Lake St. Croix Nutrient Loading and Ecological Health Assessment

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Final Project Report

Lake St. Croix Nutrient Loading and Ecological Health Assessment

Submitted to the St. Croix River Association

June 3, 2013

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EXECUTIVE SUMMARY

Located on the Minnesota-Wisconsin state boundary, Lake St. Croix is a naturally-impounded riverine lake encompassing the lower 40 kilometers (25 miles) of the St. Croix River before its confluence with the Mississippi River at Prescott, WI. Increasingly, Lake St. Croix is subject to nuisance algal blooms. These changes are thought to be due to expanding urban and agricultural inputs. Despite reductions in point-source nutrient inputs (Edlund et al. 2009), nutrient concentrations still exceed water-quality guidelines (MPCA and WDNR 2012). In addition, though concentrations of total phosphorus in the lake at Stillwater, MN have declined gradually over 30-years of monitoring, concentrations of chlorophyll-a have not declined over the same period (Lafrancois et al. 2009). Reductions in the primary nutrient phosphorus are not leading to reductions in a key ecological response variable, algal productivity. The observed ecological decoupling within the lake was the driving force behind this project.

The project had two overall goals: 1) to improve existing phosphorus mass balance estimates for Lake St. Croix, and 2) to develop an improved understanding of the relationship between water quality and ecological health of Lake St. Croix. This project was conceived as an interagency team effort, capitalizing on the expertise of the team members. To accomplish the goals, the project was divided into tasks that would be managed by a lead agency for each task:

- Task 1: Install and operate a flow gage on the St. Croix River at Stillwater, MN (USGS)
- Task 2: Collect monthly depth profiles at seven sites within Lake St. Croix (MCES)
- Task 3: Continue water quality monitoring at river and stream sites (MCES)
- Task 4: Conduct experiments to evaluate the ecological health of Lake St. Croix (USGS)
- Task 5: Conduct data analysis, and increase data integration and accessibility (SMM)
- Task 6: Complete progress reporting and final project reporting (SMM)

<u>Task 1 Findings: Improved Discharge Method</u>

The USGS established an index-velocity gaging station for the St. Croix River at Stillwater, MN (05341550) on September 9, 2011. A complete discharge record was collected by the USGS Minnesota Water Science Center according to USGS protocols and policies. Daily discharge values were published in the USGS Minnesota Water Science Center Annual Data Report (available online at http://wdr.water.usgs.gov/). In addition, daily discharges from USGS gaging stations at the St. Croix River at St. Croix Falls, WI (05340500), the Apple River near Somerset (05341500), and the St. Croix River at Stillwater (05341550) were used to compare measured discharges at Stillwater with discharges computed from previously-used methods (Lafrancois et al. 2009), using simple linear regression analyses. Regression outputs indicated that the previous method of estimating discharge at Stillwater resulted in errors in the timing and magnitude of discharge (Pearson's R^2 = 92.1%). It was determined that a predictor equation that incorporates a one-day time lag into $Q_{\text{St. Croix Falls}}$ data produces the strongest relationship to the measured index velocity discharge and the least amount of unexplained variation (Pearson's R^2 = 98.2%). The provisional regression equation to predict daily discharge for the St. Croix River at Stillwater, MN is as follows:

$$Q_{\text{Stillwater}} = 0.9706*(Q_{\text{St. Croix Falls}}) + 680.6135$$

Additional linear regression analyses will be performed to improve and finalize historical flow estimates at Stillwater, and the results of additional loading analyses will be published in a separate USGS Scientific Investigations Report (SIR).

Task 2 Findings: Stratification of Lake Profiles

Under the direction of MCES, seven sites on Lake St. Croix have been monitored by trained citizen volunteers since 1999. The monitoring plan includes May-October biweekly sampling for nutrients and chlorophyll, temperature, clarity, and resource quality assessments. Starting in August 2008, two volunteers, Jim and Roberta Harper, took on the task of obtaining additional water quality information at the seven Lake St. Croix volunteer monitoring sites. On a monthly basis during the May-October period each year, the Harpers have been collecting samples for nutrient and phytoplankton analysis and measuring depth profiles of temperature, pH, conductivity, and dissolved oxygen. The monthly depth profiles of temperature and dissolved oxygen showed that classic lake stratification occurs in this riverine lake system, where warm oxic surface waters in the epilimnion overlie cool anoxic bottom waters in the hypolimnion, persisting every year between June and September. Variations in this pattern depend on flow dynamics, in both the long-term flow response to climate conditions of that year (e.g., low-flow years or high-flow years) and the short-term flow response to stormflow events. Low-flow years exhibit a shallow epilimnion of highly oxygenated waters, while high-flow years exhibit a deep epilimnion of moderately-oxygenated waters. Large storm events sustained over multiple days appeared to disrupt the stratification pattern, with the high flows driving the bottom of the epilimnion deeper into the lake profile and mixing upward some of the low-oxygen waters from the hypolimnion.

Task 3 Findings: Temporal and Spatial Patterns in Water Quality

This report focused on the biweekly water quality variables that were assigned water quality standards during the Lake St. Croix TMDL process: Secchi depth clarity (SD>1.4 meters), total phosphorus (TP<40 μg/L), and viable chlorophyll-a (VChla<14 μg/L). Exceedances of these standards increased in high-flow years, indicating that the long-term achievement of these standards need to be assessed over multi-year periods that average-out annual flow variability. The mean summer (June-September) values appeared to be dependent on annual flow conditions. In low-flow conditions (e.g., 2009) clarity was high, total phosphorus was low, and chlorophyll-a was high. In high-flow conditions (e.g. 2011) clarity was low, total phosphorus was high, and chlorophyll-a was low. These results suggest that annual algal growth (chlorophyll) benefits from the high clarity of a low-flow year, despite lower nutrient concentrations. However, the mean summer values of clarity, total phosphorus, and chlorophyll were not consistent during moderate-flow years (e.g., 2008, 2010, 2012), indicating more complexity in the relationships between clarity, nutrients, and algal growth.

To assess spatial variability, the mean annual (Oct-Sep) discharge and mean summer (Jun-Sep) values of Secchi depth clarity (SD), total phosphorus (TP) concentration, and viable chlorophyll-a (VChla) concentration were plotted for each year (2008-2012) at each of the seven MCES lake monitoring sites. Secchi depth clarity showed a downstream trend of increasing clarity for all years, perhaps due to particulates settling out of suspension toward the downstream end of the lake. Total phosphorus showed a downstream trend of decreasing concentrations for most years except 2008, perhaps for the same reason. Chlorophyll-a showed a downstream trend of decreasing concentrations through Pool 1(from SC-1 to SC-3), then increased again in Pool 2 (at SC-4), before continuing its downstream trend of decreasing chlorophyll concentrations. For TP and VChla, interannual variability exceeded spatial variability, pointing to flow as a driving mechanism in both phosphorus delivery and algal response.

To assess seasonal variability, we plotted the monthly mean discharge at Stillwater and the monthly mean values of SD, TP, and VChla, for May-October during 2008-2012. Total phosphorus concentrations seemed to exhibit little seasonality, with monthly means clustered around 49 μg/L, the mean for the study period. In contrast, chlorophyll showed clear seasonality, with concentrations increasing through the summer months to peak values near 20 μg/L in August and September. Secchi depth clarity peaked at 1.6 meters in June/July, after the peak discharges during April/May had decreased, but before the low-flow peak algal growth period of August/September.

Task 4 Findings: Drivers of Ecological Functioning

Phytoplankton samples collected by the Harpers were analyzed by Dr. Jeff Janik, a phytoplankton taxonomist, who identified the abundance and biomass of nine algal groups to genus and species. Like chlorophyll concentrations, maximum total abundance observed in any given year depended on that year's flow conditions: the highest total abundances were observed in low-flow years (e.g., 2008 and 2009). Low-flow conditions were conducive to maximum algal growth, while high-flow conditions reduced water residence time in the pools so as to limit algal growth. Among the algal groups, blue-green algae (BGA) dominated the whole lake summer (June-September) total abundance during 2009-2012 (2008 was omitted as a partial summer); BGA accounted for about 75% of the total abundance in a low-flow year (e.g., 2009) and 50% of the total abundance in a high-flow year (e.g., 2011). To assess overall spatial variability, we plotted the summer (June-September) percent abundances at the seven MCES lake monitoring sites during 2009-2012. Although BGA dominated at all sites, the lake-like lower pools showed abundances of haptophyta (flagellates) in low-flow conditions, and abundances of haptophyta, cryptophyta, and chlorophyta (green algae) in high-flow conditions.

To assess the affect of nutrients on algal growth, we conducted bioassays on phytoplankton samples grown in lakewater samples from the four deep-pool sites, collected during five seasons in 2011-2012. The bioassays included four experiments across a gradient of nutrient enrichment: an experimental control of ambient lakewater with no additional nutrients, and three combinations of additional nutrients: nitrogen (N), phosphorus (P), and both (NP). Addition of both nutrients together (NP) significantly increased growth rates over ambient controls in both years at all sites, providing seasonal estimates of maximum, nutrient-sufficient growth rates at each location. Maximum growth rates were consistent across seasons in Pools 1 and 2. In contrast, maximum growth rates in the lower two pools were lowest in the spring and increased through summer and autumn. Comparisons of the N and P experiments revealed whether nitrogen or phosphorus were the limiting nutrient in algal growth during the sampled seasons of 2011 and 2012. All of the pools were P-limited during high-flow conditions in 2011 and in the spring of 2012, but moved towards N-limitation or co-limitation by late summer in August of 2012. Three of the four pools (1, 2, and 4) were N-limited during the summer of 2012. This key finding, that both P-limitation and N-limitation are occurring, points to the need for a greater understanding of the dynamics of both nutrients within this riverine lake.

To determine the spatial variability in the potential for internal phosphorus loading released from lake sediments, we incubated sediment cores in the laboratory, under both oxic and anoxic conditions, that had been collected intact from the four deep-pool sites. Rates of phosphorus release rates were much lower in the sediments from the upper Pools 1 and 2, compared to the higher rates observed in the sediments from the lower Pools 3 and 4. Sediments deposited in the upper pools are likely to be heavier coarse-grained clastics, while sediments deposited in the lower pools are more likely to be fine-grained fractions that favor phosphorus release from those sediments. In contrast, Robertson and Lenz (2002) assumed lower rates of internal loading in the lower pools of Lake St. Croix.

To estimate the relative influence of internal loading on total phosphorus loading to Lake St. Croix, we used dissolved oxygen profiles to estimate the extent and duration of anoxia at the sediment-water interface across Lake St. Croix during summer (June-September) 2012. In order to achieve the most accurate yet conservative estimate, we merged the upper pool release rates of Robertson and Lenz (2002) along with the lower pool release rates measured summer 2012. The merged internal load estimate of 9117 kg is greater than the 7927 kg reported by Robertson and Lenz (2002). In low-flow conditions, this internal load would account for 21.6% of the total phosphorus summer load to Lake St. Croix, while in high-flow conditions it would account for 5.9% of total loads.

Several aspects of this project are part of ongoing efforts, which will build on the accomplishments of this project, and have been extended for more in-depth work with additional funding:

1) The USGS will expand the historical assessment of St. Croix River flow at Stillwater, MN and Prescott, WI, into a citable reference, a new Scientific Investigations Report (Ziegeweid and Magdalene, in press) that will update the estimates of historical flows and loads at these two sites.
2) MCES will continue to support the Lake St. Croix Volunteer Monitoring Program, and the USGS will continue to provide funding for the volunteers to measure monthly lake profiles.
3) The USGS will upgrade the static one-dimensional BATHTUB model of Lake St. Croix (Robertson and Lenz 2002) to a dynamic two-dimensional CE-Qual-W2 model of Lake St. Croix (Kiesling et al., in press), which will enhance the understanding of the ecological functioning of Lake St. Croix and enable scenario testing using predictive relationships.
4) The St. Croix Basin Water Resources Planning Team has identified four projects for implementation, via the Stillwater Bridge Mitigation funding: 1) the USGS will conduct three years of operation and maintenance of the St. Croix River at Stillwater, MN gage, 2) three years of load monitoring at three key Wisconsin tributaries to improve the nutrient mass balance for Lake St. Croix, and 3) continuous monitoring of key water quality variables at deep pool sites to improve understanding of ecological functioning in Lake St. Croix, and 4) SMM will prepare a State of the Lake report (Magdalene et al., in press), to evaluate 1999-2013 progress toward achieving the phosphorus reduction goal by 2020.
5) The St. Croix River Association is pursuing the development of an interactive mapping application on the web, which will bring public accessibility of St. Croix data closer to reality.

In each case, the current work proved valuable enough (and complex enough) that decisions were made to continue the work in a more in-depth manner than originally conceived. Therefore, better information will be the ultimate result.

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ABBREVIATIONS

ADCP	acoustic Doppler current profiler
ADVM	acoustic Doppler velocity meter
C	degrees Celsius
cfs	cubic feet per second (ft^3/s) rate of stream discharge
Chl-a	chlorophyll-a
DO	dissolved oxygen
DOC	dissolved organic carbon
F	degrees Fahrenheit
ft	feet, English unit of length
GMT	Greenwich mean time
in	inches
km	kilometers
m	meters
mg/L	milligrams per liter
MCES	Metropolitan Council Environmental Services
MPCA	Minnesota Pollution Control Agency
NHx-N	ammonium + ammonia-nitrogen
NOx-N	nitrogen oxides, nitrate+nitrite-nitrogen
NPS	National Park Service
NWS	National Weather Service
RM or rm	river mile on the St. Croix above the confluence with the Mississippi River
s	second, SI unit of time
SCWRS	St. Croix Watershed Research Station
SD	secchi depth clarity
SMM	Science Museum of Minnesota
TCMA	Twin Cities metropolitan area
TMDL	total maximum daily load
TP	total phosphorus
TSS	total suspended solids
USGS	United States Geological Survey
USACE	United States Army Corps of Engineers
VChla	viable chlorophyll-a
WDNR	Wisconsin Department of Natural Resources
WWTF	wastewater treatment facility

ACKNOWLEDGEMENTS

First and foremost our thanks to Minnesota State Senator Katie Sieben, who sponsored legislation specifically to fund comprehensive water monitoring in Lake St. Croix, which made this project possible. Thanks to Deb Ryun and the St. Croix River Association, a long-time organization that only gets stronger over the years, sharing with others the beauty of the Namekagon and St. Croix Rivers.

Special thanks to the citizen scientists who regularly volunteer their time and energy to monitor the health of Lake St. Croix via Metropolitan Council Environmental Services (MCES) Citizen-Assisted Monitoring Program (CAMP): Harry and Cecilia Martin, Rich and Sheryl Lindholm, Rick Meierotto, Mayme Johnson at the Carpenter Nature Center, and especially Jim and Roberta Harper, retired 3M engineer and chemist driven to excellence by intellectual curiosity, who have dodged barges and weathered the storms.

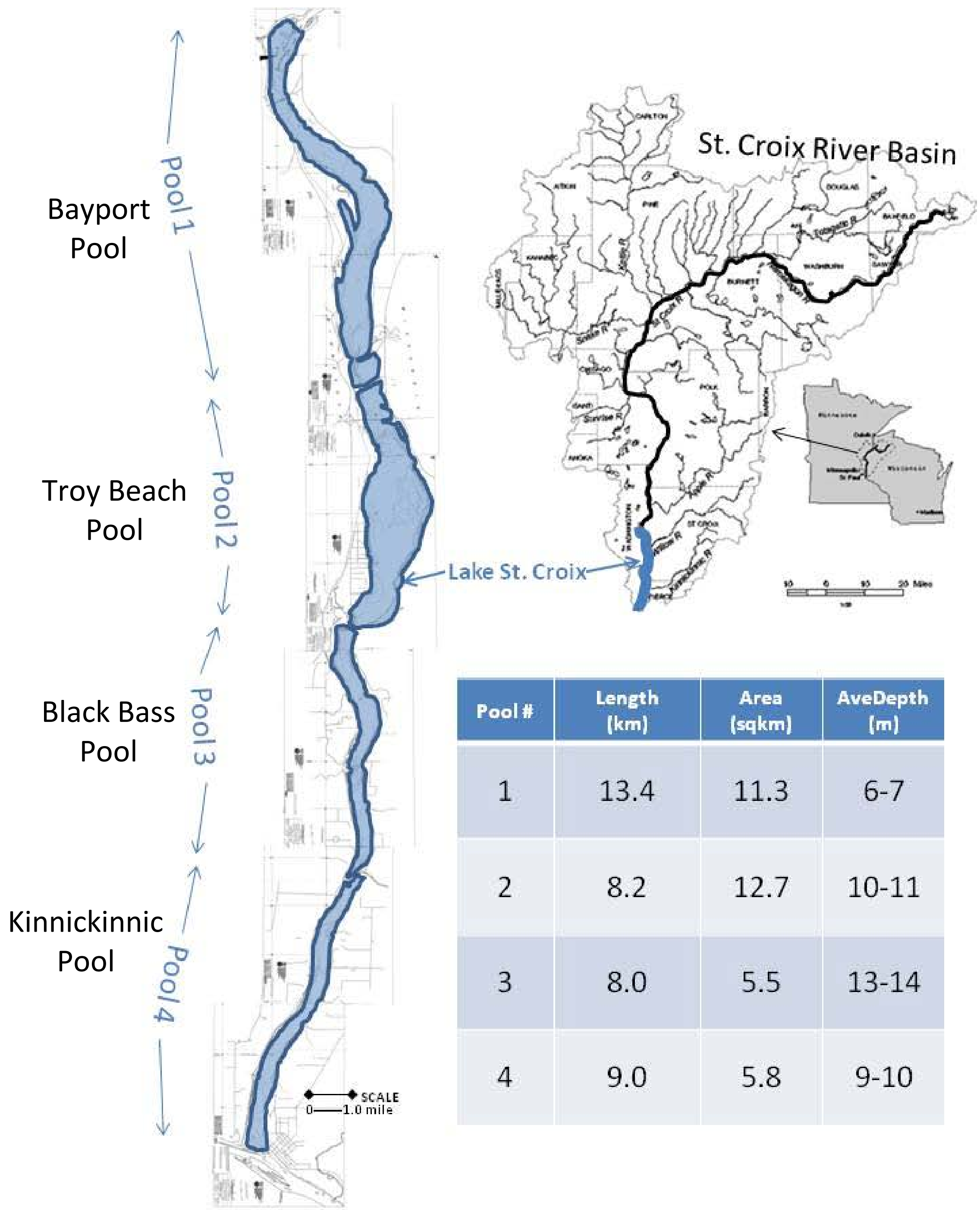
Valuable technical expertise and assistance was rendered by Dr. Jeff Janik for phytoplankton taxonomy, Mimi Wallace for phytoplankton bioassays, Sarah Elliot for water quality processing and field assistance, and Molly Trombley for field assistance. Our thanks to all.

INTRODUCTION

Lake St. Croix is a naturally-impounded riverine lake encompassing the lower 40 kilometers (25 miles) of the St. Croix River before its confluence with the Mississippi River at Prescott, WI. The lake is comprised of four successive pools (Figure 1) sitting within the widened and deepened valley of the lower St. Croix River, constricted at its mouth by a peninsula of glacial deposits re-worked by the Mississippi River. Lake St. Croix is the receiving water body for direct runoff, three minor trout streams (Browns, Silver, and Valley creeks), two major tributaries (Willow and Kinnickinnic rivers), and the St. Croix River upstream of Stillwater, MN. In addition, Lake St. Croix is the downstream integrator of a 20,100 square kilometer (7,760 square mile) basin that encompasses portions of the States of Minnesota and Wisconsin. The St. Croix River and its northern tributary, the Namekagon River, have been protected as the St. Croix National Scenic Riverway since 1968. Early federal and state protection efforts focused on preventing degradation of the mainstem river; non-degradation designations restricted waste discharges to the river (Holmberg et al. 1997, Payne et al. 2002). Over time, state and local protection efforts have expanded up into the basin, especially with the recognition of the importance of Lake St. Croix as a water quality integrator of land and water management practices. Today, we acknowledge that the health of Lake St. Croix is a measure of the health of the entire St. Croix River Basin.

Increasingly, Lake St. Croix is subject to nuisance algal blooms. These changes are thought to be due to expanding urban and agricultural nutrient inputs. Despite reductions in point-source phosphorus inputs (Edlund et al. 2009), phosphorus and chlorophyll concentrations still exceed water-quality standards (MPCA and WDNR 2012). Therefore, in 2008, Lake St. Croix was placed on Minnesota's 303(d) list of impaired waters for eutrophication due to excess phosphorus. Based on this impairment, a TMDL plan was prepared by the MPCA and WDNR (reference), with approval by the Environmental Protection Agency. The TMDL plan set a phosphorus reduction goal based on research into historical ecological conditions in Lake St. Croix (Davis 2004). However, previous phosphorus load calculations used St. Croix River discharge values that were estimated at St. Croix Falls, Wisconsin, which is well upstream of the inlet to Lake St. Croix. In addition, previous assessments of phosphorus impacts were made based on the assumption of primarily oxic conditions in Lake St. Croix (Robertson and Lenz 2002). However, recent depth profiles demonstrate the prevalence of anoxic conditions in the hypolimnion of all four Lake St. Croix pools.

The goal of this project was to improve the understanding of how nutrient loading affects the ecological health of Lake St. Croix. Installation of a streamflow gaging station on the St. Croix River at Stillwater provided more accurate mainstem flow data. Sediment-core incubation experiments improved the accuracy of internal phosphorus loading within the Lake St. Croix pools. Increased vertical-depth profiling and water-quality sampling provided more information about lake conditions and within-lake processes controlling nutrient concentrations. Finally, phytoplankton bioassays and zooplankton grazing studies identified patterns of nutrient limitation and biological control mechanisms of nutrients in Lake St. Croix.



Pool #	Length (km)	Area (sqkm)	AveDepth (m)
1	13.4	11.3	6-7
2	8.2	12.7	10-11
3	8.0	5.5	13-14
4	9.0	5.8	9-10

Figure 1. Location of Lake St. Croix and the basic morphometry of its four pools.

LOCATION AND SETTING

Landscape in the region of the study area was carved by huge volumes of glacial runoff, causing the incision of bedrock and creating steeper valley slopes than the gentle grasslands to the west. The valleys of large rivers, like the St. Croix and the Mississippi, were deepened and widened by glacial rivers, and side tributaries delivered copious amounts of reworked glacial deposits as deltas prograding into the large rivers (Montz et al. 1989, Payne et al. 2002). These processes resulted in a unique landscape feature within the region: natural riverine lakes impounded by tributary deltas at their downstream ends (e.g., Lake Pepin and Lake St. Croix). Similarly, Lake St. Croix is divided into four sub-basins by the deltas of side-valley tributaries along its course: the Willow River delta forms the downstream boundary of Pool 1 (Bayport Pool), the Valley Creek delta forms the downstream boundary of Pool 2 (Troy Beach Pool), the Kinnickinnic River delta forms the downstream boundary of Pool 3 (Black Bass Pool), and the sediment deposits at the confluence of the St. Croix and Mississippi Rivers forms Pool 4 (Kinnickinnic Pool).

The St. Croix River basin spans approximately 20,100 square kilometers (7,760 square miles) and is dominated by forests, pastures, and croplands (Heiskary and Vavricka 1993, Larson et al. 2002). The 2006 National Land Cover Dataset (NLCD) identified 66% of the basin area as forest and wetland cover, and 26% as agricultural cover (crop and pasture). Urban lands comprise less than 5% of the basin (Minnesota Forest Resources Council, in press). Prior to European settlement, the St. Croix basin was dominated by forests, peatlands, and prairie grasslands (Niemela and Feist 2000, Payne et al. 2002). The basin was heavily logged from the mid 1800s to the early 1900s, when agriculture began to replace logging as the primary land use. The combination of logging and agriculture reduced wetlands and altered the landscape of the basin. Land use practices in the St. Croix River basin shifted around the middle of the twentieth century as larger urban areas developed and agricultural practices became modernized. These land use changes, resulting in increased stormwater runoff from urban and agricultural areas, as well as the need for wastewater treatment. The resulting affects on water quality led to development of environmental protections starting in the late 1960s and early 1970s. However, because the St Croix River is in close proximity to the Minneapolis/St. Paul, MN metropolitan area, the basin continues to receive increased pressure for developmental and recreational use. Recreational use of the river doubled from 1973 to 1995, with nearly one million visitors annually as of 1995 (National Park Service 1995).

The St. Croix River basin experiences a typical Upper Midwest climate. The 30-year (1981-2010) average air temperature is 44.2 degrees Fahrenheit and the average annual precipitation is 29.6 inches at St. Croix Falls, MN (Figure 2).

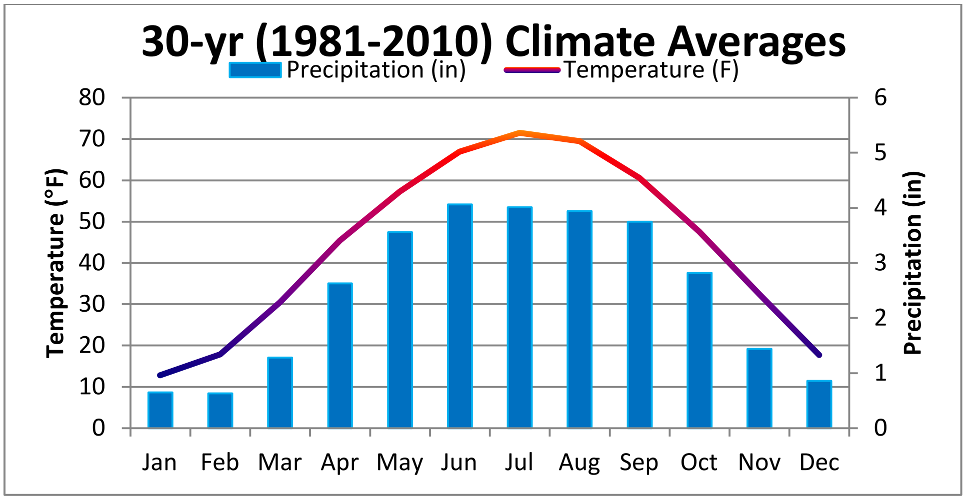


Figure 2. Thirty-year (1980-2010) average monthly air temperature (°F) and precipitation (in) at St. Croix Falls, WI. Data from NOAA.

Lake Morphometry and Flow Routing

Lake St. Croix has a surface area of 35 square kilometers (13.5 (sq. mi.), a mean depth of 9.7 meters (31.8 feet), and a water residence time of 15-50 days (MPCA and WDNR 2012). However, the four pools have a range of characteristics that influence their individual lake ecology (Table 1).

Pool 1(Bayport Pool), at the northern end of Lake St. Croix, is the longest and shallowest pool (Table 1) of Lake St. Croix. Pool 1 receives the combined runoff from the entire St. Croix River basin above Stillwater, MN. This mainstem St. Croix River input is the single largest source of flow to Lake St. Croix. As the river enters Lake St. Croix, the sudden decrease in water velocity causes the sediment load from the river to settle out, resulting in shallow areas above the Stillwater bridge. Also, near the head of the pool are two small tributary inputs that flow through suburban Washington County MN, Browns Creek and Silver Creek, coldwater trout streams. Outfalls from the MCES St. Croix Valley Wastewater Treatment Plant and from the Xcel Energy Allen S. King Generating Station are the largest point source inputs to Lake St. Croix; both enter Pool 1 from Washington County. Willow River, the large tributary input from St. Croix County, WI enters Pool 1 at its downstream end. Pool 1 also receives nonpoint source runoff along its length, dominated by urban runoff from the cities of Stillwater and Bayport, MN and Hudson, WI. Bayport Pool 1 terminates at the narrows created by the re-working of the sediments of the Willow River delta.

Pool 2 (Troy Beach Pool) begins below the Willow River narrows, but above the I-94 Bridge. Pool 2 is the widest pool with the largest surface area, making it the most lake-like of the Lake

St. Croix pools. The Hudson Wastewater Treatment Facility outfall, located near the I-94 Bridge, is the only permitted point source input to Pool 2. Nonpoint source inputs include municipal runoff from Hudson, WI, and from Lakeland and Lake St. Croix Beach, MN. Troy Beach Pool 2 terminates at the large deltaic complex of Valley Creek, entering from Washington County, MN and re-worked on the opposite (east) bank into Catfish Bar, which extends into the lake from the Wisconsin shoreline, further constricting Lake St. Croix at this location. There is a narrow navigation channel connecting Pool 2 and Pool 3, most visible in bathymetric maps (Figure 3).

