elevation. These distributed tributary flows can be positive or negative; large positive values were determined to correlate with large precipitation events, whereas negative values usually were during the driest portions of the calibration period. Based on daily means, <1 percent of the total water flow was accounted for by the distributed tributary flows. A comparison between the simulated and measured water-surface elevations for Madison Lake is shown in figure 6, with MAE and RMSE values of <0.02 m.

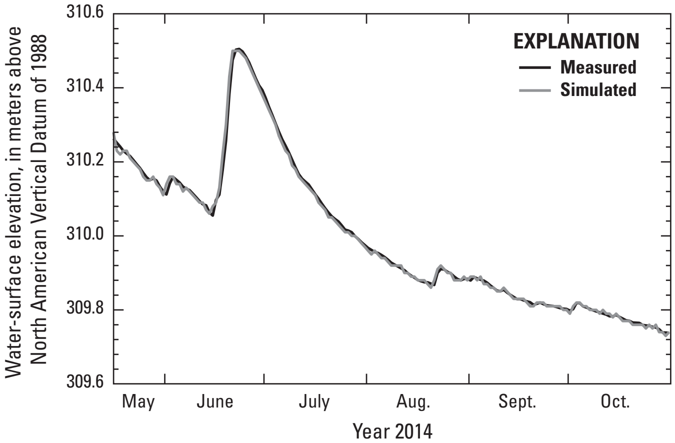


Figure 6. Simulated and measured water-surface elevations for Madison Lake, May 15 to November 1, 2014.

Pearl Lake

The initial attempt to achieve a water balance for Pearl Lake used the two gaged tributaries, the southwest corner inlet and the Mill Creek inlet (table 1), as the sole inflows for the calibration period of May 14–November 13, 2014; however, the simulated water-surface elevation was below the measured water-surface elevation, which indicated additional water sources to the lake. A distributed tributary flow for Pearl Lake was added to all segments equally and as with Madison Lake, these distributed tributary flows can be positive or negative. Based on daily means, approximately 2 percent of the total water flow was accounted for by the distributed tributary flows. A comparison between the simulated and measured water-surface elevations for Pearl Lake is shown in figure 7, with MAE and RMSE values of <0.02 m.

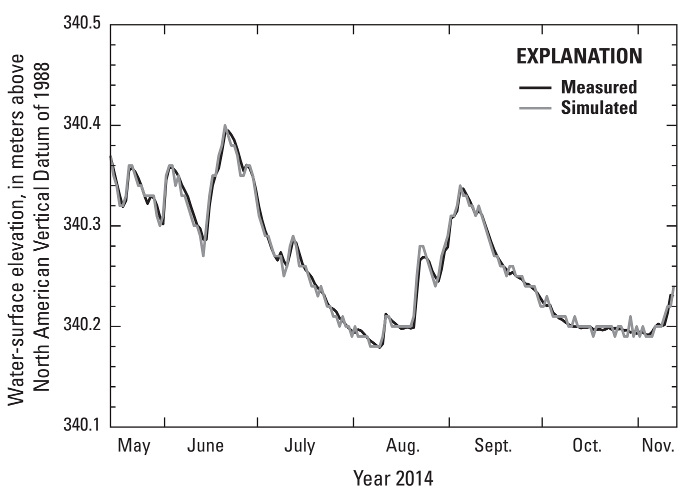


Figure 7. Simulated and measured water-surface elevations for Pearl Lake, May 14 to November 13, 2014.

Temperature

A critical calibration step is the water temperature calibration because of the effect of temperature on water density. Water temperature is a key metric to determine the accuracy of the model's calibration. Boundary conditions that affect water temperature include sediment temperature, initial lake water temperature, and inflow water temperature. Meteorological effects include air temperature, wind velocity, wind direction, and solar radiation. Because solar radiation was not directly available for any of the lake models, an internal calculation within the model was made based on the amount of cloud cover and the latitude/longitude. Wind effects can be further augmented by the wind sheltering coefficient (WSC), controlled through a separate input file, which takes into account the effects of boundary factors, such as topography and shoreline tree cover, on wind mixing. Several hydraulic parameters also affect water temperature. For example, the amount of reradiated heat back to the water column from solar radiation that penetrates the entire water column is controlled by the TSEDF parameter (table 4), a hydraulic coefficient. Another set of critical parameters includes the extinction coefficients (EXH2O, EXSS, EXOM, EXA1, EXA2, EXA3, EXA4, EXM1), which specify the water absorption of light and other ancillary extinction coefficients for organic suspended solids, inorganic suspended solids, algae, and macrophytes (table 4).

Madison Lake

In Madison Lake, the principal temperature calibration targets were data from several continuous profiles collected by thermistors at three depths in the epilimnion (1-, 3-, and 4-m depths), three depths in the transitional zone between the epilimnion and the hypolimnion (5.5-, 7-, and 9-m depths), and three depths in the hypolimnion (11-, 13-, and 16.5-m depths) at the southwest deep point site (figs. 8–10; table 5). Eight of the nine depths had MAE and RMSE values <0.90 and <1.10 °C, respectively. For most of the continuous profile depths, the simulated temperatures were approximately 1.0 to 1.5 °C warmer than the measured temperatures until late July. After late July, the simulated and measured temperatures tracked closely to each other with the exception of the 11-m depth; the 11-m depth took longer to equilibrate to the measured temperature. Overall, the deeper locations (transitional zone and the hypolimnion) shown in figures 9 and 10 (with the exception of 11-m depth, fig. 10), had simulated profiles that approximated the measured temperatures with better accuracy than the shallower locations (fig. 8). The MAE values ranged from 0.36 to 0.73 °C and the RMSE values ranged from 0.51 to 0.94 °C for the depths at or below 5.5 m, not including the 11-m depth. For the epilimnion depths shown in figure 8, the CE-QUAL-W2 model's inherent thermal stability does not as easily account for short-term shifts in shallow mixing in comparison to long-term shifts, which could have accounted for temperature offsets between simulated and measured values. The WSC was adjusted from 0.50 to 0.95 to try to compensate for the shallow temperature offsets; however, these adjustments were used sparingly given the lack of measured data to support frequent adjustments in the WSC. Of the three different depth classes, the transitional zones shown in figure 9 had the lowest MAE and RMSE values; the transitional zone between the epilimnion and hypolimnion generally can have larger deviations between simulated and measured values, so these low statistical measures demonstrated the strong temperature calibration for Madison Lake.

Secondary calibration targets for Madison Lake were several continuous thermistors at one depth in the epilimnion (1-m depths) and two depths in the transitional zone between

Table 5. Summary of values of mean absolute error (MAE) and root mean square error (RMSE) for calibration runs for Madison Lake and Pearl Lake.

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

Constituent	Site name	Common name in report	Depth (meters)	Number of compared data points	MAE	RMSE
		Madison Lake				
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	1	170	0.82	1.07
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	3	170	0.78	1.02
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	4	170	0.71	0.92
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	5.5	170	0.73	0.94
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	7	170	0.57	0.72
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	9	170	0.52	0.66
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	11	170	1.50	1.83
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	13	170	0.65	0.83
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	16.5	170	0.36	0.51
Water temperature, °C	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	Multiple	103	0.53	0.68
Water temperature, °C	Madison Lake northeast deep point near Madison Lake, Minn.	Northeast deep point	1	70	1.23	1.40
Water temperature, °C	Madison Lake northeast deep point near Madison Lake, Minn.	Northeast deep point	5	70	0.94	1.14
Water temperature, °C	Madison Lake northeast deep point near Madison Lake, Minn.	Northeast deep point	9	70	1.28	1.48
Dissolved oxygen, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	Multiple	103	0.68	1.15
Chlorophyll *a*, µg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	22.2	26.4
Chlorophyll *a*, µg/L	Madison Lake northeast deep point near Madison Lake, Minn.	Northeast deep point	2	5	25.9	30.6
Dissolved orthophosphate as phosphorus, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	0.01	0.02
Dissolved ammonia as nitrogen, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	0.18	0.31
Dissolved nitrate plus nitrite as nitrogen, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	0.05	0.08

the epilimnion and the hypolimnion (5- and 9-m depths) at the northeast deep point site (fig. 11; table 5). Similar to the southwest deep point site, the shallower simulated depth in the epilimnion deviated from the measured values more than the deeper profiles. The MAE values for all three depths ranged from 0.94 to 1.28 °C; RMSE values for all three depths ranged from 1.14 to 1.48 °C. Part of the reason for the larger MAE and RMSE values was that the temperature records at the northeast deep point site were only available until late July. The same period for the southwest deep point site showed more deviation between simulated and measured temperature than the latter portion of the year.

Table 5. Summary of values of mean absolute error (MAE) and root mean square error (RMSE) for calibration runs for Madison Lake and Pearl Lake.—Continued

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

Constituent	Site name	Common name in report	Depth (meters)	Number of compared data points	MAE	RMSE
Total Kjeldahl as nitrogen, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	0.29	0.33
Total phosphorus as phosphorus, µg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	82	86
Total phosphorus as phosphorus, µg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	16.5	5	54	69
Total phosphorus as phosphorus, µg/L	Madison Lake northeast deep point near Madison Lake, Minn.	Northeast deep point	2	5	79	80
Bacillariophyta/crysophyta, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	0.74	1.00
Chlorophyta, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	0.18	0.20
Cyanophyta, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	1.81	1.95
Haptophyta/cryptophyta, mg/L	Madison Lake southwest deep point near Madison Lake, Minn.	Southwest deep point	2	5	0.13	0.15
		Pearl Lake				
Water temperature, °C	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	1.6	184	0.60	0.78
Water temperature, °C	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2.1	184	0.63	0.80
Water temperature, °C	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2.8	184	0.58	0.78
Water temperature, °C	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	3.8	184	0.57	0.72
Water temperature, °C	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	4.8	184	0.70	0.97
Water temperature, °C	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	5.4	184	0.87	1.22
Water temperature, °C	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	Multiple	30	0.71	0.95
Dissolved oxygen, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	Multiple	30	1.17	1.98
Chlorophyll *a*, µg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	0.93	1.16
Dissolved orthophosphate as phosphorus, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	<0.01	<0.01
Dissolved ammonia as nitrogen, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	0.02	0.02
Dissolved nitrate plus nitrite as nitrogen, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	0.03	0.04
Total Kjeldahl as nitrogen, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	0.17	0.19
Total phosphorus as phosphorus, µg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	6	7
Total phosphorus as phosphorus, µg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	4.5	6	13	17
Bacillariophyta/crysophyta, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	0.08	0.10
Chlorophyta, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	0.05	0.06
Cyanophyta, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	0.22	0.28
Haptophyta/cryptophyta, mg/L	Pearl Lake deep point near Marty, Minn.	Pearl Lake deep point	2	6	0.13	0.14

Simulated water temperatures in Madison Lake were compared to vertical profiles of lake water temperatures at the southwest deep point site, generally collected during MNDNR water-quality sampling trips. A total of eight dates are shown in figure 12. Similar to the continuous temperature profiles, low MAE and RMSE values provided additional confidence in the model's ability to predict water temperature. For Madison Lake, the model consistently attained MAE and RMSE values <1.0 °C for all eight dates, with several values <0.5 °C. For the combined vertical profiles, the MAE and RMSE values were 0.53 and 0.68 °C, respectively (table 5). In addition, the location and slope of the simulated thermocline matched the measured thermocline. Similar to the continuous temperature profiles for depths of 1 to 16.5 m (table 5), the simulated temperatures for Madison Lake were warmer than the measured temperatures for the earlier vertical profiles through the July 9 vertical profile (fig. 12).

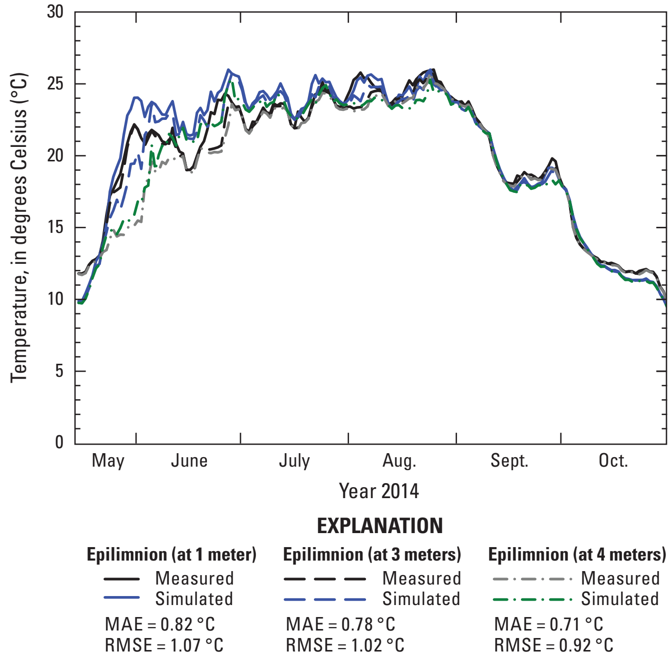


Figure 8. Simulated and measured water temperature for the three different depths (1, 3, and 4 meters) in the epilimnion at Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

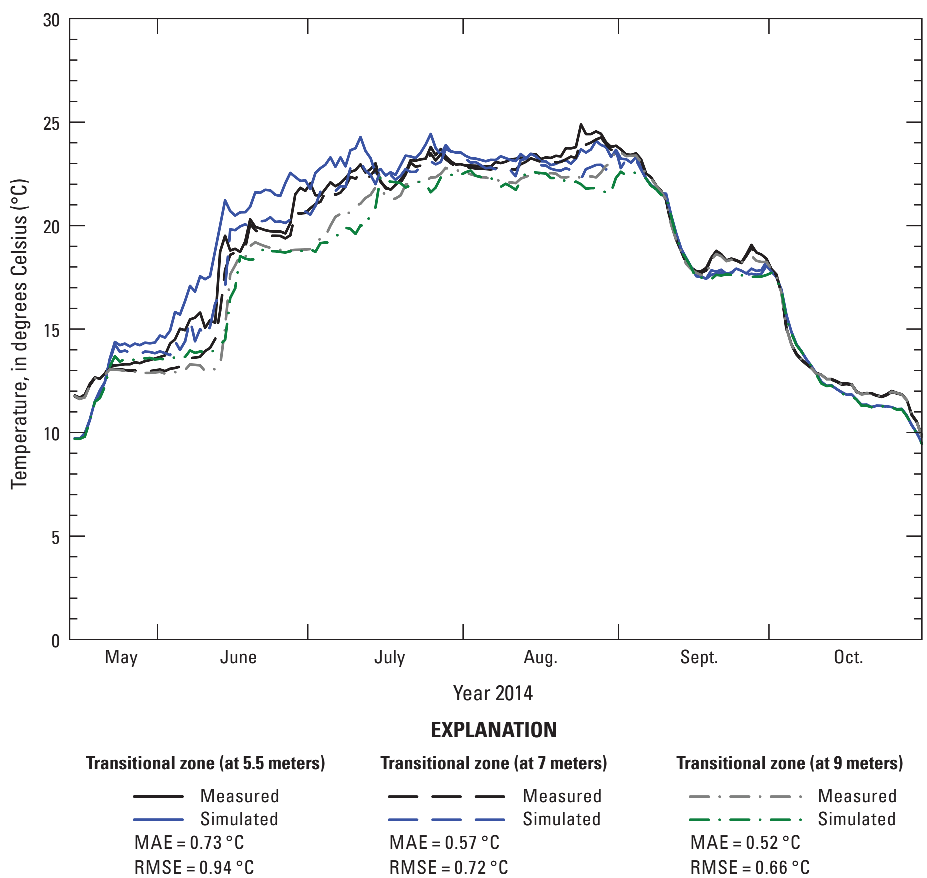


Figure 9. Simulated and measured water temperature for the three different depths (5.5, 7, and 9 meters) in the transitional zone between the epilimnion and the hypolimnion at Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

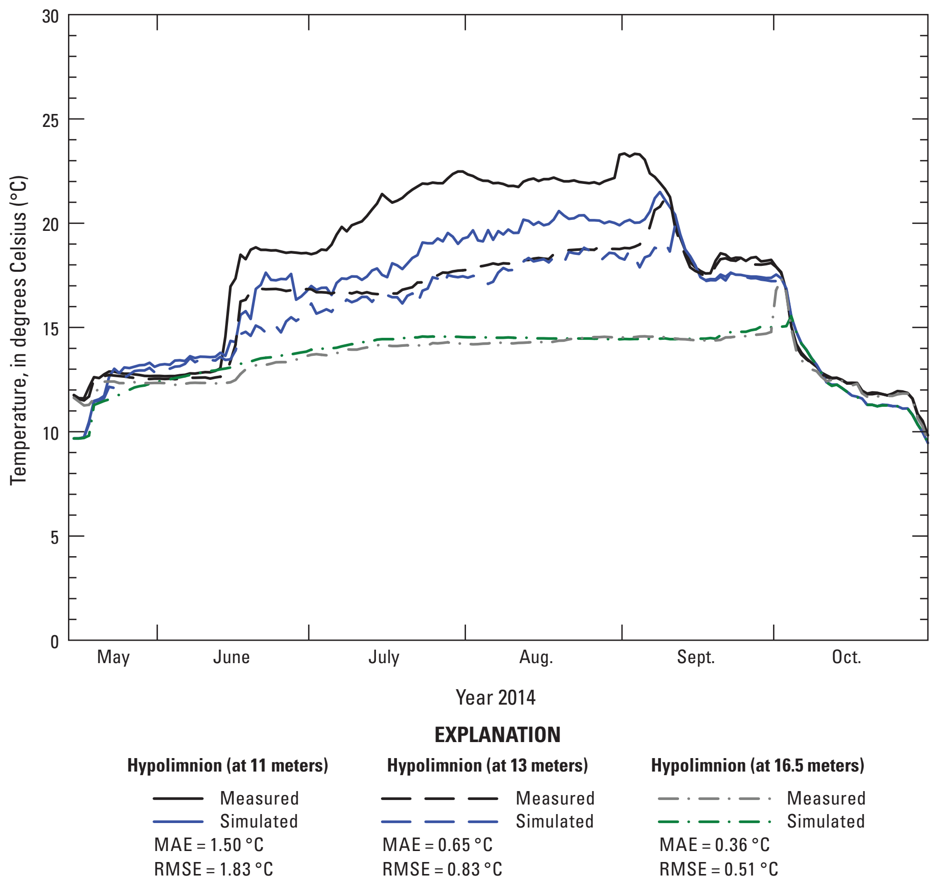


Figure 10. Simulated and measured water temperature for the three different depths (11, 13, and 16.5 meters) in the hypolimnion at Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

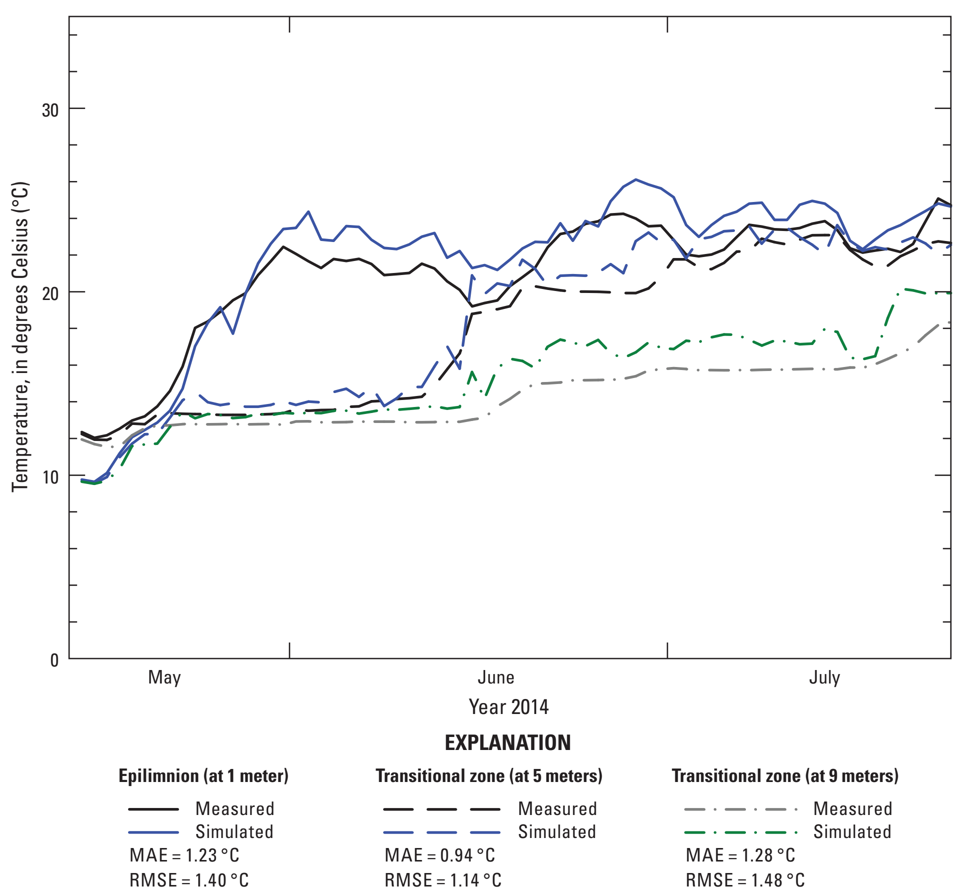


Figure 11. Simulated and measured water temperature for three different depths (1, 5, and 9 meters) for the Madison Lake northeast deep point near Madison Lake, Minnesota, May 15 to July 23, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