Though short and narrow, Pool 3 (Black Bass Pool) is the deepest pool of Lake St. Croix. Immediately downstream of the Valley Creek deltaic deposits, the pool drops off to the lake's greatest depth of nearly 30 meters. Pool 3 receives nonpoint source runoff from Afton, MN. A small trout stream, Trout Brook, enters Pool 3 in the area of Afton State Park, MN. Black Bass Pool 3 terminates at the large delta formed by the tributary input of the Kinnickinnic River in Pierce County, WI.

Pool 4 (Kinnickinnic Pool) begins at the Kinnickinnic narrows created by the Kinnickinnic River delta, and is shallow at its head due to re-working of the deltaic deposits. The landscape around Pool 4 is less developed, although the pool receives a portion of the nonpoint source runoff from Prescott, WI. Kinnickinnic Pool 4 terminates at a large spit or peninsula across the mouth of the St. Croix River formed by the re-working of sediments deposited by the Mississippi River at its confluence with the St. Croix River.

Table 1. Characteristics of Lake St. Croix pools.

Pool #	[1]Upstream end (rm)	[1]Downstream end (rm)	Length (mi)	Length (km)	Width Ave (km)	[2]Area (km^2)	Depth [3]Ave (m)
1	25.0	16.7	8.3	13.4	0.8	11.3	**6-7**
2	16.7	11.6	5.1	8.2	1.5	12.7	10-11
3	11.6	6.6	5.0	8.0	0.7	5.5	13-14
4	6.1	0.5	5.6	9.0	0.6	5.8	9-10

[1]From USACE (2011) navigational channel maps, river mile (rm) refers to mile markers along thalweg
[2]From Robertson and Lenz (2002)
[3]From Robertson and Lenz (2002), range based on dry-wet years

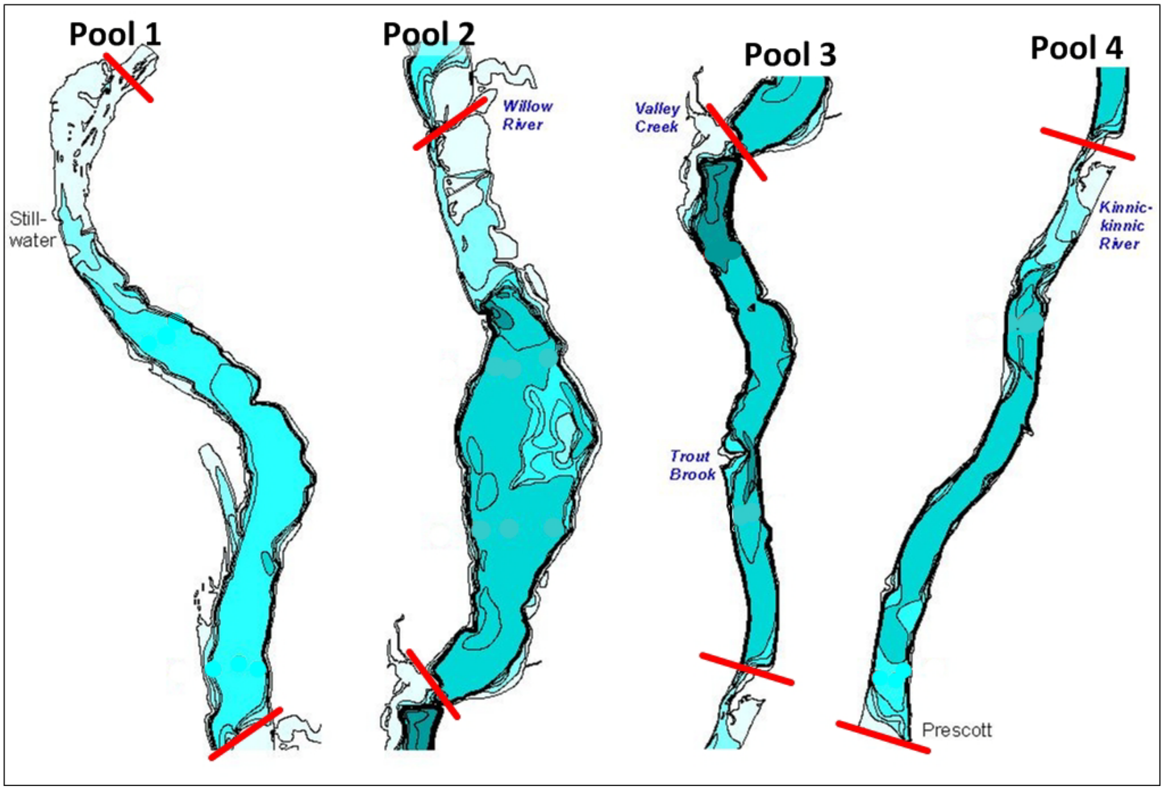


Figure 3. Bathymetric map of the Lake St. Croix pools. Red segments indicate pool boundaries. (modified from Triplett, personal communication).

BACKGROUND

Previous Work

Numerous efforts have been made to study and manage Lake St. Croix. Metropolitan Council Environmental Services (MCES) has been collecting routine water quality data in Lake St. Croix since 1976 to help determine compliance with state water quality standards and criteria, assess the performance and effectiveness of the MCES St. Croix Valley Wastewater Treatment Plant, obtain information on the sources and water quality impacts of nonpoint source pollutants, and document long-term trends and changes in water quality. Despite reductions in point source phosphorus inputs, phosphorus and chlorophyll concentrations in Lake St. Croix still exceed water quality standards.

In 1994, several regional, state and federal agencies collaborated to form the St. Croix Basin Water Resources Planning Team (hereafter, Basin Team) with the goal of investigating water resource issues in the basin. Lenz et al. (2003) monitored tributary and mainstem nutrient loads across the basin in 1999, and Robertson and Lenz (2002) modeled sediment and nutrient loading to the pools of Lake St. Croix. Triplett et al. (2003) used lake sediment coring to determine historical loading rates of sediment and phosphorus to Lake St. Croix. A phosphorus loading rate of 360 tons per year, derived from conditions that were last recorded in lake cores during the 1940s, was expected to restore Lake St. Croix to benthic algal dominance.

Based on those results, the Basin Team recommended a new phosphorus goal for Lake St. Croix (Davis, 2004), which was endorsed via an interstate agreement in 2006. In 2008, the Minnesota Pollution Control Agency (MPCA) added Lake St. Croix to the state’s 303(d) list of impaired waters for eutrophication due to elevated levels of phosphorus and chlorophyll. Finally, the MPCA and Wisconsin Department of Natural Resources (WDNR) developed an interstate Total Maximum Daily Load (TMDL) plan to address the eutrophication impairment in Lake St. Croix (MPCA and WDNR 2012), including ongoing components of civic engagement, implementation, and load monitoring.

Previous Limitations

Early efforts to characterize the ecological health of Lake St. Croix were limited by: 1) the downstream influence of the Mississippi River, which has prevented the use of traditional flow measurement methods, and 2) an assumption that river processes dominate lake processes, that Lake St. Croix primarily functions as a river, not as a lake. First, the drainage area of the Mississippi River as it passes the mouth of the St. Croix River is over four times larger than the St. Croix River Basin itself, and rainfall distribution between the St. Croix and Mississippi River basins can vary depending on weather patterns. Occasionally, the Mississippi River delivers such a large volume of water past the mouth of the St. Croix River that it can cause a hydrologicdamming backwater effect in Lake St. Croix, which interferes with traditional discharge measurement methods. Streamflow gaging of the St. Croix River above the St. Croix Falls dam, beyond the reach of these backwater effects, was started by the USGS in 1902. For over a century, the flows through Lake St. Croix were estimated from the St. Croix Falls gage, 83 kilometers (52 miles) upstream from its outlet and 42 kilometers (27 miles) upstream from its inlet, and from the dam on the Apple River, one of the larger tributaries entering from Wisconsin. Important sediment and phosphorus mass loading work in the 1990s had to rely on these older methods for flow estimates. Only in very recent years has new flow measurement technology become available that can account for the backwater effects. In 2007, the USGS was able to install an Acoustic Doppler Velocity Meter (ADVM) flow gage at the mouth of the St. Croix River, measuring actual outflow from the St. Croix Basin for the first time. However, inflows to Lake St. Croix at Stillwater, MN continued to be estimated from the St. Croix Falls and Apple River gages. A new ADVM gaging station on the St. Croix River at Stillwater, Minnesota is ranked as a high priority by the Basin Team (VanderMuelen et al. 2010) because gaged streamflows would provide the data necessary to improve calculations of phosphorus loads at the mainstem inflow of Lake St. Croix and to track progress toward the goal of a 20% reduction in phosphorus loading to Lake St. Croix. Installation of a flow gage at Stillwater will enable correlations between USGS flow data at Stillwater, MN and St. Croix Falls, WI, improving the accuracy of historical and future load calculations. While the MCES staff had begun taking individual flow measurements for the development of a stage-discharge rating curve, it would be a long time before sufficient data was gathered to complete the rating curve.

The second barrier to the accurate characterization of the health of Lake St. Croix was an assumption that riverine processes were dominating lacustrine processes in Lake St. Croix, in particular that river mixing would prevent lake stratification in the elongated flow-through waterbody. Lake stratification was expected to occur rarely, and only in low flow summer conditions. However, the first documented occurrence of stratification and anoxia in deep pools of Lake St. Croix occurred in August 2006, near the downstream end of Lake St. Croix. Anoxic conditions likely affect the water chemistry of Lake St. Croix. The mobility of phosphorus under anoxia was especially concerning, given the problems with eutrophication that Lake St. Croix was already experiencing. Previously, internal loading was assumed to be minimal under presumed oxic conditions (Robertson and Lenz 2002), and the presence of anoxic conditions may result in substantial internal loading from the lakebed into overlying waters. In 2008, MCES volunteers initiated a record of monthly temperature and dissolved oxygen (DO) depth profiles at seven volunteer monitoring sites located longitudinally along the length of Lake St. Croix. The prevalence of stratification and anoxic conditions in the deep pools was surprising to water resource managers in Minnesota and Wisconsin. Stratification and anoxia were often found to last from June through September, and water resource managers began to think of Lake St. Croix as a warm river sliding over four isolated cold pools. Collected data invalidated the assumption that water in Lake St. Croix remained mixed and oxygenated, and therefore the mass balance of phosphorus and the effect of nutrient cycling on ecological health in Lake St. Croix, needed to be re-assessed.

Problem Statement

Using the long-term record from MCES, Lafrancois et al. (2009) confirmed by seasonal Kendall analysis that concentrations of the stressor variable total phosphorus (TP) inflow to Lake St. Croix have declined by an average of 0.2 μg/L per year during the 1976-2004 period, but the response variable chlorophyll-a has lagged behind these improvements (Figure 4). Although turbidity, total suspended solids (TSS) and TP have all declined in Lake St. Croix, chlorophyll-a loads have not changed, and chlorophyll-a concentrations have significantly increased in the outlet of Lake St. Croix at Prescott, WI (Lafrancois et al. 2009).

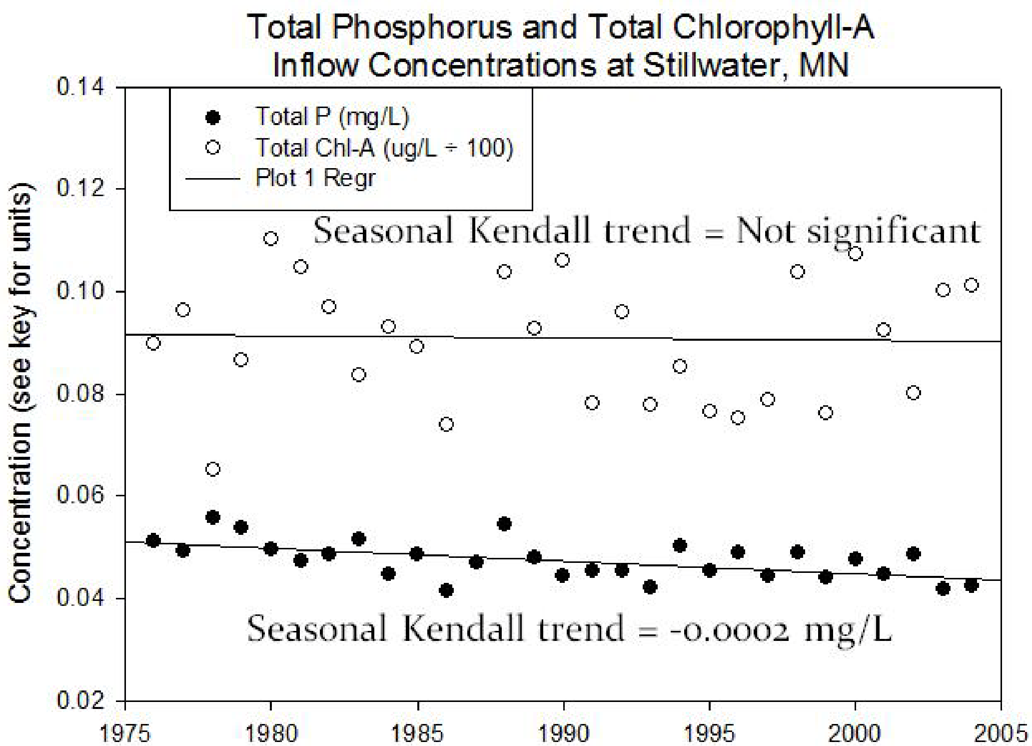


Figure 4. Seasonal Kendall trends for total phosphorus and total chlorophyll-a concentrations in the St. Croix River at Stillwater, MN, 1976-2004. Modified from Lafrancois et al. (2009).

Recent studies have begun to examine long-term water quality trends in Lake St. Croix (Robertson and Lenz 2002, Lafrancois et al. 2009). Changes observed in long-term nutrient data likely are the result of improved point-source management (Lafrancois et al. 2009). Although phosphorus concentrations remain well above pre-1950s levels (Triplett et al. 2009), strict waste water treatment facility (WWTF) regulations and a ban on phosphorus-containing laundry detergents have helped to reduce point-source phosphorus loads (Kloiber 2004, Lafrancois et al. 2009). Similarly, nitrification processes used by WWTPs to reduce NHx-N contributions have increased point-source NOx-N contributions above and within Lake St. Croix. From 1976 to 2004, total nitrogen (TN) and ammonia/ammonium (NHx-N) concentrations have decreased significantly at the inlet to Lake St. Croix, but total nitrogen concentrations have increased significantly at the outlet. In addition, nitrate+nitrite-nitrogen (NOx-N) concentrations have increased at both the inlet and the outlet. Previous studies indicate that the highest NOx-N concentrations occur at base flow in the two sub-watersheds with the most agriculture (Lenz et al. 2003, Lafrancois et al. 2009). In addition, decreasing total nitrogen concentrations at the inlet and increasing total nitrogen concentrations at the outlet indicate that non-point sources below the inlet of Lake St. Croix are contributing substantially to nutrient inputs.

Although extensive water-quality data have been collected, additional information is needed to improve mass balance estimates and understand relationships between available nutrients and biological responses in Lake St. Croix. Current mainstem inflows are estimated mathematically using discharges measured upstream at St. Croix Falls, Minnesota and adding estimated discharges from the Apple River between St. Croix Falls and Stillwater. In addition, estimates of internal loading have been based on the recently-challenged assumption of dominantly oxic conditions in Lake St. Croix (Robertson and Lenz 2002). The limiting nutrient of Lake St. Croix also has been debated. Robertson and Lenz (2002) used nitrogen to phosphorus (N:P) ratios to support the claim that lake St. Croix is phosphorus-limited. However, other studies argued that despite recent declines, phosphorus concentration remain high and are unlikely to limit algal growth (Larson et al. 2002, Lafrancois et al. 2009).

Project Goals, Tasks, and Objectives

This project had two overall goals: 1) to improve existing phosphorus mass balance estimates for Lake St. Croix, and 2) to develop an improved understanding of the relationship between water quality and ecological health of Lake St. Croix.

This study was designed to provide the Basin Team with the information necessary to better understand the mechanisms of nutrient loading in Lake St. Croix. Monthly depth profile data provided further information on the spatial and temporal extent of lake-bottom hypoxia. Short-term sediment-core incubations provided information about nutrient release rates at the sediment-water interface. Water-quality and phytoplankton bioassay data helped document patterns in both nutrient limitation and in phytoplankton responses to nutrient availability. In addition, zooplankton-grazing experiments examine both the seasonal intensity of zooplankton-grazing mortality on phytoplankton and rates of algal biomass loss.

This study also provides the information necessary to develop a predictive, bio-physical model, which will aid the Basin Team in evaluating functional responses to TMDL implementations.

The evaluation of functional responses will demonstrate whether the targeted 20% reduction in phosphorus loading to Lake St. Croix can reduce nuisance algal blooms and restore Lake St. Croix to benthic algal dominance. This information will allow the Basin Team to adjust management plans accordingly.

In order to accomplish the objectives in a timely manner, the proposed study was divided into a series of tasks. The tasks were divided among the collaborating agencies, and each task had specific objectives relating to the overall goals of the study. The tasks (and lead agency responsible for the task) are as follows:

Task 1 (USGS): Install and operate a flow gage on the St. Croix River at Stillwater, MN

The objectives of this task were to:

1) install USGS flow gage, similar to the index velocity gage installed at Prescott, WI,
2) collect at least 10 discharge measurements in order to develop an index velocity rating at Stillwater, and
3) operate and maintain flow gage for two years.

Without a gaging station at Stillwater, the mainstem portion of inflow loading to Lake St. Croix is unknown and must be estimated using scaled values from the St. Croix Falls flow record (e.g., Lafrancois et al. 2009). Gage installation will provide accurate discharge data at the inlet of Lake St. Croix. A full-time stream gage will be installed on the St. Croix River at Stillwater, Minnesota as soon as weather conditions allow, providing an immediate flow record. The gage will be installed by experienced USGS stream-gaging personnel, and the gage will be operated according to standard USGS procedures (Rantz et al. 1982). A minimum of 10 discharge measurements is required to develop an index-velocity rating curve, based on a stage-area relationship for a surveyed cross-section of the St. Croix River at Stillwater, MN (Rickman 2011).

Task 2 (USGS): Monthly Depth Profiles at Seven Sites within Lake St. Croix

The objectives of this task were to:

1) measure depth profiles of temperature, conductivity, and dissolved oxygen, and
2) collect depth-integrated samples of phytoplankton.

Two MCES volunteers were hired as USGS contracted observers to monitor Lake St. Croix pools at seven sites, on a monthly basis during the open-water season. Temperature, conductivity, and dissolved-oxygen will be measured with a YSI water-quality probe. The monthly lake profile data gathered by contracted observers will give us more information on the spatial and temporal extent of lake-bottom anoxia. Depth-integrated phytoplankton surface samples will be collected from the top two meters at the same time as the vertical depth profiles. These observers have conducted this work on a volunteer basis since 2008, so the observers are well-trained in sample collection and safety protocols. This proposal includes laboratory analysis of the phytoplankton samples collected during the previous two field seasons, thereby expanding the project outcome in a cost-effective manner.

Task 3 (MCES): Lake St. Croix Water Quality Monitoring

The objectives of this task were to:

1) oversee work done by contracted observers,

2) analyze water quality samples collected from Lake St. Croix pools,

3) collect and analyze water quality samples from the St. Croix River at Stillwater and Prescott,

4) analyze water quality samples collected from three Washington County tributaries, and

5) conduct loading analysis, using FLUX, for the two river and three tributary sites.

As part of its responsibilities, the MCES conducts extensive water quality monitoring of streams and rivers in Washington County, MN. The data derived from these efforts will be a valuable part of the overall project goals, improved loading estimates and improved understanding of ecological health.

Task 4 (USGS): Ecological Health of Lake St. Croix

The objectives of this task were to:

1) assess the phytoplankton nutrient-dependent growth rates to determine baseline response to an increased concentration of the limiting nutrient,

2) assess the seasonal intensity of zooplankton grazing mortality on phytoplankton,

3) assess the use of particulate nutrient ratios (i.e., carbon : nitrogen : phosphorus ratios) to document the pattern of nutrient limitation,

4) estimate algal biomass loss rates based on zooplankton grazing, and

5) estimate the potential for internal nutrient loading using short-term sediment core incubations.

The ecological health of Lake St. Croix will be evaluated by developing a predictive relationship between limiting nutrient supply and planktonic chlorophyll. The predictive relationship will be based on the results of phytoplankton growth bioassays, zooplankton-grazing experiments, and short-term sediment-core incubations, all of which will be conducted by the USGS. Patterns of nutrient limitation will be documented by using phytoplankton bioassays to measure particulate nutrient ratios (i.e., C:N:P ratios). In addition, phytoplankton bioassays will be used to assess phytoplankton nutrient-dependent growth rates. Zooplankton-grazing experiments will assess both the seasonal intensity of zooplankton-grazing mortality on phytoplankton and rates of algal biomass loss. Sediment-core incubations estimate the potential for internal nutrient loading.

Task 5 (SMM): Data Analysis, Integration, and Accessibility

The objectives of this task were to:

1) correlate new USGS flow data with USACE stage data, back-calculate historical flows from historical stage data, and assess accuracy of previous flow estimates scaled from St. Croix Falls record,

2) update historical nutrient loading at Stillwater,

3) update phosphorus mass balance of Lake St. Croix, using updated mainstem inflows and updated internal loading (no change to tributary inflows),

3) develop new framework for reporting annual Lake St. Croix assessments, enabling more timely progress reports to TMDL stakeholders,

4) initiate coordinated inter-agency water monitoring database for Lake St. Croix, likely within MSAccess, and

5) improve public access to Lake St. Croix monitoring data, possibly through a map interface on a basin website.

Data analyses will be performed by the agency responsible for each task. The USGS will be responsible for working the flow records for the new and currently operating gages during the study period (Rantz et al. 1982). In addition, the USGS will be responsible for analyzing the data collected during phytoplankton bioassays, zooplankton-grazing experiments, and sediment-core incubations. The MCES will analyze collected vertical-depth profile and water-quality data and conduct load analyses. The remaining analyses will be performed by the St. Croix Watershed Research Station (SCWRS).

Task 6 (SMM): Progress Reporting and Final Project Reporting

The objectives of this task were to:

1) prepare progress reports on a quarterly basis,

2) prepare a final project report that integrates the flow, water quality, and biotic studies into an assessment of the ecological health of Lake St. Croix during the open water season,

3) present project results at the 2012 St. Croix Research Rendezvous, and

4) submit a manuscript for publication in an environmental journal.

METHODS

Pool Morphometry

Pool lengths were determined from the USACE (2011) mapping of the St. Croix River that identifies the central channel (thalweg) of the St. Croix. The length of the river upstream from its confluence with the Mississippi River was marked on the USACE maps according to the length of the St. Croix thalweg from its intersection with the thalweg of the Mississippi River. That intersection occurs at rivermile 811.3 of the Mississippi River (above the confluence with the Ohio River) such that the US10 bridge at Prescott, WI crosses at rivermile 0.3 of the St. Croix River. Pool length depends on the delineation of upstream and downstream boundaries. The upstream terminus is defined by a sharp widening of the river channel that often has shallow depth due to infilling from decreased stream power at the widened point (e.g., the upstream end of Pool 1). The downstream terminus of a pool is defined by a sharply narrowed channel of flow, though not necessarily narrowed valley walls (e.g.., the downstream end of Pool 2).

River Discharge (USGS)

Historical Estimation Methods

In the past, USGS discharge gaging efforts were limited to free-flowing systems where water level is the only variable needed to accurately predict discharge. Low stream gradient within Lake St. Croix, combined with high flow levels in the larger Mississippi River at the mouth of the St. Croix, can create backwater conditions and limit outflow from the St. Croix River, confounding the linear relationship between water level and discharge in the lower St. Croix. Older technologies were not able to accurately predict discharge in backwater conditions. Therefore, flows through Lake St. Croix were estimated from discharges measured at upstream gages (Kloiber 2004, Triplett et al. 2009, Lafrancois et al. 2009). St. Croix River flows at Stillwater, MN ($Q_{Stillwater}$) were estimated from the gage on the St. Croix River at St. Croix Falls, WI ($Q_{St.\ Croix\ Falls}$, USGS#05340500, measured since 1892) and the gage on the Apple River at Somerset, WI (Q_{Apple}, USGS#05341500, measured since 1914) as follows:

$$Q_{Stillwater} = Q_{St.\ Croix\ Falls} + Q_{Apple} \qquad [1]$$

on days when both stations were measured and recorded. The longterm flow ratio of the Apple gage relative to the St. Croix Falls gage was 0.108 or 11%, such that during gaps in the Apple River record, flows at Stillwater were estimated by:

$$Q_{Stillwater} = 1.11^{*}\ Q_{St.\ Croix\ Falls} \qquad [2]$$

Outflows from Lake St. Croix at Prescott, WI were estimated from the above estimated flows at Stillwater plus measured tributary flows to Lake St. Croix, particularly two Wisconsin rivers, the Willow (Q_{Willow}, USGS#05341752) and Kinnickinnic ($Q_{Kinnickinnic}$, USGS#05342000) Rivers, as follows:

$$Q_{Prescott} = Q_{Stillwater} + Q_{Willow} + Q_{Kinninckinnic} \quad [3]$$

on days when both stations were measured and recorded. The shortened period of record for these two stations have flow ratios with St. Croix Falls of 0.037 and 0.032, respectively (Lafrancois et al. 2009), such that during gaps in the Willow and Kinnickinnic River records, outflows at Prescott were estimated by:

$$Q_{Prescott} = 1.18 * Q_{St.\ Croix\ Falls} \quad [4]$$

Though being the best estimates at the time, errors due to unmeasured flows were significant to an unknown degree. Unmeasured portions of Lake St. Croix inflow at Stillwater included: 1) Trout Brook in Osceola, WI, and 2) direct runoff from Minnesota into the St. Croix River between Taylor Falls and Stillwater, including Lawrence Creek. Unmeasured portions of Lake St. Croix inflows below Stillwater included: 1) tributary flows to Lake St. Croix from Minnesota trout streams, Browns Creek, Silver Creek, Valley Creek, and Trout Brook, and 2) groundwater recharge to Lake St. Croix.

Conversion of Stage Gage to ADVM Index Velocity Gage

In order to more accurately determine the flow from the St. Croix River into Lake St. Croix, an index velocity continuous discharge site was established for the St. Croix River at Stillwater, MN (USGS #05341550). An index velocity station is ideally suited for this location, because backwater effects from the confluence of the St. Croix and Mississippi Rivers prohibit the use of a typical stage-discharge relationship to accurately predict the discharge at Stillwater (Levesque and Oberg 2012). Instead of measuring stage and calculating discharge from a stage-discharge rating relationship, an index velocity station measures stage and stream velocity using an acoustic Doppler velocity meter (ADVM). Continuous discharges are calculated from collected water level and velocity data using a stage-area rating that estimates river cross-sectional area and an index velocity-mean velocity rating that predicts average river velocity from ADVM readings. The steps are as follows:

1. Survey a standard cross-section of the stream,
2. Develop rating relationship between stage (H) and standard cross-sectional area (A),
3. Continuously measure stage (H) and index velocity (V_{index}) using an ADVM,
4. During discrete events, measure discharge (Q_m) at the standard cross-section,

5. Calculate measurement velocity (V_m) = Q_m /A,
6. Develop rating relationship between index velocity (V_{index}) and measurement velocity (V_m),
7. Using the established ratings, convert H to A and V_{index} to V_m,
8. Calculate predicted discharge (Q_p) = (A)*(V_m),
9. Re-measure the standard cross-section annually to check for changes to channel morphology, and,
10. Continue to check accuracy of index velocity rating by making measurements every six weeks and during dramatic changes in flow and water levels.

The index velocity station was developed by expanding an existing stage-only site (St. Croix River at Stillwater, MN, station STLM5) that is operated by the U.S. Army Corps of Engineers (USACE). The USACE gage is mounted to the downstream side of the Minnesota State Highway 36 Bridge at Stillwater, MN (hereafter, Stillwater Liftbridge). The expansion to an index velocity station occurred on September 9, 2011. The ADVM was installed at the Mulberry Point Yacht Harbor, approximately 0.25 miles upstream of the Stillwater Liftbridge (Figure 5). The ADVM communicated data to the USACE gage for transmission over the internet using radios that were connected to the ADVM and the USACE gage. Stage, velocity, and discharge data were collected with methods and instrumentation in accordance with USGS policies (Rantz et al. 1982, Kennedy 1983; Turnipseed and Sauer 2010, Fallon et al. variously dated).

ADVM Gage Installation

The expansion from a stage-only site to an index velocity flow measurement site started on September 9, 2011. A side-looking acoustic Doppler velocity meter (ADVM) was installed upstream of the Stillwater Liftbridge at the Mulberry Point Yacht Harbor. The ADVM was directed horizontal and normal to river flow. The ADVM was connected to a radio bridge that communicates with another radio bridge that was installed at the USACE stage gage on September 9, 2011.

The ADVM was configured and operated according to guidelines established by Levesque and Oberg (2012). The clock was synchronized within two seconds of Greenwich Mean Time (GMT). In addition to the horizontal velocity sensors, a vertical acoustic sensor on the ADVM collected stage and water temperature data. The acoustic stage sensor does not meet USGS accuracy standards, so data from the USACE transducer were used as the primary stage data for developing ratings. However, acoustic stage data were substituted as primary stage data during periods of missing or bad USACE transducer record. Because the ADVM was located close to a marina with frequent boat traffic, a 2.0 m blanking distance was set to minimize interference and signal noise. Then the ADVM was programmed to collect four velocity cells, each 2.0 m long, meaning that the ADVM collected velocity data from 2.0 m to 10.0 m into the channel. An averaging interval of 720 s (12 min) was used with a sampling interval of 900 s (15 min), providing 180 s for the data to be transmitted from the ADVM to the USACE gage.

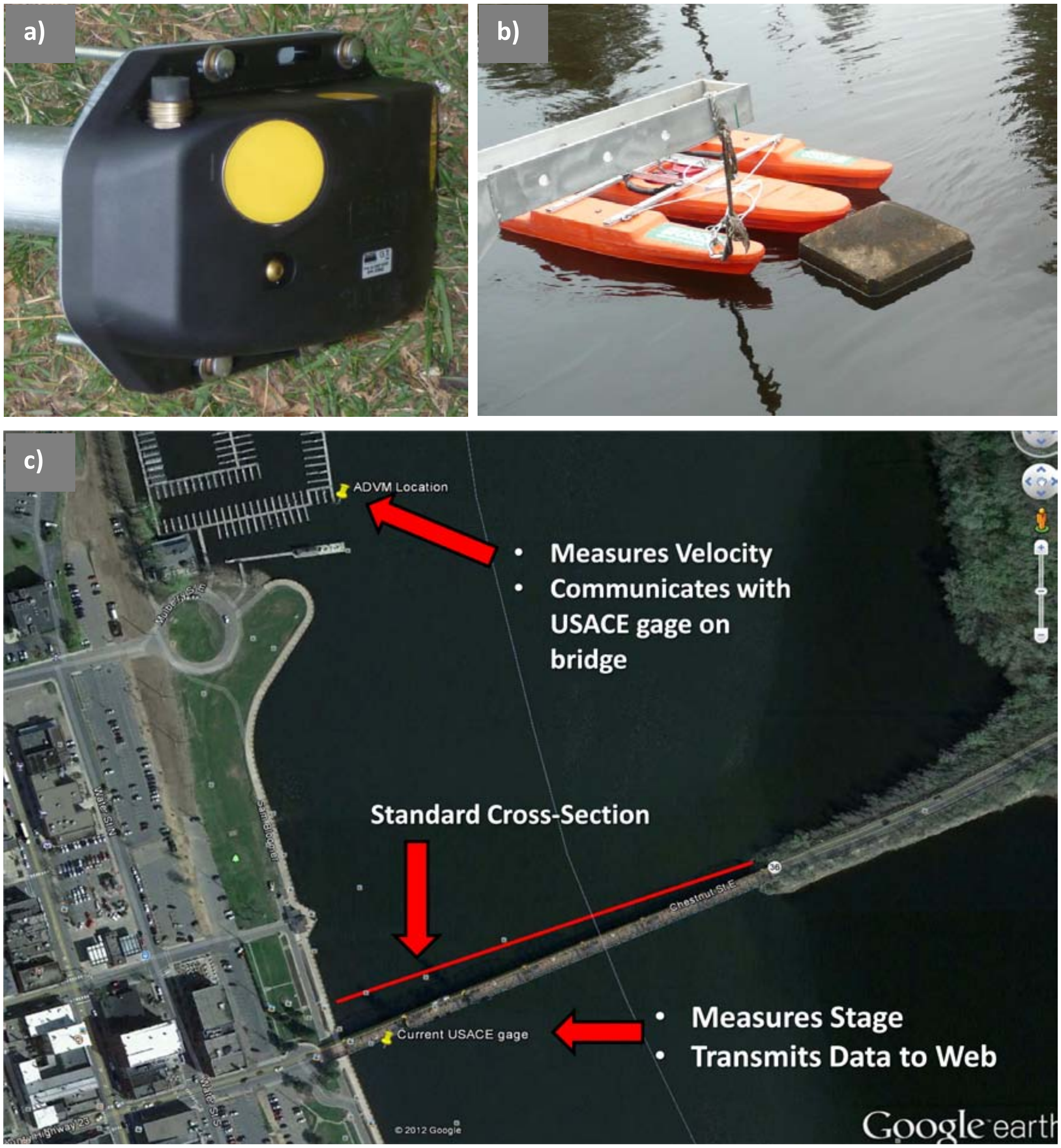


Figure 5. USGS ADVM gage station, a) ADVM, b) ADCP, c) map showing relative locations of USGS gage, USACE gage, and standard cross-section.

Datum Checks and Corrections

During site visits, transducer stage readings were checked against an outside reference reading obtained using a wire-weight gage, which was also mounted to the downstream side of the Stillwater Liftbridge. Stage readings for the wire-weight gage and the transducer were set by surveying water levels based on a benchmark of known elevation (Kennedy 2010). In 2012, the USACE discovered a discrepancy in the datum information for the benchmark used to set gage stage readings. On December 11, 2012, the USACE ran survey levels from another benchmark of known elevation (1912 datum) to check the accuracy of the wire-weight gage and the transducer. The survey levels run using the 1912 datum elevations resulted in a stage increase of 0.12 ft. The wire-weight gage and the transducer were reset to reflect this change in stage. In addition, a + 0.12 ft data correction was applied to all field readings and continuously-recorded stage readings from the start of the index velocity period until the readings were adjusted on December 11, 2012. Finally, the + 0.12 ft data correction was incorporated into the stage-area and index velocity ratings used to determine continuous discharges.

Standard Cross-section for Stage-Area Rating Development

The next step in determining continuous discharges for the St. Croix River at Stillwater was to select a standard cross-section (Figure 6) for stage-area rating development. A standard cross-section was selected on the upstream side of the Stillwater Liftbridge, within the narrows of the

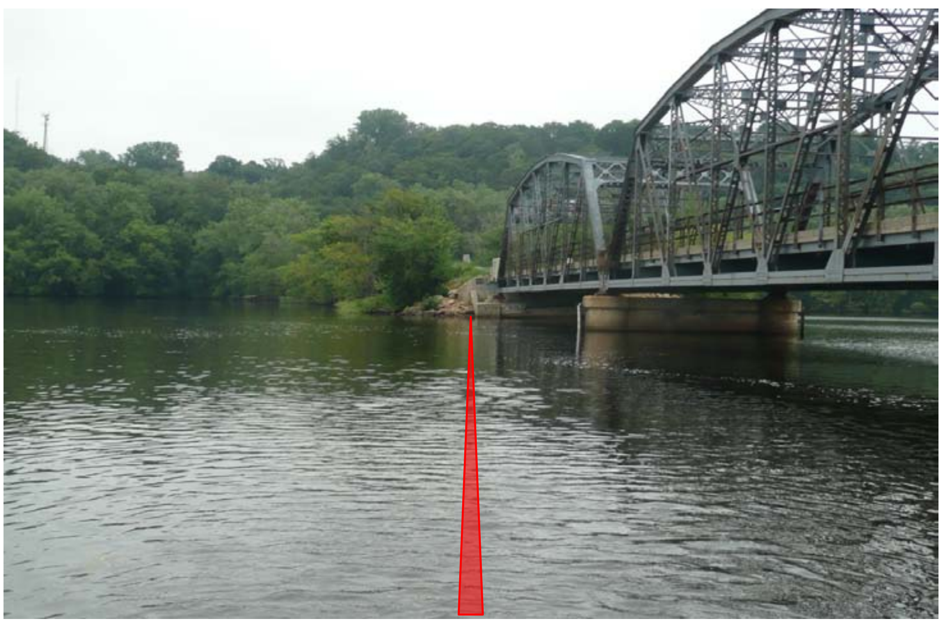


Figure 6. View along standard cross-section (red line) on north side of Stillwater Liftbridge, looking eastward from west bank.

bridge, which could be associated with the USACE stage data. The standard cross-section extends from the left bank (looking downstream), at the rip-rapped edge of water approximately 15 ft upstream of the Stillwater Liftbridge, to the right bank, at a concrete retaining wall on the downstream end of a concrete park staircase approximately 90 ft upstream of the Stillwater Liftbridge (Figure 7). The majority of the cross-section was measured on July 28, 2011 using an acoustic Doppler current profiler (ADCP) at a stage of 681.03 ft (1912 datum). The left bank was vertical enough to estimate the shape of the left bank for remaining stages (up to 690.00 ft), but on the right bank, water spills over the concrete retaining wall at an approximate elevation of 677.42 ft, extending the standard cross-section over a road, another retaining wall, and a series of steps in a city park (Figure X). On March 21, 2012, survey levels were used to measure overbank elevations up to a stage of 690.00 ft. Data from the ADCP and surveyed level measurements were incorporated into the stage-area rating using the program AreaComp2, developed to calculate the area associated with the river channel at any given stage in the standard cross-section.

Figure 7. Right bank spill-over at standard cross-section.

Index Velocity Rating Development

The next step in determining continuous discharges for the St. Croix River at Stillwater was to develop an index velocity rating, which describes the relationship between index velocities (V_{index}) measured by the ADVM and the measurement velocities (V_m) obtained during discrete discharge measurement events. Discrete discharge measurements were made according to guidelines established by Mueller and Wagner (2009). For each discrete discharge measurement, V_m was determined by dividing the measured discharge by the standard cross-sectional area for the given stage, which was determined from the stage-area rating. Discrete discharge measurements were made at regular intervals and also during periods of dramatic changes in flow caused by precipitation events. Having discrete discharge measurement data over a variety of stages, velocities, and seasons incorporated much of the variability in flow conditions, making the index velocity rating for the St. Croix River at Stillwater more robust to environmental changes that might affect the accuracies of the predicted discharges. Several regression models were evaluated to determine which model best fit the collected discharge data, including simple linear, compound linear, and multiple linear regression models (Levesque and Oberg 2012).

Computation of Continuous Discharge

After the stage-area rating and index velocity rating were developed, rated discharges were determined for each discrete discharge measurement by multiplying the standard cross-sectional area (A) by the V_{index} collected by the ADVM during the measurement event. The percent differences between rated and measured discharges were compared to determine the accuracy of the rating over the range of observed flow conditions. Finalized ratings were used to calculate continuous discharges using stage and index velocity data collected at 15-minute intervals by the joint USGS/USACE gaging station.

Estimation of Missing Discharge Data

Daily discharges for the St. Croix River at Stillwater ($Q_{Stillwater}$) were estimated for periods of bad or missing data. Routed discharges from the St. Croix River at St. Croix Falls, WI (05340500) and the Apple River near Somerset (05341500) were used as a guide to help estimate bad or missing periods.

Comparison to Previous Flow Estimates

A continuous discharge gaging station had not been established on the St. Croix River at Stillwater, MN, prior to this project, thus estimated historical daily discharges at Stillwater were used to estimate nutrient loading into Lake St. Croix as part of the phosphorus TMDL established for Lake St. Croix. Historical daily discharges at Stillwater were estimated using measurements from the nearest upstream USGS gages (equations [1] or [2]). In addition to unmeasured portions of contributing flows that were discussed earlier, historical methods of estimating discharge at Stillwater did not incorporate a time lag to account for water residence and travel time between the upstream gages and Stillwater. In order to improve previous flow and nutrient loading estimates, daily discharge values from the index velocity gage were compared to several methods of estimating Stillwater discharge using upstream gage data. Estimation methods included [1] and [2], plus just $Q_{\text{St. Croix Falls}}$, all assessed with and without a one-day time lag. The strengths of the relationships between these estimation methods and the index velocity method were evaluated using linear regression analyses. Only days with measured daily discharges at all three gaging stations (05340500, 05341500, and 05341550) were used to compare methods of determining discharge at Stillwater. A USGS Scientific Investigations Report (SIR) will more thoroughly evaluate how time lags can be used to improve historical flow estimates. Several linear regression models will be developed using flow data collected from the St. Croix River at St. Croix Falls, WI (05340500), the Apple River near Somerset, WI (05341500), and the St. Croix River at Stillwater, MN (05341550). The regression model that most accurately predicts measured flows at Stillwater (05341550) will be used in the SIR (Ziegeweid and Magdalene, in press) to improve estimates of historical flows at Stillwater (05341550), and the improved flow estimates will be used to recalculate estimates of historical loads to Lake St. Croix. Publication of the SIR will mark the completion of Task 5 from the project proposal.

River Water Quality (MCES)

Since 1976, MCES river water quality samples were collected from the St. Croix River at Stillwater, MN and at Prescott, WI, using a vertical Van Dorn sampler, a stainless steel sampler, or a polypropylene sampler from one meter below the surface. The sampler was rinsed with water from the site before sample collection, and then the sample was poured into a laboratory-cleaned sample container, placed in a cooler on ice, and transported to the MCES laboratory located at 2400 Childs Road in St. Paul for processing and analysis. The MCES laboratory is certified under the State of Minnesota's laboratory certification program, and uses analytical procedures that are based on methods approved by the EPA, ASTM, or APHA. The frequency of sampling and analysis varied by analyte. Most water quality variables were sampled and analyzed on a biweekly basis (Kloiber 2004)

Stream Water Quality (MCES)

Stormwater runoff in both urban and rural areas carries nonpoint source pollutants from diverse and widely scattered sources to Metropolitan Area streams and rivers. Stream monitoring was conducted near the mouths of Browns Creek, Silver Creek, and Valley Creek to: 1) determine the extent of nonpoint source pollutant loading from tributaries to the St. Croix River, 2) provide the information necessary for development of target pollutant loads and/or total maximum daily load

(TMDL) plans for these tributary watersheds, and 3) evaluate the effectiveness of watershed best management practices for reducing nonpoint source pollution and improving water quality in streams and rivers. Automated measurements of water stage, in conjunction with site-specific rating curves, were used to estimate flow rates in all streams. During runoff events, automated water samples and occasional grab samples were obtained for laboratory analysis of a variety of nonpoint source pollutants. During baseflow conditions, grab samples were obtained for laboratory analysis of water quality variables (MCES 2011).

Lake Water Quality (MCES)

The Citizen-Assisted Monitoring Program (CAMP) is an MCES-managed program where citizen volunteers monitor the water quality of the nine-county Twin Cities Metropolitan Area lakes since 1999. Seven volunteer monitoring sites are located in the St. Croix River, spanning all four pools of Lake St. Croix: SC-1 and SC-2 are located in shallow Pool 1, SC-3 is located at the Willow narrows, SC-4 and SC-5 are located in the wider Pool 2, SC-6 is located at the upstream end of Pool 3, and SC-7 is located toward the downstream end of Pool 4 (Figure 8). At each site on a bi-weekly basis during open water season (May-October), a volunteer collects a surface water sample for total phosphorus, total Kjeldahl nitrogen, and chlorophyll-a analyses at the MCES laboratory. Volunteers also obtain surface temperature, Secchi depth clarity measurements, and provide some user perception information about the physical and recreational condition of Lake St. Croix. The main purpose of CAMP is to provide lake and watershed managers with water quality information that will contribute to proper management of water resources, and help document water quality status and trends. An added benefit of the program is the volunteer's increased awareness of the conditions in Lake St. Croix, which has fostered local efforts to protect lakes and promote support for lake management. It should be noted that a new Stillwater bridge will be constructed crossing the river at SC-1, and starting June 2012, the SC-1 monitoring site was moved a few hundred feet north to SC-1N. Whether this move makes a difference in long-term water quality trends will be determined with time.

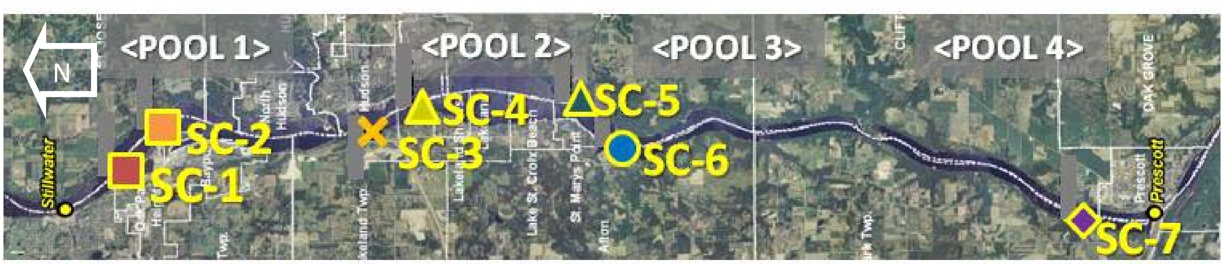


Figure 8. MCES citizen volunteer lake monitoring sites located on Lake St. Croix, sampled biweekly for nutrients and chlorophyll, monthly for depth profiles, and monthly for phytoplankton species.

Lake Ecology (MCES and USGS)

Monthly Phytoplankton Sampling

As a special extension on the MCES CAMP program, two retired 3M scientists volunteered to collect monthly samples of phytoplankton (algae) and to measure water quality variables (temperature, specific conductance, pH, and dissolved oxygen) in depth profiles down to 22 meters with a YSI meter at the seven lake monitoring sites in Lake St. Croix (Figure 8). The phytoplankton samples were collected in glass bottles and preserved with Lugol's solution. The monthly samples were analyzed in batches by Dr. Jeffrey Janik, a phytoplankton taxonomist, who identified the algae to genus- and specie-level, reported abundance (cells/ml) and biomass (mg/m^3), and summarized into nine major algal groups.

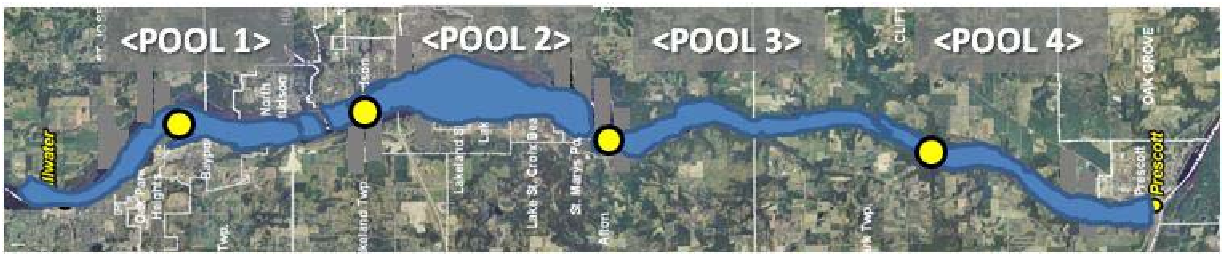


Figure 9. USGS deep pool monitoring sites located in Lake St. Croix, sampled seasonally for depth profiles, phytoplankton, zooplankton, and sediment cores.

USGS Plankton Collection and Overview of Experiments

For the phytoplankton growth bioassays, whole-water samples were collected quarterly from the mixed surface-water layer at four study sites located in the major pools in the lake (Figure. 9), filtered through 153 micron Nitex screen, placed in coolers to keep the phytoplankton at near-ambient lake temperature, isolated from light, and transported to the USGS laboratory within 4 hours of collection. Once at the lab, bioassay samples were placed in an environmental chamber at ambient temperature and under a 14hr:10hr light/dark cycle at saturating light levels.

Phytoplankton Nutrient-Dependent Growth Assays

The spatial and temporal patterns of phytoplankton N and P limitation in the four major pools were studied using nutrient-dependent phytoplankton growth bioassays (Britton and Greeson. Growth experiments consisted of four treatments: (1) ambient nutrient control without the addition of nutrients; (2) addition of N as nitrate (NO_3); (3) addition of P as phosphate (PO_4);

and (4) addition of N and P together. All experiments were conducted using 153 micron Nitex filtered water. Additions of N and P were made using algal growth media stock solutions from DYIII growth media (Lehman 1976, University of Texas 2009).

Once Nitex-filtered water arrived at the USGS MN WSC laboratory, a single composite sample was divided into four separate containers. Each container was enriched with nutrients if appropriate, mixed, and then allocated to replicate 125 milliliter (ml) bottles. All replicate bottles were incubated in an environmental growth chamber set to ambient lake temperature. Light was provided at 180 to 210 microeinstein/s/m^2 on a 14 hour : 10 hour light:dark cycle.

Phytoplankton growth rates were estimated using the acclimated growth rate method (Brand and others 1981, Leboulanger and others 2006). In this method, changes in algal biomass in the replicate bottles were estimated from daily measurements of in vivo fluorescence (IVF) of algae chlorophyll a (Chl-*a*). Chl-*a* IVF (Leboulanger and others 2006) was measured daily with a Turner model 10-AU fluorometer and fit to the following exponential growth model:

$$N_t = N_o \, e^{(rt)} \qquad [5]$$

where

N_t is phytoplankton biomass (indicated by Chl-*a* IVF concentration) after time t

N_o is initial phytoplankton biomass (indicated by Chl-*a* IVF concentration)

e is base of natural logarithms

r is phytoplankton growth rate, per day

t is time, in days.

Taking the natural logarithm (ln) of both sides of eq. 1 and solving for r results in:

$$r = (\ln[N_t] - \ln[N_o])/t \qquad [6]$$

After computing the phytoplankton growth rates, differences between the least-square mean growth rates resulting from the four treatments were assessed using an analysis of variance (ANOVA) F-test (Helsel and Hirsch 2002). Statistically significant differences among the treatment means were assessed usung a Bonferroni multiple means comparison test (Gotelli and Ellison 2004). This multiple means tests was done to indicate which growth rate(s) were different from one another at the $\alpha = 0.05$ level. The ANOVA and Bonferroni tests were conducted using Statistica software (Statistica 2008).

Zooplankton-Grazing Experiments

Zooplankton grazing experiments was conducted by trained personnel at the USGS MN WSC laboratory. Zooplankton grazing experiments measure phytoplankton mortality rates as a function of zooplankton density. In this method, zooplankton abundances (density) were manipulated by concentrating zooplankton and adding them to experimental treatment, thereby creating a gradient of grazer densities. The rate at which algal biomass disappeared from the treatments was a function of grazer density plus any algal growth that occurs during the experiment. These two population growth parameters were estimates as part of the method (Lehman and Sandgren 1985).

Experimental treatments was prepared in 150 ml bottles with the natural phytoplankton assemblage from each site (starting volume about 100 ml), and increasing additions of grazers (zooplankton) were then added to each treatment except for the controls. Experiments will consist of four treatments: (1) a control treatment with no grazers added; (2) a 1X treatment with about 25 grazers added; (3) a 2X treatment with about 50 grazers added; and (4) a 3X treatment with about 75 grazers added. Nutrients (0.15 mM nitrate and 0.015 mM phosphate) were added to all treatments to prevent phytoplankton nutrient limitation and to remove the potential for nutrient regeneration by zooplankton. After 24–48 hours, the experiment was ended, and the 2X and 3X treatments were sieved to collect the >200 µM fraction, which was preserved with 10% formalin for enumeration of zooplankton organisms. Each treatment, including controls, was filtered and extracted for fluorometric Chl-*a* measurement (Welschmeyer 1994). Chl-*a* concentrations was used to indicate phytoplankton biomass.

Seasonal grazing rates were estimated for all sites by phytoplankton mortality rate (rather than growth rate as in the phytoplankton bioassays):

$$r_m = (\ln[N_t] - \ln[N_o])/t \qquad [7]$$

where

r_m is phytoplankton mortality rate, per day

N_t is phytoplankton biomass, as measured by extracted Chl-*a* concentration and/or *In-vivo* fluorescence (IVF) after time t

N_o is initial phytoplankton biomass, and

t is time, in days.

Figure 10 shows the results of a zooplankton grazing experiment on the natural plankton assemblages collected from the near-dam location in a central Texas reservoir (Wallace and

Kiesling, *in prep*). Phytoplankton mortality rate (r_m) was plotted as a linear function of zooplankton density, and the slope of the linear regression is the grazing rate (G), which is 0.11 day^{-1} (Figure. 10).

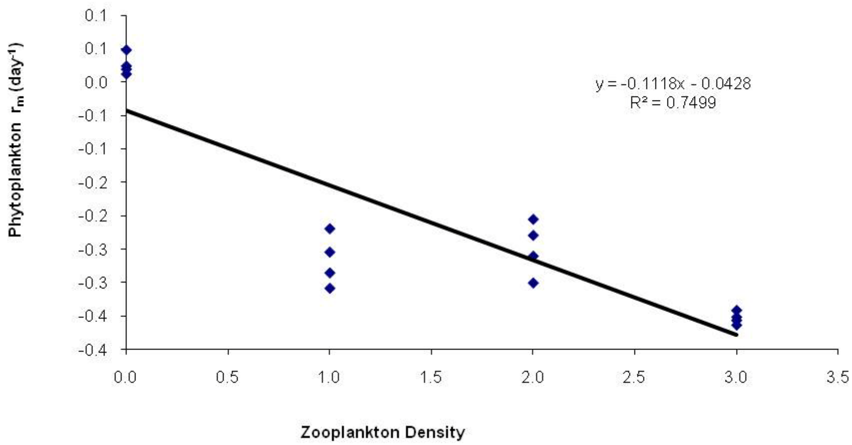


Figure 10. Zooplankton grazing plot from Lake Marble Falls, Texas, November 2007 (Wallace and Kiesling, *in prep*).

Sediments

Sediment-Core Incubations

Sediment nutrient-release rates were calculated as the difference between final and initial concentrations scaled to sediment surface area and duration of incubation. Sediment samples were collected using a Wildco Model 191-A12 stainless steel box corer (Figure 11; Box size: 150 X 150 X 230 mm) fitted with an acrylic sleeve (Christensen and others, *in press*). The box corer was lowered to approximately three meters above the surface of the sediment using a small crane fitted with a cable retrieve. The box corer was released and allowed to free-fall into the sediment, allowing for full penetration into the sediments of the lake while maintaining four to six centimeters of overlying water. Box cores were brought on to the deck of the boat and immediately sub-sampled using polycarbonate push cores (Figure 12). Four replicate push cores were collected simultaneously from each box core, sealed with polyvinyl chloride (PVC) caps, and stored on ice in an upright position for transport to the laboratory.

Figure 11. Wildco Model 191-A12 box corer with liner deployed for recovery of sediments for push cores (photos by R. Kiesling USGS).

The replicate push cores were collected using four-cm diameter, 30-cm long thin-walled polycarbonate tubes (Figure 12). Each core was capped on the top and bottom with tight-fitting PVC caps to preserve ambient conditions. After collection, cores were transported to a laboratory facility and incubated in an environmental chamber under ambient temperature. The push-core samplers allowed the cores to be collected intact and undisturbed, preserving the sediment-water interface throughout the incubation period. Sample water was replaced with particle-free water collected from overlying anoxic water collected at depth. Initial nutrient concentrations were measured from overlying water following a one-hour incubation period in the environmental chamber. Cores were incubated for 48 hours (Penn and others 2000). After incubation, overlying water was decanted from each core, filtered through 0.45 micron filters, and frozen until analysis for dissolved phosphorus. Phosphorus release rates were calculated as the difference between final and initial concentrations in the overlying water.

Figure 12. Wildco Model 191-A12 box corer with push cores (photos by R. Kiesling USGS).

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RESULTS AND DISCUSSION

River Discharge

During late 2010 and early 2011, high flow conditions delayed the installation of the Stillwater index velocity gage until autumn 2011. The first steps were to identify and measure a standard cross-section to be associated with the USACE stage readings (Figure 13) and to develop a stage-area rating for the cross-section (Figure 13).