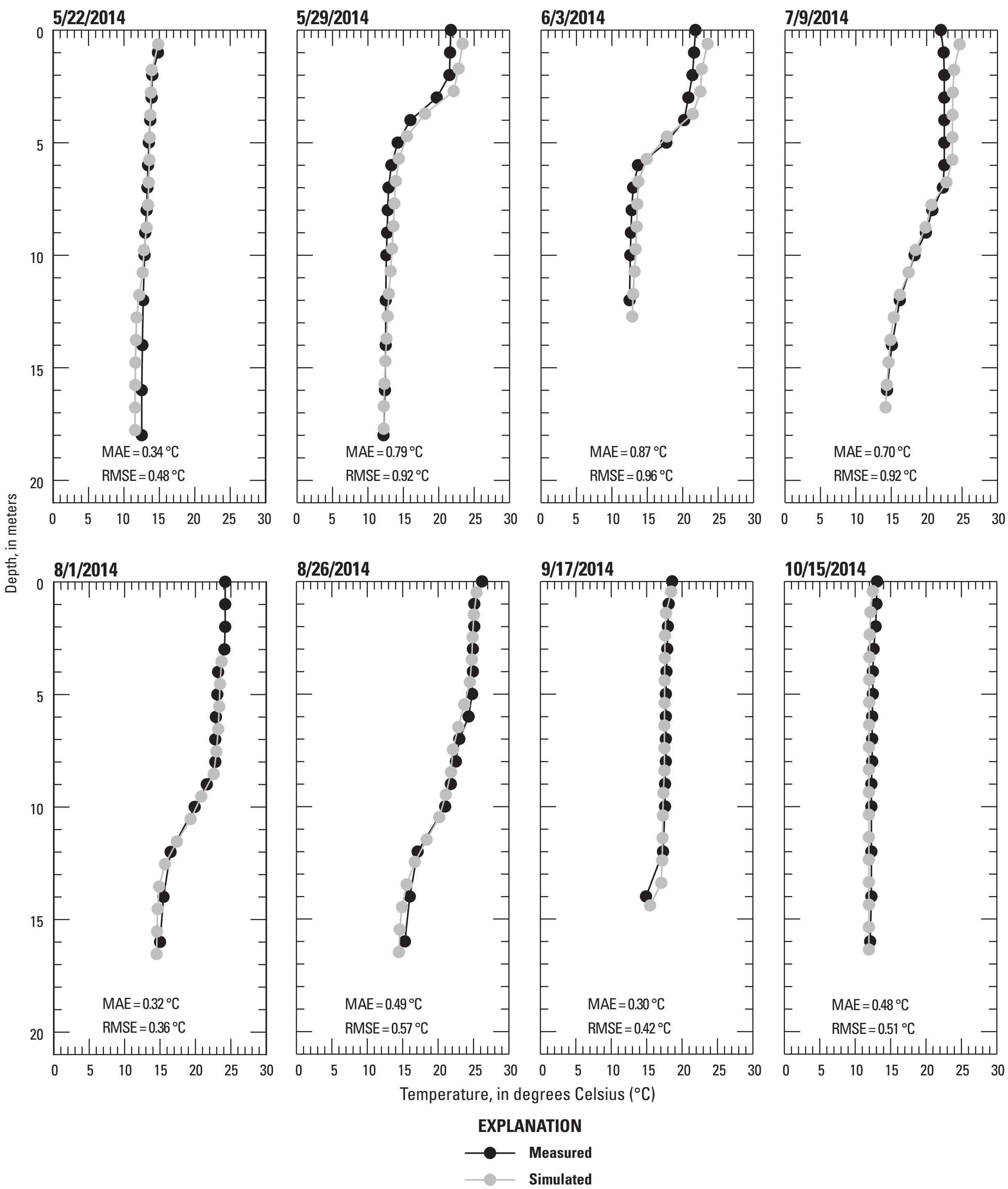


Figure 12. Simulated and measured water temperature for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minnesota, for eight dates in 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Pearl Lake

In Pearl Lake, the principal temperature calibration targets were data from several continuous profiles collected by thermistors at three depths in the shallow mixed layer (1.6-, 2.1-, and 2.8-m depths) and three depths in the deep mixed layer (3.8-, 4.8-, and 5.4-m depths) at the Pearl Lake deep point site (figs. 13 and 14, respectively; table 5). Five of the six depths had MAE and RMSE values <0.80 and <1.00 °C, respectively. For all six depths, the MAE values ranged from 0.57 to 0.87 °C, and the RMSE values ranged from 0.72 to 1.22 °C. Throughout the year, temperatures were well-mixed in Pearl Lake, and the simulated temperatures tracked closely to the measured temperatures. Given the shallow depth of Pearl Lake compared to Madison Lake, wind mixing seemed to be able to account for the minor differences near the lake bottom (figs. 13 and 14). The largest difference between the simulated and measured temperatures occurred for the deepest mixed layer location (5.4 m) in early August (fig. 14). The WSC was adjusted from 0.80 to 1.00 to try to account for the small temperature offsets at depth; however, compared to the larger adjustment range for Madison Lake, these adjustments were small and lend theoretical support to the ease of shallow lake mixing.

Simulated water temperatures in Pearl Lake also were compared to vertical profiles of lake water temperatures at the Pearl Lake deep point collected by the MNDNR. A total of six dates are shown in figure 15. Similar to the Madison Lake continuous temperature profiles, low MAE and RMSE values provided additional confidence in the model's ability to predict water temperature. The calibration fit between measured and simulated temperatures earlier in 2014 was not as close as the fit for Madison Lake, particularly in the deeper portion of the mixed layer near the lake bottom. Similar to the deepest continuous profile at 5.4 m (fig. 14), the shallow depths of Pearl Lake caused wind energy to mix warmer surface water too efficiently to the bottom of the lake. Therefore, the warmer temperatures near the bottom of the lake could have been dissipated by a lower WSC; however, the cooler temperatures would have further offset the DO concentrations at depth as described in the "Dissolved Oxygen" section. For Pearl Lake, the last four temperature vertical profiles (after June 24) had consistently low MAE and RMSE values <1.0 °C, whereas the first two dates (May 27 and June 24) were greater than or equal to 1.0 °C (fig. 15). For the combined vertical profiles, the MAE and RMSE values were 0.71 and 0.95 °C, respectively (table 5).

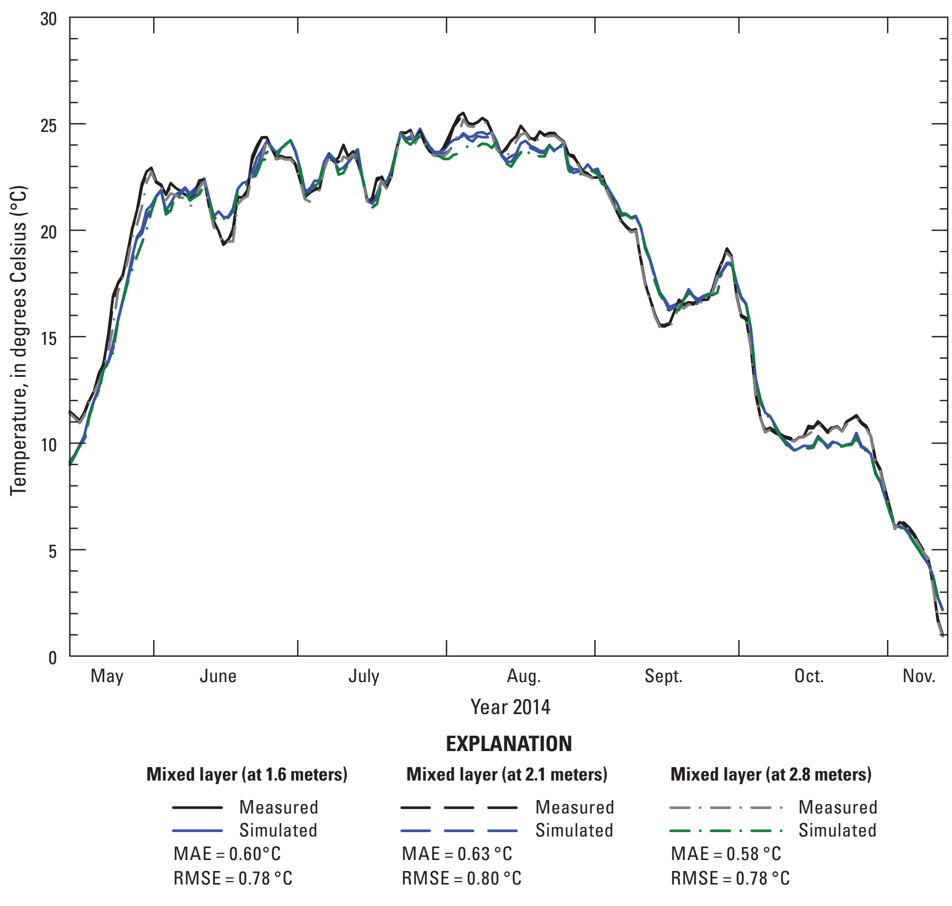


Figure 13. Simulated and measured water temperature for the three different depths (1.6, 2.1, and 2.8 meters) in the shallow mixed layer at the Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

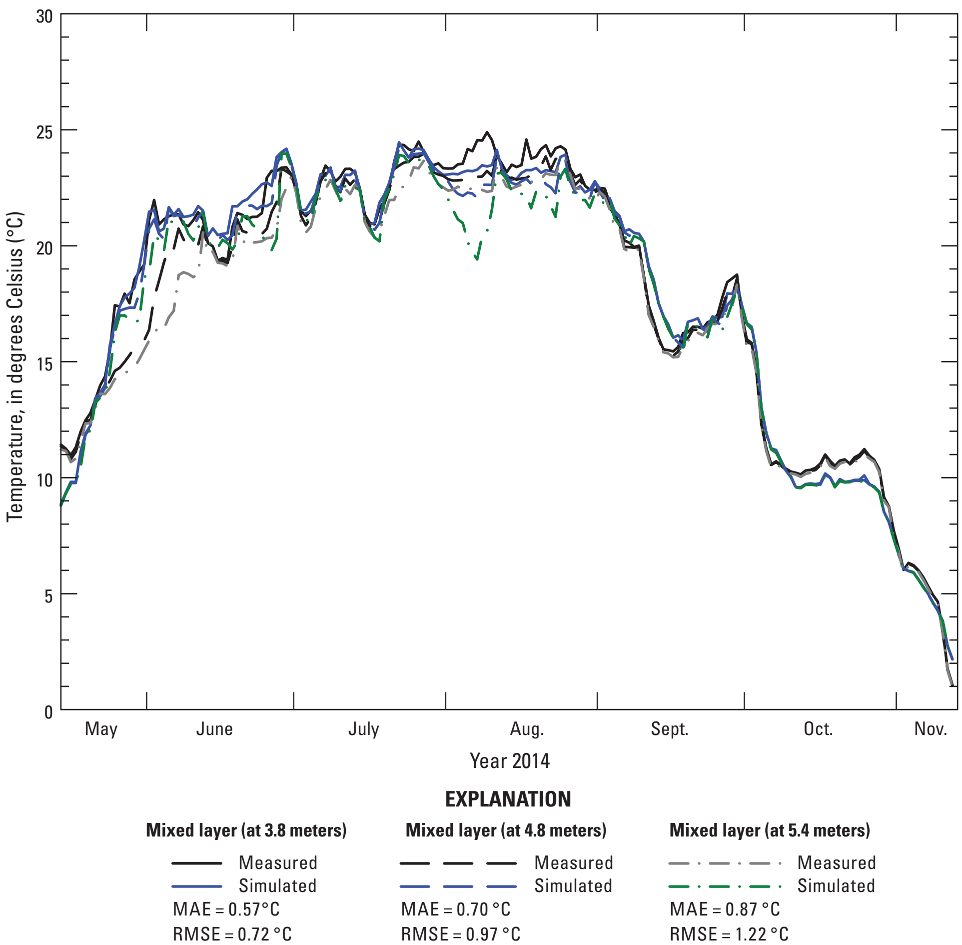


Figure 14. Simulated and measured water temperature for the three different depths (3.8, 4.8, and 5.4 meters) in the deep mixed layer at the Pearl Lake Deep Point near Marty, Minnesota, May 14, to November 13, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Dissolved Oxygen

Fish species and other aquatic organisms cannot survive without adequate DO. 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. For example, Jacobson and others (2008) evaluated the lethal oxythermal niche boundary for ciscoes in several Minnesota lakes and determined that lethal temperatures decreased for lower lethal DO concentrations. 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. Along with temperature, DO is a key metric to illustrate the accuracy of the model's calibration. 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 listed in table 4. With such complex interactions, especially when simultaneously trying to dynamically model algal communities, several hundred iterations were required for both of the final lake CE-QUAL-W2 models.

With varying success, the Madison Lake and Pearl Lake models captured the trajectories of DO concentrations at multiple depths over time, which indicated that the models were accurately simulating the underlying metabolic processes in each lake. Specific examples of the model capabilities presented in the following subsections include comparisons between simulated and measured vertical profile data for the midwater oxygen maximum earlier in the year for Madison Lake and the declining trend of DO with depth in Pearl Lake. Both cases illustrated that the internal trophic dynamics for these lakes are substantial factors affecting much of the observed biogeochemistry.

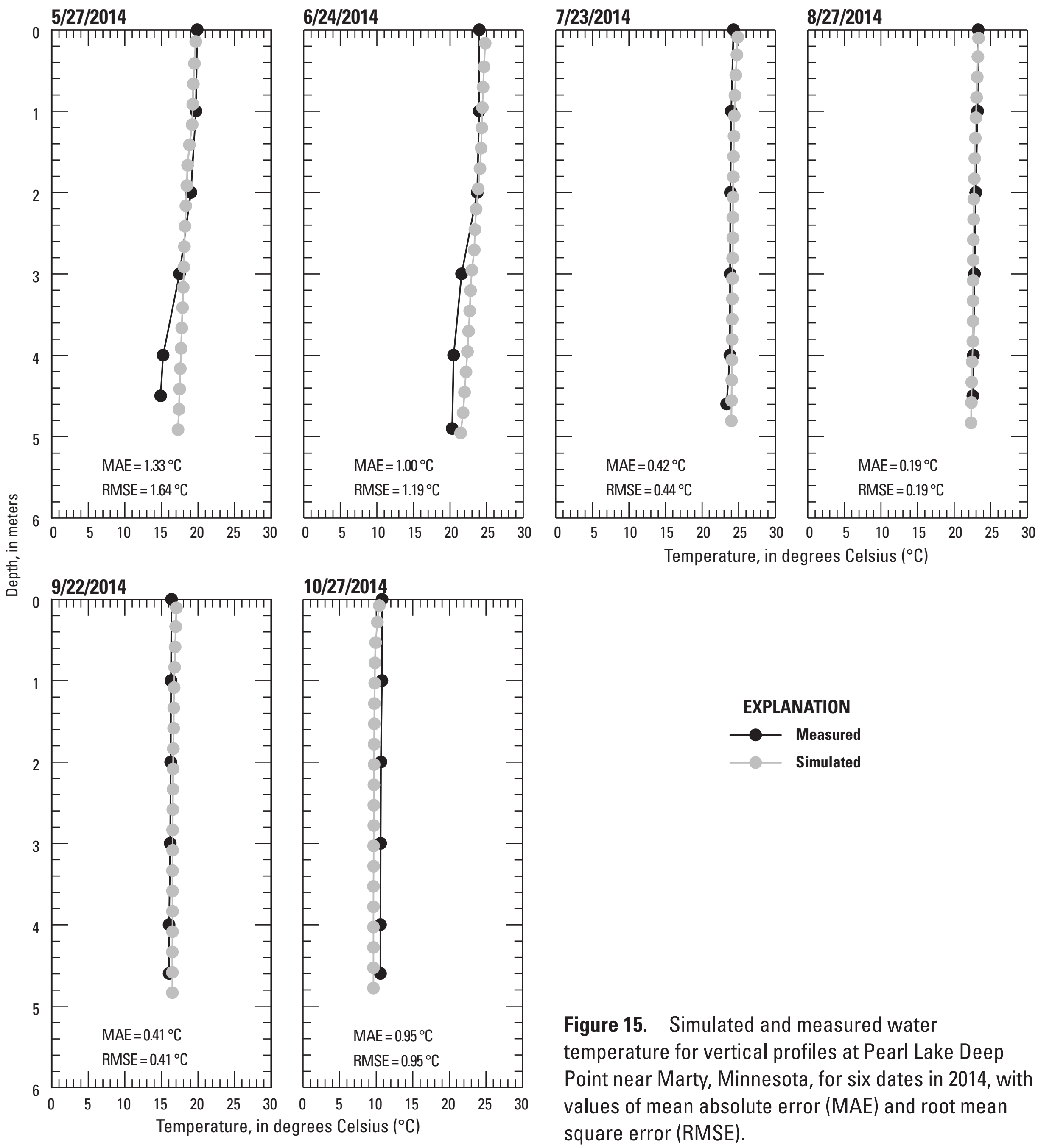


Figure 15. Simulated and measured water temperature for vertical profiles at Pearl Lake Deep Point near Marty, Minnesota, for six dates in 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Madison Lake

For the DO calibration of the Madison Lake model, 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 in 2014. Generally, DO measurements were recorded for each meter below water surface. Simulated and measured DO concentrations are shown for a total of eight dates in figure 16. Overall, the simulated DO concentrations tracked the measured concentrations from the southwest deep point site. Generally, where the greatest change in DO occurred, the simulated concentrations matched the depth and slope of the measured concentrations. 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 (<0.4 mg/L). The same simulated maximum DO was still shown on June 3, but the measured values reflected more mixing from the surface to approximately 5 m by this time, causing larger MAE and RMSE values (greater than 1.0 mg/L). Essentially, the simulated DO profiles preserved the greater midwater maximum on June 3 but also had a larger hypolimnetic oxygen deficit. The deterioration of the hypolimnetic oxygen levels occurred between May 29 and June 3; these minimal hypolimnion oxygen levels 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. 16) and lake water temperature (fig. 12). 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. With the last profile on October 15, the differences between the simulated and measured DO concentrations were close and consistently about 6 mg/L throughout the entire water column; the MAE and RMSE values for this date were 0.22 and 0.29 mg/L, respectively (fig. 16). For the combined vertical profiles, the MAE and RMSE values were 0.68 and 1.15 mg/L, respectively (table 5).

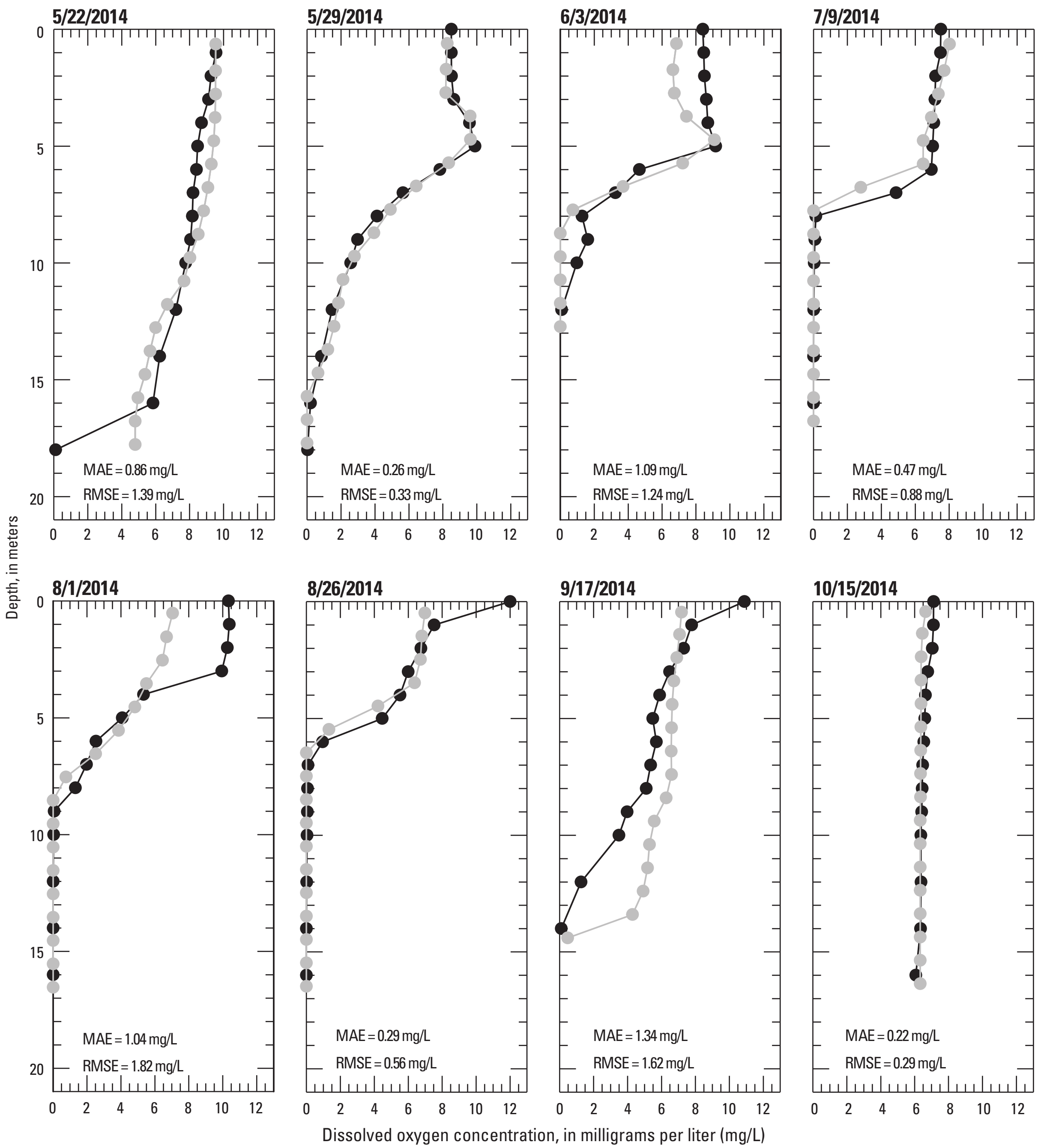


Figure 16. Simulated and measured dissolved oxygen concentration for vertical profiles at Madison Lake southwest deep point near Madison Lake, Minnesota, for eight dates in 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

A complex interaction between processes has a strong effect on limnological DO concentrations. For example, increased algal production tends to increase lake DO concentrations, whereas heterotrophic consumption of the larger algal blooms increases oxygen demand (thereby depleting oxygen) (Wetzel, 2001). The decay rates of the different organic matter pools, such as parameters that control the labile, refractory, and the labile-to-refractory DO matter decay rates (parameters LDOMDK, RDOMDK, and LRDDK, respectively, in table 4), were low compared to previous rates used for other lake models such as Lake Carlos and Elk Lake (Smith and others, 2014); however, the organic matter decay rates were similar to rates used for Trout Lake (Smith and others, 2014). Decay rates have the strongest effect on the DO concentrations in the hypolimnion. Sediment oxygen demand (parameter SOD, table 4) also was large for Madison Lake, set at 2.5 mg/L, which can greatly alter the DO profiles in the entire lake but particularly in the hypolimnion. The nitrate decay rate (parameter NO3DK, table 4) was set to 0.0575 per day, which is similar to the CE-QUAL-W2 default rate (Cole and Wells, 2015). Transitions between different algal communities affected DO, which are described in the “Algae” section. Algal dynamics played a large part in altering the DO dynamics, and given that Madison Lake calibration data existed for four different algal divisions (bacillariophyta/crysophyta [diatoms], chlorophyta [green algae], cyanophyta [blue-green algae], and haptophyta/cryptophyta [flagellates]) (appendix table 2–1), a large effort was spent in the algal community calibration while preserving DO profile dynamics.

Pearl Lake

For the DO calibration of the Pearl Lake model, the principal calibration targets were the lake profile data from the Pearl Lake deep point site, available from monthly vertical DO profiles collected by MNDNR personnel during water-quality sampling trips in 2014. The DO measurements were recorded for each meter below the water surface. Simulated and measured DO concentrations are shown for a total of six dates in figure 17. Overall, the simulated DO concentrations were similar to the measured concentrations from the Pearl Lake deep point site, particularly the four profiles from July until October. Throughout the entire calibration period, simulated DO decreases with depth were restrained, which compared well to the measured data later in the year but caused large offsets for the first 2 months. The MAE and RMSE values were large on May 27 (2.81 and 2.92 mg/L, respectively), and on June 24, the MAE and RMSE values were even larger (3.11 and 3.79 mg/L, respectively). Later in the year, the simulated concentrations matched the depth and shallow slope of the measured DO concentrations, with MAE and RMSE values <0.5 mg/L. For the combined profiles, the MAE and RMSE values were 1.17 and 1.98 mg/L, respectively (table 5); however, excluding the first two profiles, the MAE and RMSE values were 0.28 and 0.38 mg/L, respectively.

Although the general trend towards slightly lower DO concentrations in the deeper portions of the mixed layer, and therefore the lake, was simulated by the model, several potential causes existed for the poor model fit for the May 27 and June 24 vertical DO profiles. These potential causes included the lack of simulated algal growth that would cause DO supersaturation in the shallower mixed layer, lack of organic matter decomposition in the deeper portions of the lake, inadequately low macrophyte respiration rates, large overwinter SOD, and thorough wind mixing of DO by the CE-QUAL-W2 model. Of the potential causes, simulated DO supersaturation in the shallow mixed layer was not supported by the measured algal biomass or chlorophyll *a* concentrations. It was possible that an earlier bloom of green or blue-green algae was missed or not captured with the measured data, and therefore a greater algal biomass simulation of at least one of these groups would have caused larger DO values in the shallow mixed layer. Also, a quick die off of these early algal blooms would have quickly sank towards the bottom of the lake and caused organic matter decomposition, leading to simulated hypoxic to anoxic conditions that were measured in late May and June. The simulated DO also could have been better simulated with a greater initial concentration of organic matter in the lake, leading to more decomposition, but this was only weakly supported by the measured data. A third mechanism for better simulated DO profiles earlier in the year would be greater respiration rates for algae and macrophytes, which could be better explored with a calibration dataset of macrophyte growth. Large overwinter SOD also could cause measured DO concentrations well below the simulated concentrations because the model does not have the capability with the zero-order SOD model to dynamically alter SOD rates. For some lakes with large organic matter decompositions, such as Pearl Lake, overwinter SOD can be very large before wind mixing fully mixes the lake (Cross and Summerfelt, 1987). The final mechanism that could explain the lack of fit earlier in the year is thorough wind mixing of DO by the CE-QUAL-W2 model. The ability of the model to simulate greater DO in the shallow mixed layer while simultaneously simulating hypoxic to anoxic conditions in the deeper mixed layer, in a shallow lake of only 5.5 m, might be limited.

Algae

The paradigm of four general algal communities or groups was pursued rather than a more diverse species-specific modeling regime. This was partially because algal group calibration beyond four groups can be problematic for a CE-QUAL-W2 model given the sensitivity to algal group dynamics and the uncertainty in model parameterization beyond four different algal groups (Cole and Wells, 2015). The four algal groups or divisions included were (1) bacillariophyta and crysophyta (hereafter referred to as “diatoms”); (2) chlorophyta (green algae; hereafter referred to as “green algae”); (3) cyanophyta (blue-green algae; hereafter referred to as “blue-green algae”); and (4) haptophyta and cryptophyta (hereafter referred to as “flagellates”). Rather than including zooplankton as a separate group or groups, the zooplankton grazing dynamics were captured within algal specific constants such as the algal growth rate (parameter AG, table 4) and the algal mortality rate (parameter AM, table 4). Algal growth temperature ranges and the fractions of growth within the temperature ranges (parameters AT1 through AT4 and parameters AK1 through AK4, table 4) were different across all four algal groups, as were the algal growth rates (parameter AG, table 4) and the light saturation intensities at the maximum photosynthetic rate (parameter ASAT, table 4). The main guidance for the algal groups was provided by other CE-QUAL-W2 modeling efforts, such as the previous sentinel lake models (Smith and others, 2014).

Madison Lake

The simulated distribution of four primary algal groups at 2 m below the water surface is shown in figure 18 for the model segment containing the southwest deep point site in Madison Lake (segment 7, fig. 2). Diatoms were the first group to peak, as shown with the Madison Lake simulated and

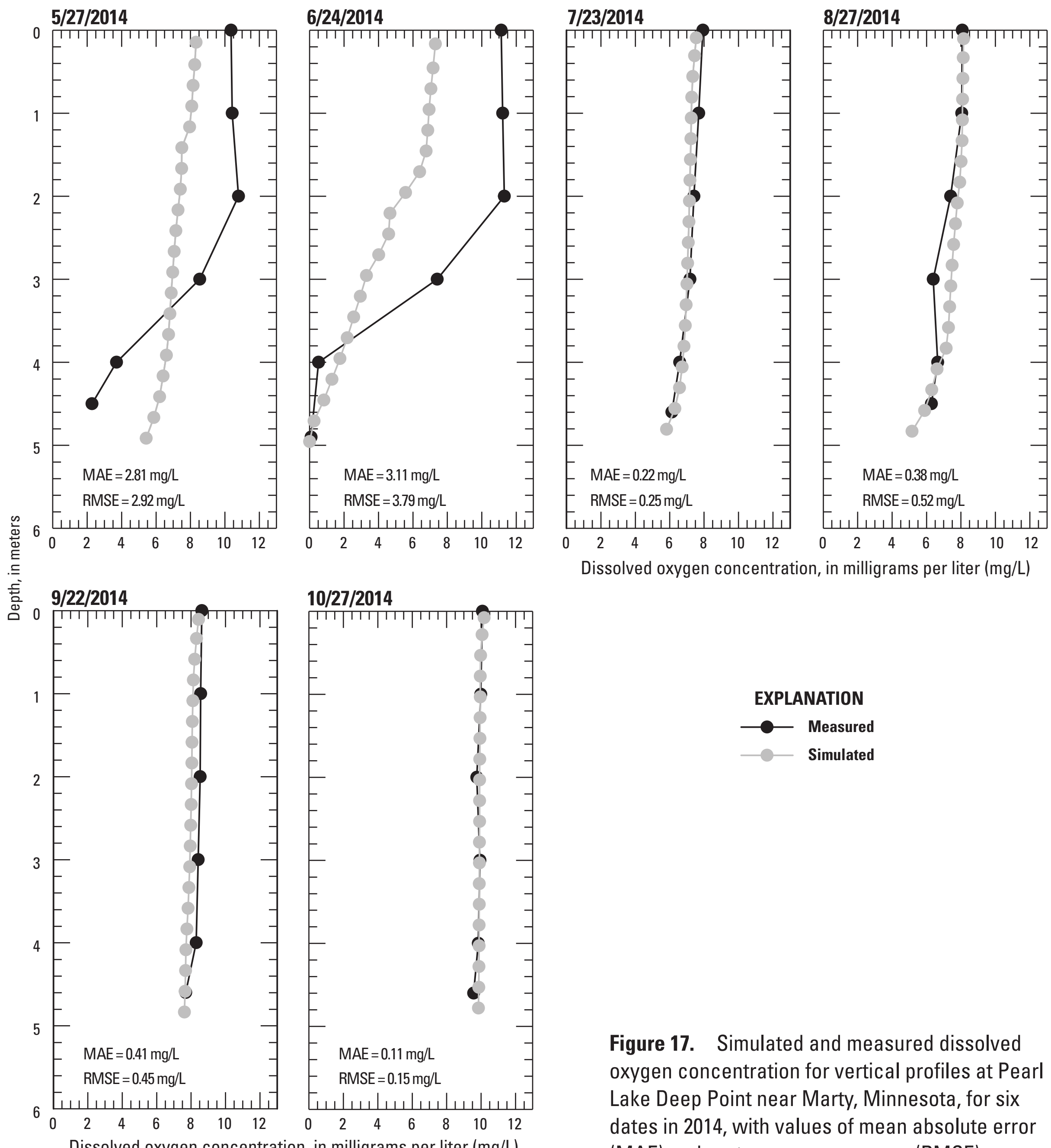


Figure 17. Simulated and measured dissolved oxygen concentration for vertical profiles at Pearl Lake Deep Point near Marty, Minnesota, for six dates in 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

measured values. Diatoms commonly peak earlier in the year (Sigee, 2005). The simulated diatom values peaked by the end 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, such as the growth rates that were tied to temperature ranges, the algal half-saturation constant for nitrogen-limited growth (parameter AHSN), the algal settling rate (parameter AS), and the algal light saturation intensity. The temperature range for diatom growth was lower than the other three algal groups, so once the lake warmed by early June, the diatoms were outcompeted by the other three 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.

Blue-green algae were the next group to succeed the diatoms. Similar to the other three groups, blue-green algae were simulated as one group despite several different species measured in the lake; splitting this group into more specific groups could be warranted in future modeling efforts to better explain the blue-green algal succession. For this calibrated Madison Lake model, the blue-green algae were parameterized to allow for greater growth rates. The algal half-saturation constants for nitrogen-limited growth (parameter AHSN, table 4, 0.002)

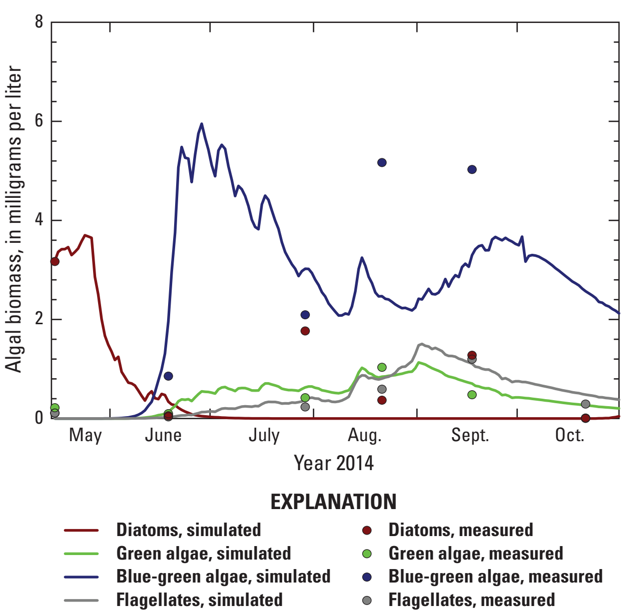


Figure 18. Simulated and measured algal group distributions (diatoms, green algae, blue-green algae, and flagellates) for the 2-meter depth at Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014.

were low relative to the other three groups, which ranged from 0.025 to 0.030, and were one of the explanatory variables for the large simulated blue-green values throughout most of the calibration period (table 4). Also, the blue-green algae had a wide temperature range for maximum algal growth (between 20 and 32 °C, table 4). The other three groups had narrow bands ranging from 4 to 6 °C wide. The blue-green algae also has a low algal light saturation intensity of 60 watts per square meter (W/m^2), similar to green algae but much lower than 150 W/m^2 for the diatoms.

The other two algal community groups, green algae and flagellates, had similar growth rates and patterns for the simulated and measured values. 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. 18). The maximum algal growth temperature range was similar for both groups, with 22 to 28 °C and 24 to 28 °C for the green algae and flagellates, respectively. Of the four groups, the flagellates had the lowest algal light saturation intensity of 20 W/m^2 (table 4). Otherwise, as shown in table 4, 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 were similar to measured algal biomass concentrations with the exception of the previously described deviation for diatoms later in the year. Also, the simulated blue-green algal concentrations did not match the large measured values in August and September. Part of the discrepancy for both groups is that algal growth is known to vary considerably through time for phytoplankton groups in nature (Marañón and others, 2000); additionally, one sample point in time might not capture the general trend over time.

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 19 (segment 7, fig. 2); additionally, the simulated and measured values of the chlorophyll *a* concentrations are shown for the northeast deep point site (segment 5, fig. 2). Measured chlorophyll *a* data primarily were collected in the surface layer at approximately 2 m below the water surface as part of the monthly MNDNR water-quality sampling trips. Overall, the simulated values were a fairly good approximation of the measured values, although this was not reflected with the large MAE and RMSE values (fig. 19; table 5). The peak measured values for segment 5 in September were not captured by the model; additionally, the low measured chlorophyll *a* concentrations in October were not captured by the model for either segment 5 or 7.

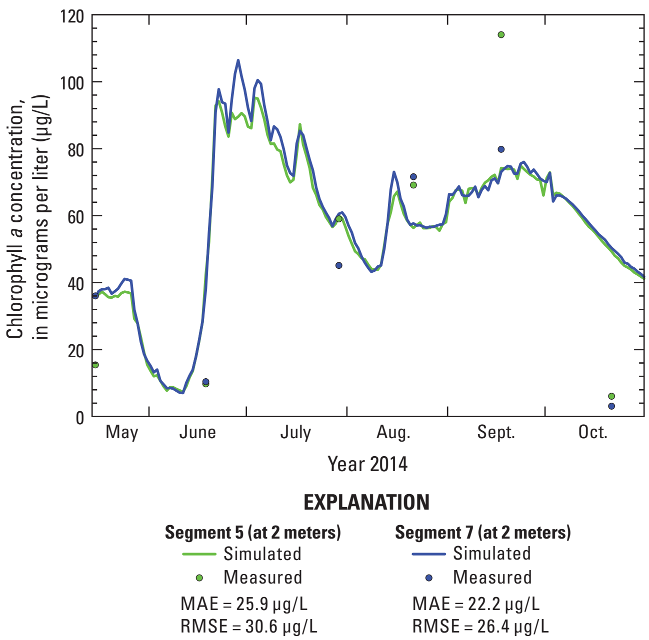


Figure 19. Simulated and measured chlorophyll *a* concentrations for the 2-meter depth at Madison Lake northeast deep point near Madison Lake, Minnesota, (segment 5) and Madison Lake southwest deep point near Madison Lake, Minnesota, (segment 7) in Madison Lake, May 15 to November 1, 2014.

Pearl Lake

The simulated distribution of four primary algal groups at 2 m below the water surface is shown in figure 20 for the model segment containing the Pearl Lake deep point site (segment 4, fig. 3). Compared to Madison Lake, Pearl Lake had a better fit between the simulated algal biomass to the measured algal biomass, as shown in figure 20 and indicated by the low MAE and RMSE values (table 5). The simulated distribution shows that all four groups started growing at approximately the same time period but at different rates. In Madison Lake, diatoms were the first group to peak, whereas in Pearl Lake, three of the four groups peaked at approximately 0.5 mg/L in early to mid-June, with the exception of a low abundance of green algae; however, the blue-green algae continue to peak and dominate the lake algal biomass for the entire summer, which also was supported with the measured algal biomass data. Green algae began to peak in mid-September, before beginning to disappear at approximately the same rate as blue-green algae and the diatoms. The final group to peak, late in the simulation period, was the flagellates in early October.

Several factors controlled the dominance of the simulated blue-green algae; these factors also are potential explanations for the measured blue-green algae blooms. Pearl Lake, in comparison to Madison Lake, has greater clarity based on Secchi depths. The mean Secchi depth was 2.0 m for Pearl Lake in 2008–9 (Anderson and others, 2012), whereas the mean depth for Madison Lake was between 0.25 and 0.40 m in 2006 and 2008 (Lindon and others, 2010). With the greater light penetration during the summer months, the blue-green algae were parameterized as higher light specialists with the other three groups set up as low light specialists by adjusting the light saturation intensity (parameter ASAT, table 4). The blue-green algae had a large algal light saturation intensity of 120 W/m^2, compared to a range of 30 to 45 W/m^2 for the other three groups. The low algal half-saturation constants for blue-green algae for nitrogen-limited growth (0.002) relative to the other three groups, which ranged from 0.01 to 0.014, is another possible factor explaining the large simulated blue-green values throughout most of the simulation cycle. Blue-green algae had a wide temperature range for maximum algal growth, between 17 and 28 °C, which covered the lake water temperature range for most of the summer months.

Similar to Madison Lake, the chlorophyll *a* concentration data for Pearl Lake were used as a secondary check on the simulated algal biomass concentrations. Simulated and measured values of the chlorophyll *a* concentrations are shown in figure 21 for the Pearl Lake deep point site. The Pearl Lake chlorophyll *a* data were collected with the same method and at the same depth (2 m below the water surface) as Madison Lake. Overall, the simulated values were a close approximation of the measured values, also reflected with the low MAE and RMSE values (fig. 21; table 5) of 0.93 and 1.16 micrograms per liter (µg/L), respectively.

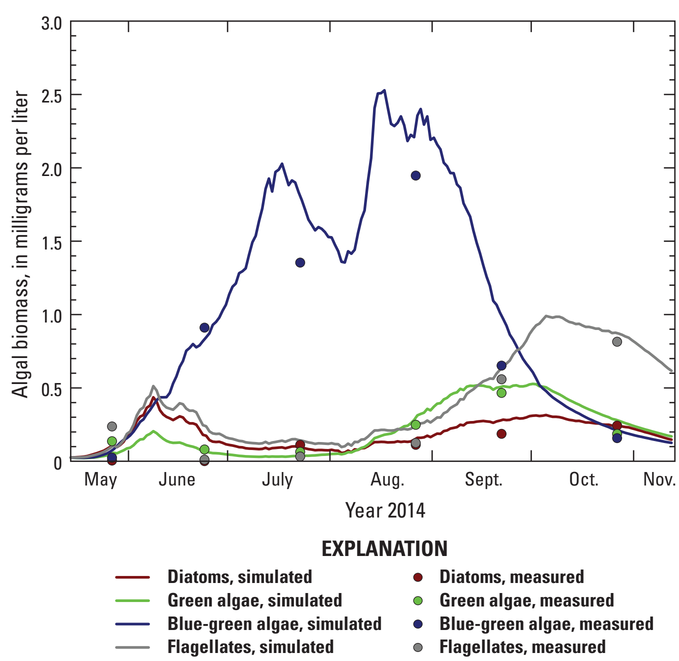


Figure 20. Simulated and measured algal group distributions (diatoms, green algae, blue-green algae, and flagellates) for the 2-meter depth at Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014.

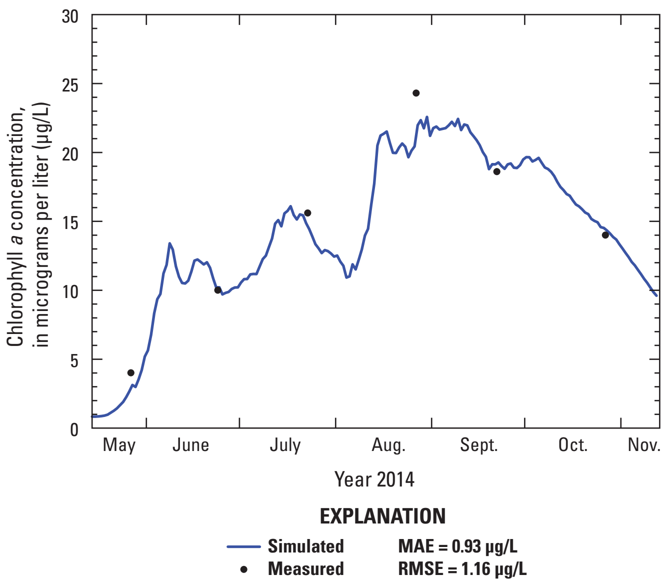


Figure 21. Simulated and measured chlorophyll *a* concentrations for the 2-meter depth at Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Macrophyte Growth

For Madison Lake and Pearl Lake, the macrophyte growth model was run to account for the large macrophyte growth documented on both lakes (Lindon and others, 2010; Anderson and others, 2012). Although there was not a calibration dataset, inclusion of modeled macrophyte growth could account for more realistically simulated nutrient and DO dynamics. As shown in table 4, most of the macrophyte growth parameters were kept at default rates with the exception of 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). For Pearl Lake, a greater percentage of littoral area exists and has well-documented macrophyte communities throughout the lake (Anderson and others, 2012); therefore, these parameters were adjusted to allow for a greater amount of macrophyte growth in Pearl Lake in comparison to Madison Lake. In addition, with Madison Lake, the parameters were optimized in such a way to account for the greater macrophyte growth documented in the littoral areas (Lindon and others, 2010).

Nutrients

Nutrients in both lakes 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 the season with a larger flux of ammonia later in the year, whereas Pearl Lake had a larger flux of nitrate mid-summer without the large mid-summer flux of ammonia. These loads 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). In-lake processing of the nutrients is the major factor controlling nutrient concentrations. The focus for evaluating the model calibration was three constituents of nitrogen and two constituents of phosphorus: nitrate plus nitrite, ammonia, total Kjeldahl nitrogen, orthophosphate, and total phosphorus.

Sources and sinks are largely the same for both lakes. Madison Lake and Pearl Lake have fairly small flows from two different inflows and seem 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 both lakes (Lindon and others, 2010; Anderson and others, 2012). An important distinction between the two agricultural lakes is the ratio of the drainage basin to lake area, which is 24:1 for Pearl Lake but only 4:1 for Madison Lake. Also, the forest land-use percentage is different between the two different lakes: the Pearl Lake drainage basin has a forest cover of 15 percent, whereas the Madison Lake drainage basin has a forest cover of 2 percent. Generally, basins with a larger percentage of forest or other undeveloped land cover will have lower nutrient loads relative to basins with a larger ratio of agricultural land use (U.S. Geological Survey, 1999).