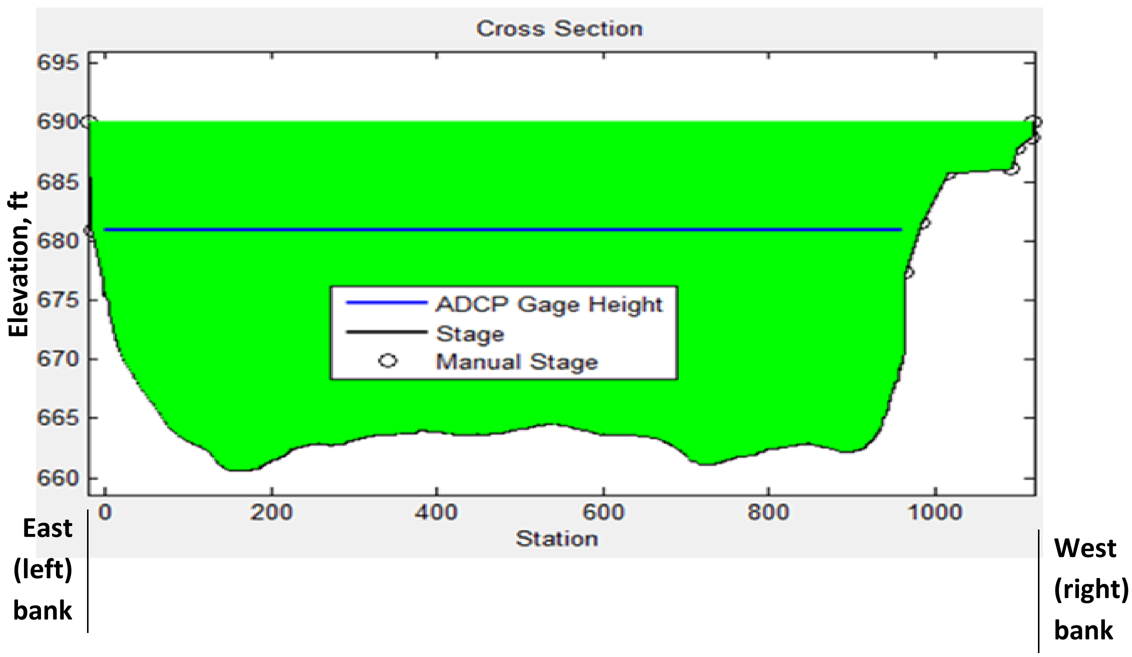

Figure 13. Standard cross-section of St. Croix River at Stillwater, MN, facing downstream.

Stage Data

The USACE stage transducer provided adequate stage data for most of the period of record, with some exceptions. When the ADVM was first installed on September 9, 2011, an improper configuration caused the ADVM to overwrite the USACE transducer data. This issue was corrected when the gage equipment was re-programmed on September 21, 2011. In addition, from October 8, 2012 through the end of the approved record on November 15, 2012, the USACE transducer was inoperable because of maintenance work being done to support pillars on the Stillwater Liftbridge. The ADVM acoustic stage data were substituted as primary stage data during periods of missing USACE transducer records.

Data corrections between continuously collected stage data and outside wire-weight readings were applied accordingly. Data corrections ranged from -0.02 ft to -0.04 ft during the period of record, with a correction of -0.04 ft observed throughout most of the period of record. However, the +0.12 ft correction resulting from the conversion to 1912 datum elevations produced actual applied data corrections ranging from +0.08 ft to +0.10 ft during the period of record.

Stage-Area Rating

Based on the ADCP survey done on July 28, 2011 (Figure 13) and the overbank level survey done on March 21, 2012, AreaComp2 was used to develop stage-area rating 1.0. The AreaComp2 output of rating 1.0 was used to determine the area of the standard cross-section at 0.10 ft intervals. The AreaComp2 output was then used to develop a linear equation that describes the relationship between stage (ft) and area (ft^2) (Figure 14). The area of the standard cross-section can be predicted from the stage (1912 datum) using the following equation ($p < 0.0001$, $df = 150$, $R^2 = 0.9993$):

$$\text{Area} = 1{,}018.66(\text{Stage}) - 676{,}837.62 \qquad [8]$$

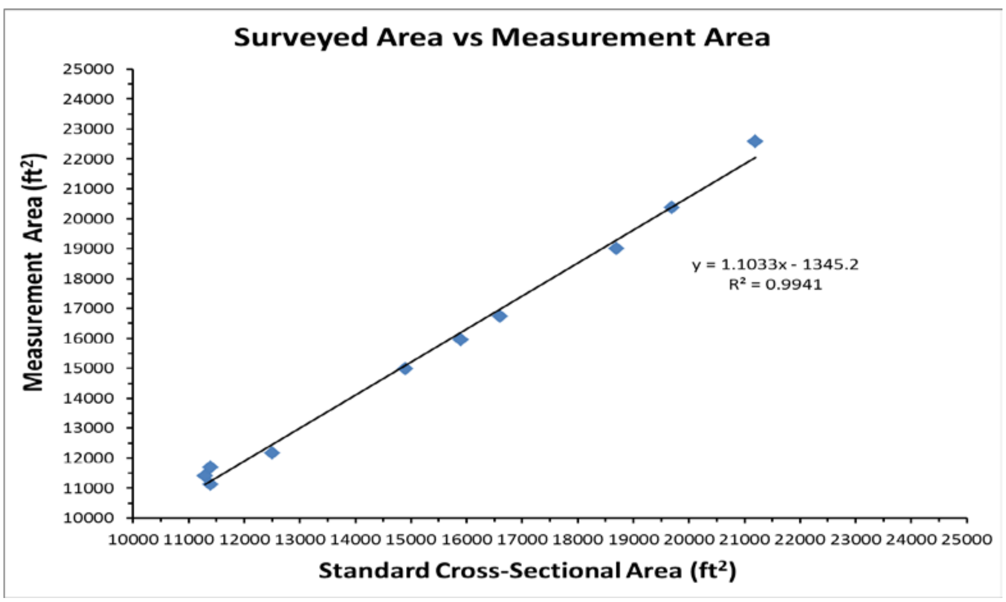

Figure 14. Stage-area rating curve fit for the standard cross-section at Stillwater Liftbridge.

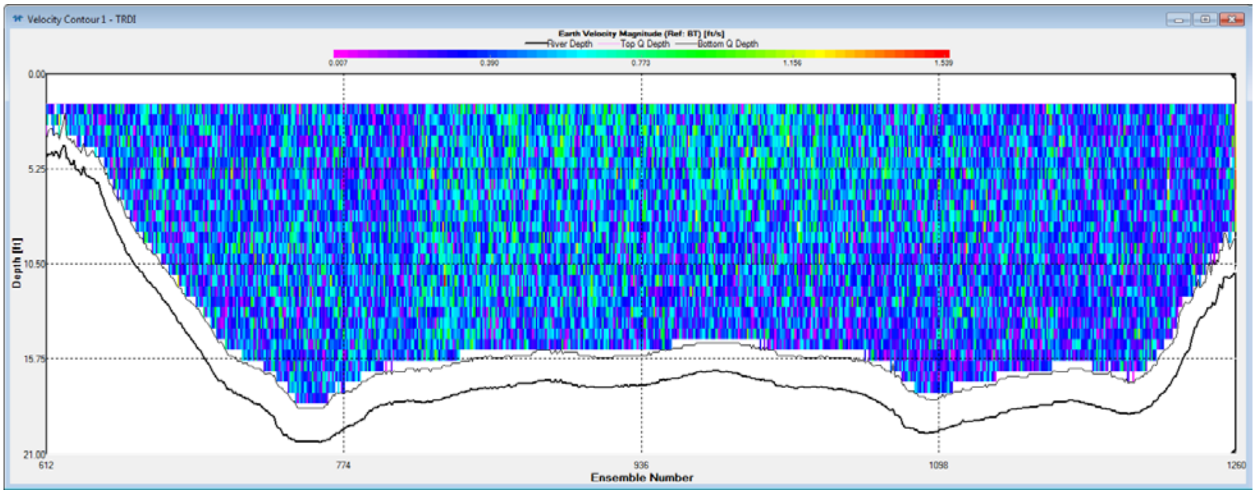

Figure 15. Cross-section of ADCP discrete flow velocity measurement event on July 28, 2011.

Index Velocity Rating

From September 9, 2011 through November 15, 2012, 11 discrete discharge measurements (Figure 15, for example) were made to help develop the index velocity rating. Discharge measurements were made and processed according to methods outlined by Mueller and Wagner (2009). Nine of the 11 discharge measurements were used in the final index velocity rating. The discharge measurement made on March 21, 2012 was excluded from the rating because a temporary power failure at the ADVM caused missing values from one hour before the measurement to one hour after the measurement. The discharge measurement made on May 14, 2012 was made during a period of rapidly changing stage and flow conditions. Residual analysis indicated that this measurement did not fit the trend of the other discharge measurements, so this measurement was excluded as an outlier.

The ADVM collected velocity data in four 2.0-m long cells, spanning from 2.0 m to 10.0 m horizontally in the channel from the ADVM. Data from the velocity cells were compared individually against velocities from discrete discharge measurements to determine which velocity cell had the strongest predictive relationship to measured velocity data. Based on residual analyses, velocity cell four (8.0 to 10.0 m from the ADVM) had the strongest predictive relationship to discrete measured velocity data.

Several linear regression models were compared to determine which model most accurately described the relationship between ADVM velocities (V_{index}) and mean measured velocities (V_{mean}). A simple linear model assumes a uniform relationship between V_{index} and V_{mean} at all observed stages. A compound linear model assumes that the relationship between V_{index} and V_{mean} changes at a breakpoint that reflects a change in cross-sectional area. The relationship is consistently linear above and below this breakpoint, but the slope of the line segments above and

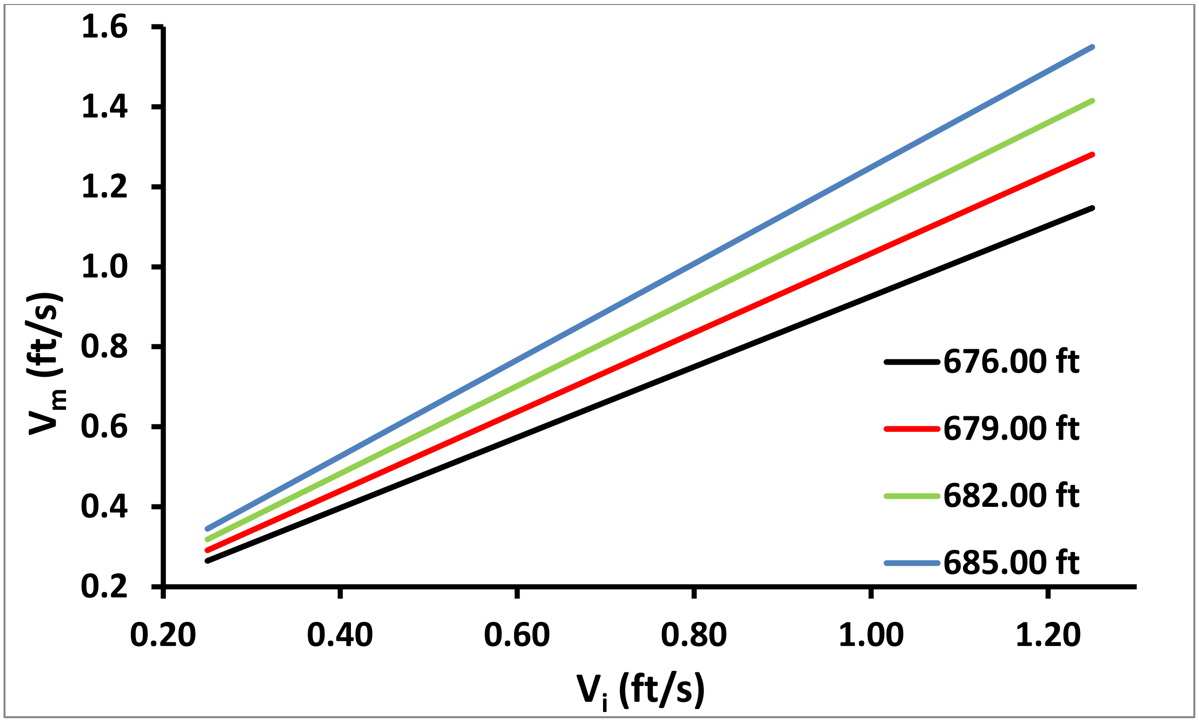

Figure 16. Theoretical output of the index velocity multiple regression rating equation and the effect of stage on the relationship between index velocity and mean streamflow velocity, demonstrated using stages of 676, 679, 682, and 685 feet.

below this point changes in relation to a change in the cross- sectional area, like the concrete retaining wall on the right bank. A multiple linear model assumes that V_{index} and stage interact to significantly affect the discharge. Based on model comparisons, a multiple linear regression model was chosen to describe the relationship between V_{index}, V_{mean}, and stage (Figure 16). The index velocity rating equation used to predict ($p = 0.03$, $df = 7$, adjusted $R^2 = 0.9975$) is as follows:

$$V_{mean} = 0.0440 - 23.3186(V_{index}) + 0.0358(V_{index}*stage) \quad [9]$$

Calibration of Discharge Computation Model

Stage-area rating 1.0 and the index velocity rating equation were used to compute rated discharges for discrete discharge measurements. The nine measurements used to develop the ratings were all within 11.6% of rated discharges (Table 2). Rated discharges were within 8% of measured discharges for eight of the nine measurements, and rated discharges were within 5% of measured discharges for seven of the nine measurements. In addition, for the measurement made on March 21, 2012, which was not included in the index velocity rating, the rated discharge was within 1.6% of the measured discharge. Only the measurement made on May 14, 2012, which was not included in the index velocity rating, had a rated discharge that differed substantially from the measured discharge (-17.3%). However, this measurement was made during a period of rapidly changing stage and flow conditions. In addition, measured discharges on May 8, 2012 and May 17, 2012 were within 3.9% and 6.7% of rated discharges, respectively. Therefore, we are confident that we can exclude the measurement from May 14, 2012 and still accurately describe the hydrology of the time period surrounding the measurement.

Table 2. Percent differences between rated (predicted) and measured discharges for the St. Croix River at Stillwater, MN.

Date	Stage (ft)	Q_{rating} (cfs)	Q_{meas} (cfs)	% Difference
11/17/2011	675.48	2860	2930	2.3
3/21/2012	676.58	8680	8540	-1.6
5/8/2012	680.14	10800	11200	3.9
5/14/2012	680.78	12100	10000	-17.3
5/17/2012	679.15	7030	7500	6.7
5/29/2012	683.93	28500	28700	0.5
5/31/2012	685.73	35300	35800	1.3
6/28/2012	682.92	14500	13800	-4.5
8/10/2012	675.39	2150	2400	11.6
9/12/2012	675.53	2130	2050	-3.7
11/15/2012	675.25	2740	2710	-1.2

Index Velocity Rating Quality Assurance

According to new guidelines outlined by the USGS Office of Surface Water (OSW), at least 20 discrete discharge measurements should be made for a multiple linear regression index velocity rating. However, for the duration of data collection allowed in this project, only 11 measurements were able to be made. We feel that the index velocity rating equation developed from the 11 measurements provides the best estimates of daily discharge for the purpose of this report. Funding for the index velocity gaging station will be continued beyond the duration of this project, and the index velocity rating equation will be updated after sufficient additional measurements are made at the site. Daily discharges for the St. Croix River at Stillwater (site 05341550) were published in the 2012 USGS Minnesota Water Science Center Annual Data Report (http://wdr.water.usgs.gov/adrgmap/index.html).

Estimated Discharge Data

During the worked period of record (September 9, 2011 through November 15, 2012), there were several periods of bad or missing data. Daily discharge values were estimated for these periods, using a one-day time lag on routed data from the St. Croix River at St. Croix Falls and the Apple River near Somerset, WI as a guide. December 2, 2011 through March 30, 2012 were estimated because of ice conditions that affected the accuracy of the velocity data being collected. April 26, 2012 through May 7, 2012 were estimated because a power failure at the ADVM resulted in no data being collected on these days. Other estimated days with insufficient continuous data for measured daily values include the following: September 11, 18, 29, 2011; October 18, 2011; April 16, 2012; July 17, 2012, August 16, 2012; September 7, 17, 22, 2012; October 5-6, 11, 14, 25, 2012; November 7-10, 15, 2012.

Comparison to Previous Flow Estimates

Prior to installation of an index velocity gaging station on the St. Croix River at Stillwater (05341550), either [1] or [2] was used to estimate daily discharges at Stillwater, and the estimated discharges were used to calculate mainstem phosphorus loading into Lake St. Croix. Following installation of the index velocity gage, daily discharges were measured directly, and measured discharges were used to calculate mainstem phosphorus loads into Lake St. Croix. In addition, we used measured daily discharges from the index velocity gaging station to assess the accuracy of previous equations for predicting daily discharges in the St. Croix River at Stillwater. Finally, we incorporated a time lag of one day into the comparisons, to account for transport time of the 42 km (26 miles) between St. Croix Falls and Stillwater, as a possible improvement on previous estimates. For the period of measured discharges at Stillwater, we predicted discharge using three equations ([1], [2], and only $Q_{St.CroixFalls}$) without a time lag, and three equations with a one-day time lag (Table 3). The resulting comparisons were fit using simple linear regression equations; a comparison between the correlation coefficients of these linear regressions are presented in Table 3. Based on the results of these simple linear regression analyses, both of the previously-used predictor equations seem highly correlated with the actual measured discharge. However, it appears that there is greater variance (i.e, lower correlation) associated with adding flow to $Q_{St.CroixFalls}$ without a time lag (Eqtns 1 & 2 in Table 3), than adding a one day time lag without additional flow (Eqtn 6 in Table 3).

Table 3. Comparison of Pearson's correlation coefficient (R^2) for simple linear regression fit to equations predicting flow in the St. Croix River at Stillwater, MN. (p-value < 0.0001, df = 239).

Predictor equation	R^2	Predictor equation with time lag	R^2
1) $Q_{St. Croix Falls} + Q_{Apple}$	0.91917	4) [$Q_{St. Croix Falls} + Q_{Apple}$] + 1 day time lag	0.98163
2) 1.11 x $Q_{St. Croix Falls}$	0.92106	5) [1.11 x $Q_{St. Croix Falls}$] + 1 day time lag	0.98219
3) $Q_{St. Croix Falls}$	0.92106	6) $Q_{St. Croix Falls}$ + 1 day time lag	0.98219

The predictor equation that incorporates a one-day time lag into $Q_{St.\ Croix\ Falls}$ data produces the strongest relationship to the measured index velocity discharge and the least amount of unexplained variation. The provisional regression equation to predict daily discharge for the St. Croix River at Stillwater is as follows:

$$Q_{Stillwater} = 0.9706(Q_{St.\ Croix\ Falls}) + 680.6135 \qquad [10]$$

In order to further assess the accuracy of old and new estimation methods, discharges predicted using the two previously-used equations ([1] and [2]) were plotted against the actual measured discharge at Stillwater in Water Year 2012 (Figure 17). In addition, Figure 17 compares an improved estimation method using a predictor equation that incorporates a time lag (the best-fit regression Equation [10] above). Based on Water Year 2012, the previously-used methods tended to underestimate base flows and overestimate peak flows measured at Stillwater. In addition, substantial changes between base flows and spring thaw/precipitation events occurred earlier in the predicted hydrographs compared to actual measured hydrographs. Graphical data from the hydrographs provide for support for incorporating a time lag into historical estimates of Stillwater discharges. Furthermore, errors in the historical estimates of discharge likely propagated into errors in phosphorus-loading calculations at Stillwater. The regression fit time-lagged equation [10] is a much better fit to measured discharge at Stillwater. However, upon closer inspection of the residuals generated during the regression analysis, which are asymmetric about the x-axis (Figure 18), this simple linear regression might be improved by stratification with discharge. That is, the estimation method should be modified using compound regression, or three separate linear regressions dependent on discharge, delineated at approximately 7,000 cfs and 30,000 cfs (Figure 18). Therefore, the SIR (Ziegeweid and Magdalene, in press) will further refine the equations that will improve historical flow and phosphorus-loading estimates at Stillwater.

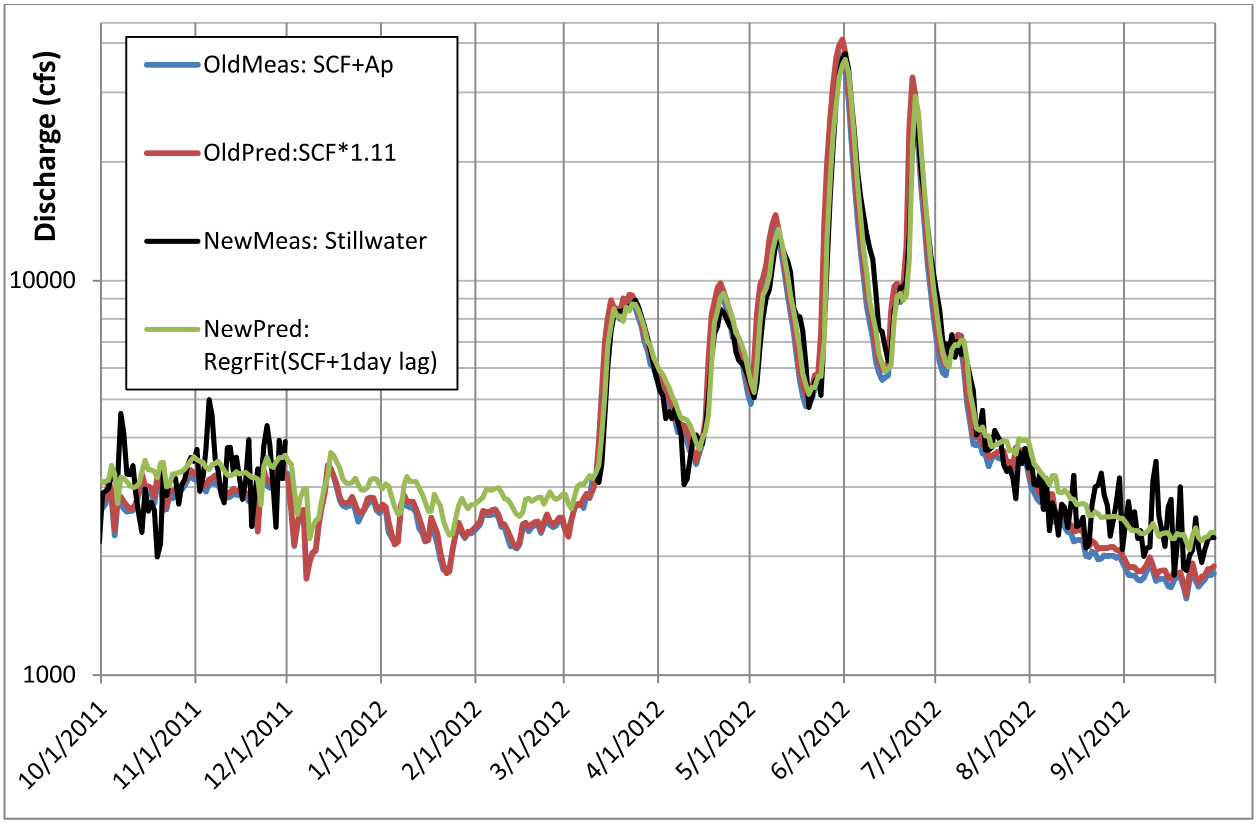

Figure 17. Comparison of estimation methods for discharge in the St. Croix River at Stillwater, MN, Water Year 2012 (October 1, 2011 – September 30,2012). Comparison is between the old measurement method (equation [1]), the old prediction method (equation [2]), the new measurement method (USGS#05341550), and the new prediction method (equation [10]). Gap in measured discharge at Stillwater during under-ice conditions (Dec 2011-Mar 2012) will likely be estimated following further analysis. Equation [10] is provisional, subject to revision in the upcoming SIR (Ziegeweid and Magdalene, in press).

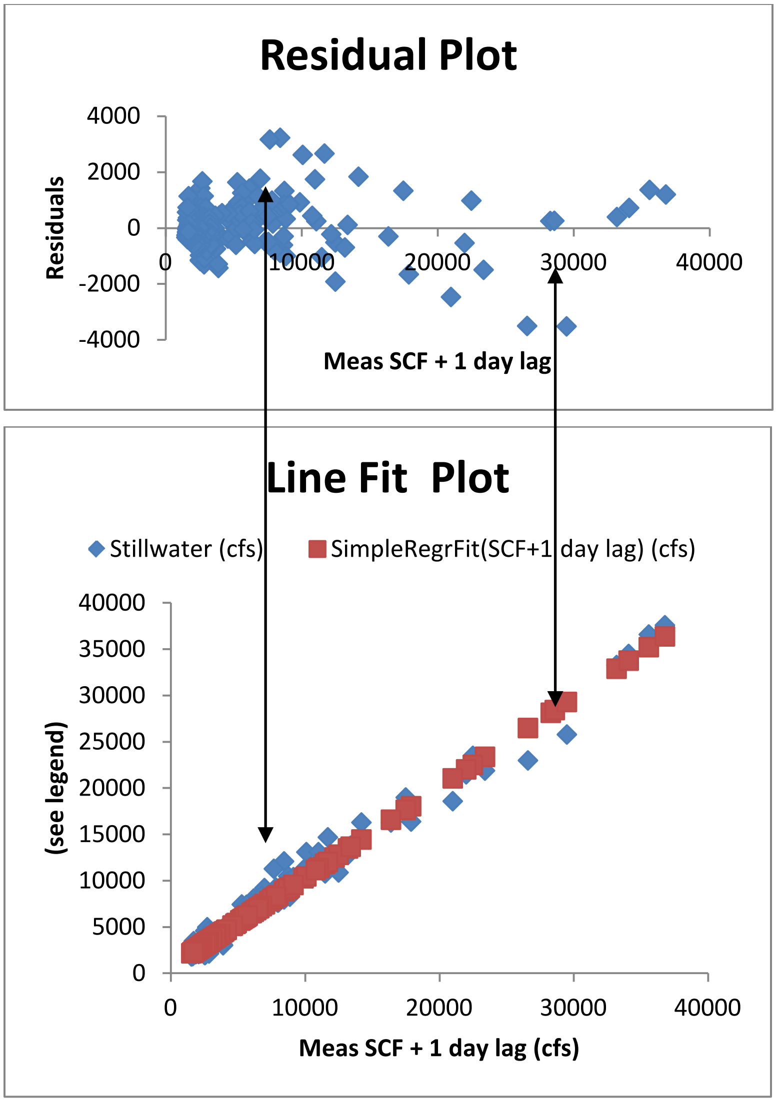

Figure 18. Statistical analysis results for using a simple linear regression equation to predict discharge on the St. Croix River at Stillwater, MN.

Characterization of climate variables and flow during the 2008-2012 period

Mean annual air temperature and total annual precipitation were compared to 30-year (1981-2010) mean values in Table 4, using four National Weather Service stations located across the St. Croix Basin: Hinckley, MN (NWS#213793), St. Croix Falls, WI (NWS#477464), Stillwater, MN (NWS #218039), and River Falls, WI (NWS#477226). The mean annual air temperature during the study period appeared to be on a warming trend at all stations. With respect to total annual precipitation, most of the 2008-2012 study period was drier than normal, with only 2010 exceeding the 30-year mean annual rainfall, except for the Hinckley station, which was also wetter than normal in 2008-2009.

Table 4. Mean annual air temperature (°F) and total annual precipitation (inches) 2008-2012, compared with 30-year (1981-2010) mean values at Hinckley, MN (NWS#213793), St. Croix Falls, WI, (NWS#477464), Stillwater, MN (NWS#218039) and River Falls, WI (NWS#477226).

Variable	2008	2009	2010	2011	2012	30-year Mean
Mean Air Temp (°F)						
Hinckley	38.0	38.8	41.7	41.3	45.7	41.1
St. Croix Falls	40.5	41.4	43.6	43.1	46.4	43.4
Stillwater	42.2	43.0	45.2	44.5	48.2	48.0
River Falls	40.8	41.6	44.0	43.4	46.9	43.6
Total Precip (in)						
Hinckley	38.2	35.1	42.9	27.8	28.8	31.8
St. Croix Falls	24.0	23.7	36.3	23.7	28.4	32.1
Stillwater	27.1	28.3	36.8	31.7	28.9	34.5
River Falls	27.1	31.0	41.4	27.0	28.2	31.8

To characterize the 5-year study period with respect to the mean or expected flow levels, we compared daily measured discharges at three St. Croix River gages to the 30-year mean daily discharge at St. Croix Falls (Figure 19). The period of 2008-2012 represents a transition from a low flow period to a high period. Before mid-2010, only a few peak flows exceeded the 30-year mean flows; baseflows were well below the 30-year mean. After mid-2010, even the baseflows at St. Croix Falls were often above the 30-year mean, with the exception of late autumn in 2011 and 2012. Though generally higher flows occurred in those years, it is possible that the landscape in the St. Croix Basin was not recharged during those two autumns.

Figure 20 shows greater seasonal detail for Stillwater, using monthly mean air temperatures, monthly total precipitation, and daily discharges at Stillwater, compared with the 30-year mean values. Because discharge was not measured on the St. Croix River at Stillwater until October 2011, the preceding period (starting January 2008) and the 30-year mean values were calculated using St. Croix Falls discharge and equation [10]. The plot of monthly temperatures supports the observation of below-normal temperatures on a warming trend during the study period. In particular, the first four winters (January 2008, January 2009, January 2010, and January 2011) had temperatures 7-18 °F below normal. Figure 20 also highlights the importance of autumn rainfall for reaching and sustaining peak flows in the spring and summer: only autumn 2010 had normal rainfall, and only 2011 had sustained peak discharges.

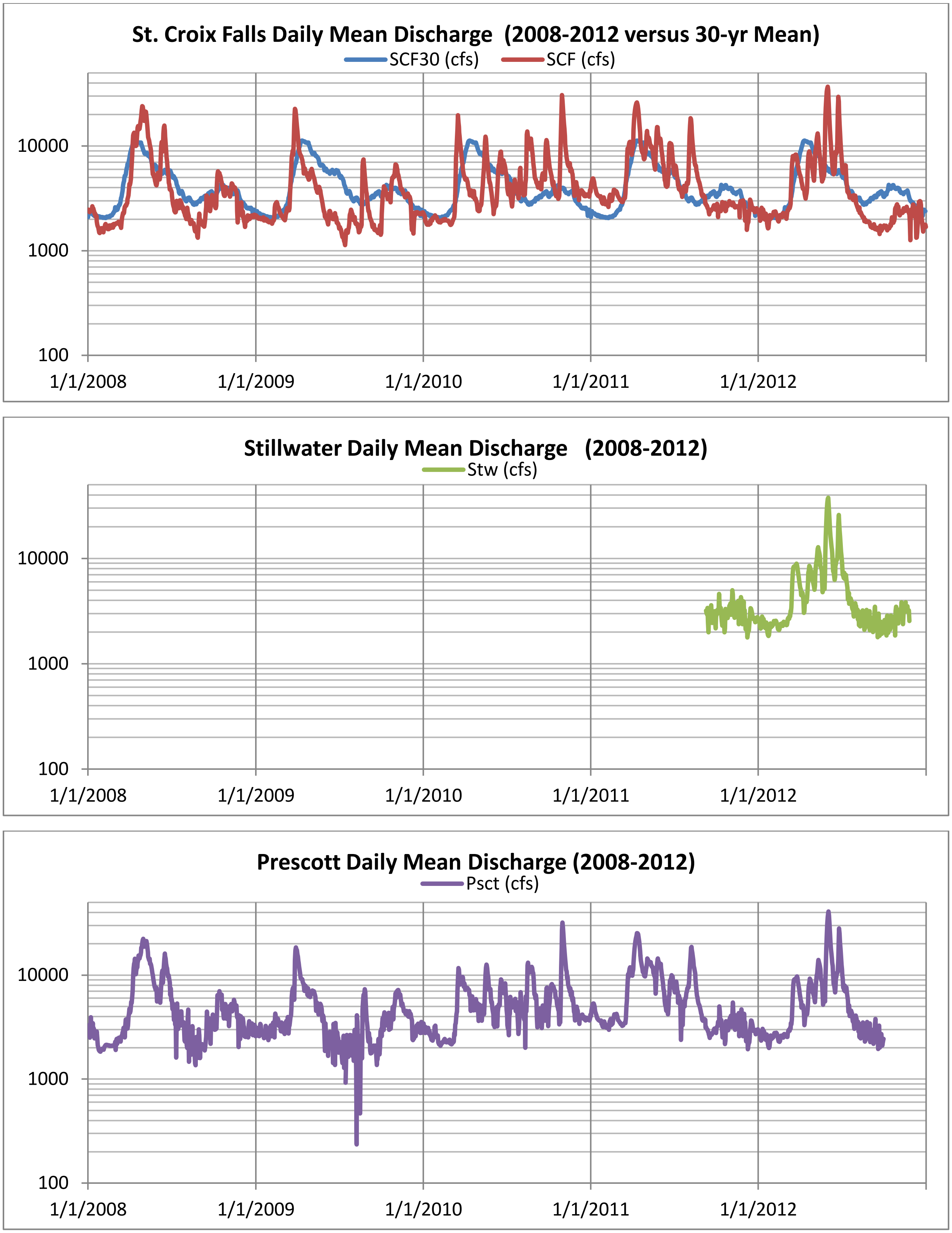

Figure 19. Daily mean of measured discharges (cfs) at a) St. Croix Falls, b) Stillwater, and c) Prescott, for 2008-2012, compared to 30-year mean (1980-2010) at St. Croix Falls.

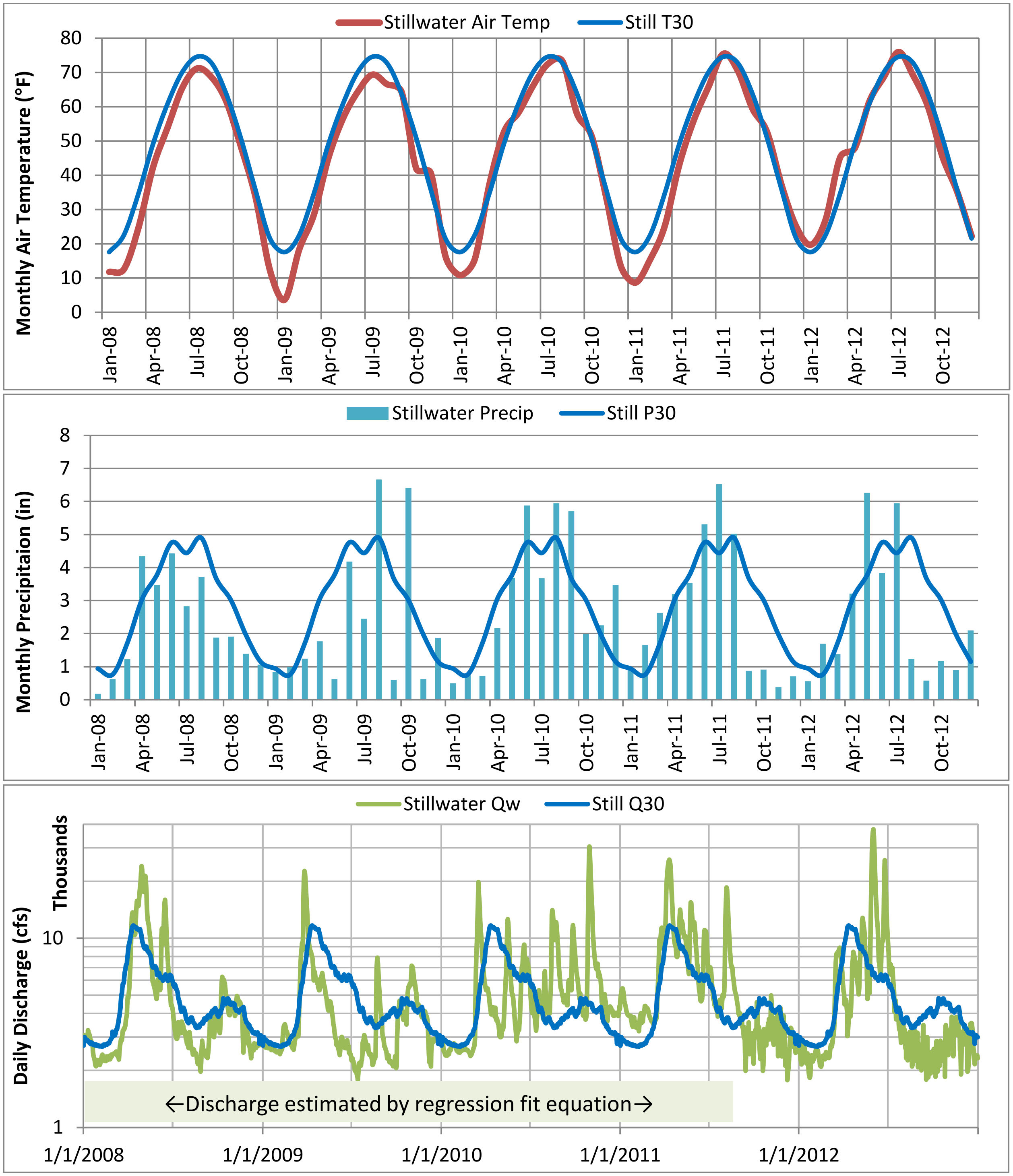

Figure 20. Monthly mean air temperature (°F), monthly total precipitation (in) and daily discharge (cfs), 2008-2012, each compared with 30-year (1981-2010) mean values at Stillwater, MN (NWS#218039 and USGS#05341550).

Monthly Profiles of Lake Water Quality

The original monitoring protocol for lake profiles was focused on the open water season (May-Oct), though questions arose as to the behavior of Lake St. Croix in the winter. Therefore, we measured temperature and dissolved oxygen depth profiles on two occasions: at SC-2 on January 11, 2010 and at deep hole SC-6 on February 24, 2012. At SC-2 on January 11, 2010, the lake demonstrated a classic winter thermal profile--frozen (0 °C) at the surface and the densest water (4 °C) at depth. On both occasions, the lake was fully oxygenated at depth (Figure 21). Further research is needed to determine whether this condition is sustained through the winter, or if anoxia develops under thick ice-cover, causing ecological stress (Fang and Stefan 2000).

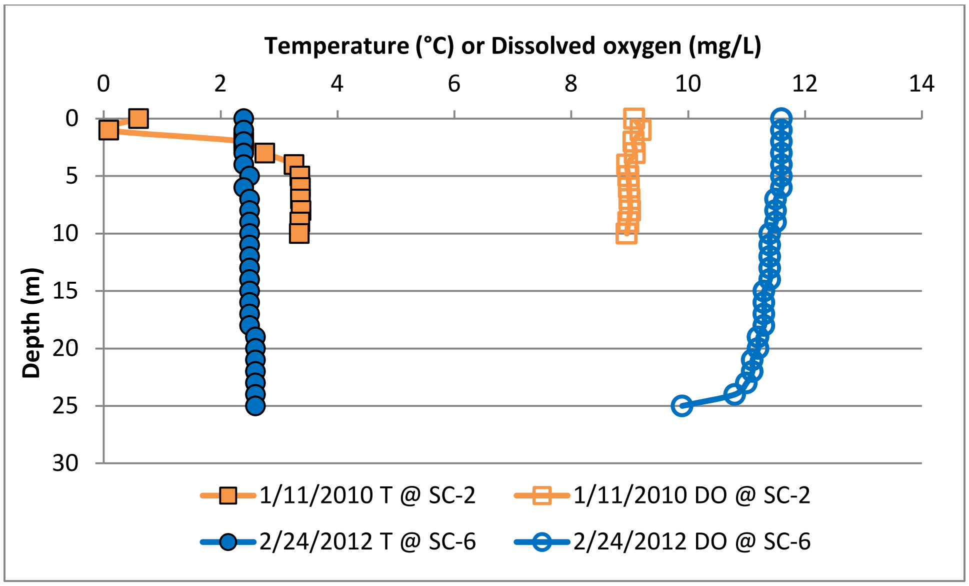

Figure 21. Depth profiles of temperature (°C) and dissolved oxygen (mg/L) at two MCES volunteer monitoring sites: SC-2 on January 11, 2010 and SC-6 on February 24, 2012.

We plotted the temperature and dissolved oxygen profiles for each month (May-Oct) of 2008-2012 (Figures 22-26). The daily measured and 30-year mean (1981-2010) discharge (USGS #05341550) and maximum air temperature (NWS #218039) is also shown, to lend context to the depth profile results; months during the study period that had air temperatures and discharges close to the long-term 30-year mean values were expected to typify the long-term behavior of that month. In contrast, when temperature or discharge measurements deviated widely from the 30-year mean values, deviations in the profile behavior in that month was expected to reveal the influence of that variable. Since 2012 was the only full year of measured discharge at Stillwater, we used St. Croix River discharge at St. Croix Falls, WI with the best-fit regression equation [10] to estimate discharge at Stillwater, MN for 2008- 2011 and for the 30-year expected mean discharge at Stillwater.

Seasonal Patterns

May: Among the three years that had profile measurements in May (2010-2012), measured air temperatures and discharges were very different from the long-term 30-year means for Stillwater (Figures 24-26). As a result, the three profile measurements may not represent long-term behavior in the month of May. Generally, the lake monitoring sites were weakly but inconsistently stratified with respect to temperature and dissolved oxygen. Water temperatures were typically 16-18 °C at the surface, declining to 10-12 °C at depth. Relative to the winter profiles in Figure 20, the deep pools warmed from 4 °C in January to 10-14 °C in May, while DO decreased from 9-10 mg/L in January down to 6-8 mg/L or lower in May.

June: Among the four years that had profile measurements in June (2009-2012), measured air temperatures and discharges were closest to the 30-year long-term June mean values in 2010 and 2011. Generally, lake-like stratification was more developed by June. Water temperatures were typically 20-24 °C at the surface, declining to about 12 °C at depth. The thermocline was at about 10-12 m depth. Thermal stratification was less well established, due to warmer temperatures in the deep pools in June, when preceded either by low flows (e.g., June 2009) or by high flows (e.g., June 2012). By June, dissolved oxygen had begun to stratify, with marked oxyclines in all deep pool sites. The deepest pools were anoxic by mid-June, when preceded by low- to moderate-flows (2009, 2010, and 2011).

July: Air temperature and discharge were close to the long-term 30-year mean in July of 2010, 2011, and 2012, therefore the long-term average July should be represented by those years. Water temperatures were typically 27 °C at the surface, declining to 12 °C at depth. In years of low flow (e.g., 2008-2009), or when preceded by a high-flow June (e.g., 2012), the deepest pools were warmer at about 16 °C. Relative to the previous month, the oxycline in July had risen in the water column at all pools. By the month of July, the oxycline also developed a spatial pattern: the oxycline was positioned at greater depths in the water column with distance downstream, and the oxycline thickened from SC-1 to SC-6. The monitoring site SC-7 breaks from this downstream trend, with a thinner and higher oxycline than SC-6, possibly due to its lateral position within the lake, off the main flow channel.

August: Profile measurements in August 2010 and August 2012 were preceded by air temperatures and discharges close to the long-term 30-year mean values, therefore the long-term average August should be represented by those years. Relative to the preceding month, water temperatures in August were a little warmer at the lake surface (28-32 °C), but unchanged in the deep pools. As a result, the thermocline had been driven deeper in the water column between July and August. During low flow years (e.g., 2008, 2009), thermal profiles were tightly clustered with respect to site; there was little difference in the shape and depth of the thermocline. In contrast, there was greater difference between sites regarding the depth of the thermocline during moderate- to high-flow years (e.g., 2010-2012). The pool surfaces were usually well oxygenated in August, with DO in the range of 8-10 mg/L. The oxycline rose to the highest position of the year, for all pools, though the same spatial pattern established between sites in July persisted into August: the thermocline was pushed progressively deeper from SC-1 to SC-6, rose again at SC-7. By August, all deep pools were anoxic below 10 m depth.

September: Profile measurements were taken across a range of stages of the autumn turnover process, depending on the relative timing of measurement date and seasonal cooling. In September 2009, profile measurement apparently occurred before turnover, in that both T and DO profiles had changed very little since the previous month. In September 2012, profiles in the upper pools had changed little, but both thermocline and oxycline had been driven deeper, compared to the previous month. In September 2011, the lake has clearly initiated autumn turnover when profile measurements were taken; compared to the previous month, surface temperatures had not changed much, but the thermocline had been driven deeper at all sites, and the oxygen profiles had shifted. The upper sites were more oxygenated than the lower sites and the surface layer of the lower sites were uniformly mixed above the oxycline, compared to the previous month. In September 2008, the surface temperatures had cooled relative to the previous month, and both thermocline and oxycline had been driven toward the bottom of the deepest pools, mixing hypoxic waters upward in the water column. A spatial pattern developed, a downstream trend of lower DO concentrations. Finally, the late September 2010 profile was measured after the completion of autumn turnover. Both thermocline and oxycline had disappeared, allowing for surface-to-bottom mixing of hypoxic bottom waters within each pool, possibly along with downstream advection, produced vertical dissolved oxygen profiles that decreased in concentration from upstream to downstream.

October: By October, thermal profiles were vertical at all sites, with the exception of a persistent 4-6 °C warming in the upper surface two meters at SC-2 near the power plant outfall. The spatial pattern of temperature profiles indicates a slight (3-5 °C) increasing downstream trend. October oxygen profiles have the opposite pattern: a decreasing downstream trend. As mixing progressed during the preceding month, hypoxic waters were mixed downstream, and the lake became less oxygenated proceeding downstream. The DO concentrations at SC-7 in October were sometimes less than 6 mg/L, and under low-flow conditions approached the water quality standard of 5 mg/L (e.g., October 2009).

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Relationship between lake profiles and river discharge

The 2008-2012 study period included a range of flow conditions, starting within a period of low-flow conditions (since 2006), and switching to high-flow conditions in mid-2010. During mid-summer low-flow conditions (e.g., July-September 2009) the dissolved oxygen profile exhibits a thin (2-4 m) surface layer of highly-oxygenated (>8 mg/L) waters underlain by a thick, gradual oxycline ending by 8-14 meters depth in anoxic bottom waters. During mid-summer normal-flow conditions (e.g., July 2011 or July 2012), the dissolved oxygen profile exhibits a thicker (6-8 m) surface layer of moderately-oxygenated (6-8 mg/L) waters underlain by a steeper oxycline ending by 10-16 meters depth in anoxic bottom waters. During mid-summer high-flow conditions (e.g., May-June 2011, May-June 2012) the dissolved oxygen profile exhibits a very thick (10-14 m) surface layer of moderately-oxygenated (6-8 mg/L) waters. When preceded by moderately-high flows, the oxygenated surface layer was underlain by a steep oxycline ending by 16-22 meters depth in anoxic bottom waters (e.g., June 2011). When preceded by very-high flows (e.g., June 2012), the oxygenated surface layer was underlain by a truncated oxycline ending in hypoxic bottom waters (>2 mg/L).

In general, high-flow conditions in Lake St. Croix appear to force downward the well-mixed surface layer, shearing off the cooler, less oxygenated layers below and mixing them upward. This is most clearly demonstrated in the August 2011 profiles (Figure 25). The shallower sites (e.g., SC-1, SC-2, and SC-7) that had been thermally-stratified the previous month, until the high discharges in August 2012 pushed the thermocline deeper to the point that those sites de-stratified with respect to temperature. The surface temperatures, instead of warming that August, cooled off from June 2012 as the cooler water in the bottom of those shallow pools was sheared off and mixed upward. Sites SC-1 and SC-2 in Bayport Pool 1maintained their oxygen stratification from June 2012, but site SC-7 in Kinnickinnic Pool 4was affected by the deeper mixing of high discharge conditions and the bottom-layer hypoxia that mixed upward caused decreased dissolved oxygen in the entire profile at SC-7 in August 2012.

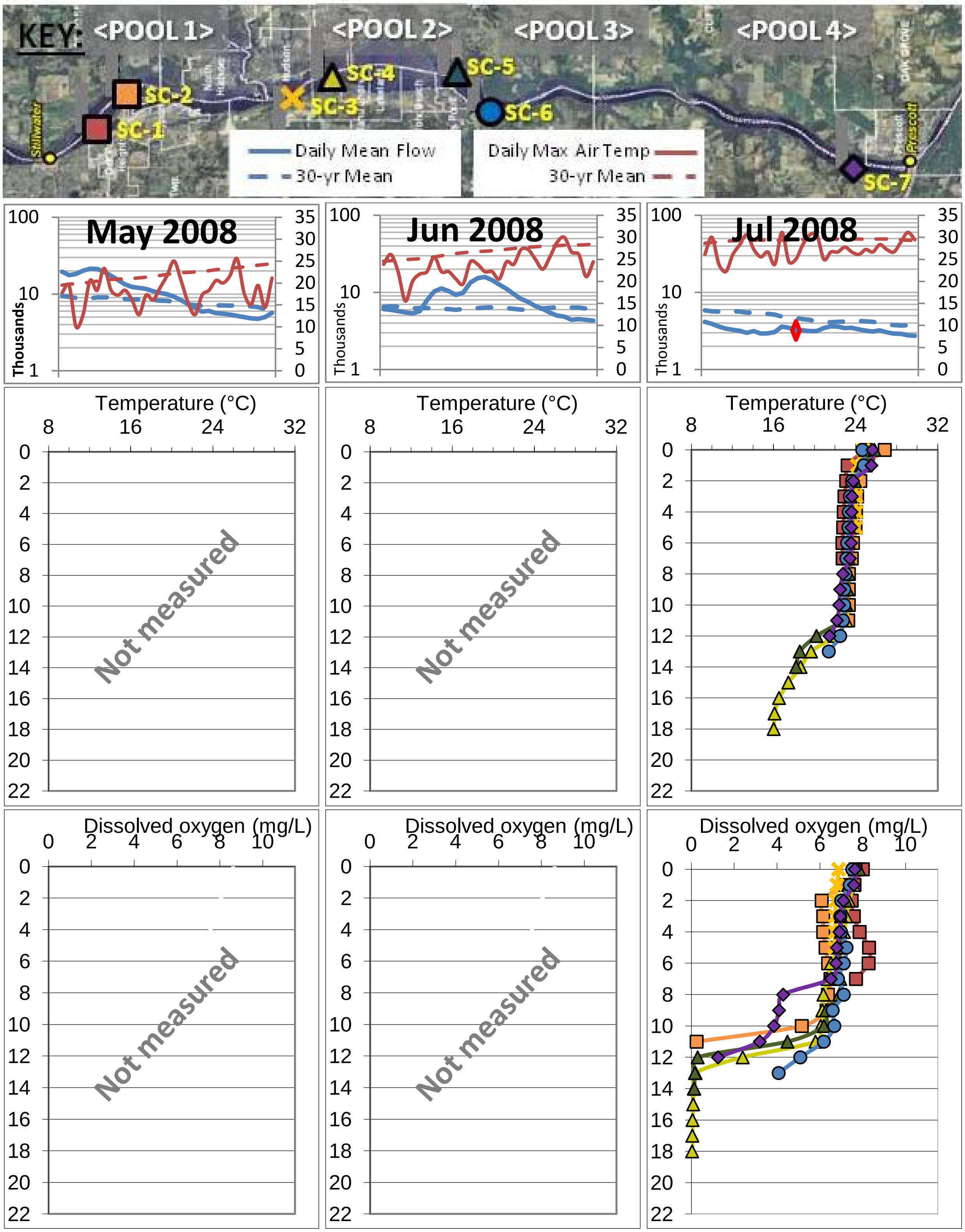


Figure 22a. Estimated daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, July 2008. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

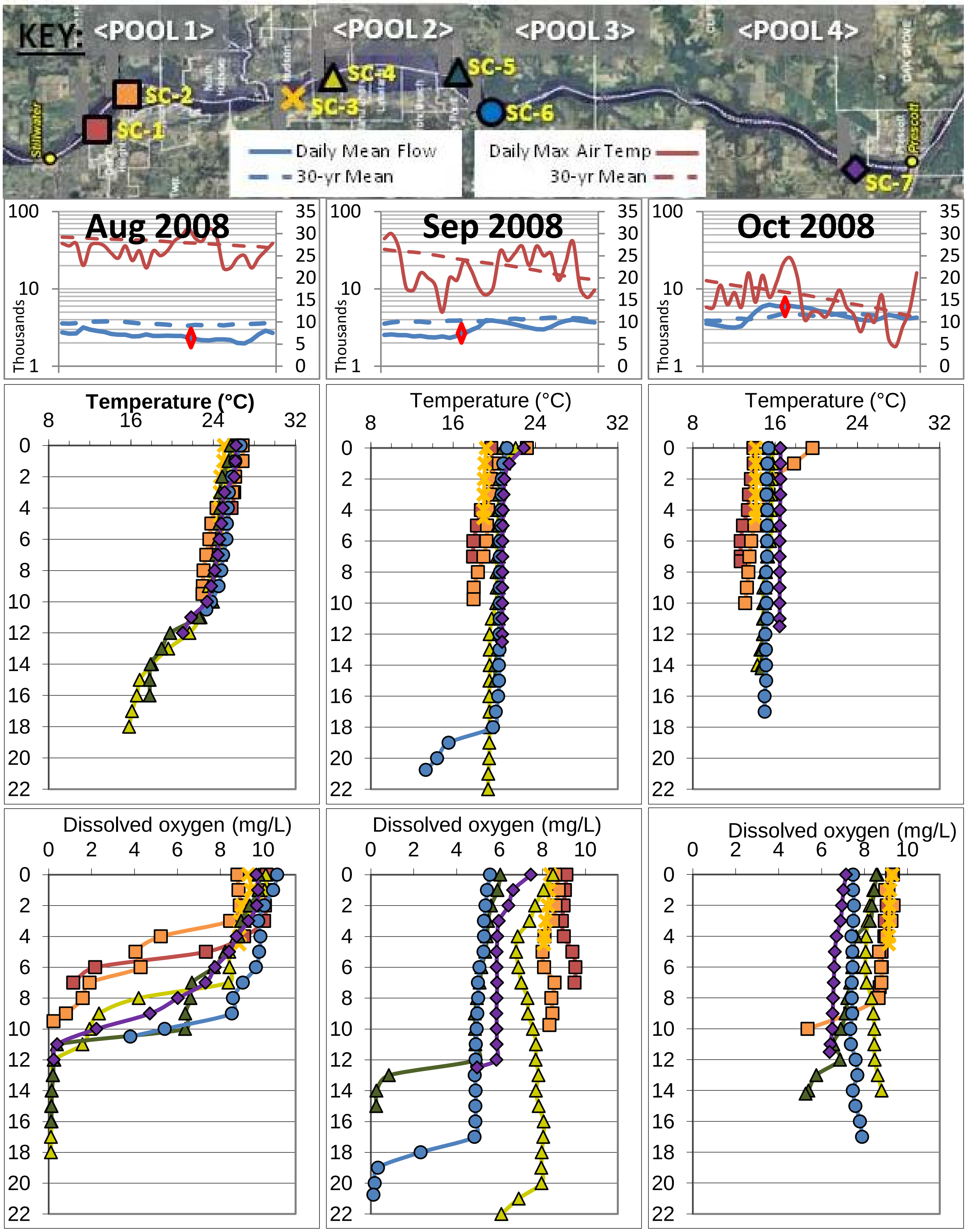


Figure 22b. Estimated daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, August-October 2008. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

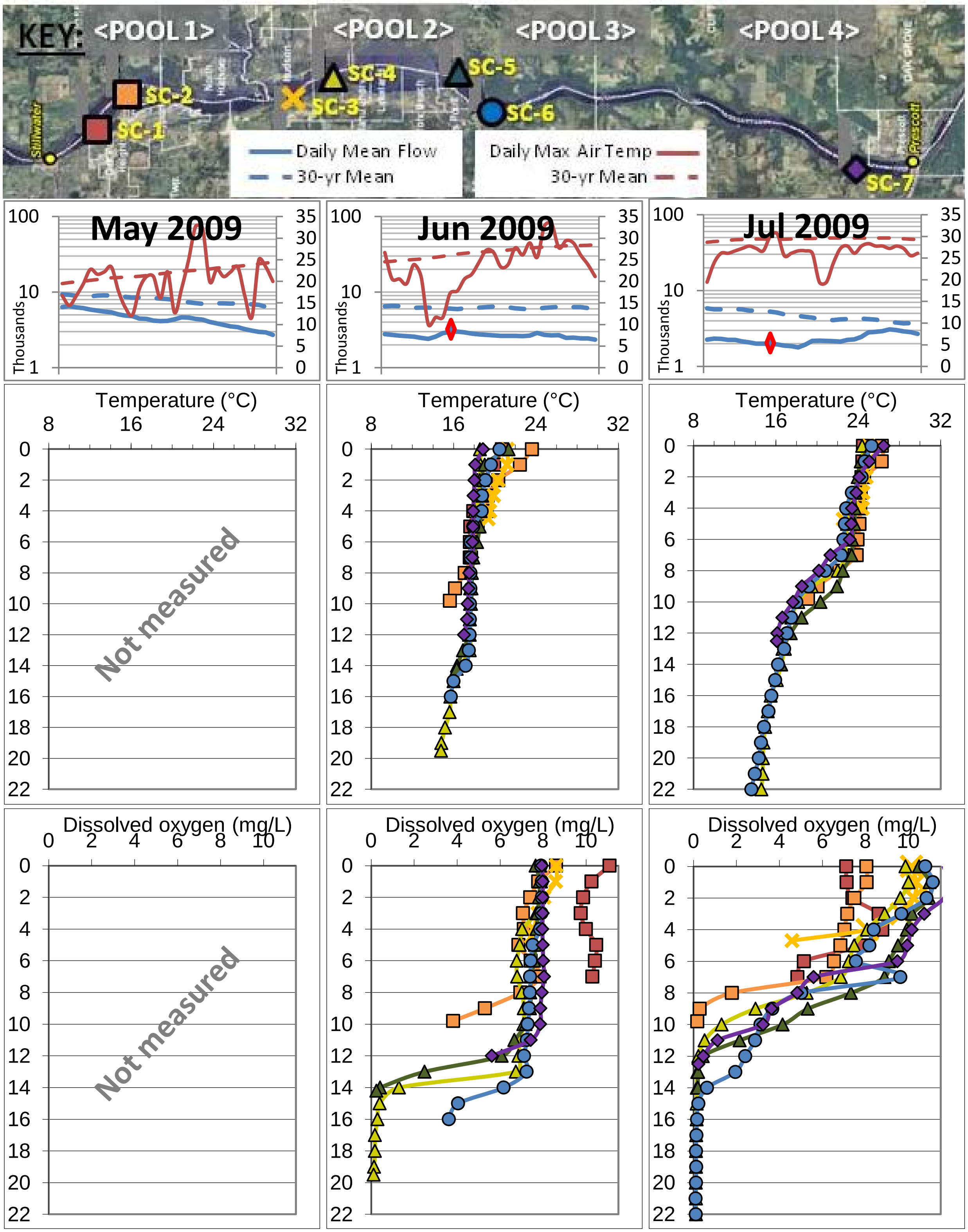


Figure 23a. Estimated daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, Jun-July 2009. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