For nitrate plus nitrite, sources include all inflows and ammonia nitrification; sinks include denitrification (in the water column and sediments), algal uptake, and lake outflow (Cole and Wells, 2015). For ammonia as nitrogen (ammonia [NH_3] and ammonium [NH_4^+]), sources include all inflows, decay of all organic nitrogen pools, sediment release under anaerobic conditions, and algal respiration; sinks include nitrification, algal uptake, and lake outflow (Cole and Wells, 2015). For orthophosphate, sources include all inflows, decay of all organic matter pools, sediment release under anaerobic conditions, and algal respiration; sinks include particles settling with adsorbed phosphorus, algal uptake, and lake outflow (Cole and Wells, 2015). 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). 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 in Madison Lake and six samples in Pearl Lake).

Madison Lake

Dissolved ammonia and dissolved nitrate plus nitrite distributions in Madison Lake were largely affected by the inflows and the lake hydrodynamics. The simulated and measured concentrations of dissolved ammonia as nitrogen at 2 m below the water surface are shown in figure 22 for the model segment containing the southwest deep point site in Madison Lake (segment 7, fig. 2). Few differences in the measured dissolved ammonia concentrations were noted among the epilimnion locations (2-m depth) in the lake from July through September. The measured value in June was greater than the mid- to late summer measured values, and the late October sample had the largest measured dissolved ammonia concentration for the entire year. Algal uptake of available ammonia was fairly rapid in the simulation and actual lake, 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. Compared to earlier in the year, inflow loads were also low during the mid- to late summer period. Earlier in the year, algal growth rates in the simulation and the actual lake were not large enough to incorporate the available ammonia, so the simulated and the single measured values in June were greater than in May. Late in the model simulation, the simulation indicated that ammonia began to accumulate in the lake but could not account for the large concentration of 0.88 mg/L measured on October 21. To account for this discrepancy, an adjustment could have been made to the stoichiometric equivalent of nitrogen for the different algal groups to artificially low values, the ammonia release rate from sediments could have been set higher, or the ammonia decay rate could have been lowered; however, any of these adjustments would have caused the other simulated values to be too large in comparison to the four other measured values. The MAE and RMSE values for dissolved ammonia were large compared to the other sentinel lake simulations (Smith and others, 2014), mostly due to the mismatch to the late October sample; the MAE and RMSE values were 0.18 and 0.31 mg/L, respectively (fig. 22; table 5).

Simulated and measured dissolved nitrate plus nitrite concentrations are shown in figure 23 for the Madison Lake southwest deep point site. Additional nitrogen depletion was simulated for dissolved nitrate plus nitrite concentrations during the mid- to late summer period that only started to recover towards the end of the simulation period in October, although this was not supported by the measured data. The simulated increase in dissolved nitrate plus nitrite concentrations that occurred in early to mid-June was caused by greater nitrate concentrations in the two inflows and by ammonia nitrification of the earlier ammonia influx in the late spring. Towards the end of the simulation period, the dissolved nitrate plus nitrite concentration became increasingly depleted without a steady source of ammonia for nitrification, likely because of nitrate decay. The MAE and RMSE values for dissolved nitrate plus nitrite were 0.05 and 0.08 mg/L, respectively (fig. 23; table 5), which are in line with previous calibrations for other sentinel lakes (Smith and others, 2014).

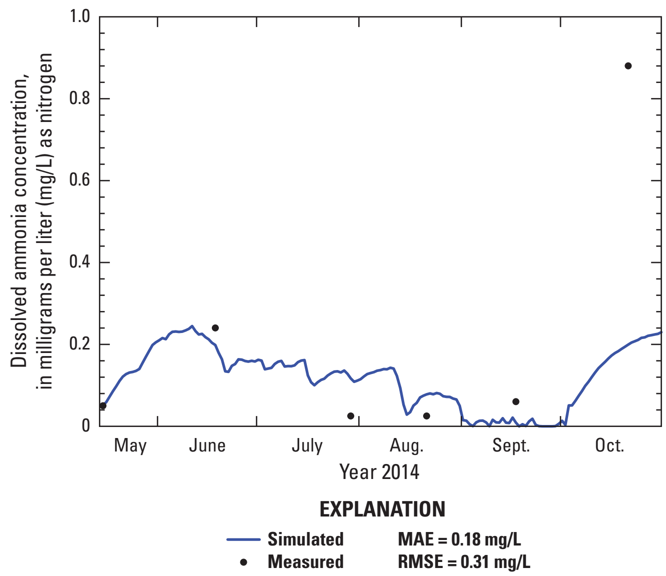


Figure 22. Simulated and measured dissolved ammonia concentrations at 2 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

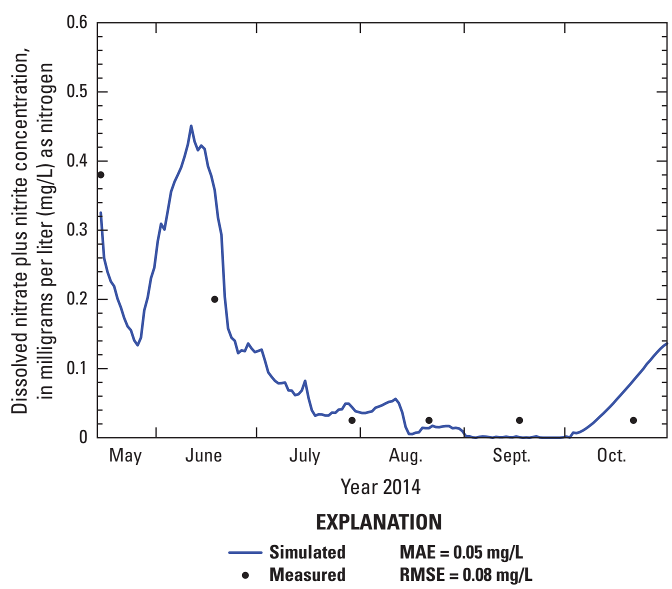


Figure 23. Simulated and measured dissolved nitrate plus nitrite concentrations at 2 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Dissolved orthophosphate concentrations in the Madison Lake measured data were stable for most of the year (fig. 24). The simulated orthophosphate concentrations were considerably more variable due to the algal dynamics of the lake and the cycling of nutrients through the various organic pools, algal communities, and the lake's simulated macrophyte community. Also, despite replenishment by organic matter decay and inflows, algal uptake is fairly rapid; therefore, the succession through various algal communities with different phosphorus requirements would also cause variability. At the end of the simulation period, a steady increase in dissolved orthophosphate concentrations occurred primarily because of the lack of demand by algae and macrophytes. The MAE and RMSE values of 0.01 and 0.02 mg/L, respectively, for dissolved orthophosphate were low overall because of the low concentrations with a good fit between the simulated and measured values (fig. 24; table 5).

Simulated and measured concentrations are shown for total Kjeldahl nitrogen in figure 25. The MAE and RMSE values for total Kjeldahl nitrogen were 0.29 and 0.33 mg/L, respectively (fig. 25; table 5). The measured data indicate a fairly 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, with the exception of a steady decrease in total Kjeldahl nitrogen towards the end of the simulation period (fig. 25). This decrease was likely because of the overall decay of the simulated organic matter pools and the decrease in simulated total algal biomass.

Total phosphorus (fig. 26) was affected by the same factors as total Kjeldahl nitrogen but was a much smaller pool and an overall smaller portion of algal biomass; in the case of total phosphorus, the epilimnion and hypolimnion locations are shown because measured hypolimnion values were available for total phosphorus. In the epilimnion, the measured total phosphorus concentrations were stable but the simulated concentrations 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 total phosphorus occurred (greater than 1,100 µg/L) for the simulated and measured concentrations (fig. 26) throughout the simulation period starting in late May until mid-September, at which time the simulated and measured concentrations dropped precipitously to the baseline of approximately 130 µg/L. The likely explanation for the large phosphorus concentrations in the simulated and measured values (fig. 26) was the large release rates in phosphorus from the lake sediments. The MAE values for the epilimnion (2-m depth) and hypolimnion (16.5-m depth) were 82 and 54 µg/L, respectively; the RMSE values for the epilimnion (2-m depth) and hypolimnion

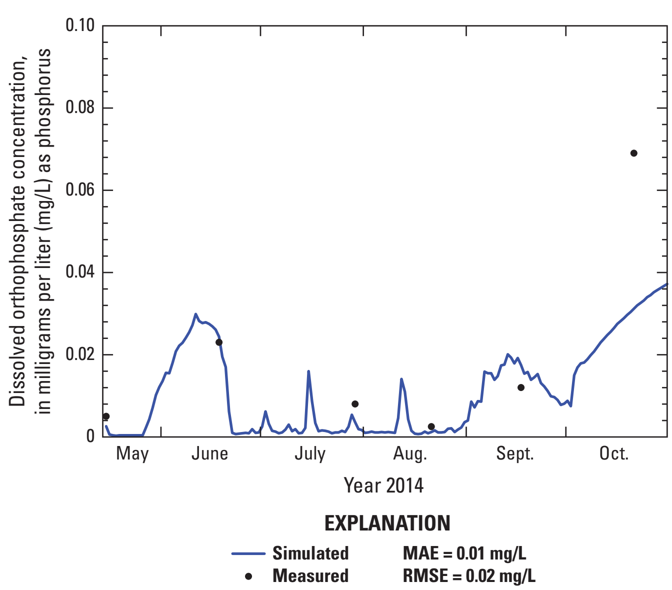


Figure 24. Simulated and measured dissolved orthophosphate concentrations at 2 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

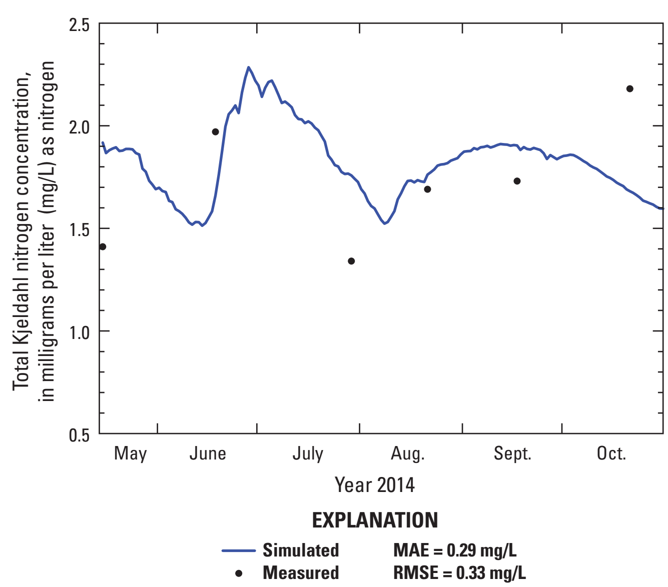


Figure 25. Simulated and measured total Kjeldahl nitrogen concentrations at 2 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

(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 of the lake water, which redistributed the concentrated total phosphorus to the entire lake volume.

For Madison Lake, the large nutrient loads, particularly nitrate plus nitrite, were tied back to basin processes. Large nitrate plus nitrite loads from the two inflow sites and the distributed tributary flow (mainly groundwater) were the initial source of nitrate plus nitrite. Generally, basins with a large relative percentage of agricultural land use have larger concentrations of nitrogen and phosphorus species (Nolan and others, 1997; U.S. Geological Survey, 1999; Schilling and others, 2008). The small drainage basin to lake area of approximately 4:1 indicated that natural reduction processes for these nutrients were limited (Fraterrigo and Downing, 2008). Without heavily controlled agricultural best management practices to reduce these external loads, such as those practices highlighted in the Minnesota agricultural best management practices handbook (Minnesota Department of Agriculture, 2012), the receiving waters of the lake would likely have greater nitrate loads.

Once in the lake, combined with consistent delivery of other nutrients such as phosphorus, algal growth and the recycling of nutrients proceeded. As the algae and macrophytes died, the decomposition would liberate ammonia and other nutrients, leading to a feedback loop. In addition, as the decaying organic matter sank, the deeper mixed layer and the hypolimnion became increasingly hypoxic. This caused the release of sediment-bound phosphorus that would initialize more algal growth. Between the external nutrient loading, internal nutrient loading from sediment release of phosphorus, and the organic matter decomposition of the algal and macrophyte biomass, even more algal and macrophyte growth was initiated. This recycling feedback between active growth and decomposition caused the series of algal blooms and the greater nutrient concentrations. Although ammonia, nitrate, and total Kjeldahl nitrogen generally decreased from the beginning of the simulation period to the end, total phosphorus in the hypolimnion continued to increase throughout most of the simulation period; however, lake overturn caused the phosphorus to become readsorbed and sink to the bottom of the lake, causing the total phosphorus concentrations in the lake to decrease, albeit the concentrations were greater than 130 µg/L. At the total phosphorus peak, concentrations were greater than 1,000 µg/L (1 mg/L). Because Madison Lake has a lake residence time on the order of 3–4 years (Lindon and others, 2010), this cycle has the potential to continue until drastic nutrient load reductions occur within the basin.

Pearl Lake

Dissolved ammonia and nitrate plus nitrite distributions in Pearl Lake were low compared to Madison Lake and largely affected by the algal growth dynamics. The simulated and measured concentrations of dissolved ammonia as nitrogen at 2 m below the water surface are shown in figure 27 for the model segment containing the Pearl Lake deep point site (segment 4, fig. 3). The measured dissolved ammonia

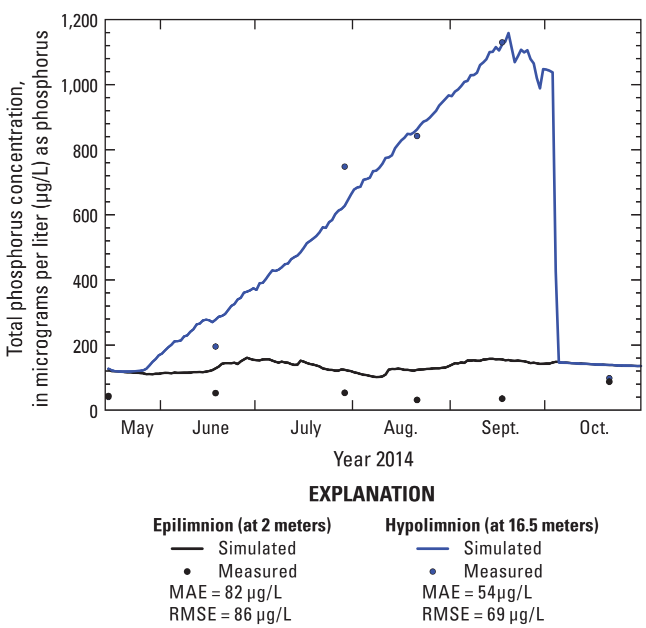


Figure 26. Simulated and measured total phosphorus concentrations at 2 meters and 16.5 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

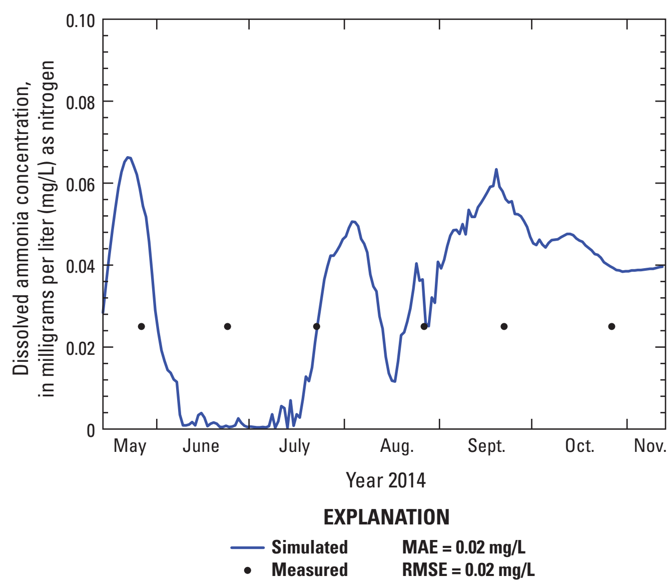


Figure 27. Simulated and measured ammonia concentrations at 2 meters below the water surface in model segment 4 containing the Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

concentrations did not vary throughout the simulation period, whereas the simulated ammonia concentrations covered a range of from near zero to greater than 0.06 mg/L. Similar to Madison Lake, algal uptake of available ammonia was fairly rapid in the simulation and actual lake measurements, with replenishment by organic matter decay and inflows. Because blue-green algae have lower nitrogen stoichiometric requirements (Williams and Burris, 1952), simulated ammonia concentrations did recover during certain periods in the summer months because the blue-green algae did not require as much nitrogen (part of their nitrogen requirements were incorporated from atmospheric nitrogen). The MAE and RMSE values for dissolved ammonia, both at 0.02 mg/L, were low compared to Madison Lake (fig. 27; table 5).

Simulated and measured dissolved nitrate plus nitrite concentrations are shown in figure 28 for the Pearl Lake deep point site. Additional depletion was simulated for dissolved nitrate plus nitrite concentrations during the early summer period that only started to recover briefly in late July and again beginning in September. The simulated increase in dissolved nitrate plus nitrite concentrations that began in late July coincided with greater nitrate concentrations in the two inflows, particularly from the southwest corner inlet and the distributed tributary flow that was set up to mimic the incoming loads by way of the southwest corner inlet. Towards the end of the simulation period, the dissolved nitrate plus nitrite concentration steadily increased due to the slightly larger incoming concentrations (averaging between 2 and 3 mg/L) and the steady source of nitrate from ammonia nitrification. The MAE and RMSE values for dissolved nitrate plus nitrite were 0.03 and 0.04 mg/L, respectively, which are similar to previous calibrations for other sentinel lakes and the Madison Lake calibration (fig. 28; table 5).

Measured dissolved orthophosphate concentrations in Pearl Lake were low for the simulation period (fig. 29). The simulated orthophosphate concentrations, similar to Madison Lake, were dependent on the algal dynamics of the lake and the cycling of nutrients, particularly phosphorus, through the various organic pools, algal communities, and the lake's simulated macrophyte community. At the end of the simulation period, simulated and measured dissolved orthophosphate concentrations remained steady. The orthophosphate MAE and RMSE values were both <0.01 mg/L because of the good fit and the low dissolved orthophosphate concentrations (fig. 29; table 5).

Simulated and measured concentrations are shown for total Kjeldahl nitrogen (fig. 30). The MAE and RMSE values for total Kjeldahl nitrogen were 0.17 and 0.19 mg/L, respectively (fig. 30; table 5). The measured data indicate a less dynamic range than Madison Lake, from approximately 0.6 to 1.0 mg/L. A peak in measured total Kjeldahl nitrogen concentrations occurred in late August and again in late October. The August peak nearly coincided with a simulated peak in total Kjeldahl nitrogen, whereas the late October peak occurred when the simulated total Kjeldahl nitrogen had a steady decrease towards the end of the simulation period. The simulated peak in total Kjeldahl nitrogen was mostly due to the simulated blue-green algae peak in late August; additionally, the steady decrease towards the end of the simulation coincides with the blue-green algae decrease.

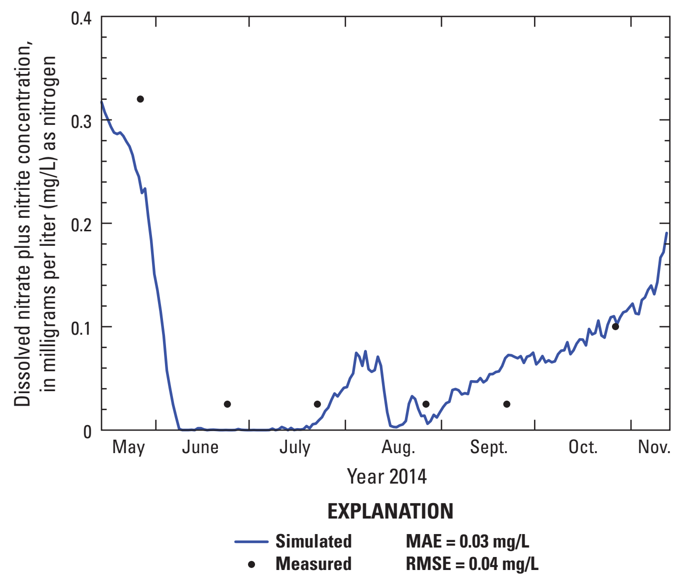


Figure 28. Simulated and measured nitrate plus nitrite concentrations at 2 meters below the water surface in model segment 4 containing the Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

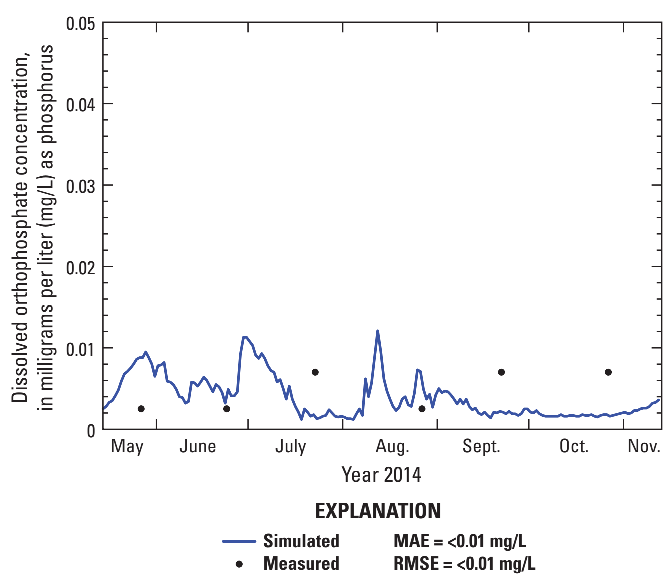


Figure 29. Simulated and measured orthophosphate concentrations at 2 meters below the water surface in model segment 4 containing the Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

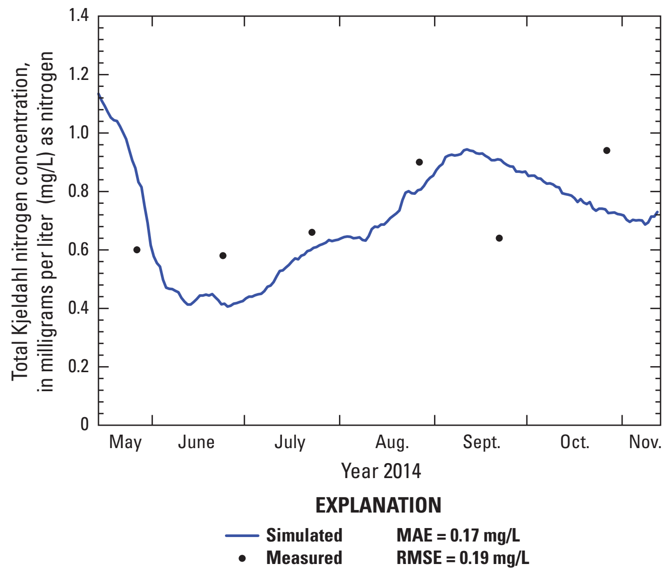


Figure 30. Simulated and measured total Kjeldahl nitrogen concentrations at 2 meters below the water surface in model segment 4 containing the Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

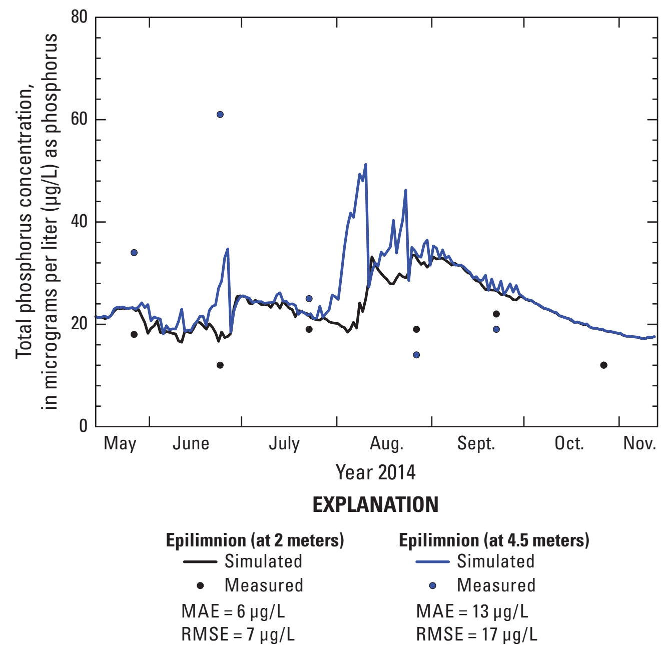


Figure 31. Simulated and measured total phosphorus concentrations at 2 meters and 4.5 meters below the water surface in model segment 4 containing the Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with values of mean absolute error (MAE) and root mean square error (RMSE).

Measured total phosphorus concentrations were stable (between 10 and 20 µg/L) in the shallow mixed layer at 2-m depth, with slightly greater simulated total phosphorus concentrations (fig. 31); however, the simulated total phosphorus concentrations did follow the general trend of the measured total phosphorus concentrations. In the deeper part of the mixed layer at 4.5-m depth, the simulated total phosphorus concentrations do not capture the early peaks in measured total phosphorus concentrations of 34 and 65 µg/L for the May 27 and June 24 data collections, respectively. Larger total phosphorus concentrations in the deeper portion of the mixed layer earlier in the year would lend support to the theory of greater organic matter concentrations, as hypothesized in the "Dissolved Oxygen" section; however, without supporting evidence from a simultaneously collected total Kjeldahl nitrogen sample, it is not known if this was the cause of the greater measured total phosphorus concentrations. The MAE values for the shallow mixed layer (2-m depth) and deeper mixed layer (4.5-m depth) were 6 and 13 µg/L, respectively; the RMSE values for the shallow mixed layer (2-m depth) and deeper mixed layer (4.5-m depth) were 7 and 17 µg/L, respectively (fig. 31; table 5). Compared to Madison Lake, Pearl Lake did not have similar large total phosphorus concentrations.

Pearl Lake does not have nutrient concentrations as large as those present in Madison Lake. By presentation, many of the Pearl Lake figures appear to have large nutrient concentrations but the scales are smaller than those used for Madison Lake figures. Although external loading did provide nutrients to sustain greater algal growth and blue-green algae blooms, the differences in basin characteristics between the two lakes were likely a reason for the lower nutrient concentrations. Pearl Lake has a larger drainage basin to lake area ratio of 24:1, with approximately 15 percent forest cover. Additionally, for internal dynamics, the lake residence time is on the order of 1–2 years so the lake can conceivably cycle nutrients out of the lake faster. Although nutrient concentrations in Pearl Lake are smaller than those in Madison Lake, Pearl Lake has large nutrient concentrations that cause persistent blue-green algal blooms throughout the summer, as well as late-season green algae and flagellate blooms. Therefore, considerations in regards to basin management processes mentioned for Madison Lake, such as the necessity for external load reductions to limit algal blooms, also pertain to Pearl Lake.

Model Limitations

A full understanding of model limitations is necessary to better evaluate the performance of any water-quality model. Because the CE-QUAL-W2 model is laterally averaged, processes that could impose variations perpendicular to the primary flow axis of the lake will not be represented in the model. The CE-QUAL-W2 model vertically averages within a layer, although the discretization into 1-m (or smaller) segments likely is a sufficient representation of the vertical variability within the lake. Water-quality limitations include the simplification of a complex aquatic ecosystem into a series of kinetic reactions expressed in source and sink terms (Cole and Wells, 2015). Also, 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. Specific water-quality modules within the CE-QUAL-W2 model have shortcomings that include the SOD, which is user-defined and is decoupled from the water column, and SOD variation only occurs with temperature. A complete sediment diagenesis model, with fully integrated sediment kinetics and the sediment-water interface, does currently exist within the CE-QUAL-W2 V4.0 model; however, without adequate physical measurements, the complete sediment diagenesis model is not recommended for usage (Cole and Wells, 2015). The zooplankton module was also not included in this study because of sparse data during the calibration period; instead, the zooplankton effects were accounted for within the parameterization scheme of SOD and the algal dynamics as an attempt to address this deficiency.

Not only do data limitations exist, but structural selections such as segment geometry, the number of vertical layers, and the numerical transport scheme can potentially impose a bias in the outcome of the model. Boundary conditions are not fixed in nature but are limited in model development by the availability of data. In addition, extrapolation of the data was necessary to fit the requirements of the CE-QUAL-W2 model. For example, water-quality data were linearly interpolated between sampling dates.

Fish Habitat Suitability for Cool-Water and Warm-Water Species

Models frequently are used to simulate fish responses to changes in climate and land-use patterns. For example, statistically based models have used predicted climatic data and land-use data to demonstrate habitat losses that could cause localized extinctions of smallmouth bass in midwestern, large-river, flood-plain ecosystems (Peterson and Kwak, 1999). In Minnesota, fish thermal and DO-based habitats were simulated in lakes over a range of current and proposed climate change scenarios using the MINLAKE96 program (Fang and others, 1999). Fang and others (1999) used good-growth habitat areas and good-growth habitat volumes as metrics to evaluate habitat units on a normalized scale, and the results demonstrated that fish habitat parameters depended more strongly on geometry and less on trophic state in inland, temperate lakes; however, the MINLAKE96 program does not use a carbon-based, trophic-dynamic model to predict fish habitat change nor does it include basin effects.

The CE-QUAL-W2 model, however, can be used to simulate biological responses to changing environmental conditions. Researchers have used the CE-QUAL-W2 model to simulate distribution and survival of white sturgeon over varying hydrologic conditions (Sullivan and others, 2003), the movements of blueback herring in a southern impoundment (Nestler and others, 2002), as well as ongoing work by the USGS to simulate cisco habitat in Minnesota lakes under varying conditions of climate and nutrient loading (Smith and Kiesling, 2016).

In order to evaluate fish lake habitat suitability for cool-water and warm-water species, the CE-QUAL-W2 model results were evaluated for total volume of good-growth habitat (a lower and upper good-growth range), optimal growth habitat, and lethal oxythermal habitat. These sometimes overlapping ranges were developed from guidance in Fang and others (1999). The evaluation criteria given in this report were based on the mean range given in Fang and others (1999), which was developed from the earlier work of Eaton and others (1995) and based on entire fish guilds grouped into cold water, cool water, and warm water. For Madison and Pearl Lakes, examples of important cool-water fish, particularly game fish, include northern pike (*Esox lucius*), walleye (*Sander vitreus*), and black crappie (*Pomoxis nigromaculatus*) (Minnesota Department of Natural Resources, 2016c, 2016d). Examples of important warm-water fish include bluegill (*Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), and smallmouth bass (*Micropterus dolomieu*) (Minnesota Department of Natural Resources, 2016c, 2016d).

Lower cool-water good-growth habitat was defined as between 13.2 and 18.2 °C, upper cool-water good-growth habitat was defined as 27.7 to 28.8 °C, optimal cool-water growth habitat was defined as between 24.0 and 25.7 °C, and the lethal temperature range was set to above 28.0 °C. For all of the different ranges for cool-water fish, the DO minimum was set at 3 mg/L. Lower warm-water good-growth habitat was defined as between 17.7 and 22.5 °C, upper warm-water good-growth habitat was defined as 31.3 to 34.7 °C, optimal warm-water growth habitat was defined as between 27.0 and 32.0 °C, and the lethal temperature range was set to above 32.3 °C. For all of the different ranges for warm-water fish, the DO minimum was set at 2.5 mg/L; all of the limits were based on the work from Fang and others (1999). Sustained temperatures beginning at the high end of the upper good-growth habitat temperature ranges, combined with low DO, can be lethal.

Madison Lake

The combined good-growth (lower and upper) and optimal growth habitats for cool-water fish during the simulation period from May 15 to November 1, 2014, are shown in figure 32. Mainly due to the temperature requirements for these ranges, good-growth habitat range was limited to outside of the months of July and August. After mid-June and before early September, optimal growth habitat existed in as much as about 70 percent of the lake volume according to the criteria. Although much of the lower lake depths were hypoxic, most of the lake volume was under favorable DO conditions and would cross into optimal growth habitat during the warmer periods of the summer.

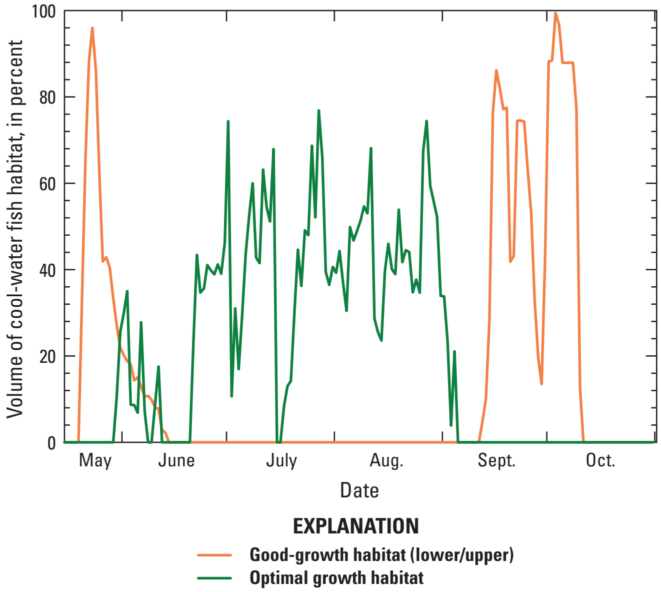


Figure 32. Good-growth (combined lower and upper ranges) and optimal growth habitat, by percentage of total lake volume, for cool-water fish in Madison Lake, May 15 to November 1, 2014.

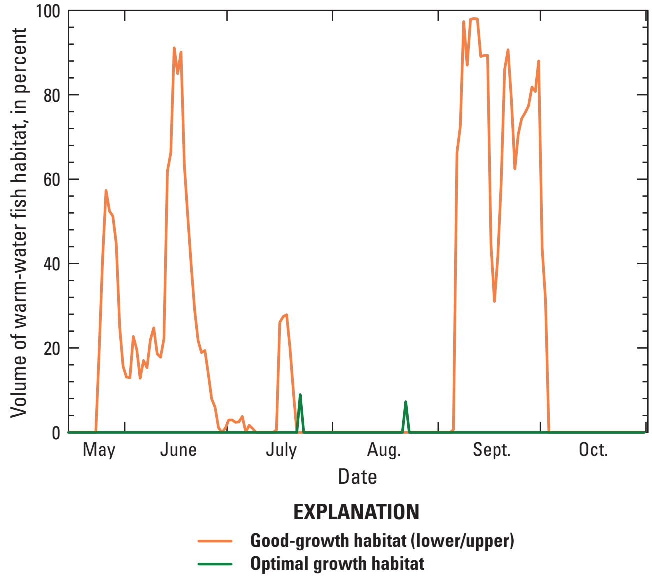


Figure 33. Good-growth (combined lower and upper ranges) and optimal growth habitat, by percentage of total lake volume, for warm-water fish in Madison Lake, May 15 to November 1, 2014.

The combined good-growth (lower and upper) and optimal growth habitats for warm-water fish during the simulation period from May 15 to November 1, 2014, are shown in figure 33. Much of the lake did contain good-growth habitat through mid-June and again after early September; however, unlike the cool-water fish, optimal growth habitat was sparse, mainly because the lake is not warm enough for warm-water fish to thrive, at least by the criteria set by Fang and others (1999). For cool-water and warm-water fish, the lake volume did not contain any lethal oxythermal habitat.

Pearl Lake

The combined good-growth (lower and upper) and optimal growth habitats for cool-water fish during the simulation period from May 14 to November 13, 2014, are shown in figure 34. Mainly due to the temperature requirements for these ranges, the good-growth habitat range was limited to a short period in late May and approximately 3 weeks from mid-September to early October. From mid-June to the end of August, optimal growth habitat existed in as much as about 90 percent of the lake volume according to the criteria. Although small portions of lower lake depths were hypoxic, most of the lake volume was under favorable DO conditions and would cross into optimal growth habitat during the warmer periods of the summer.

The combined good-growth (lower and upper) and optimal growth habitats for warm-water fish during the simulation period from May 14 to November 13, 2014, are shown in figure 35. Much of the lake did contain good-growth habitat from mid-May through mid-July and again after late August, but little optimal growth habitat existed in Pearl Lake. Similar to Madison Lake, the lake volume did not contain any lethal oxythermal habitat for either cool-water or warm-water fish.

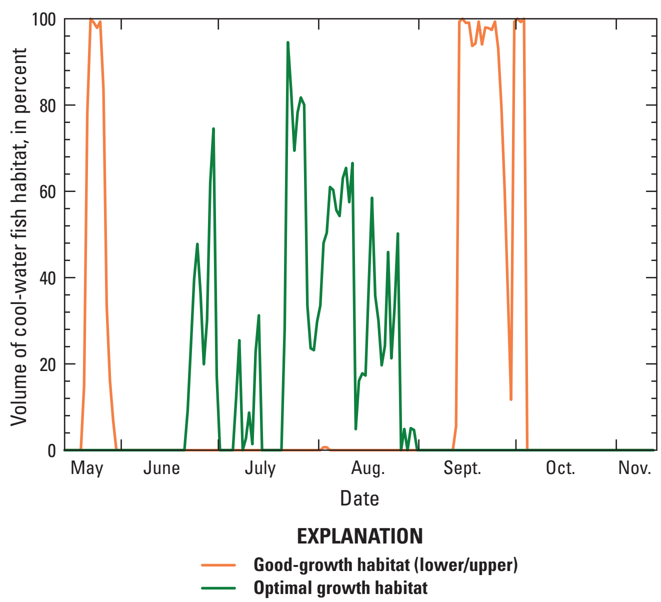


Figure 34. Good-growth (combined lower and upper ranges) and optimal growth habitat, by percentage of total lake volume, for cool-water fish in Pearl Lake, May 14 to November 13, 2014.

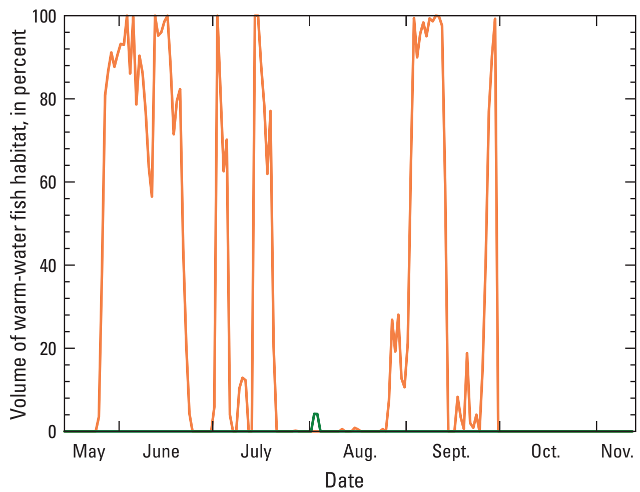


Figure 35. Good-growth (combined lower and upper ranges) and optimal growth habitat, by percentage of total lake volume, for warm-water fish in Pearl Lake, May 14 to November 13, 2014.

Sensitivity Analysis

A sensitivity analysis was completed to understand the effects of controlled departures in the calibrated model parameters and input nutrient loads on the model results. Because of the numerous calibrated parameters in both of the lake models (table 4), only seven different constituents were altered in the sensitivity analysis. For each of the following parameters or input loads, the calibrated lake model value was increased by 20 percent and decreased by 20 percent: WSC, inflow phosphorus, phosphorus sediment release rate, inflow nitrogen, inflow organic matter, SOD, and the extinction coefficient. In the case of the extinction coefficient, all of the component light extinction coefficients (table 4) were adjusted including the light extinction coefficients for pure water (parameter EXH20), inorganic suspended solids (parameter EXSS), organic suspended solids (parameter EXOM), the four different algal groups (diatoms [parameter EXA1], green algae [parameter EXA2], blue-green algae [parameter EXA3], and flagellates [parameter EXA4]), and the macrophyte coefficient (parameter EXM1). During model development and calibration, a more robust but less controlled sensitivity analysis was undertaken to attain a final calibrated model, meaning that more than these seven different constituents underwent sensitivity analysis; however, the seven constituents chosen for this analysis were determined to be some of the most sensitive parameters or input loads, which is similar to previous CE-QUAL-W2 lake models (Green and others, 2003; Sullivan and Rounds, 2004; Galloway and Green, 2006; Galloway and others, 2008; Smith and others, 2014).

With the exception of the WSC, six of the seven parameters or input loads were utilized to better understand the model response to parameters that can be directly affected by basin processes. These sensitivity analyses operated as land-use change scenarios because alterations in agricultural practices, for example, could potentially increase or decrease nutrient loads. Three of the seven parameters (inflow nitrogen, inflow organic matter, and inflow phosphorus) were related to input loads to the lake, so the connection to basin processes was more direct. Of the other four parameters, the following parameters have some indirect connections to basin processes: sediment release rates of phosphorus, SOD, and the extinction coefficients. The sediment release rate of phosphorus can be related to the available phosphorus in the sediments, which, if the available phosphorus was large due to legacy phosphorus loading (Marsden, 1989; James and others, 2015), under favorable conditions, more phosphorus can be released back into the water column. The SOD is the oxygen consumed by organisms in the sediment, so in shallow to moderately deep lakes rich in organic matter (such as eutrophic lakes), SOD can be large (Molongoski and Klug, 1980; Cross and Summerfelt, 1987), whereas in an oligotrophic lake this effect would not be as important. Extinction coefficients are related to light extinction caused by different solids in the water column, such as inorganic suspended solids, organic suspended solids, algal biomass, and macrophyte biomass (Cole and Wells, 2015). In a lake with larger concentrations of algal biomass, such as a eutrophic or hypereutrophic lake because of nutrient loading, extinction coefficients can be directly related back to basin processes (Carlson, 1977).

Madison Lake

Daily values for the southwest deep point site in Madison Lake (segment 7, fig. 2) were averaged into a single value for three different depths (2 m, 5 m, and 15 m) for temperature and DO. The summary of the minimum, maximum, and median values of water temperature and DO for each of three depths is shown in table 6. Additionally, departures from the baseline (calibrated model) to the sensitivity analysis, shown as percent change from the baseline (calibrated) value, were completed for the following parameters with a 20-percent increase and a 20-percent decrease: WSC, inflow phosphorus, phosphorus sediment release rate, inflow nitrogen, inflow organic matter, SOD, and the extinction coefficient.

Water temperature in the Madison Lake was most sensitive to alterations in the WSC (table 6). The WSC adjusted the resultant wind speed, which affected the amount of mixing in the vertical dimension and thereby the depth of the thermocline over time. Decreases in the WSC resulted in lower wind speeds and led to a shallower thermocline and greater water temperatures at the lake surface. Increases in the WSC resulted in greater wind speeds and led to a deeper thermocline and lower water temperatures at the lake surface.

Table 6. Summary of sensitivity analysis for water temperature and dissolved oxygen, in percent change from calibration run, for the segment containing the Madison Lake southwest deep point near Madison Lake, Minnesota, and the segment containing Pearl Lake Deep Point near Marty, Minnesota. The following depths are presented: 2 meters, 5 meters, and 15 meters (note: 15 meters only in Madison Lake). Also, the baseline (calibrated) values for the minimum, maximum, and median are shown for both lakes in the same segments.