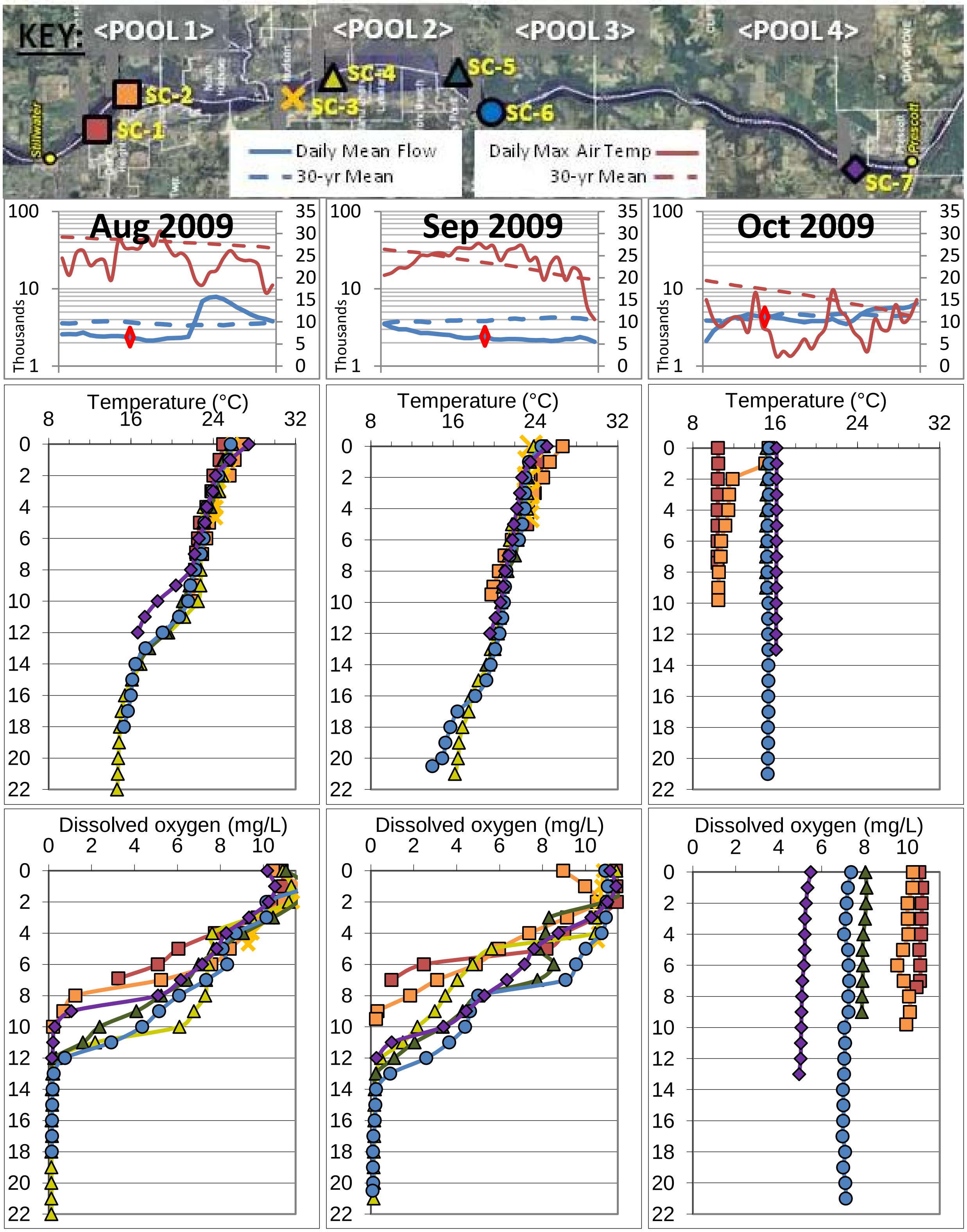


Figure 23b. Estimated daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, August-October 2009. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

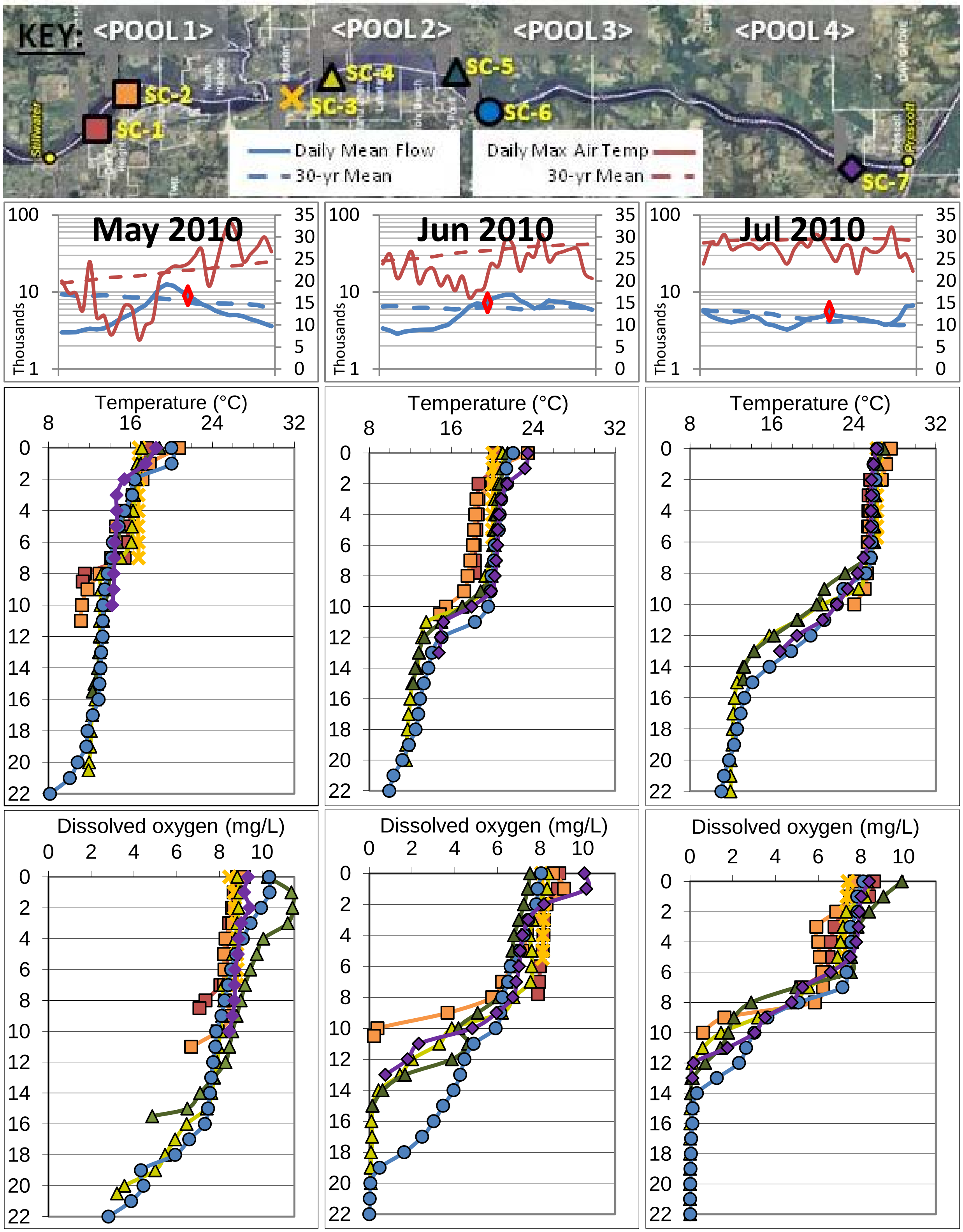


Figure 24a. Estimated daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, May-July 2010. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

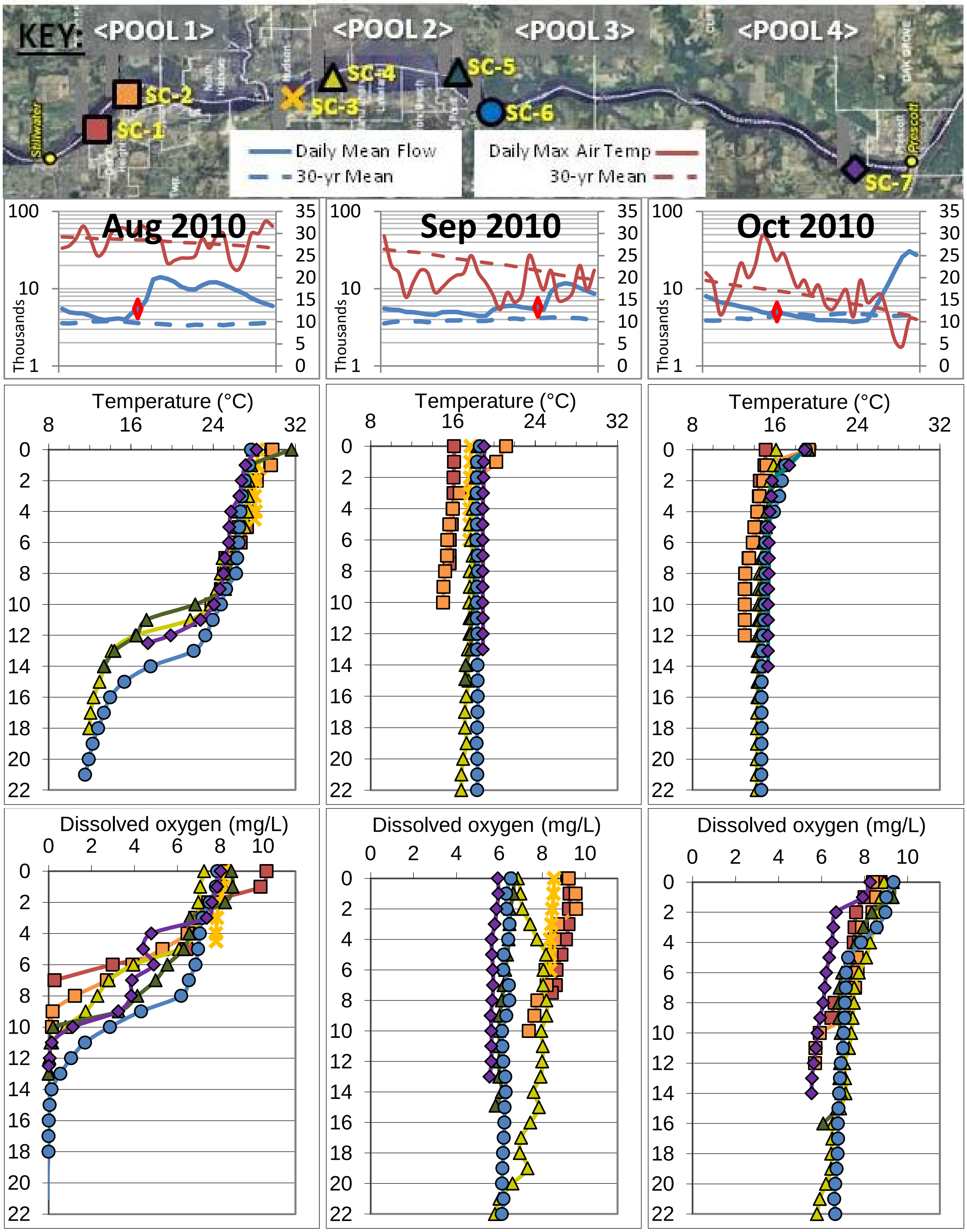


Figure 24b. Estimated daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, August-October 2010. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

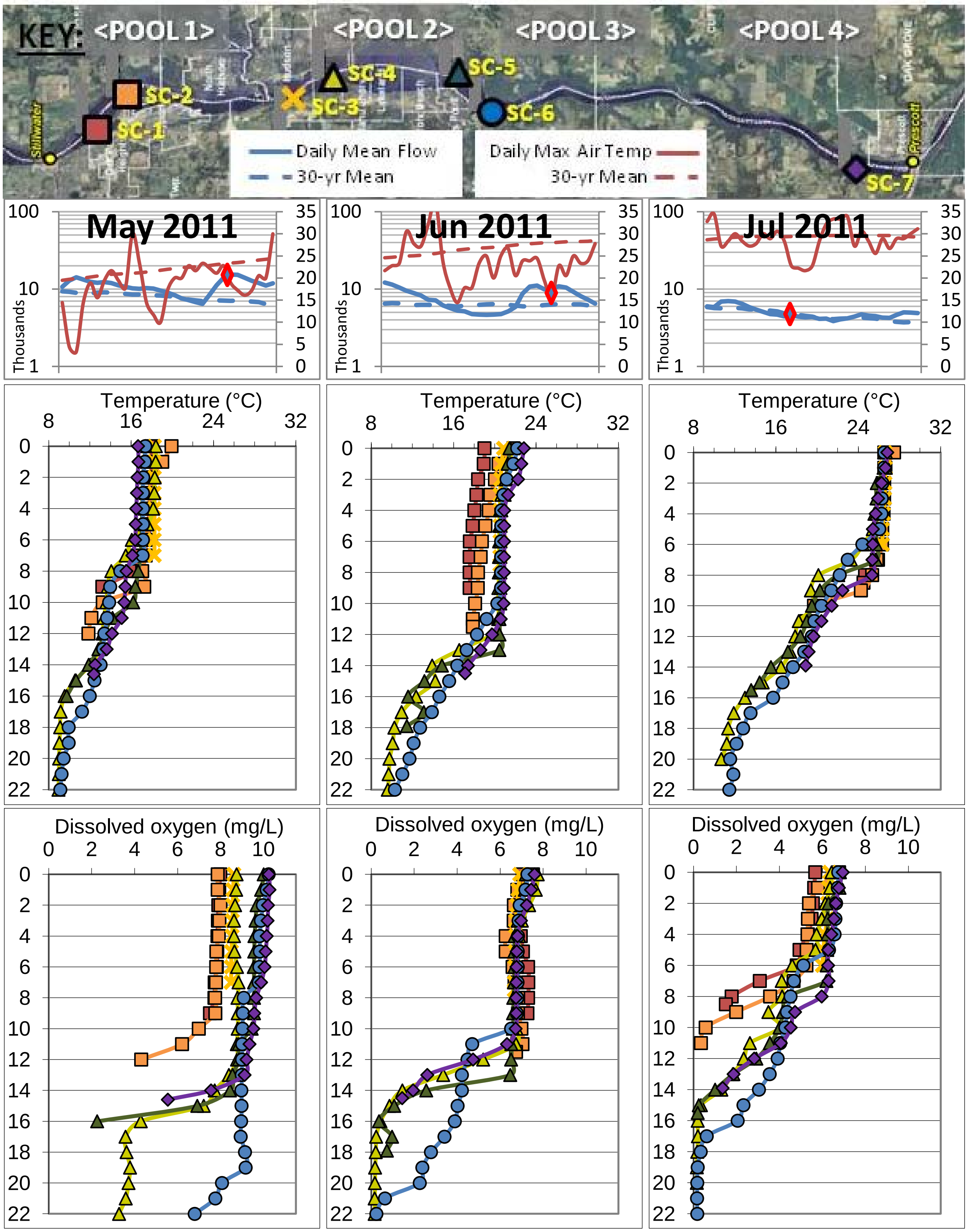


Figure 25a. Estimated daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, May-July 2011. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

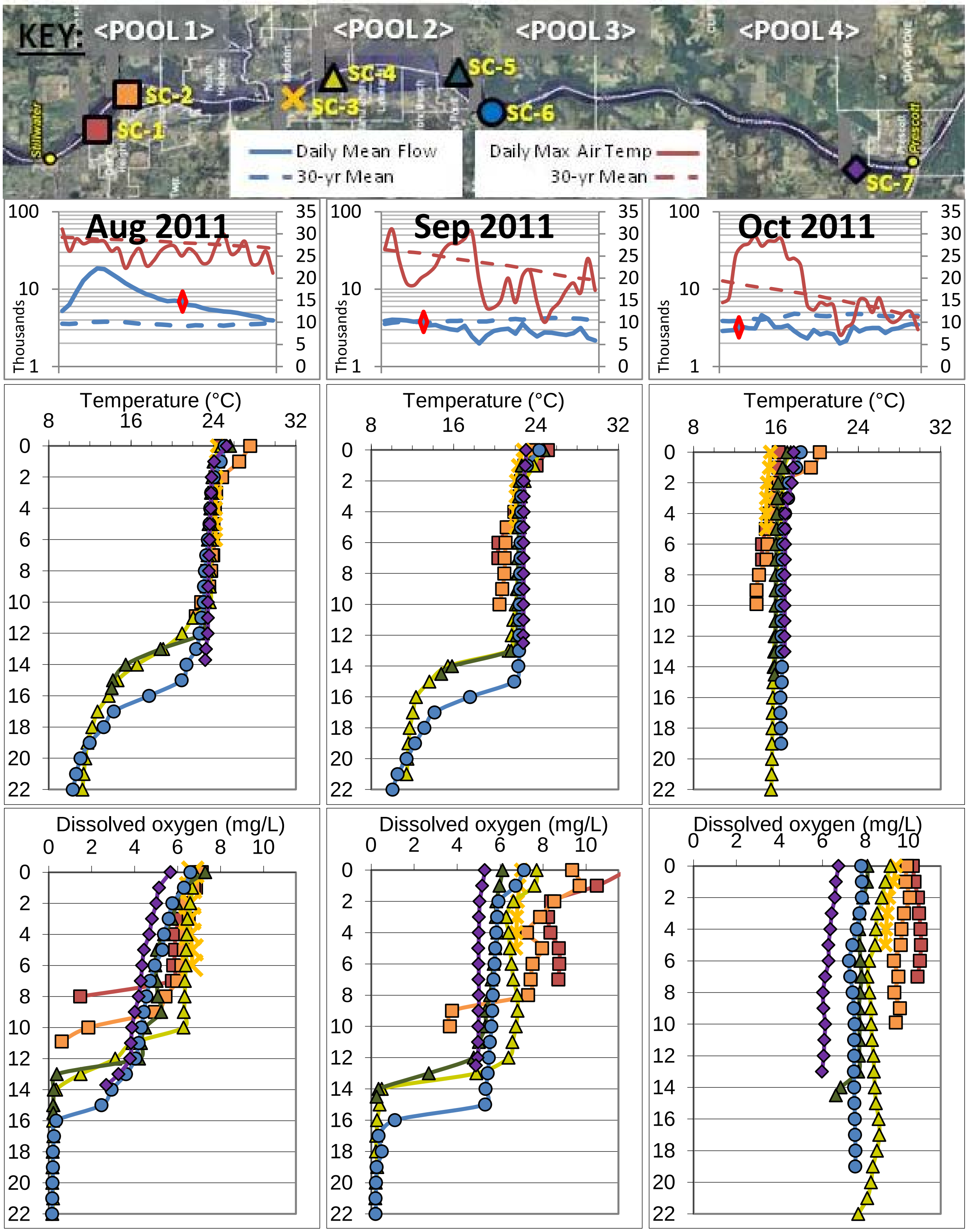


Figure 25b. Estimated daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, August-October 2011. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

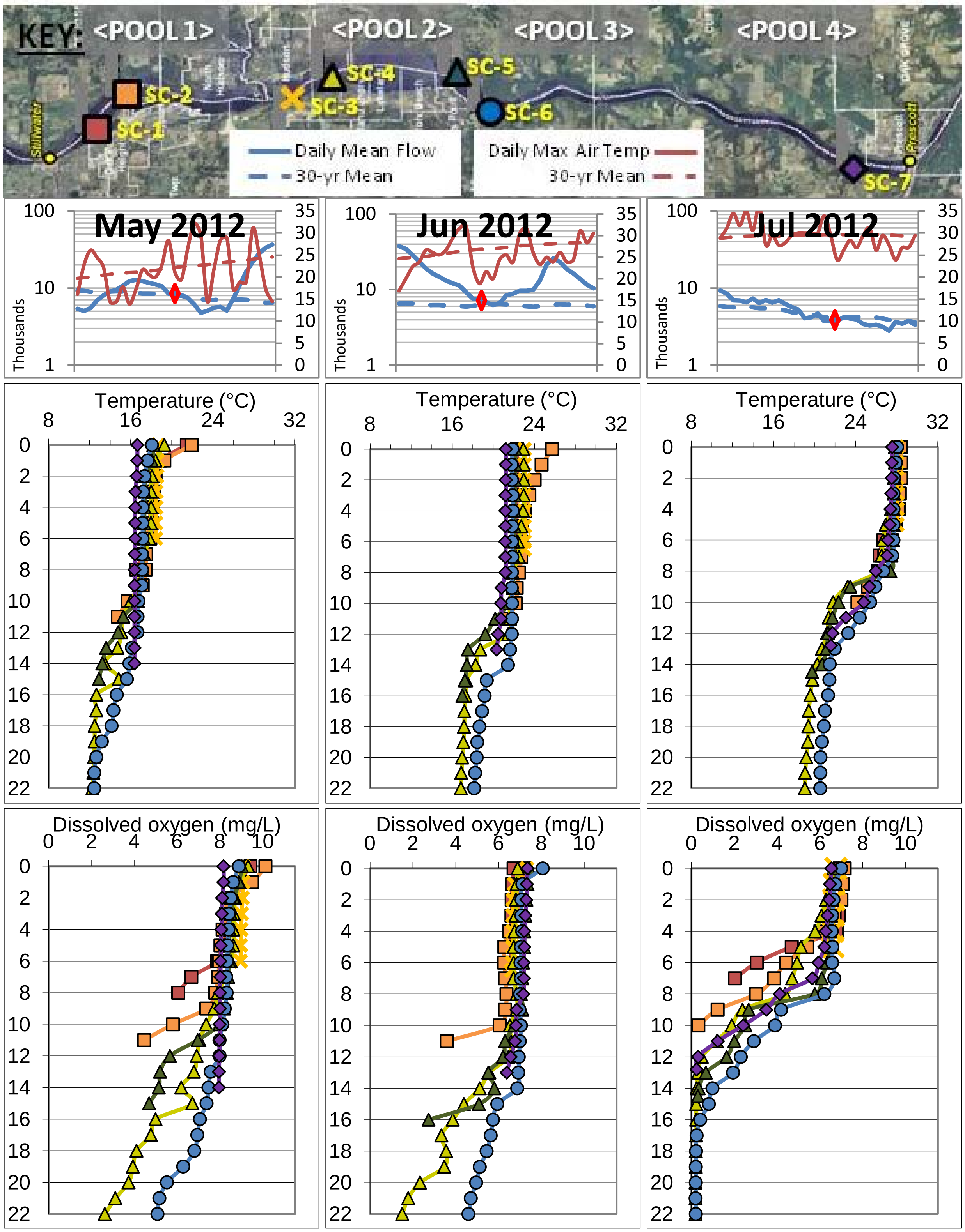


Figure 26a. Measured daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, May-July 2012. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

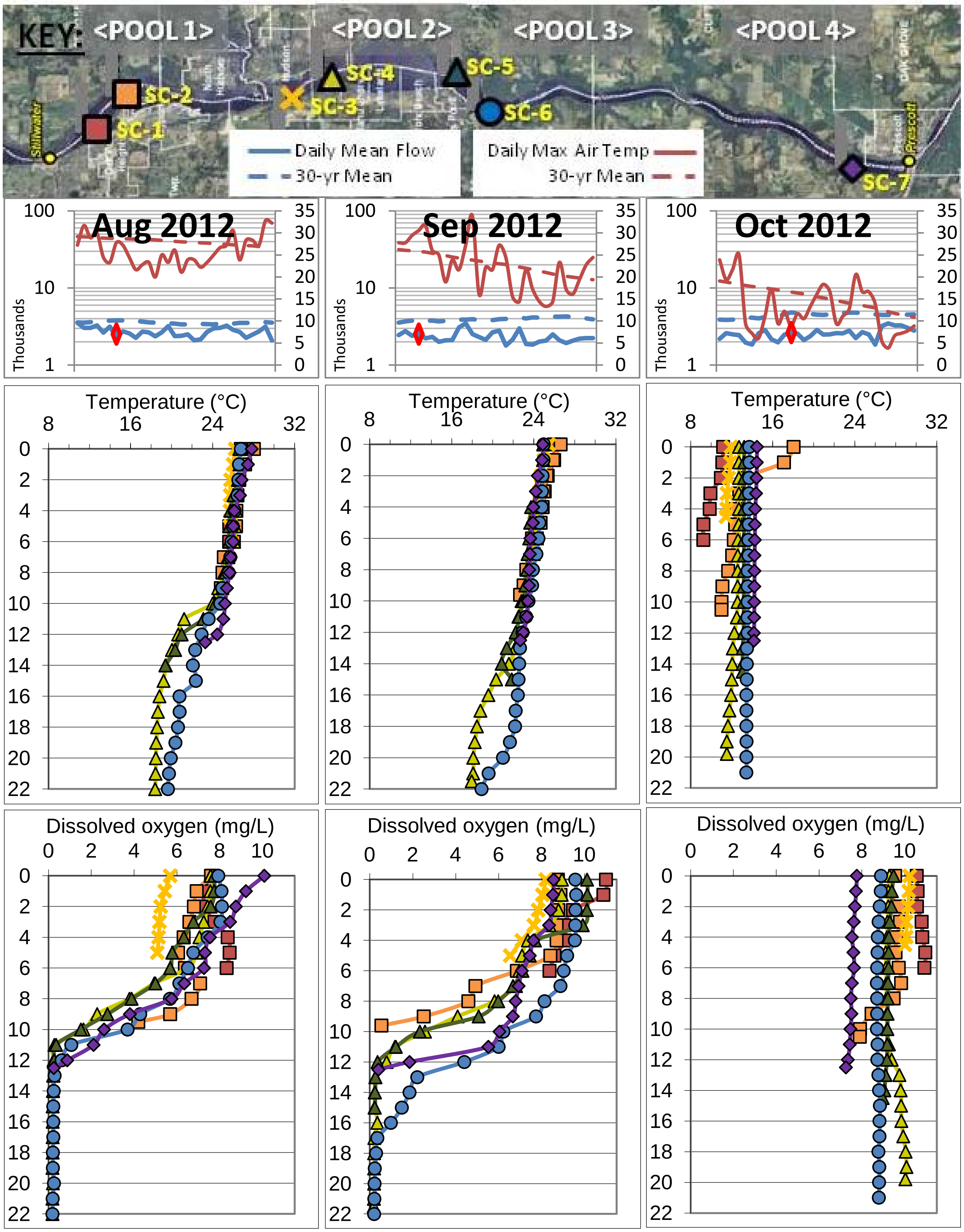


Figure 26b. Measured daily discharge (cfs) and maximum daily air temperature (°C), at Stillwater compared with 30-yr (1981-2010) mean values, and temperature (°C) and dissolved oxygen (mg/L) depth profiles at seven lake monitoring sites, August-October 2012. Diamond symbol on hydrograph indicates date of profile measurement. Refer to map for key to site symbols. Depth in meters.