[The following depths are shown: 2 meters, 5 meters, and 15 meters (note: 15 meters only in Madison Lake). Also, the baseline (calibrated) values for the minimum, maximum, and median are shown for both lakes in the same segments. mg/L, milligram per liter; --, not applicable]

Parameter	Input, in percent change from calibrated value	Water temperature (degrees Celsius)			Dissolved oxygen (mg/L)		
		2 meters	5 meters	15 meters	2 meters	5 meters	15 meters
Madison Lake							
Baseline, minimum	--	9.5	9.5	9.5	4.5	0.0	0.0
Baseline, maximum	--	26.0	25.4	15.7	11.4	9.9	8.0
Baseline, median	--	23.0	21.5	14.5	7.1	6.2	0.0
Madison Lake model output, in percent change from baseline (calibrated) value							
Wind sheltering coefficient	-20	1.9	-2.2	-1.9	0.5	-17.1	-14.6
	+20	-2.0	-0.2	4.7	-2.9	5.1	22.8
Inflow phosphorus	-20	0.0	0.0	-0.0	-0.7	-0.2	-0.7
	+20	-0.0	-0.1	0.0	0.7	-0.0	0.4
Sediment release rate, phosphorus	-20	0.0	0.0	-0.0	-0.8	-0.3	-0.6
	+20	-0.0	-0.1	0.0	0.2	-0.5	-0.0
Inflow nitrogen	-20	-0.0	-0.0	0.0	0.0	0.0	0.2
	+20	-0.0	-0.0	0.0	-0.0	-0.2	-0.4
Inflow organic matter	-20	0.0	-0.0	0.0	0.2	0.5	0.4
	+20	-0.0	-0.0	0.0	-0.2	-0.5	-0.3
Sediment oxygen demand	-20	0.0	0.0	-0.0	4.8	9.2	8.7
	+20	-0.0	-0.1	0.1	-6.6	-10.8	-9.6
Extinction coefficient	-20	0.1	0.5	-0.3	4.0	8.8	0.7
	+20	-0.1	-0.4	0.3	-3.8	-8.7	-1.5
Pearl Lake							
Baseline, minimum	--	2.2	2.2	--	5.1	0.0	--
Baseline, maximum	--	24.6	24.1	--	12.7	12.6	--
Baseline, median	--	21.2	20.5	--	7.7	6.3	--
Pearl Lake model output, in percent change from baseline (calibrated) value							
Wind sheltering coefficient	-20	2.8	0.8	--	-1.6	-21.1	--
	+20	-2.8	-2.0	--	1.8	12.0	--
Inflow phosphorus	-20	-0.0	-0.0	--	-0.4	-0.2	--
	+20	-0.0	-0.0	--	0.4	0.0	--
Sediment release rate, phosphorus	-20	-0.0	-0.0	--	-0.8	-0.1	--
	+20	0.0	-0.0	--	0.8	0.2	--
Inflow nitrogen	-20	-0.1	-0.1	--	0.3	0.3	--
	+20	0.0	0.0	--	-0.3	-0.5	--
Inflow organic matter	-20	-0.0	-0.0	--	0.0	0.1	--
	+20	0.0	-0.0	--	-0.1	-0.2	--
Sediment oxygen demand	-20	-0.1	-0.0	--	1.9	7.4	--
	+20	0.0	-0.0	--	-1.9	-6.4	--
Extinction coefficient	-20	-0.6	-0.2	--	-0.3	3.0	--
	+20	0.2	-0.0	--	-0.7	-2.2	--

Perturbations in DO reflected the dynamic nature of DO as it is affected by many different processes, in particular alterations from the baseline (calibrated) model that affect algal growth dynamics. Dissolved oxygen was more sensitive than water temperature to alterations in the wind sheltering due to similar reasons as water temperature. Decreases in the WSC resulted in lower wind speeds and led to a shallower DO chemocline. Increases in the WSC resulted in more mixing of greater DO lake water near the surface to deeper parts of the mixed layer and into the hypolimnion, causing the DO chemocline to move deeper into the lake, particularly in July and early August. A 20-percent increase in the extinction coefficient caused a net decrease in DO for all three depths (table 6) because increases in the extinction coefficient had a negative effect on algal growth rates, causing less overall oxygen production. The late September blue-green algal bloom was most affected by this suppression of algal growth, having a two-fold effect: (1) less oxygen production in the shallow epilimnion; (2) less organic matter decomposition in the hypolimnion, causing greater DO levels. Lowering the extinction coefficient by 20 percent, on the other hand, led to more algal growth of green algae and flagellates, which led to more epilimnion DO, although it did not seem to have much of an effect deep in the lake at 15 m. Finally, SOD was a major sink for DO, with increases in SOD leading to lower DO concentrations and decreases in SOD leading to greater DO concentrations.

Nutrient concentrations were affected by several parameters or input loads (table 7). Dissolved ammonia and dissolved nitrate plus nitrite were most affected by increases in the WSC, inflow nitrogen, SOD, and the extinction coefficient (table 7). The increased WSC affected ammonia and nitrate as less algal growth, likely due to greater temperatures at the limit or beyond the maximum growth ranges, led to less photosynthesis (uptake) (fig. 36) and therefore larger ammonia pools (fig. 37). Declines in the WSC did not have as much of an effect on dissolved ammonia or dissolved nitrate plus nitrite, although an increase in dissolved ammonia did occur, which was due to an earlier season suppression of algal growth that caused a short-term increase in dissolved ammonia. Inflow nitrogen increases and decreases led directly to comparable increases or decreases in the dissolved ammonia and dissolved nitrate plus nitrite pools. An SOD increase led to less dissolved ammonia and dissolved nitrate plus nitrite, whereas SOD decreases led to even larger gains in both of those pools. Finally, the decrease in the extinction coefficient had a larger effect on dissolved ammonia and dissolved nitrate plus nitrite than an increase in the extinction coefficient. The greater light penetration with decreased extinction coefficients led to less blue-green algal growth, similar to the effects mentioned earlier; in this case, less algal growth led to less nutrient uptake.

Dissolved orthophosphate and total phosphorus tracked together in terms of the effects of the different sensitivity parameters (table 7), albeit at different percent changes because of the relative sizes of these pools. Dissolved orthophosphate was only a portion of the total phosphorus, so any percent change in orthophosphate was generally exaggerated in comparison to total phosphorus. Two sensitive parameters, although not as large as expected, were the changes in the sediment release rate of phosphorus and the changes in the inflow phosphorus. Dissolved orthophosphate and total phosphorus pools increased with a 20-percent increase in either of these two parameters, and both pools decreased with a 20-percent decrease. Dissolved orthophosphate and total phosphorus pools were more sensitive to WSC, SOD, and the extinction coefficients. The WSC increase caused more phosphorus burial as lake overturn and mixing happened earlier in the year; a decrease in the WSC, and thereby wind energy, caused a shallower DO chemocline, which led to more phosphorus release into the water column (fig. 38). With SOD increases, more phosphorus was released into the water column and vice versa for SOD decreases. Dissolved orthophosphate was affected by extinction coefficient increases because less algal growth led to more orthophosphate in the water column.

Pearl Lake

Daily values for the Pearl Lake deep point site in Pearl Lake (segment 4, fig. 3) were averaged into a single value for two different depths (2 m and 5 m) for temperature and DO. The summary of the minimum, maximum, and median values of water temperature and DO for both of the depths is shown in table 6. Additionally, departures from the baseline (calibrated model) to the sensitivity analysis, indicated by percent change from the baseline (calibrated) value, were completed for the same parameters as used for Madison Lake.

Water temperature in the Pearl Lake was most sensitive to alterations in the WSC, mainly for the upper mixed layer. Decreases in the WSC resulted in lower wind speeds and led to a shallower thermocline and greater water temperatures at the lake surface. Increases in the WSC resulted in greater wind speeds and led to a deeper thermocline and lower water temperatures at the lake surface, which is a similar effect as indicated for Madison Lake.

Similar to Madison Lake, DO was more sensitive than water temperature to wind sheltering alterations. Decreases in the WSC resulted in lower wind speeds and led to a shallower DO chemocline. Less DO at 2 m was due to a substantial increase in blue-green algae (fig. 39) throughout the summer months, which eventually died off and caused massive organic matter decomposition in the deeper mixed layer. Increases in the WSC resulted in more mixing of greater DO lake water near the surface to the deeper mixed layer near the bottom of the lake. Of the remaining parameters, only SOD had a percent change greater than 5 percent, with increases in SOD leading to lower DO concentrations and decreases in SOD leading to greater DO concentrations (table 7).

Nutrient concentrations were affected by several parameters or input loads (table 7); however, compared to Madison Lake, these effects were subdued because the overall load into Pearl Lake and the internal lake dynamics were much smaller for nutrients. Dissolved ammonia and dissolved nitrate plus

Table 7. Summary of sensitivity analysis for the water-quality parameters of dissolved orthophosphate, dissolved ammonia, dissolved nitrate plus nitrite, dissolved oxygen, and total phosphorus. Values shown are the percent change from the calibration run, based on the volume-weighted averages for the segment containing Madison Lake southwest deep point near Madison Lake, Minn. and the segment containing Pearl Lake Deep Point near Marty, Minn.

[Values shown are the percent change from the calibration run, based on the volume-weighted averages for the segment containing Southwest Deep Point in Madison Lake and the segment containing Pearl Lake Deep Point in Pearl Lake. Also, the baseline (calibrated) volume-weighted averages for the minimum, maximum, and median are shown for both lakes in the same segments. mg/L, milligram per liter; µg/L, microgram per liter]

Parameter	Input, in percent change from calibrated value	Dissolved ammonia (mg/L as nitrogen)	Dissolved nitrate plus nitrite (mg/L as nitrogen)	Dissolved orthophosphate (mg/L as phosphorus)	Total phosphorus (µg/L as phosphorus)
Madison Lake					
Baseline, minimum	--	0.0	0.0	0.0	116.0
Baseline, maximum	--	0.2	0.4	0.1	213.0
Baseline, median	--	0.2	0.1	0.0	167.5
Madison Lake model output, in percent change from baseline (calibrated) value					
Wind sheltering coefficient	-20	14.3	-1.9	103.2	28.6
	+20	26.4	50.5	-45.6	-17.3
Inflow phosphorus	-20	8.1	13.8	-4.4	-2.2
	+20	-8.5	-11.0	6.1	2.3
Sediment release rate, phosphorus	-20	11.4	13.2	-19.3	-6.4
	+20	-5.2	-4.9	24.7	6.9
Inflow nitrogen	-20	-9.6	-17.1	2.9	0.3
	+20	8.7	19.3	-1.7	-0.1
Inflow organic matter	-20	-1.6	-0.7	0.0	-2.6
	+20	1.4	0.7	0.9	2.7
Sediment oxygen demand	-20	21.1	31.9	-22.9	-8.4
	+20	-8.1	-11.5	37.4	11.0
Extinction coefficient	-20	23.1	16.0	-12.6	-3.2
	+20	-7.4	0.3	16.7	3.5
Pearl Lake					
Baseline, minimum	--	0.0	0.0	0.0	17.2
Baseline, maximum	--	0.1	0.3	0.0	38.3
Baseline, median	--	0.0	0.1	0.0	23.9
Pearl Lake model output, in percent change from baseline (calibrated) value					
Wind sheltering coefficient	-20	13.1	0.8	72.3	27.4
	+20	-14.6	-14.7	-25.9	-17.3
Inflow phosphorus	-20	-2.6	-1.6	-7.7	-4.6
	+20	2.0	3.1	9.3	4.1
Sediment release rate, phosphorus	-20	-4.5	-0.5	-19.2	-10.3
	+20	3.2	3.5	26.0	11.3
Inflow nitrogen	-20	-5.6	-5.8	7.9	-0.5
	+20	2.5	8.2	-3.9	0.5
Inflow organic matter	-20	-2.2	1.0	-0.9	-4.4
	+20	2.1	0.7	1.1	3.9
Sediment oxygen demand	-20	-6.9	3.1	-25.7	-14.7
	+20	4.1	0.6	44.4	19.1
Extinction coefficient	-20	-6.0	0.0	-25.3	-17.0
	+20	2.8	-10.4	35.8	16.3

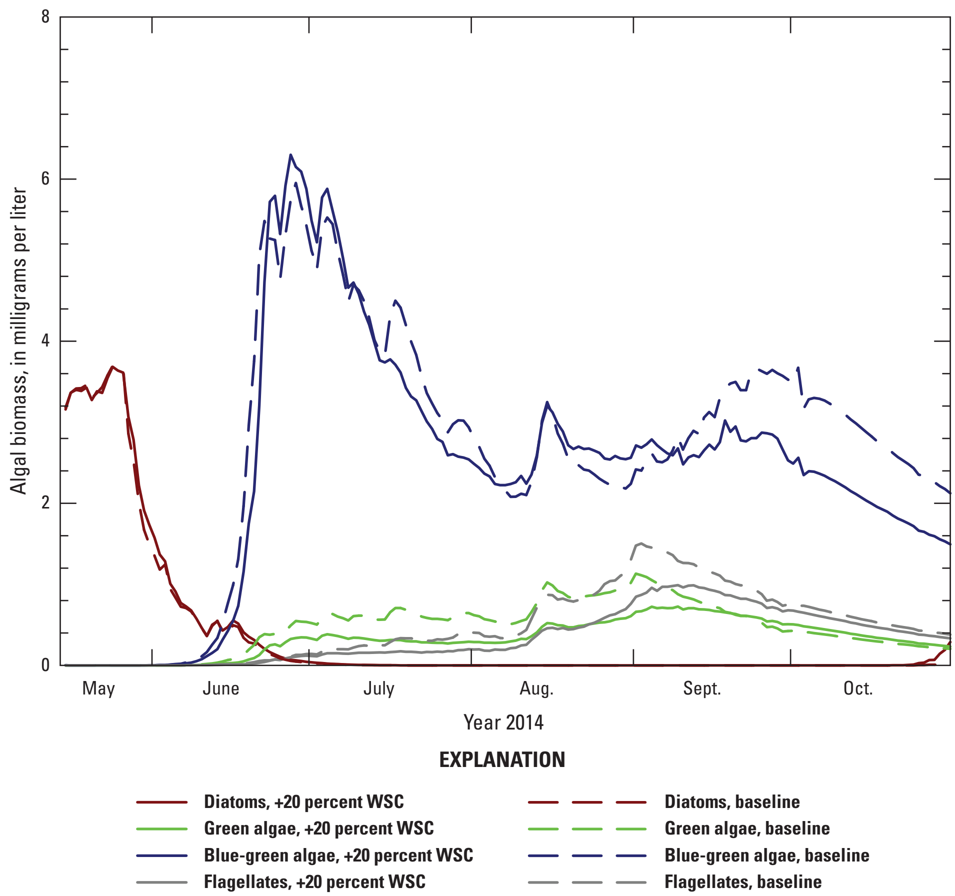


Figure 36. Simulated algal group distributions (diatoms, green algae, blue-green algae, and flagellates) for the 2-meter depth at the Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with a 20-percent increase in the wind sheltering coefficient (WSC) and the baseline (calibrated) model.

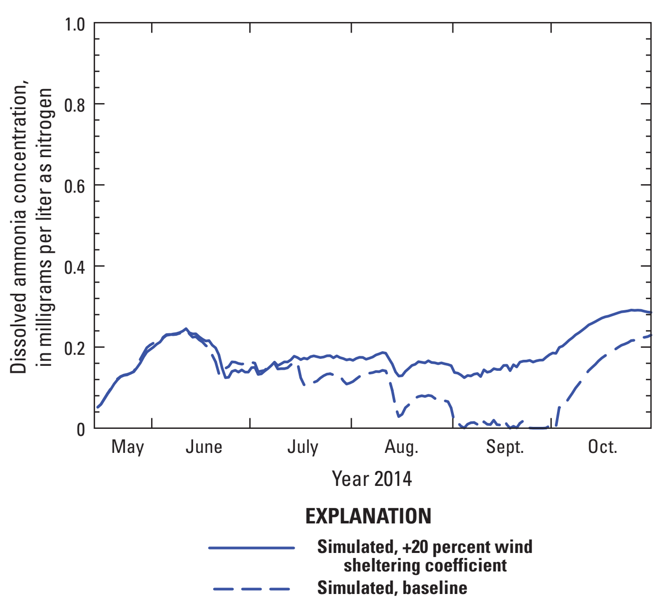


Figure 37. Simulated dissolved ammonia concentrations at 2 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with a 20-percent increase in the wind sheltering coefficient and the baseline (calibrated) model.

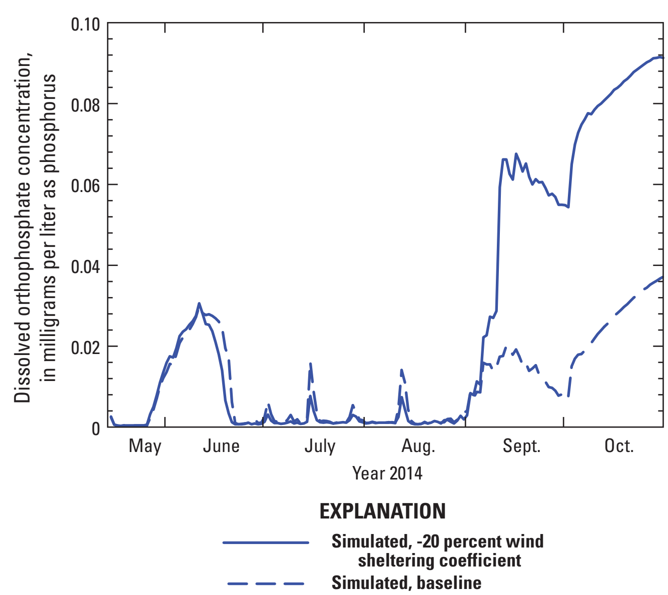


Figure 38. Simulated dissolved orthophosphate concentrations at 2 meters below the water surface in model segment 7 containing the Madison Lake southwest deep point near Madison Lake, Minnesota, May 15 to November 1, 2014, with a 20-percent decrease in the wind sheltering coefficient and the baseline (calibrated) model.

nitrite were affected by increases in the WSC, inflow nitrogen, and the extinction coefficient (table 7), whereas a decrease in the extinction coefficient affected dissolved ammonia more than an increase. An increase in the WSC caused more flagellates growth later in the summer into September and October (fig. 40), causing a decrease in the available dissolved ammonia (fig. 41) and dissolved nitrate plus nitrite (not shown). The slightly cooler temperatures were more favorable to flagellate growth than blue-green algae. A decrease in WSC, which caused the aforementioned blue-green algal bloom (fig. 39), led to less nutrient uptake because blue-green algae require less nitrogen from the water column. As with Madison Lake, inflow nitrogen increases and decreases led directly to comparable increases or decreases in the dissolved ammonia and dissolved nitrate plus nitrite pools. Changes in SOD, either positive or negative, led to mixed changes for dissolved ammonia and dissolved nitrate plus nitrite, but to a smaller degree than Madison Lake. An increase in the extinction coefficient caused more blue-green algae growth, given its greater light saturation constant for Pearl Lake in comparison to Madison Lake, so less ammonia was required for nutrient uptake (although dissolved nitrate plus nitrite concentrations did decrease by more than 10 percent).

Similar to Madison Lake, dissolved orthophosphate and total phosphorus tracked together in terms of the effects of the different sensitivity parameters for Pearl Lake. Two sensitive parameters, more sensitive than Madison Lake, were the changes in the sediment release rate of phosphorus and the changes in the inflow phosphorus of Pearl Lake. Both pools increased with a 20-percent increase in either of these two parameters, and both pools decreased with a 20-percent decrease. Similar to Madison Lake, dissolved orthophosphate and total phosphorus in Pearl Lake were more sensitive to WSC, SOD, and the extinction coefficients than other parameters. The WSC increase caused more phosphorus burial as additional DO mixing happened throughout the year; a decrease in WSC, which caused the aforementioned blue-green algal bloom (fig. 39), led to substantially more anoxia and therefore an increase in SOD release of phosphorus. With SOD increases, more phosphorus was released into the water column and vice versa for SOD decreases. Finally, orthophosphate was affected by extinction coefficient increases because less algal growth led to more orthophosphate in the water column.

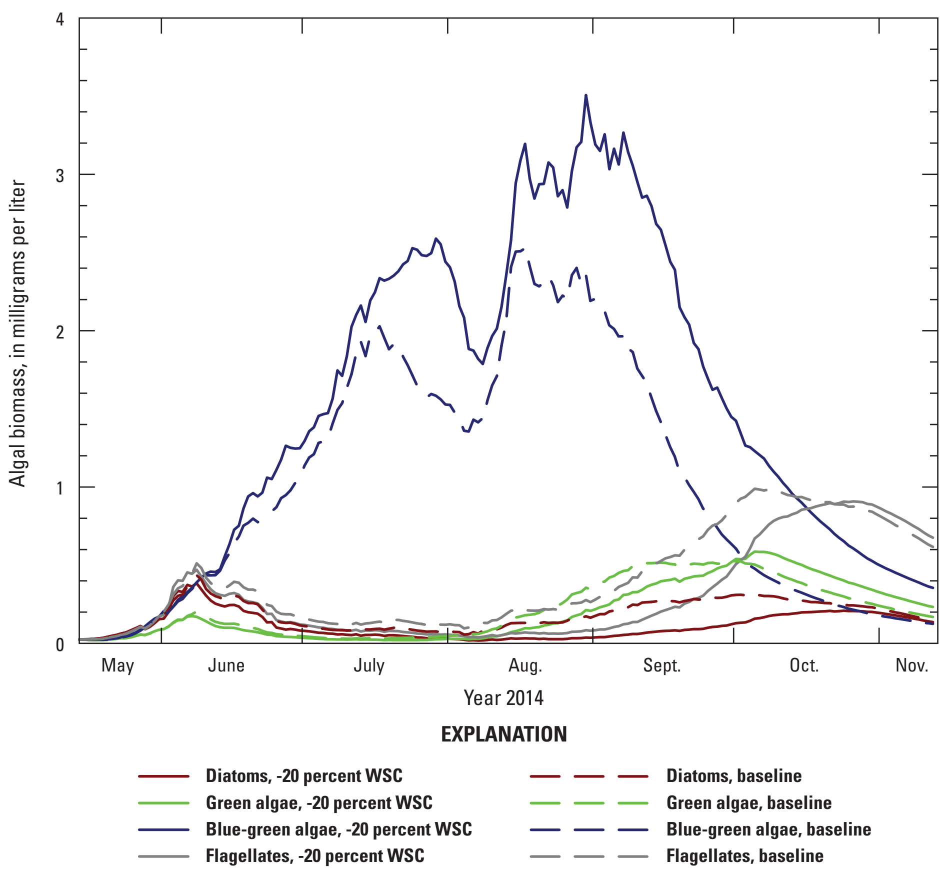


Figure 39. Simulated algal group distributions (diatoms, green algae, blue-green algae, and flagellates) for the 2-meter depth at Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with a 20-percent decrease in the wind sheltering coefficient (WSC) and the baseline (calibrated) model.

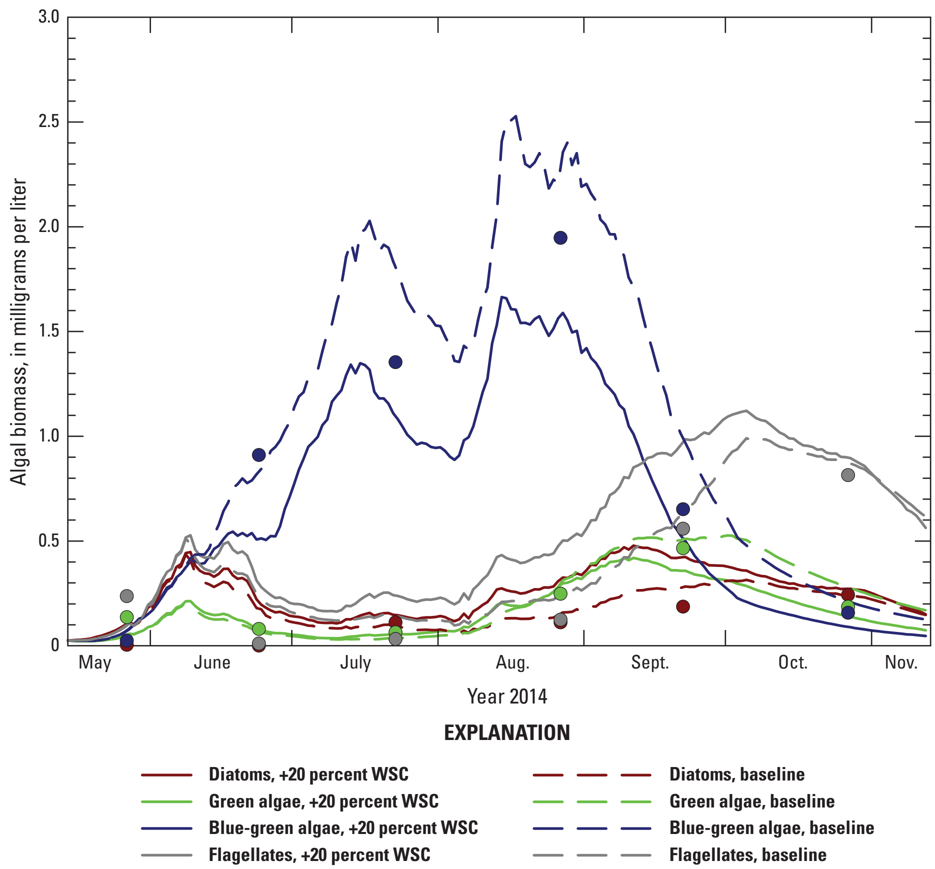


Figure 40. Simulated algal group distributions (diatoms, green algae, blue-green algae, and flagellates) for the 2-meter depth at Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with a 20-percent increase in the wind sheltering coefficient (WSC) and the baseline (calibrated) model.

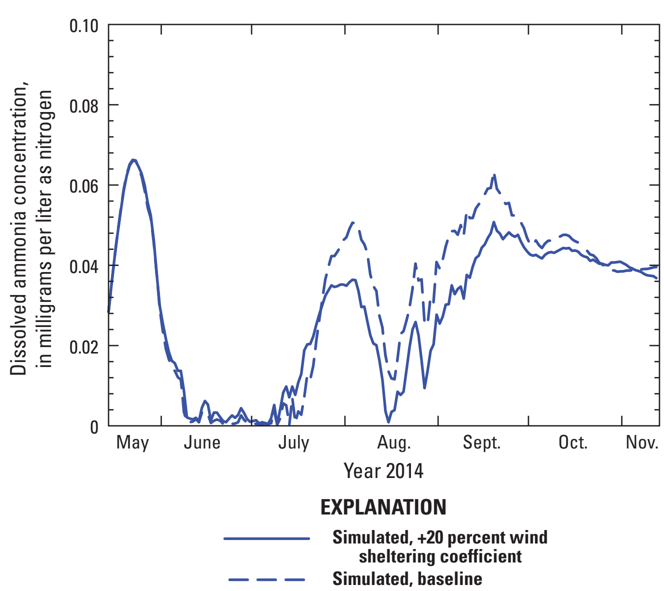


Figure 41. Simulated ammonia concentrations at 2 meters below the water surface in model segment 4 containing the Pearl Lake Deep Point near Marty, Minnesota, May 14 to November 13, 2014, with a 20-percent increase in the wind sheltering coefficient and the baseline (calibrated) model.

Summary

In Minnesota, lakes are facing substantial risks from land-use change and climate change. Although better management practices are being used on agricultural land, increased economic pressure towards high intensity row-crop agriculture challenges the paradigm of improving water quality. In recent years, water-resource scientists have been making the case for focused assessments and monitoring of “sentinel” systems to assess how these stressors affect lakes over the long term. Lakes and their contributing drainage basins are complex, and development of a mechanistic understanding of the linkage between basin-based stressors and lake metabolism is best accomplished by taking a long-term, adaptive approach towards water-resource management. Following the “sentinel” approach, the U.S. Geological Survey, in cooperation with the Minnesota Department of Natural Resources, developed predictive water-quality models for two agricultural land-use dominated lakes in Minnesota. The models were developed to assess algal community dynamics, water quality, and fish habitat suitability of these two lakes under recent (2014) meteorological conditions. The two selected lakes—Pearl Lake and Madison Lake—have abundant cool-water and warm-water fish communities but are threatened with frequent summer blue-green algal blooms that can potentially degrade fish habitat.

Sources and sinks are largely the same for both lakes. Madison Lake and Pearl Lake have fairly small flows from two different inflows and seem to have considerably large groundwater sources relative to surface inflows. Agricultural land use is the dominant land use for approximately 50 percent of the drainage basin for both lakes. Important distinctions between the two agricultural lakes are the ratio of the drainage basin to lake area, which is 24:1 for Pearl Lake but only 4:1 for Madison Lake. Also, the forest land-use percentage is different between the two different lakes: the Pearl Lake drainage basin has a forest land cover of 15 percent, whereas the Madison Lake drainage basin has a forest cover of 2 percent. Lake residence time is 1–2 years for Pearl Lake and 3–4 years for Madison Lake.

Hydrodynamics and water-quality characteristics were simulated using the CE-QUAL-W2 model, which is a carbon-based, laterally averaged, two-dimensional water-quality model. The CE-QUAL-W2 lake models address the interaction between nutrient cycling, primary production, and trophic dynamics to predict responses in the distribution of temperature and oxygen. The Madison Lake and Pearl Lake models were calibrated using data collected from May through November 2014, including at least one deep location per lake with vertical profiles of water temperature and dissolved oxygen concentration.

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. For Madison Lake, the mean absolute error and root mean square error were 0.53 and 0.68 degree Celsius, respectively, for the vertical profile comparisons. Simulated water temperatures were also matched to continuous profile data collected by thermistors at nine fixed depths; eight of the nine depths had mean absolute errors and root mean square errors of <0.90 and <1.10 degrees Celsius, respectively. For Pearl Lake, the mean absolute error and root mean square error were 0.71 and 0.95 degree Celsius, respectively, for the vertical profile comparisons. Altogether, simulated Madison Lake water temperatures tracked measured water temperatures throughout the water column; for Pearl Lake, simulated water temperature compared better to measured water temperature in the upper mixed layer than in the deeper mixed layer.

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. For Madison Lake, the mean absolute error and root mean square error were 0.68 milligram per liter and 1.15 milligrams per liter, respectively, for the vertical profile comparisons. 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. For Pearl Lake, the mean absolute error and root mean square error were 1.17 and 1.98 milligrams per liter, respectively, for the vertical profile comparisons. Pearl Lake had a total of six profiles, with the first two causing the larger errors; the mean absolute error and root mean square error for the last four profiles were 0.28 and 0.38 milligram per liter, respectively. Overall, dissolved oxygen was more sensitive than water temperature to alterations in the wind sheltering and extinction coefficients; additionally, sediment oxygen demand caused predictable changes for both lake models, with increases in sediment oxygen demand causing negative deviations in the deeper mixed layer and hypolimnion dissolved oxygen concentrations.

Algal dynamics were captured by four general groups: (1) bacillariophyta/crysophyta (diatoms), (2) chlorophyta (green algae), (3) cyanophyta (blue-green algae), and (4) haptophyta/cryptophyta (flagellates). Generally, both lake models successfully simulated algal growth dynamics throughout the year, in particular the prediction of persistent summer blue-green algae blooms in both lakes, early season diatoms in Madison Lake, and late season green algae and flagellate blooms in Pearl Lake.

For Madison Lake, the large nutrient concentrations, particularly nitrate, were tied back to basin processes. Large nitrate loads from the two inflow sites and the distributed tributary flow (mainly groundwater) were the initial sources of nitrate. Once in the lake, combined with consistent delivery of other nutrients such as phosphorus, algal growth and the recycling of nutrients proceeded. As the algae and macrophytes died, decomposition liberated ammonia and other nutrients, leading to a feedback loop. Also, as the decaying organic matter sank deeper into the lake, the deeper mixed layer and the hypolimnion became increasingly hypoxic. This increased hypoxia caused the release of sediment-bound phosphorus, which would initialize more algal growth. Between the external nutrient loading, internal nutrient loading from sediment release of phosphorus, and the organic matter decomposition of the algal and macrophyte biomass, even more algal and macrophyte growth was initiated. This recycling feedback between active growth and decomposition caused the series of algal blooms and the greater nutrient concentrations. Although dissolved ammonia, dissolved nitrate plus nitrite, and total Kjeldahl nitrogen concentrations generally decreased from the beginning of the simulation period to the end, total phosphorus in the hypolimnion continued to increase throughout most of the simulation period. The total phosphorus concentrations in the lake only decreased once the lake overturn caused the phosphorus to become readsorbed and sink to the bottom of the lake, albeit at concentrations of 130 micrograms per liter. At the total phosphorus peak, concentrations reached more than 1,100 micrograms per liter, or more than 1 milligram per liter. Pearl Lake, in comparison to Madison Lake, did not have nearly as large nutrient concentrations. Although external loading did provide nutrients to sustain greater algal growth and blue-green algae blooms, the difference in basin characteristics between the two lakes was likely a driver for lower nutrient concentrations.

Boundary factors, such as topography and shoreline tree cover, can have a substantial effect on wind mixing. Wind effects from these boundary factors were indirectly augmented through the wind sheltering coefficient. The assigned wind sheltering coefficient was determined to be a sensitive parameter that affected the amount of mixing in the vertical dimension and thereby the depth of the thermocline and dissolved oxygen chemocline over time. Sensitivity analyses were also completed to understand lake response effects through the usage of controlled departures on certain calibrated model parameters and input nutrient loads. These sensitivity analyses operated as land-use change scenarios because alterations in agricultural practices, for example, could potentially increase or decrease nutrient loads.

Available lake habitat suitable for cool-water and warm-water fish assemblages was evaluated for total volume of good-growth habitat, optimal growth habitat, and lethal oxythermal habitat. Criteria were based on thermal constraints and minimum dissolved oxygen thresholds. Overall, the fish habitat volume in general contained a large proportion of good-growth habitat and sustained period of optimal growth habitat in the summer, without any periods of lethal oxythermal habitat.

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Appendixes

Appendix 1. Elevation-Streamflow Ratings, Unit Value Streamflows, and Daily Mean Streamflows for Madison Lake Inflow and Outflow Sites

Elevation-streamflow ratings and daily mean streamflows for gaged Madison Lake inflow and outflow sites for the 2014 model calibration are shown in tables 1–1 through 1–4. Elevation is shown in water-surface elevation referenced to the North American Vertical Datum of 1988. Streamflow values are shown in cubic feet per second and cubic meters per second. The three sites shown in tables 1–1 through 1–4 are unnamed stream to Madison Lake at CR-48 near Madison Lake, Minn. (U.S. Geological Survey [USGS] station 05320130), unnamed stream between Schoolhouse and Goolsby Lakes southeast of Madison Lake, Minn. (USGS station 05320140), and Madison Lake outlet to Mud Lake south of Madison Lake, Minn. (USGS station 05320170). The first tab (table 1–1) contains the elevation-streamflow ratings for all three Madison Lake sites; the second tab (table 1–2) contains the daily mean streamflow record for unnamed stream to Madison Lake at CR-48 near Madison Lake, Minn. (USGS station 05320130), organized monthly similar to the way daily streamflow values are published in USGS Annual Water Data Reports (http://wdr.water.usgs.gov/); the third tab (table 1–3) contains the daily mean streamflow record for unnamed stream between Schoolhouse and Goolsby Lakes southeast of Madison Lake, Minn. (USGS station 05320140); the fourth tab (table 1–4) contains the daily mean streamflow record for Madison Lake outlet to Mud Lake south of Madison Lake, Minn. (USGS station 05320170).

Table 1–1. Elevation-streamflow ratings for gaged Madison Lake inflow and outflow sites.

[ft; foot; ft^3/s, cubic foot per second; m^3/s, meter per second; ---, not determined]

Elevation (ft)	Streamflow (ft^3/s)			Streamflow (m^3/s)		
	05320130	05320140	05320170	05320130	05320140	05320170
1,016.56	0.000	0.000	0.000	0.000	0.000	0.000
1,016.57	0.079	0.029	0.004	0.002	0.001	0.000
1,016.58	0.158	0.059	0.023	0.004	0.002	0.001
1,016.59	0.237	0.088	0.064	0.007	0.002	0.002
1,016.60	0.316	0.118	0.131	0.009	0.003	0.004
1,016.61	0.395	0.147	0.229	0.011	0.004	0.006
1,016.62	0.474	0.176	0.357	0.013	0.005	0.010
1,016.63	0.553	0.206	0.513	0.016	0.006	0.015
1,016.64	0.632	0.235	0.704	0.018	0.007	0.020
1,016.65	0.711	0.265	0.933	0.020	0.007	0.026
1,016.66	0.790	0.294	1.200	0.022	0.008	0.034
1,016.67	0.869	0.323	1.520	0.025	0.009	0.043
1,016.68	0.948	0.353	1.890	0.027	0.010	0.054
1,016.69	1.027	0.382	2.300	0.029	0.011	0.065
1,016.70	1.106	0.412	2.720	0.031	0.012	0.077
1,016.71	1.185	0.441	3.100	0.034	0.012	0.088
1,016.72	1.264	0.470	3.500	0.036	0.013	0.099
1,016.73	1.343	0.500	3.940	0.038	0.014	0.112
1,016.74	1.422	0.529	4.400	0.040	0.015	0.125
1,016.75	1.501	0.559	4.900	0.043	0.016	0.139
1,016.76	1.558	0.588	5.180	0.044	0.017	0.147
1,016.77	1.615	0.617	5.390	0.046	0.017	0.153
1,016.78	1.672	0.647	5.610	0.047	0.018	0.159
1,016.79	1.729	0.676	5.830	0.049	0.019	0.165
1,016.80	1.786	0.706	6.060	0.051	0.020	0.172
1,016.81	1.843	0.735	6.290	0.052	0.021	0.178
1,016.82	1.900	0.764	6.530	0.054	0.022	0.185
1,016.83	1.957	0.794	6.770	0.055	0.022	0.192
1,016.84	2.014	0.823	7.020	0.057	0.023	0.199
1,016.85	2.071	0.853	7.280	0.059	0.024	0.206
1,016.86	2.128	0.882	7.530	0.060	0.025	0.213
1,016.87	2.185	0.911	7.800	0.062	0.026	0.221
1,016.88	2.242	0.941	8.060	0.063	0.027	0.228
1,016.89	2.299	0.970	8.330	0.065	0.027	0.236
1,016.90	2.356	1.000	8.610	0.067	0.028	0.244
1,016.91	2.413	1.042	8.890	0.068	0.030	0.252
1,016.92	2.470	1.084	9.180	0.070	0.031	0.260
1,016.93	2.527	1.126	9.480	0.072	0.032	0.268
1,016.94	2.584	1.168	9.780	0.073	0.033	0.277
1,016.95	2.641	1.210	10.100	0.075	0.034	0.286
1,016.96	2.698	1.252	10.400	0.076	0.035	0.294
1,016.97	2.755	1.294	10.700	0.078	0.037	0.303
1,016.98	2.812	1.336	11.000	0.080	0.038	0.311
1,016.99	2.869	1.378	11.400	0.081	0.039	0.323
1,017.00	2.926	1.420	11.700	0.083	0.040	0.331
1,017.01	2.983	1.462	12.000	0.084	0.041	0.340
1,017.02	3.040	1.504	12.400	0.086	0.043	0.351
1,017.03	3.097	1.546	12.700	0.088	0.044	0.360
1,017.04	3.154	1.588	13.100	0.089	0.045	0.371
1,017.05	3.211	1.630	13.400	0.091	0.046	0.379
1,017.06	3.268	1.681	13.800	0.093	0.048	0.391
1,017.07	3.325	1.732	14.200	0.094	0.049	0.402
1,017.08	3.390	1.783	14.600	0.096	0.050	0.413
1,017.09	3.514	1.834	15.000	0.100	0.052	0.425
1,017.10	3.638	1.885	15.400	0.103	0.053	0.436
1,017.11	3.762	1.935	15.800	0.107	0.055	0.447
1,017.12	3.886	1.986	16.200	0.110	0.056	0.459
1,017.13	4.010	2.037	16.600	0.114	0.058	0.470
1,017.14	4.134	2.088	17.000	0.117	0.059	0.481
1,017.15	4.258	2.139	17.400	0.121	0.061	0.493
1,017.16	4.382	2.190	17.800	0.124	0.062	0.504
1,017.17	4.506	2.241	18.300	0.128	0.063	0.518
1,017.18	4.630	2.292	18.700	0.131	0.065	0.530
1,017.19	4.754	2.343	19.200	0.135	0.066	0.544
1,017.20	4.878	2.394	19.600	0.138	0.068	0.555
1,017.21	5.000	2.444	20.000	0.142	0.069	0.566
1,017.22	5.143	2.495	20.400	0.146	0.071	0.578
1,017.23	5.286	2.546	20.900	0.150	0.072	0.592

Table 1–1. Elevation-streamflow ratings for gaged Madison Lake inflow and outflow sites.—Continued

[ft; foot; ft^3/s, cubic foot per second; m^3/s, meter per second; ---, not determined]