Bi-Weekly Lake Samples of Clarity, Phosphorus and Chlorophyll

This report focused on the biweekly water quality variables that were assigned water quality standards during the Lake St. Croix TMDL process: Secchi depth clarity (SD), total phosphorus (TP), and viable chlorophyll-a (VChla). The standards are as follows: TP < 40 µg/L, VChl-a < 14 µg/L, and SD > 1.4 meters (MPCA and WDNR 2012). The annual mean discharge at Stillwater and the annual mean values of summer (Jun-Sep) biweekly sampling for the three water quality variables are shown in Figure 27, and compared to their respective water quality standards. Interestingly, our initial assumption of positive correlation between changes in TP and changes in chlorophyll-a is not evident in the annual means (Figure 27). If anything, TP and VChla would appear to be inversely correlated. Relative to 2008, when both variables met standards, TP decreases in 2009 yet VChla increases, TP increases again in 2010 yet VChla decreases, etc. In contrast, the summer mean chlorophyll-a and Secchi clarity appear to be more closely correlated, suggesting that light attenuation may be a driver of algal growth in the lake.

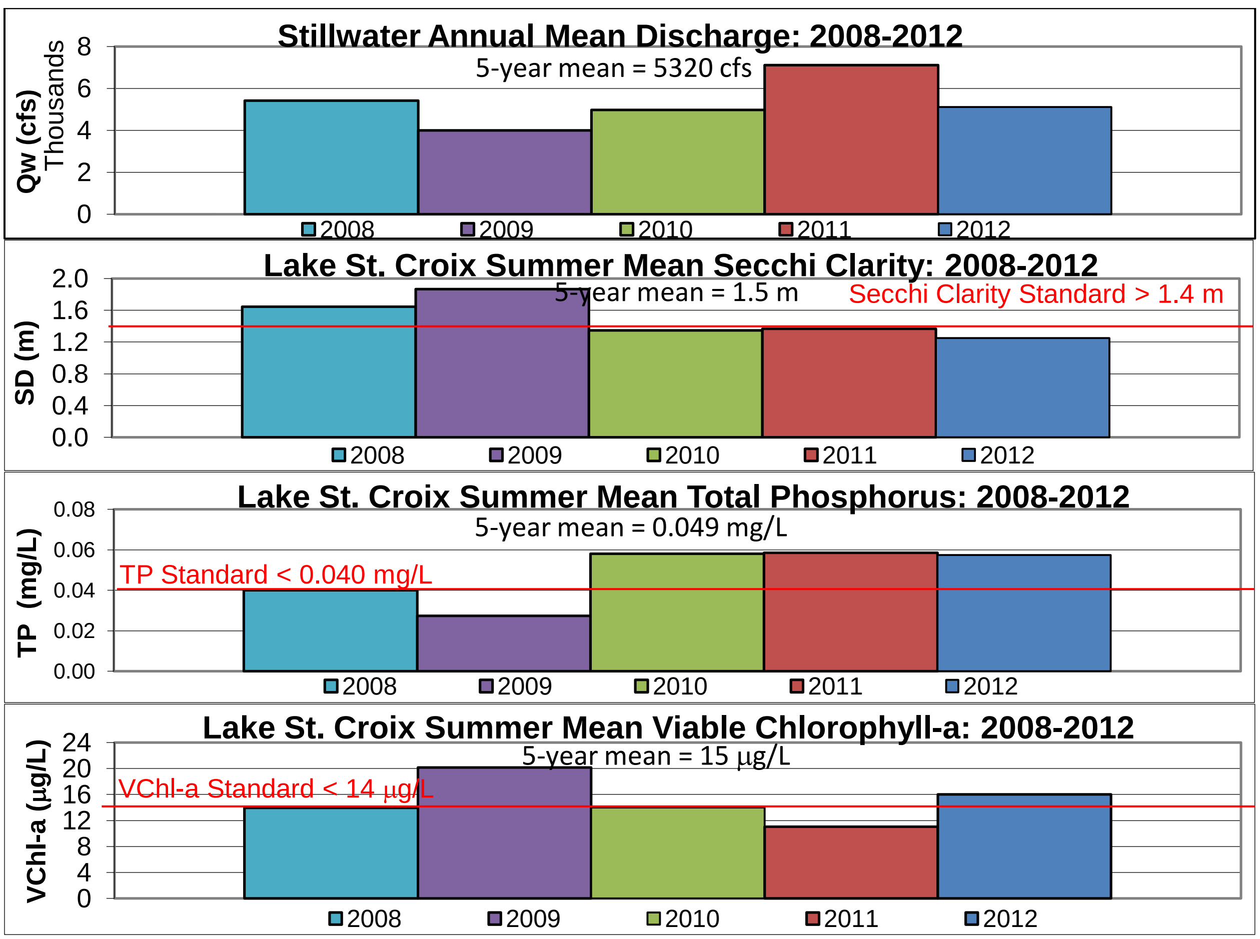


Figure 27. Annual mean of Stillwater discharge (Qw) and summer (Jun-Sep) mean of Lake St. Croix biweekly samples of total phosphorus (TP), viable chlorophyll-a (VChla), and Secchi depth clarity (SD), 2008-2012.

Sampling results at the seven MCES lake monitoring sites were compared to the standards. We calculated and plotted the percent exceedance of those standards (i.e., %SD<1.4, %TP>40, and %VChla>14) in all samples collected at each of the seven lake monitoring sites, each year 2008-2012 (Figure 28). In general, exceedances were lower in low-flow years and higher in high-flow years. During the low-flow period of 2008-2009, exceedances of the clarity and phosphorus standards were mostly less than 40%, with the exception of site SC-1 in 2009. In contrast, chlorophyll exceedances were over 40% at most sites. With the onset of high flow conditions in 2010, exceedances of the clarity standard averaged about 60% with a downstream trend of decreasing exceedances from SC-1 to SC-7. A similar downstream decreasing trend was observed for TP in high-flow conditions. In 2011, all sites exceeded the TP standard for at least 80% of the samples, and in 2012 nearly 80% of the samples. In contrast, chlorophyll was exceeded at most of the sites for less than 60% of the samples during 2010-2012. The two sites that exceeded the chlorophyll standard in more than 60% of the samples were SC-4 and SC-6, the deep-pool sites in Troy Beach Pool 2 and Black Bass Pool 3.

To assess spatial variability, the mean annual (Oct-Sep) discharge and mean summer (Jun-Sep) values of Secchi depth clarity (SD), total phosphorus (TP) concentration, and viable chlorophyll-a (VChla) concentration were plotted for each year (2008-2012) at each of the seven MCES lake monitoring sites, and compared with their respective water quality standards (Figure 29). Mean annual discharge at each of the seven sites was estimated by adding the intervening tributary flows to Stillwater flows. Compared to the 5-year average discharge at Stillwater (5320 cfs), Willow River averaged 178 cfs, Valley Creek averaged 14 cfs, and Kinnickinnic River averaged 104 cfs. These small downstream additions in discharge were minor compared to the interannual variability of St. Croix River discharge due to variability in climate variables like temperature and precipitation.

In contrast to discharge, the water quality variables more clearly demonstrated spatial variability in Lake St. Croix. Secchi depth clarity showed a downstream trend of increasing clarity for all years, from a study-period mean of 1.2 m at SC-1 to a study-period mean of 1.8 m at SC-7, perhaps due to particulates settling out of suspension toward the downstream end of the lake. Total phosphorus showed a downstream trend of decreasing concentrations for most years except 2008, perhaps for the same reason. Chlorophyll-a showed a downstream trend of decreasing concentrations through Pool 1(from SC-1 to SC-3), then increased again in Pool 2 (at SC-4), before continuing its downstream trend of decreasing chlorophyll concentrations. It is possible that the long shallow areas at the upstream ends of Pool 1 and Pool 2 encourage abundant algal growth. Another possibility is allochthonous, or external, sources of chlorophyll delivered by the mainstem St. Croix River to Pool 1, and by the Willow River to Pool 2. For SD, the within-site variability and between-site variability are very similar. However, for TP and VChla, the within-site variability is larger than the between-site variability. Interannual variability exceeded spatial variability for these two measures of water quality, pointing to flow as a driving mechanism in both phosphorus delivery and algal response.

During the low-flow period of 2008-2009 and the high-flow period of 2010-2011 in Figure 29, TP and chlorophyll appear to be inversely related at any given site, like Figure 27. However, this inverse relationship does not apply to the spatial patterns of the two variables. For both variables, the 5-year site means area highest at SC-1and decreasing downstream. The highest chlorophyll concentrations are where the lowest clarity measurements occur, indicating algal growth is not limited by light penetration alone.

To assess seasonal variability, we plotted the monthly mean discharge at Stillwater and the monthly mean values of SD, TP, and VChla, for May-October during 2008-2012 (Figure 30). The monthly mean discharges during the study period deviate from the 30-year mean (Figure 20), in that discharges in July should average closer to 5000 cfs and August discharges should be the lowest of the summer, lower than September discharges. Instead, discharges measured in July during the study period were lower than the long-term average, and discharges measured in August were higher than the long-term average.

The monthly means of the three water quality variables were plotted only when sufficient data was available (at least 20 samples each month, averaged across the sites). The month of October during 2008-2010 was not sampled sufficiently, and the 5-year mean for that month was not calculated (Figure 30). The 5-year mean Secchi depth clarity peaks in June/July, when discharge is dropping after the peak discharges in April/May, but before the low-flow peak algal growth period of August/September. However, which month in any given year experienced peak clarity was dependent on the flow conditions of that particular year: the low-flow months of each year tended to have the highest clarity of that year, while the high-flow months of each year tended to have the lowest clarity of that year. Thus, discharge and clarity seem to maintain throughout the summer months the inverse proportionality observed in the summer means (Figure 27).

Total phosphorus concentration seems to exhibit little seasonality. There is little difference between the 5-year monthly means in Figure 30; they all cluster close to the 5-year summer mean of 0.049 mg/L. However, in any given year, the proportional relationship between discharge and TP from Figure 27 is maintained through the summer months of each year, such that the between-year variability is higher than the between-month variability. Mean monthly phosphorus is controlled not so much by seasonality or monthly discharge, but by the overall discharge and climate conditions of that year. The monthly means of viable chlorophyll-a shows a clear seasonality, with concentrations increasing through the summer months to peak values in September, and though October data is sparse, from the October 2011 and October 2012 monthly mean concentrations it appears that chlorophyll likely decreases after autumn de-stratification and turnover.

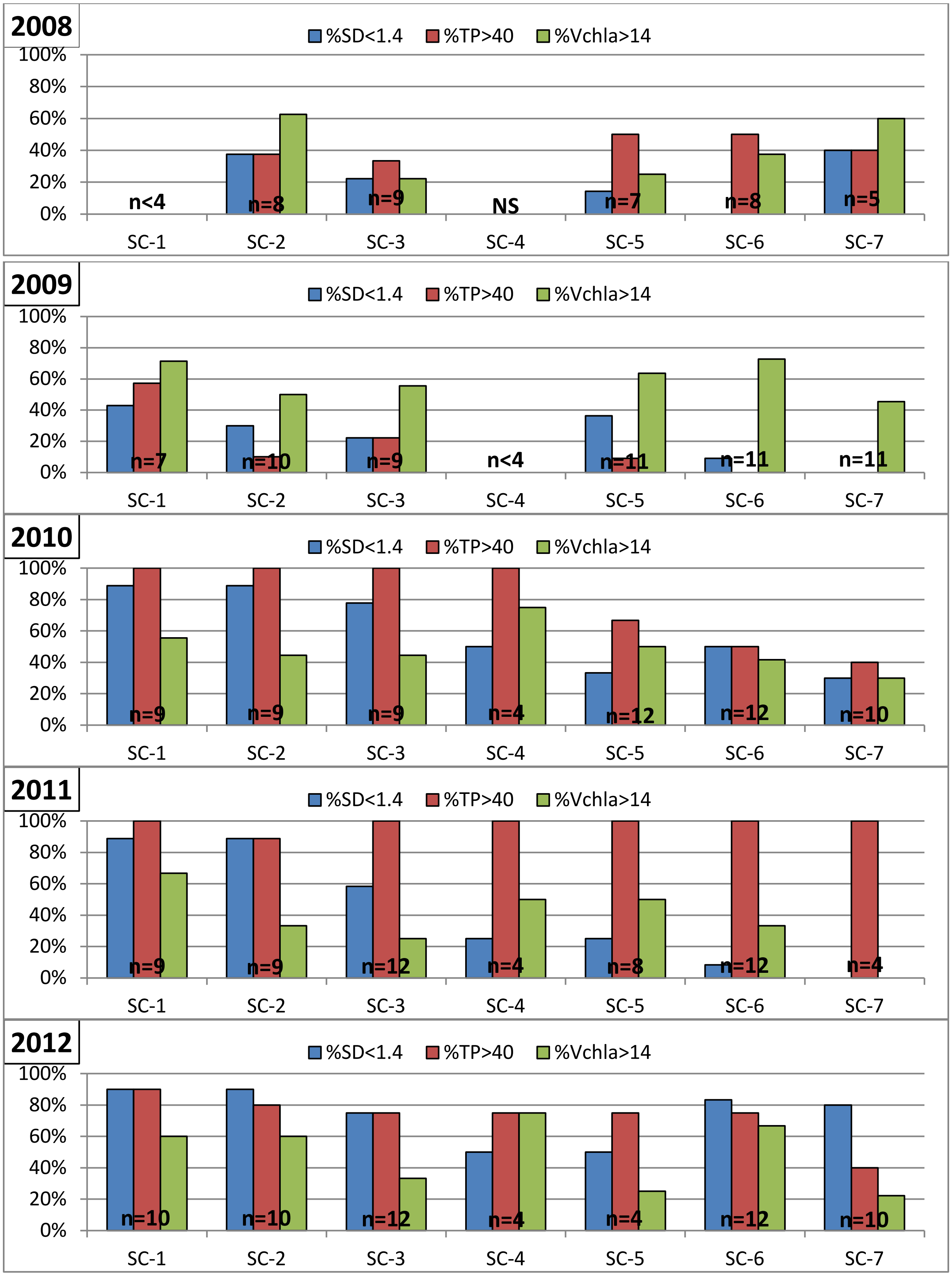


Figure 28. Percent of summer (Jun-Sep) samples that exceed water quality standards for secchi depth clarity (SD), total phosphorus (TP), and viable chlorophyll-a (Vchla) at seven MCES lake monitoring sites, 2008-2012. NS = not sampled, and n<4 indicates where a minimum of monthly sampling was not achieved.

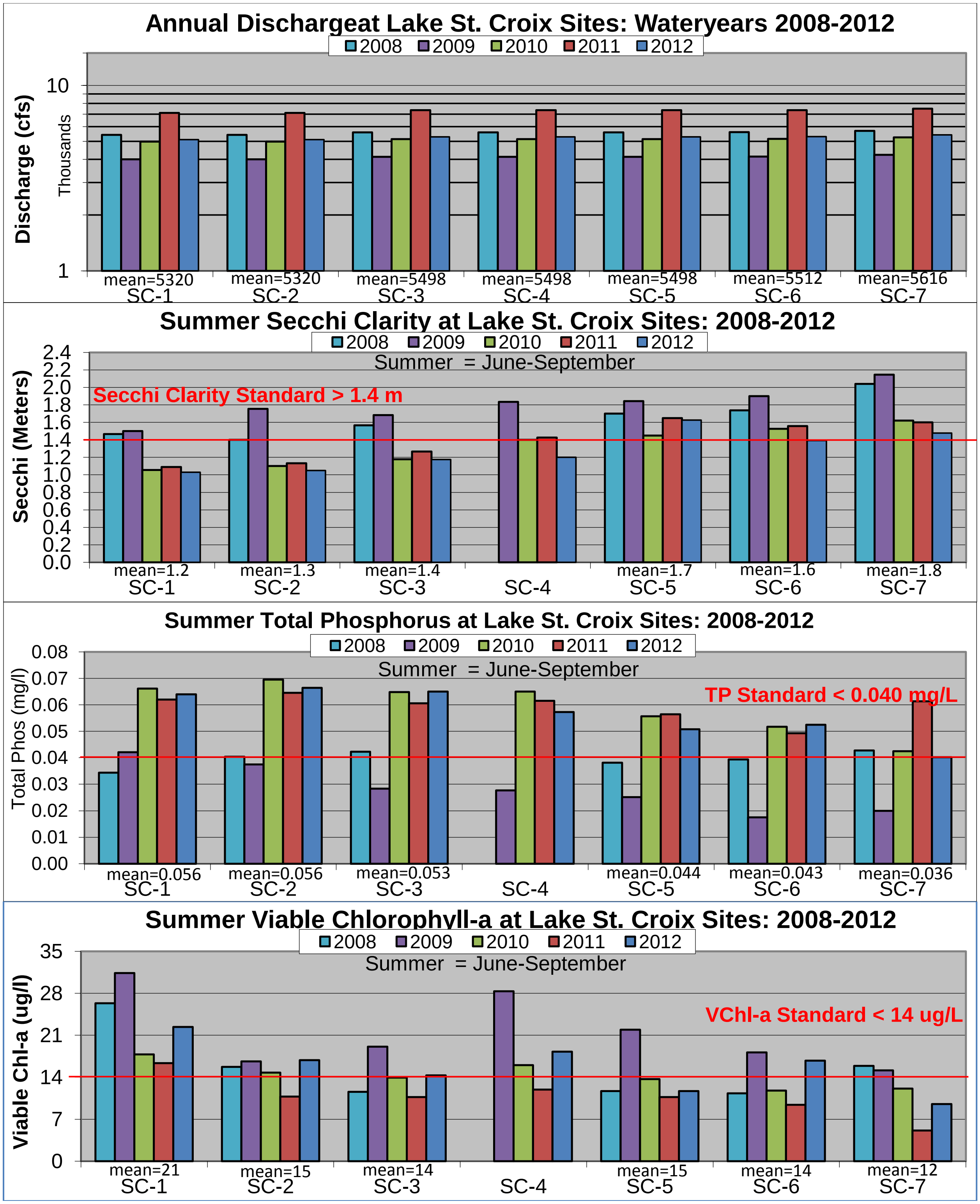


Figure 29. Mean annual (Oct-Sep) discharge and mean summer (Jun-Sep) Secchi depth, total phosphorus concentration, and viable chlorophyll-a concentration from biweekly sampling at seven MCES lake monitoring sites, 2008-2012. Site SC-4 was not monitored in 2008. The 5-year summer mean for the other six sites is listed above their site labels.

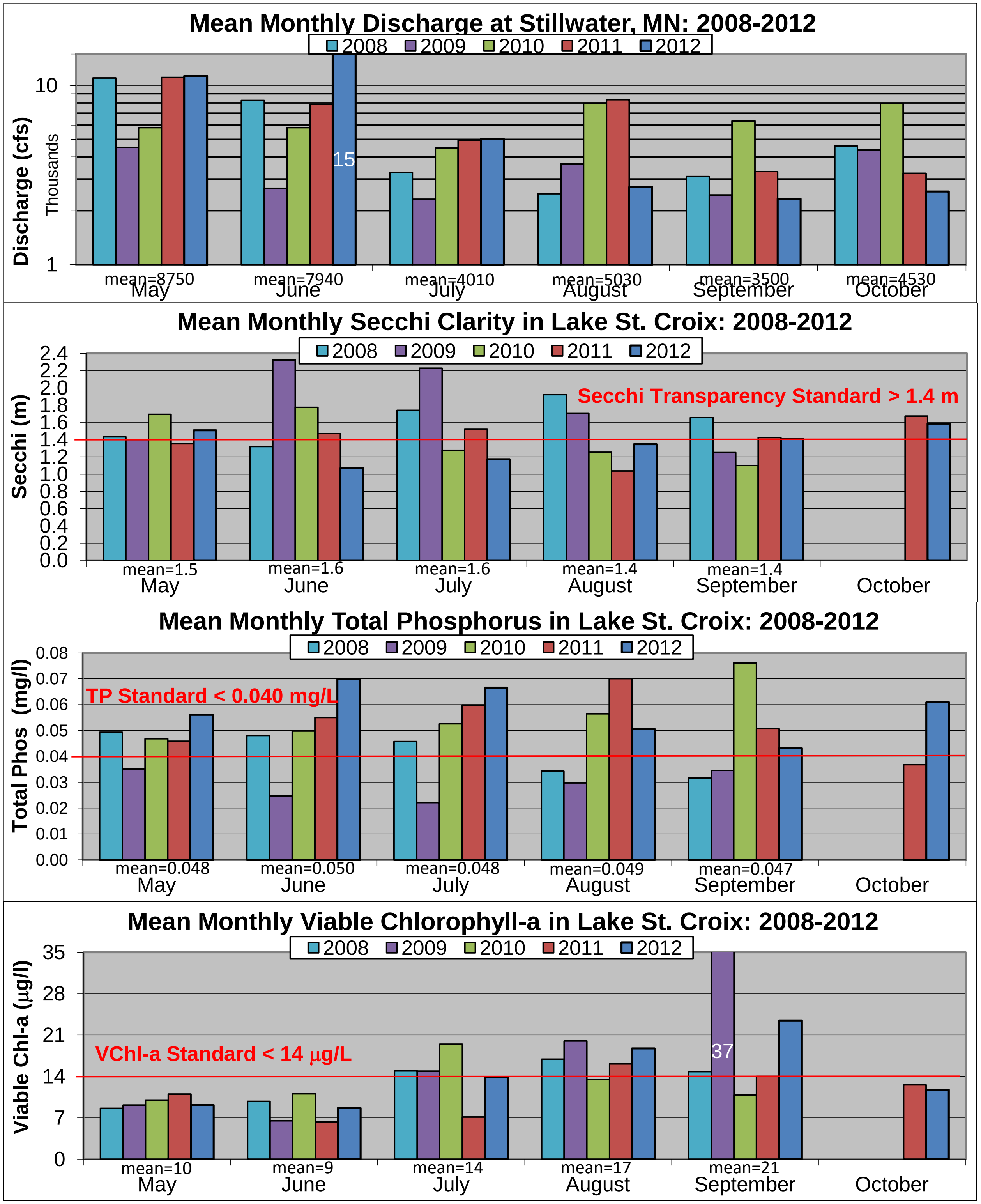


Figure 30. Mean monthly discharge at Stillwater, and Secchi depth, total phosphorus concentration, and viable chlorophyll-a concentration from biweekly sampling averaged across all MCES lake monitoring sites, 2008-2012. Too few samples were collect in October 2008-2010. The 5-year mean for the other months is listed above the month labels.

Monthly Lake Samples of Phytoplankton

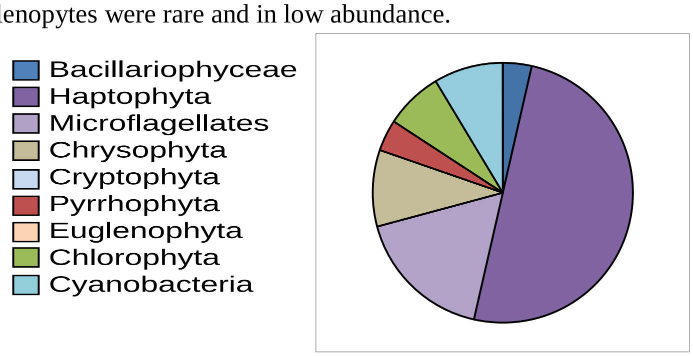
The monthly phytoplankton samples collected at the seven volunteer monitoring sites were summarized into the following nine major algal groups (Wehr and Sheath 2003):

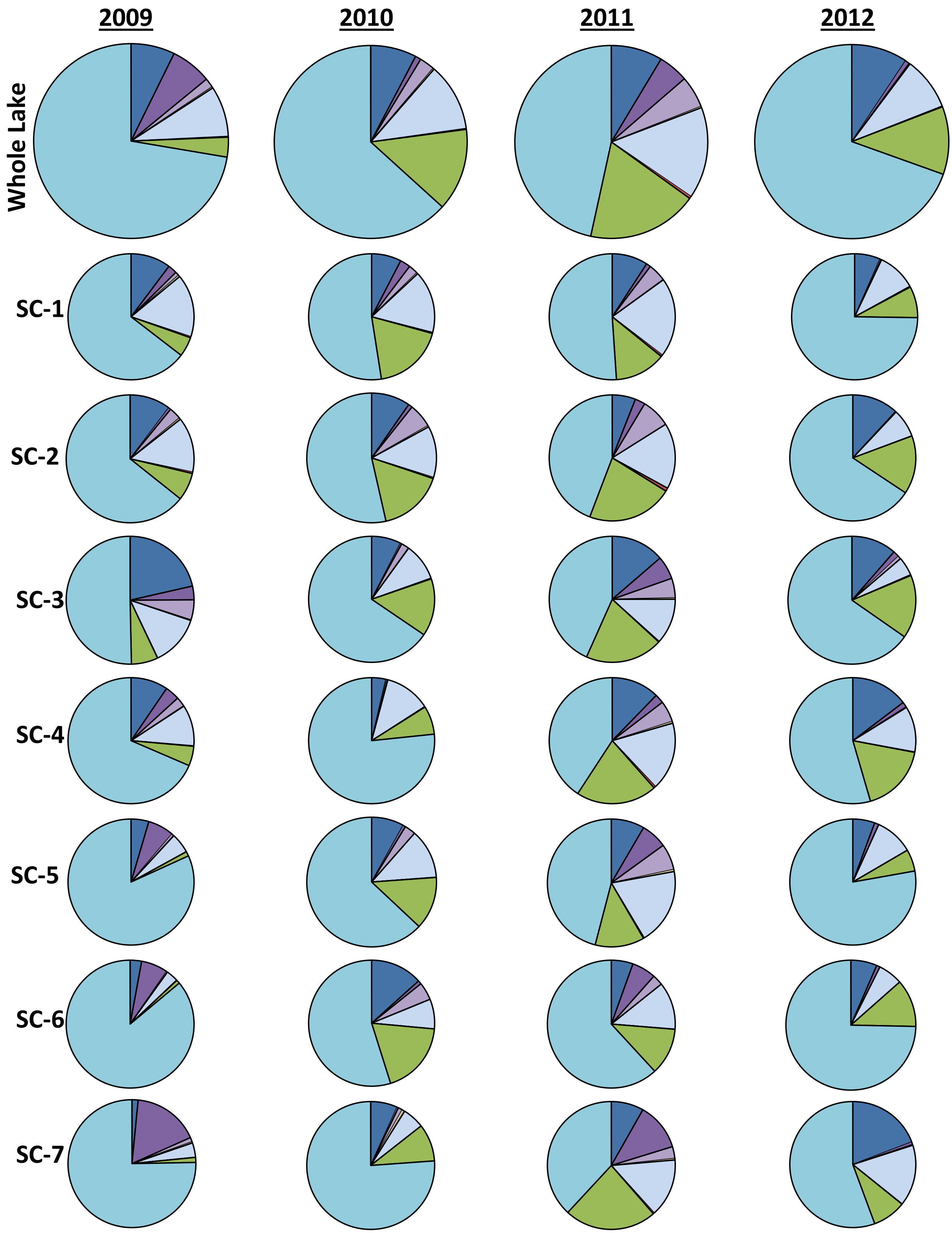
Bacillariophyceae – siliceous diatoms, tend to bloom in spring
Haptophyta – flag-like organ, some are toxic
Microflagellates – flagellates < 3μm
Chrysophyta – golden brown algae
Cryptophyta – cold-loving, dominant in early spring and late autumn, selectively grazed by zooplankton
Pyrrhophyta – red algae, cause for most taste and odor problems
Euglenophyta – considered an indicator of organic pollution
Chlorophyta – green algae
Cyanobacteria – blue-green algae, can be the source of toxins microcystins and anatoxin-a

For comparison with the summer period during which water quality standards are applicable, algal groups observed in summer (June-September) at each of the seven sites and for the the whole lake were summarized in pie diagrams for 2009-2012 (Figure 31). No phytoplankton samples were collected in June 2008, so the 2008 results were omitted from Figure 31 as a partial summer record. Among the algal groups, blue-green algae (BGA) dominated: BGA accounted for about 75% of the total abundance in a low-flow year (e.g., 2009) and 50% of the total abundance in a high-flow year (e.g., 2011). To assess overall spatial variability, we plotted the summer (June-September) percent abundances at the seven MCES lake monitoring sites during 2009-2012. Although BGA dominated at all sites, the lake-like lower pools showed abundances of haptophyta (flagellates) in low-flow conditions, and abundances of haptophyta, cryptophyta, and chlorophyta (green algae) in high-flow conditions.