Elevation (ft)	Streamflow (ft^3/s)			Streamflow (m^3/s)		
	05320130	05320140	05320170	05320130	05320140	05320170
1,017.24	5.429	2.597	21.300	0.154	0.074	0.603
1,017.25	5.572	2.648	21.800	0.158	0.075	0.617
1,017.26	5.715	2.699	22.200	0.162	0.076	0.629
1,017.27	5.858	2.750	22.700	0.166	0.078	0.643
1,017.28	6.001	2.801	23.200	0.170	0.079	0.657
1,017.29	6.144	2.852	23.600	0.174	0.081	0.668
1,017.30	6.287	2.902	24.100	0.178	0.082	0.682
1,017.31	6.430	2.953	24.600	0.182	0.084	0.697
1,017.32	6.573	3.004	25.100	0.186	0.085	0.711
1,017.33	6.716	3.055	25.600	0.190	0.087	0.725
1,017.34	6.859	3.106	26.100	0.194	0.088	0.739
1,017.35	7.000	3.157	26.600	0.198	0.089	0.753
1,017.36	7.200	3.208	27.100	0.204	0.091	0.767
1,017.37	7.400	3.259	27.700	0.210	0.092	0.784
1,017.38	7.600	3.310	28.200	0.215	0.094	0.799
1,017.39	7.800	3.525	28.700	0.221	0.100	0.813
1,017.40	8.000	3.740	29.300	0.227	0.106	0.830
1,017.41	8.200	3.955	29.800	0.232	0.112	0.844
1,017.42	8.400	4.170	30.400	0.238	0.118	0.861
1,017.43	8.600	4.385	30.900	0.244	0.124	0.875
1,017.44	8.800	4.600	31.500	0.249	0.130	0.892
1,017.45	9.000	4.815	32.000	0.255	0.136	0.906
1,017.46	9.200	5.030	32.600	0.261	0.142	0.923
1,017.47	9.400	5.245	33.200	0.266	0.149	0.940
1,017.48	9.600	5.460	33.800	0.272	0.155	0.957
1,017.49	9.800	5.675	34.400	0.278	0.161	0.974
1,017.50	10.000	6.100	35.000	0.283	0.173	0.991
1,017.51	10.250	6.391	35.600	0.290	0.181	1.008
1,017.52	10.500	6.682	36.200	0.297	0.189	1.025
1,017.53	10.750	6.973	36.800	0.304	0.197	1.042
1,017.54	11.000	7.264	37.500	0.311	0.206	1.062
1,017.55	11.250	7.555	38.100	0.319	0.214	1.079
1,017.56	11.500	7.846	38.700	0.326	0.222	1.096
1,017.57	11.750	8.137	39.300	0.333	0.230	1.113
1,017.58	12.000	8.428	39.900	0.340	0.239	1.130
1,017.59	12.250	8.719	40.600	0.347	0.247	1.150
1,017.60	12.500	9.300	41.200	0.354	0.263	1.167
1,017.61	12.750	9.655	41.900	0.361	0.273	1.186
1,017.62	13.000	10.010	42.500	0.368	0.283	1.203
1,017.63	13.250	10.365	43.200	0.375	0.294	1.223
1,017.64	13.500	10.720	43.900	0.382	0.304	1.243
1,017.65	13.750	11.075	44.500	0.389	0.314	1.260
1,017.66	14.000	11.430	45.200	0.396	0.324	1.280
1,017.67	14.250	11.785	45.900	0.404	0.334	1.300
1,017.68	14.500	12.140	46.600	0.411	0.344	1.320
1,017.69	14.750	12.495	47.300	0.418	0.354	1.339
1,017.70	15.000	13.200	48.000	0.425	0.374	1.359
1,017.71	15.292	13.680	48.700	0.433	0.387	1.379
1,017.72	15.584	14.160	49.400	0.441	0.401	1.399
1,017.73	15.876	14.640	50.200	0.450	0.415	1.422
1,017.74	16.168	15.120	50.900	0.458	0.428	1.441
1,017.75	16.460	15.600	51.600	0.466	0.442	1.461
1,017.76	16.752	16.080	52.400	0.474	0.455	1.484
1,017.77	17.044	16.560	53.100	0.483	0.469	1.504
1,017.78	17.336	17.040	53.900	0.491	0.483	1.526
1,017.79	17.628	17.520	54.600	0.499	0.496	1.546
1,017.80	17.920	18.000	55.400	0.507	0.510	1.569
1,017.81	18.212	18.560	56.200	0.516	0.526	1.591
1,017.82	18.504	19.120	57.000	0.524	0.541	1.614
1,017.83	18.796	19.680	57.800	0.532	0.557	1.637
1,017.84	19.088	20.240	58.500	0.541	0.573	1.657
1,017.85	19.380	20.800	59.300	0.549	0.589	1.679
1,017.86	19.672	21.360	60.100	0.557	0.605	1.702
1,017.87	19.964	21.920	61.000	0.565	0.621	1.727
1,017.88	20.256	22.480	61.800	0.574	0.637	1.750
1,017.89	20.548	23.040	62.600	0.582	0.652	1.773
1,017.90	20.840	23.600	63.400	0.590	0.668	1.795
1,017.91	21.132	24.275	64.300	0.598	0.687	1.821
1,017.92	21.424	24.950	65.100	0.607	0.707	1.843
1,017.93	21.716	25.625	66.000	0.615	0.726	1.869
1,017.94	22.000	26.300	66.800	0.623	0.745	1.892
1,017.95	22.295	26.975	67.700	0.631	0.764	1.917
1,017.96	22.590	27.650	68.600	0.640	0.783	1.943
1,017.97	22.885	28.325	69.400	0.648	0.802	1.965
1,017.98	23.180	29.000	70.300	0.656	0.821	1.991
1,017.99	23.475	29.675	71.200	0.665	0.840	2.016
1,018.00	23.770	30.350	72.100	0.673	0.859	2.042
1,018.01	24.065	31.025	73.000	0.681	0.879	2.067
1,018.02	24.360	31.700	73.900	0.690	0.898	2.093
1,018.03	24.655	32.375	74.800	0.698	0.917	2.118
1,018.04	24.950	33.050	75.800	0.707	0.936	2.146
1,018.05	25.245	33.725	76.700	0.715	0.955	2.172
1,018.06	25.540	34.400	77.600	0.723	0.974	2.197
1,018.07	25.835	35.070	78.600	0.732	0.993	2.226
1,018.08	26.130	35.740	79.500	0.740	1.012	2.251
1,018.09	26.425	36.410	80.500	0.748	1.031	2.280
1,018.10	26.720	37.080	81.500	0.757	1.050	2.308
1,018.11	27.015	37.750	82.400	0.765	1.069	2.333
1,018.12	27.310	38.420	83.400	0.773	1.088	2.362
1,018.13	27.605	39.090	84.400	0.782	1.107	2.390
1,018.14	27.900	39.760	85.362	0.790	1.126	2.417
1,018.15	28.195	40.430	86.324	0.798	1.145	2.444
1,018.16	28.490	41.100	87.286	0.807	1.164	2.472
1,018.17	28.785	41.770	88.248	0.815	1.183	2.499
1,018.18	29.080	42.440	89.210	0.823	1.202	2.526
1,018.19	29.375	43.110	90.172	0.832	1.221	2.553
1,018.20	29.670	43.780	91.134	0.840	1.240	2.581
1,018.21	29.965	44.450	92.096	0.849	1.259	2.608
1,018.22	30.260	45.120	93.058	0.857	1.278	2.635
1,018.23	30.555	45.790	94.020	0.865	1.297	2.662
1,018.24	30.850	46.460	94.982	0.874	1.316	2.690
1,018.25	31.145	47.130	95.944	0.882	1.335	2.717
1,018.26	31.440	47.800	96.906	0.890	1.354	2.744
1,018.27	31.735	48.470	97.868	0.899	1.373	2.771
1,018.28	32.030	49.140	98.830	0.907	1.391	2.799
1,018.29	32.325	49.810	99.792	0.915	1.410	2.826

Table 1–1. Elevation-streamflow ratings for gaged Madison Lake inflow and outflow sites.—Continued

[ft; foot; ft³/s, cubic foot per second; m³/s, meter per second; ---, not determined]

Elevation (ft)	Streamflow (ft³/s)			Streamflow (m³/s)		
	05320130	05320140	05320170	05320130	05320140	05320170
1,018.30	32.620	50.480	100.754	0.924	1.429	2.853
1,018.31	32.915	51.150	101.716	0.932	1.448	2.880
1,018.32	33.210	51.820	102.678	0.940	1.467	2.908
1,018.33	33.505	52.490	103.640	0.949	1.486	2.935
1,018.34	33.800	53.160	104.602	0.957	1.505	2.962
1,018.35	34.095	53.830	105.564	0.965	1.524	2.989
1,018.36	34.390	54.500	106.526	0.974	1.543	3.016
1,018.37	34.685	55.170	107.488	0.982	1.562	3.044
1,018.38	34.980	55.840	108.450	0.991	1.581	3.071
1,018.39	35.275	56.510	109.412	0.999	1.600	3.098
1,018.40	35.570	57.180	110.374	1.007	1.619	3.125
1,018.41	35.865	57.850	111.336	1.016	1.638	3.153
1,018.42	36.160	58.520	112.298	1.024	1.657	3.180
1,018.43	36.455	59.190	113.260	1.032	1.676	3.207
1,018.44	36.750	59.860	114.222	1.041	1.695	3.234
1,018.45	37.045	60.530	115.184	1.049	1.714	3.262
1,018.46	37.340	61.200	116.146	1.057	1.733	3.289
1,018.47	37.635	61.870	117.108	1.066	1.752	3.316
1,018.48	37.930	62.540	118.070	1.074	1.771	3.343
1,018.49	38.225	63.210	119.032	1.082	1.790	3.371
1,018.50	38.520	63.880	119.994	1.091	1.809	3.398
1,018.51	38.815	64.550	120.956	1.099	1.828	3.425
1,018.52	39.110	65.220	121.918	1.107	1.847	3.452
1,018.53	39.405	65.890	122.880	1.116	1.866	3.480
1,018.54	39.700	66.560	123.842	1.124	1.885	3.507
1,018.55	39.995	67.230	124.804	1.133	1.904	3.534
1,018.56	40.290	67.900	125.766	1.141	1.923	3.561
1,018.57	40.585	68.570	126.728	1.149	1.942	3.589
1,018.58	40.880	69.240	127.690	1.158	1.961	3.616
1,018.59	41.175	69.910	128.652	1.166	1.980	3.643
1,018.60	41.470	70.580	129.614	1.174	1.999	3.670
1,018.61	41.765	71.250	130.576	1.183	2.018	3.698
1,018.62	42.060	71.920	131.538	1.191	2.037	3.725
1,018.63	42.355	72.590	132.500	1.199	2.056	3.752
1,018.64	42.650	73.260	133.462	1.208	2.074	3.779
1,018.65	42.945	73.930	134.424	1.216	2.093	3.806
1,018.66	43.240	74.600	135.386	1.224	2.112	3.834
1,018.67	43.535	75.270	136.348	1.233	2.131	3.861
1,018.68	43.830	75.940	137.310	1.241	2.150	3.888
1,018.69	44.125	76.610	138.272	1.249	2.169	3.915
1,018.70	44.420	77.280	139.234	1.258	2.188	3.943
1,018.71	44.715	77.950	140.196	1.266	2.207	3.970
1,018.72	45.010	78.620	141.158	1.275	2.226	3.997
1,018.73	45.305	79.290	142.120	1.283	2.245	4.024
1,018.74	45.600	79.960	143.082	1.291	2.264	4.052
1,018.75	45.895	80.630	144.044	1.300	2.283	4.079
1,018.76	46.190	81.300	145.006	1.308	2.302	4.106
1,018.77	46.485	81.970	---	1.316	2.321	---
1,018.78	46.780	82.640	---	1.325	2.340	---
1,018.79	47.075	83.310	---	1.333	2.359	---
1,018.80	47.370	83.980	---	1.341	2.378	---
1,018.81	47.665	84.650	---	1.350	2.397	---
1,018.82	47.960	85.320	---	1.358	2.416	---
1,018.83	48.255	85.990	---	1.366	2.435	---
1,018.84	48.550	86.660	---	1.375	2.454	---
1,018.85	48.845	87.330	---	1.383	2.473	---
1,018.86	49.140	88.000	---	1.391	2.492	---
1,018.87	49.435	88.670	---	1.400	2.511	---
1,018.88	49.730	89.340	---	1.408	2.530	---
1,018.89	50.025	90.010	---	1.417	2.549	---
1,018.90	50.320	90.680	---	1.425	2.568	---
1,018.91	50.615	91.350	---	1.433	2.587	---
1,018.92	50.910	92.020	---	1.442	2.606	---
1,018.93	51.205	92.690	---	1.450	2.625	---
1,018.94	51.500	93.360	---	1.458	2.644	---
1,018.95	51.795	94.030	---	1.467	2.663	---
1,018.96	52.090	94.700	---	1.475	2.682	---
1,018.97	52.385	95.370	---	1.483	2.701	---
1,018.98	52.680	96.000	---	1.492	2.718	---
1,018.99	52.975	---	---	1.500	---	---
1,019.00	53.270	---	---	1.508	---	---
1,019.01	53.565	---	---	1.517	---	---
1,019.02	53.860	---	---	1.525	---	---
1,019.03	54.155	---	---	1.533	---	---
1,019.04	54.450	---	---	1.542	---	---
1,019.05	54.745	---	---	1.550	---	---
1,019.06	55.040	---	---	1.559	---	---
1,019.07	55.335	---	---	1.567	---	---
1,019.08	55.630	---	---	1.575	---	---
1,019.09	55.925	---	---	1.584	---	---
1,019.10	56.220	---	---	1.592	---	---
1,019.11	56.515	---	---	1.600	---	---
1,019.12	56.810	---	---	1.609	---	---
1,019.13	57.105	---	---	1.617	---	---
1,019.14	57.400	---	---	1.625	---	---
1,019.15	57.695	---	---	1.634	---	---
1,019.16	57.990	---	---	1.642	---	---
1,019.17	58.285	---	---	1.650	---	---
1,019.18	58.580	---	---	1.659	---	---
1,019.19	58.875	---	---	1.667	---	---
1,019.20	59.170	---	---	1.676	---	---
1,019.21	59.465	---	---	1.684	---	---
1,019.22	59.760	---	---	1.692	---	---
1,019.23	60.055	---	---	1.701	---	---
1,019.24	60.350	---	---	1.709	---	---
1,019.25	60.645	---	---	1.717	---	---
1,019.26	60.940	---	---	1.726	---	---
1,019.27	61.235	---	---	1.734	---	---
1,019.28	61.530	---	---	1.742	---	---
1,019.29	61.825	---	---	1.751	---	---
1,019.30	62.120	---	---	1.759	---	---
1,019.31	62.415	---	---	1.767	---	---
1,019.32	62.710	---	---	1.776	---	---
1,019.33	63.005	---	---	1.784	---	---
1,019.34	63.300	---	---	1.792	---	---
1,019.35	63.595	---	---	1.801	---	---

Table 1–1. Elevation-streamflow ratings for gaged Madison Lake inflow and outflow sites.—Continued

[ft; foot; ft³/s, cubic foot per second; m³/s, meter per second; ---, not determined]

Elevation (ft)	Streamflow (ft³/s)			Streamflow (m³/s)		
	05320130	05320140	05320170	05320130	05320140	05320170
1,019.36	63.890	---	---	1.809	---	---
1,019.37	64.185	---	---	1.818	---	---
1,019.38	64.480	---	---	1.826	---	---
1,019.39	64.775	---	---	1.834	---	---
1,019.40	65.070	---	---	1.843	---	---
1,019.41	65.365	---	---	1.851	---	---
1,019.42	65.660	---	---	1.859	---	---
1,019.43	65.955	---	---	1.868	---	---
1,019.44	66.250	---	---	1.876	---	---
1,019.45	66.545	---	---	1.884	---	---
1,019.46	66.840	---	---	1.893	---	---
1,019.47	67.135	---	---	1.901	---	---
1,019.48	67.430	---	---	1.909	---	---
1,019.49	67.725	---	---	1.918	---	---
1,019.50	68.000	---	---	1.926	---	---

Table 1–2. Daily mean streamflows for unnamed stream to Madison Lake at CR-48 near Madison Lake, Minnesota (USGS station 05320130), May–November 2014.

[---, not determined]

Day	Daily mean streanflows, in cubic meters per second						
	May 2014	June 2014	July 2014	August 2014	September 2014	October 2014	November 2014
1	---	0.2880	0.8678	0.0642	0.0189	---	---
2	---	0.3347	0.7976	0.0611	0.0164	---	---
3	---	0.3109	0.6643	0.0588	0.0128	---	---
4	---	0.2827	0.5930	0.0561	0.0114	---	---
5	---	0.2667	0.5374	0.0526	0.0059	---	---
6	---	0.2467	0.4909	0.0487	0.0011	---	---
7	---	0.2327	0.4426	0.0449	---	---	---
8	---	0.2167	0.3930	0.0417	---	---	---
9	---	0.1966	0.3421	0.0385	---	---	---
10	---	0.1827	0.2960	0.0354	---	---	---
11	---	0.1726	0.2705	0.0323	---	---	---
12	---	0.1641	0.2555	0.0253	---	---	---
13	---	0.1408	0.2409	0.0203	---	---	---
14	---	0.1327	0.2168	0.0150	---	---	---
15	0.5901	0.1839	0.1951	0.0119	---	---	---
16	0.5587	0.2013	0.2102	0.0087	---	---	---
17	0.5290	0.3181	0.1881	0.0067	---	---	---
18	0.4985	0.9070	0.1713	0.0079	---	---	---
19	0.4656	1.3229	0.1548	0.0046	---	---	---
20	0.4441	1.5764	0.1398	0.0008	---	---	---
21	0.4203	1.5385	0.1275	0.0265	---	---	---
22	0.3917	1.4527	0.1183	0.0302	---	---	---
23	0.3644	1.3690	0.1042	0.0274	---	---	---
24	0.3392	1.2886	0.0946	0.0268	---	---	---
25	0.3128	1.2144	0.0941	0.0237	---	---	---
26	0.2921	1.1488	0.0912	0.0164	---	---	---
27	0.2977	1.0833	0.0863	0.0111	---	---	---
28	0.2803	1.0325	0.0800	0.0075	---	---	---
29	0.2585	0.9950	0.0756	0.0137	---	---	---
30	0.2372	0.9289	0.0712	0.0133	---	---	---
31	0.2294	---	0.0676	0.0120	---	---	---

Table 1–3. Daily mean streamflows for unnamed stream between Schoolhouse and Goolsby Lakes southeast of Madison Lake, Minnesota (USGS station 05320140), May–November 2014.

[---, not determined]

Day	Daily mean streanflows, in cubic meters per second						
	May 2014	June 2014	July 2014	August 2014	September 2014	October 2014	November 2014
1	---	0.3046	1.6030	0.0457	0.0240	0.0005	---
2	---	0.3716	1.4468	0.0435	0.0229	0.0048	---
3	---	0.3381	1.2917	0.0424	0.0209	0.0102	---
4	---	0.3130	1.1397	0.0404	0.0206	0.0056	---
5	---	0.2759	1.0067	0.0381	0.0202	0.0014	---
6	---	0.2538	0.9170	0.0353	0.0173	0.0004	---
7	---	0.2391	0.8146	0.0342	0.0147	0.0006	---
8	---	0.2127	0.7192	0.0338	0.0122	---	---
9	---	0.1856	0.5822	0.0317	0.0117	---	---
10	---	0.1563	0.4804	0.0298	0.0152	---	---
11	---	0.1377	0.4203	0.0303	0.0121	---	---
12	---	0.1322	0.3854	0.0267	0.0101	---	---
13	---	0.0957	0.3459	0.0243	0.0079	---	---
14	---	0.0895	0.3113	0.0222	0.0069	---	---
15	0.9277	0.1656	0.2568	0.0204	0.0072	---	---
16	0.8540	0.2148	0.1997	0.0198	0.0055	---	---
17	0.7903	0.4586	0.1456	0.0187	0.0047	---	---
18	0.7031	1.6554	0.1211	0.0193	0.0036	---	---
19	0.6433	2.1885	0.0990	0.0187	0.0013	---	---
20	0.6243	2.5215	0.0899	0.0166	0.0041	---	---
21	0.5903	2.5847	0.0864	0.0272	0.0060	---	---
22	0.5210	2.5794	0.0832	0.0301	0.0032	---	---
23	0.4615	2.5421	0.0784	0.0283	0.0014	---	---
24	0.4033	2.4670	0.0728	0.0266	0.0008	---	---
25	0.3480	2.3551	0.0717	0.0261	0.0008	---	---
26	0.3246	2.2022	0.0696	0.0238	0.0003	---	---
27	0.3427	2.0575	0.0647	0.0215	---	---	---
28	0.3178	1.9296	0.0608	0.0200	---	---	---
29	0.2761	1.8486	0.0553	0.0218	---	---	---
30	0.2305	1.7269	0.0516	0.0225	---	---	---
31	0.2069	---	0.0482	0.0205	---	---	---

Table 1–4. Daily mean streamflows for Madison Lake outlet to Mud Lake south of Madison Lake, Minnesota (USGS station 05320170), May–November 2014.

[---, not determined]

Day	Daily mean streanflows, in cubic meters per second						
	May 2014	June 2014	July 2014	August 2014	September 2014	October 2014	November 2014
1	---	1.0276	2.5903	0.2730	0.0715	---	---
2	---	1.1306	2.3929	0.2578	0.0587	---	---
3	---	1.0863	2.1918	0.2505	0.0450	---	---
4	---	1.0419	2.0003	0.2374	0.0383	---	---
5	---	0.9785	1.8352	0.2229	0.0238	---	---
6	---	0.9477	1.7030	0.2052	0.0105	---	---
7	---	0.9343	1.5730	0.1872	0.0038	---	---
8	---	0.8889	1.4312	0.1758	0.0006	---	---
9	---	0.8377	1.2988	0.1651	0.0003	---	---
10	---	0.7866	1.1820	0.1543	0.0026	---	---
11	---	0.7422	1.1088	0.1416	0.0003	---	---
12	---	0.7296	1.0629	0.1120	0.0000	---	---
13	---	0.6430	1.0174	0.0864	---	---	---
14	---	0.6051	0.9451	0.0659	---	---	---
15	1.8487	0.8012	0.8734	0.0430	---	---	---
16	1.7639	0.8761	0.8144	0.0331	---	---	---
17	1.6840	1.2464	0.7465	0.0268	---	---	---
18	1.5994	1.8112	0.6895	0.0264	---	---	---
19	1.5457	2.8965	0.6389	0.0187	---	---	---
20	1.4994	3.7211	0.5957	0.0124	---	---	---
21	1.4256	3.9719	0.5652	0.1180	---	---	---
22	1.3528	3.9889	0.5395	0.1501	---	---	---
23	1.2831	3.9262	0.4971	0.1466	---	---	---
24	1.2043	3.8079	0.4532	0.1261	---	---	---
25	1.1311	3.6620	0.4443	0.1094	---	---	---
26	1.0875	3.4731	0.4278	0.0771	---	---	---
27	1.1111	3.2781	0.3895	0.0526	---	---	---
28	1.0786	3.1115	0.3625	0.0404	---	---	---
29	0.9805	2.9949	0.3341	0.0530	---	---	---
30	0.9215	2.8020	0.3102	0.0531	---	---	---
31	0.8776	---	0.2896	0.0406	---	---	---

Appendix 2. Relative Counts and Converted Algal Biomass for Madison Lake, Minnesota, and Pearl Lake, Minnesota

Relative counts and converted algal biomass (in milligrams per liter) for Madison Lake southwest deep point near Madison Lake, Minnesota, and Pearl Lake Deep Point near Marty, Minnesota, for 2014.

Table 2–1. Summary of relative counts and converted algal biomass for Madison Lake and Pearl Lake, May–November 2014.

[mg/L, milligram per liter]

Constituent	Common name in report	Date	Relative count	Converted algal biomass (mg/L)
		Madison Lake		
Diatoms/crysophyta	Diatoms	2014–05–14	88	3.168
		2014–06–18	5	0.054
		2014–07–29	38	1.767
		2014–08–21	5	0.369
		2014–09–17	16	1.277
		2014–10–21	1	0.003
Chlorophyta	Green algae	2014–05–14	6	0.216
		2014–06–18	9	0.098
		2014–07–29	9	0.418
		2014–08–21	14	1.033
		2014–09–17	6	0.479
		2014–10–21	3	0.009
Cyanophyta	Blue-green algae	2014–05–14	3	0.108
		2014–06–18	79	0.856
		2014–07–29	45	2.092
		2014–08–21	70	5.167
		2014–09–17	63	5.027
		2014–10–21	1	0.003
Haptophyta/cryptophyta	Flagellates	2014–05–14	3	0.108
		2014–06–18	3	0.033
		2014–07–29	5	0.232
		2014–08–21	8	0.591
		2014–09–17	15	1.197
		2014–10–21	94	0.291
		Pearl Lake		
Diatoms/crysophyta	Diatoms	2014–05–27	1	0.004
		2014–06–24	0	0.000
		2014–07–23	7	0.111
		2014–08–27	9	0.112
		2014–09–22	10	0.186
		2014–10–27	17	0.243
Chlorophyta	Green algae	2014–05–27	34	0.136
		2014–06–24	8	0.080
		2014–07–23	4	0.064
		2014–08–27	20	0.248
		2014–09–22	25	0.465
		2014–10–27	13	0.186
Cyanophyta	Blue-green algae	2014–05–27	6	0.024
		2014–06–24	91	0.910
		2014–07–23	85	1.353
		2014–08–27	157	1.946
		2014–09–22	35	0.651
		2014–10–27	11	0.157
Haptophyta/cryptophyta	Flagellates	2014–05–27	59	0.237
		2014–06–24	1	0.010
		2014–07–23	2	0.032
		2014–08–27	10	0.124
		2014–09–22	30	0.558
		2014–10–27	57	0.814

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