Algal group abundances are plotted for each year 2008-2012 (Figures 32-36). Like chlorophyll concentrations, maximum total abundance observed in any given year depended on that year's flow conditions: the highest total abundances were observed in low-flow years (e.g., 2008 and 2009). Low-flow conditions were conducive to maximum algal growth, while high-flow conditions reduced water residence time in the pools so as to limit algal growth.

Figure 31. (folowing page) Algal group proportions (% abundance) for whole lake sampling and at seven lake monitoring sites, summers (Jun-Sep) 2009-2012. 2008 was omitted as a partial summer. Key to algal groups is below. Euglenopytes were rare and in low abundance.




2009
2010
2011
2012
Whole Lake
SC-1
SC-2
SC-3
SC-4
SC-5
SC-6
SC-7

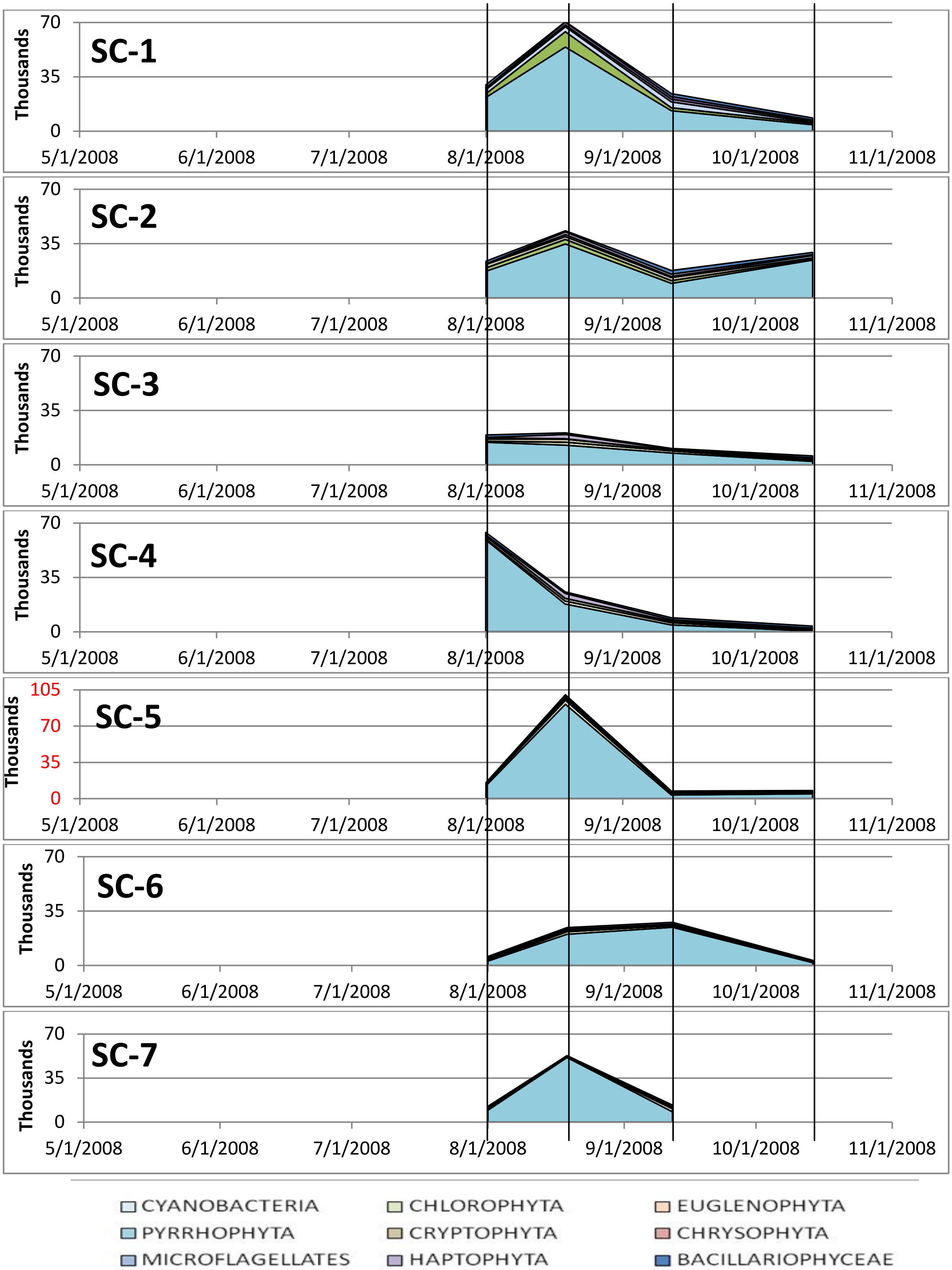


Figure 32. Algal group abundance (cells/mL) at seven MCES lake monitoring sites, August-October 2008. Note the difference in scale at SC-5.

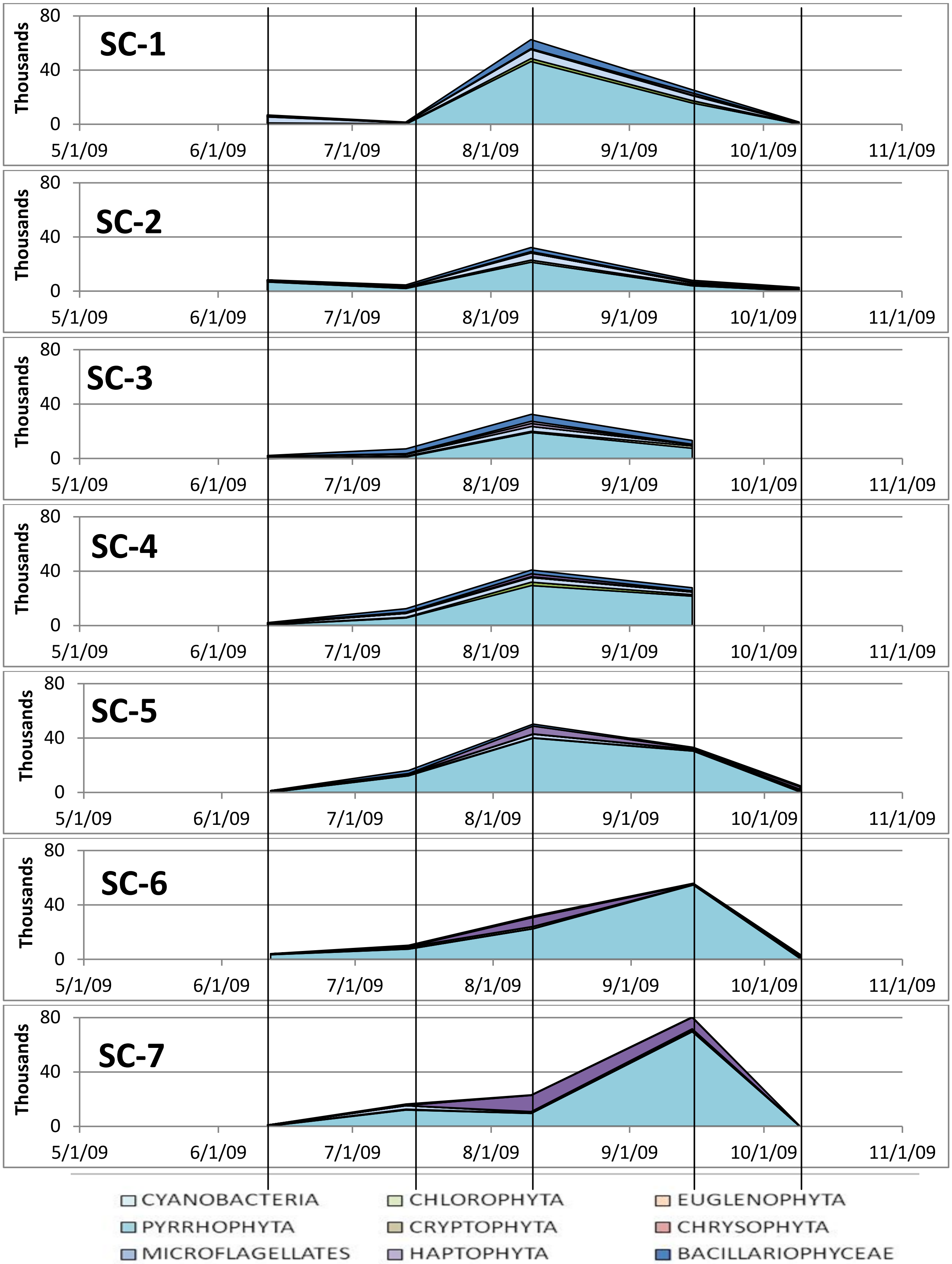


Figure 33. Algal group abundance (cells/mL) at seven MCES lake monitoring sites, June-October 2009.

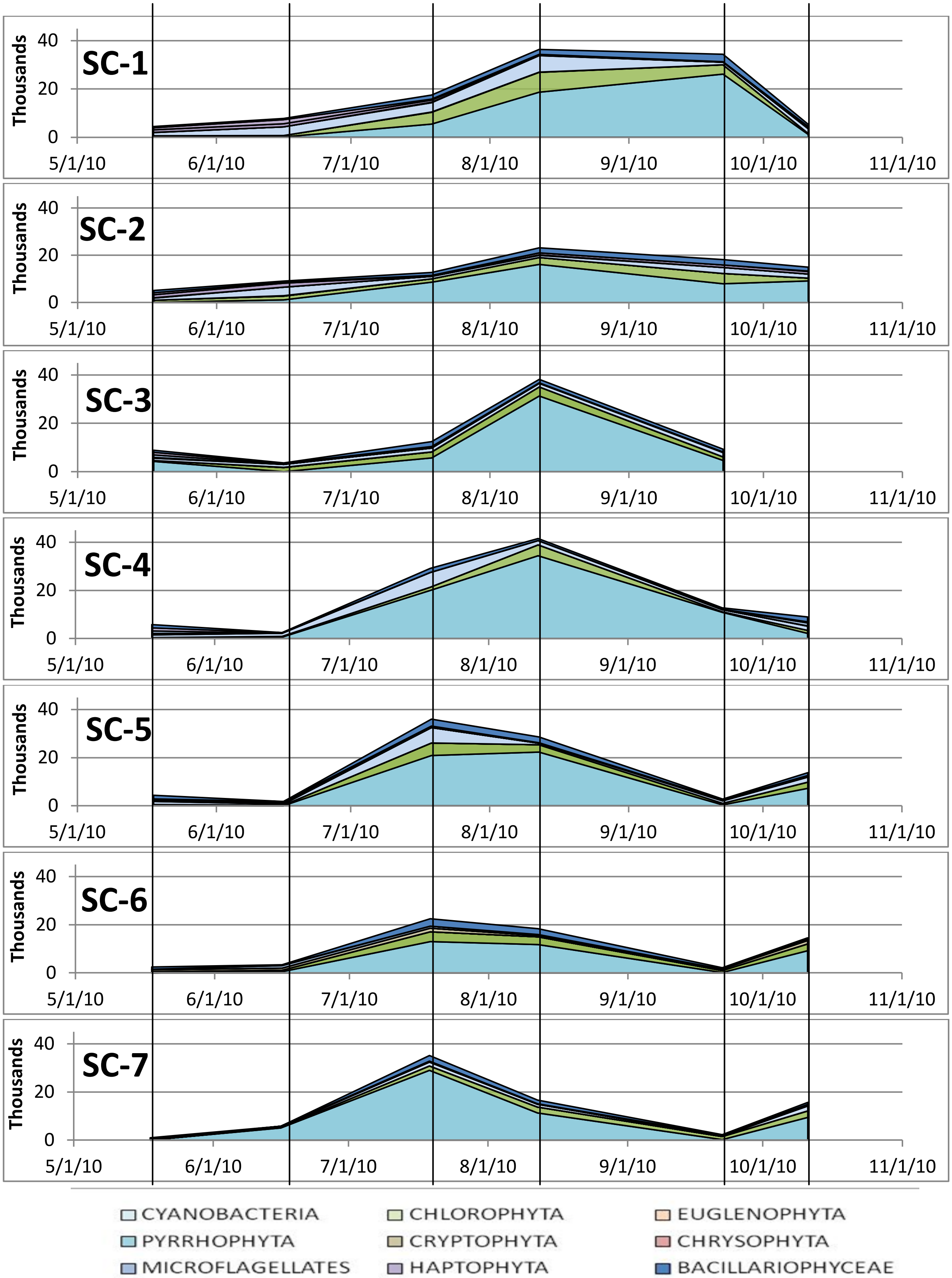


Figure 34. Algal group abundance (cells/mL) at seven MCES lake monitoring sites, May-October 2010.

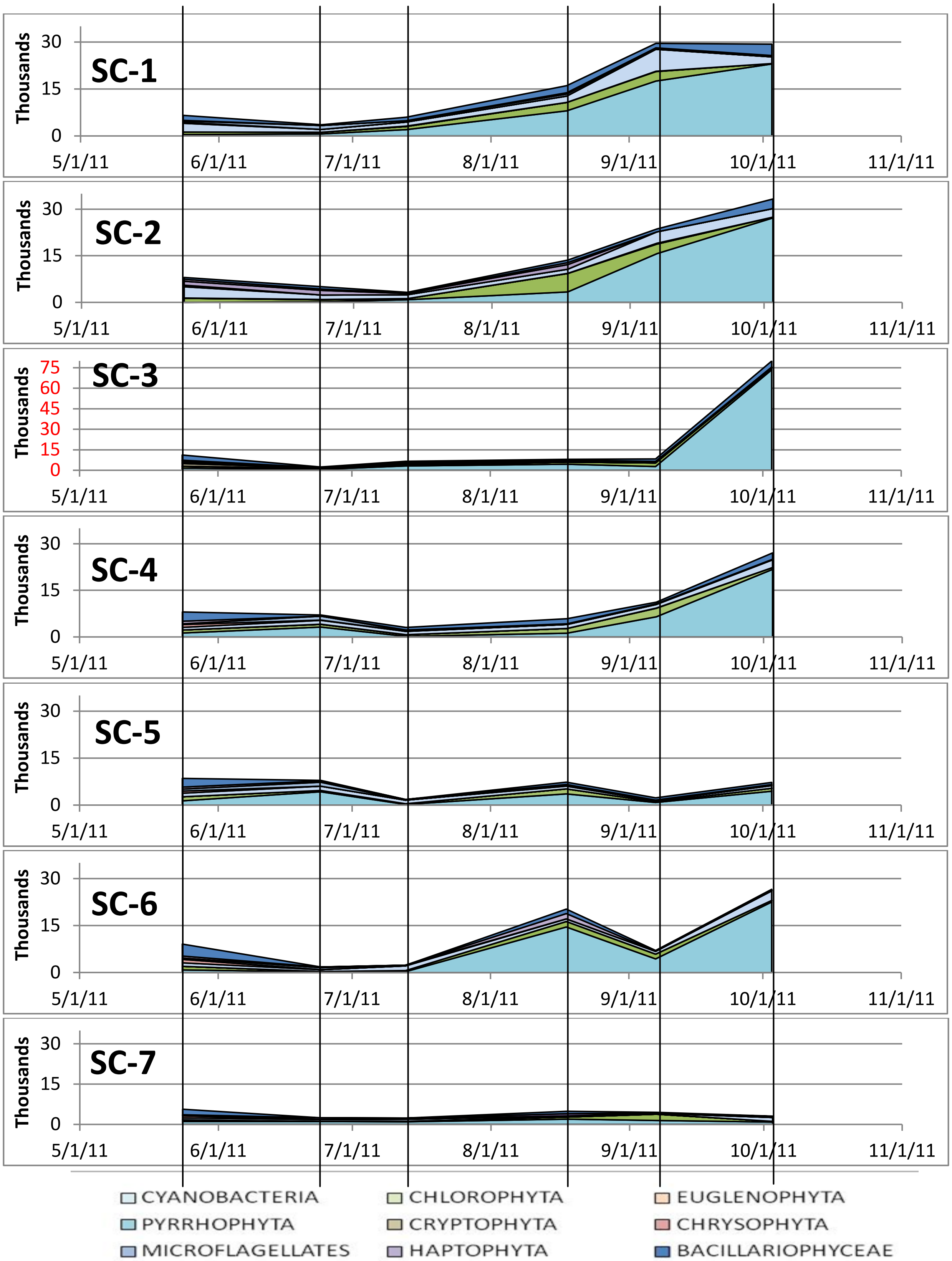


Figure 35. Algal group abundance (cells/mL) at seven MCES lake monitoring sites, May-October 2011. Note the difference in scale at SC-3.

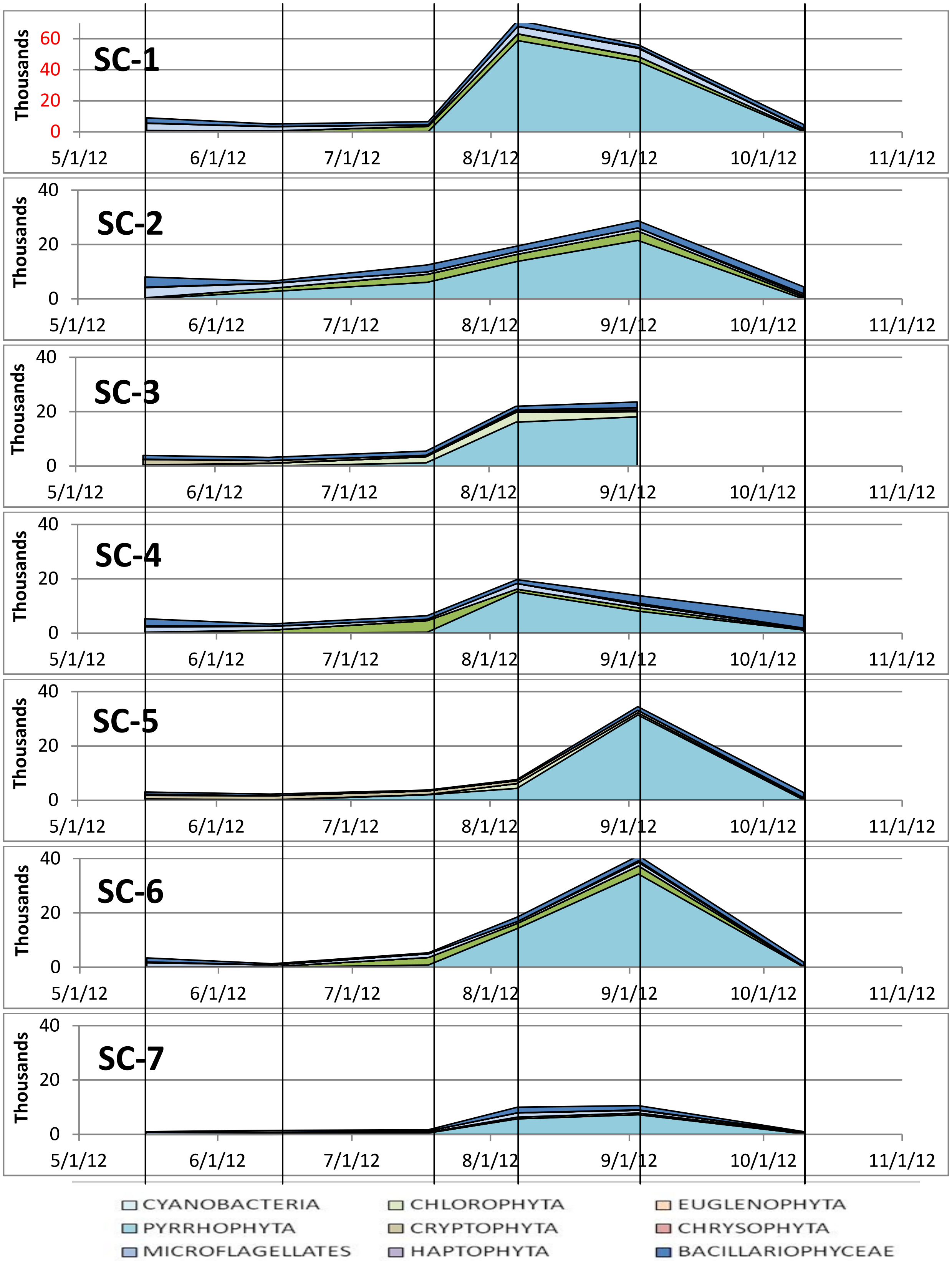


Figure 36. Algal group abundance (cells/mL) at seven MCES lake monitoring sites, May-October 2012. Note the difference in scale at SC-1.

Seasonal Deep-hole Samples of Phytoplankton

Phytoplankton nutrient-dependent growth rates were determined in the laboratory using a modified algal growth potential (AGP) bioassay protocol. Five seasonal bioassay samples were collected from the upper mixed layer of the water column in the four main pool sites during sampling of five seasons of the study period: summer and autumn of 2011 and the spring, early summer and late summer of 2012. Samples were subjected to an experimental nutrient enrichment gradient that included four treatments: an ambient control (CON), and additions of ortho-phosphate phosphorus (P), sodium nitrate nitrogen (N) or a combination of both (NP). Nutrients were added at levels sufficient to produce nutrient-saturated growth under normal conditions.

Results from each of the four bioassay treatments applied to the five seasonal samples are summarized by pool in Figure 37. Overall, results varied by site and treatment, with significant differences in the magnitude and timing of responses between 2011 and 2012. Ambient growth rates measured in the control treatments were positive in all but one experiment at one location, and results varied by year and by season. In 2011, ambient growth rates were low in summer and low and variable between the four pools in autumn. The following year, ambient growth rates were lowest in the spring of 2012 but reached their highest levels in June and August later that summer (Figure 38).

Responses to additions of N or P alone produced dynamic results. In general, P-limitation was more common in summer samples from 2011, while samples from 2012 exhibited signs of N- or P-limitation as well as unlimited growth. All of the pools were P-limited in the spring of 2012 and moved towards N-limitation or co-limitation by late summer in August of 2012. Three of the four pools (1, 2, and 4) were N-limited at one point during the summer of 2012.

Addition of both nutrients together (NP) significantly increased growth rates over ambient controls in both years at all sites, providing seasonal estimates of maximum, nutrient-sufficient growth rates at each location. Maximum growth rates are consistent across seasons in Pools 1 and 2. In contrast, maximum growth rates are lower in the spring in the lower two pools and are more variable the rest of the year when compared to the two upper pools of the lake.

The higher ambient growth rates observed in the summer of 2012 coincided with a period of reduced growth response to the P-addition treatments. The P nutrient treatment response is easier to assess if growth rate data are converted to a relative index of P limitation (PLI) shown in Figure 39. The index is a ratio of the observed ambient control growth rate compared to the P-sufficient treatment (Figure 38). A low index value indicates potential P-limited growth, while index value of one or greater indicates that ambient growth rates were unaffected by the addition of P. The index values for Lake St. Croix bioassays make it clear that additions of P alone stimulated growth in June and October of 2011. This pattern did not persist into 2012. Only the April bioassay in 2012 showed a significant response to added P (Figure 37).

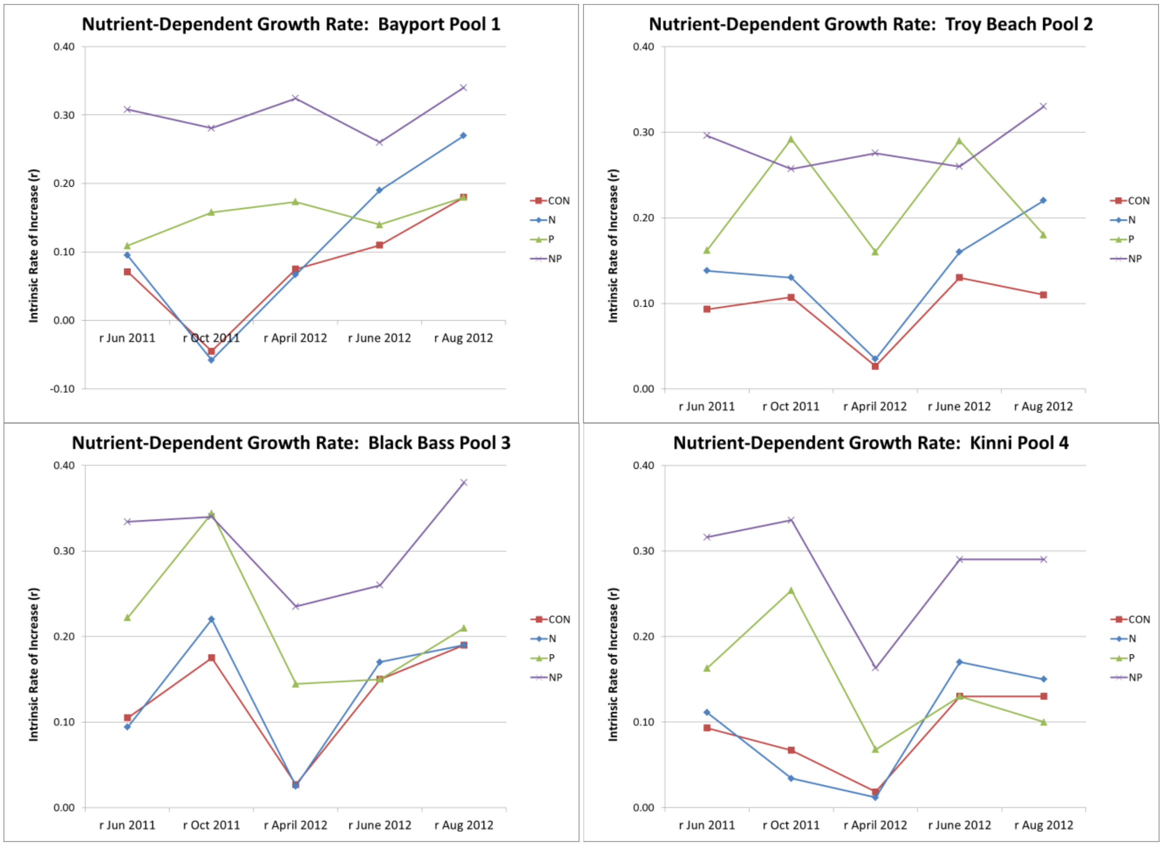


Figure 37. Results of seasonal nutrient-dependent algal growth bioassays conducted on samples collected from the four pools of Lake St. Croix. Water was collected from the mixed layer of the lake and assessed for the tested for the effect of nutrient additions of nitrogen (N), phosphorus (P) or N and P in combination (NP), compared to a control (CON). Results were used to estimate algal community growth rates (r).

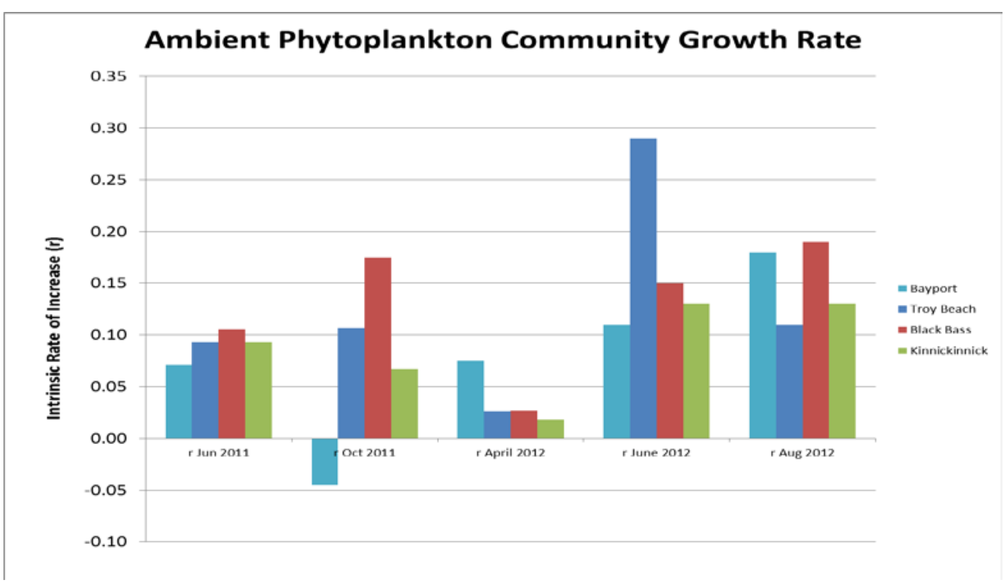


Figure 38. Ambient (control condition) phytoplankton community growth rates from the control treatments of the nutrient bioassays, showing the intrinsic rate of increase (r) in the four deep-pool sites of Lake St. Croix, for five seasonal sampling events.

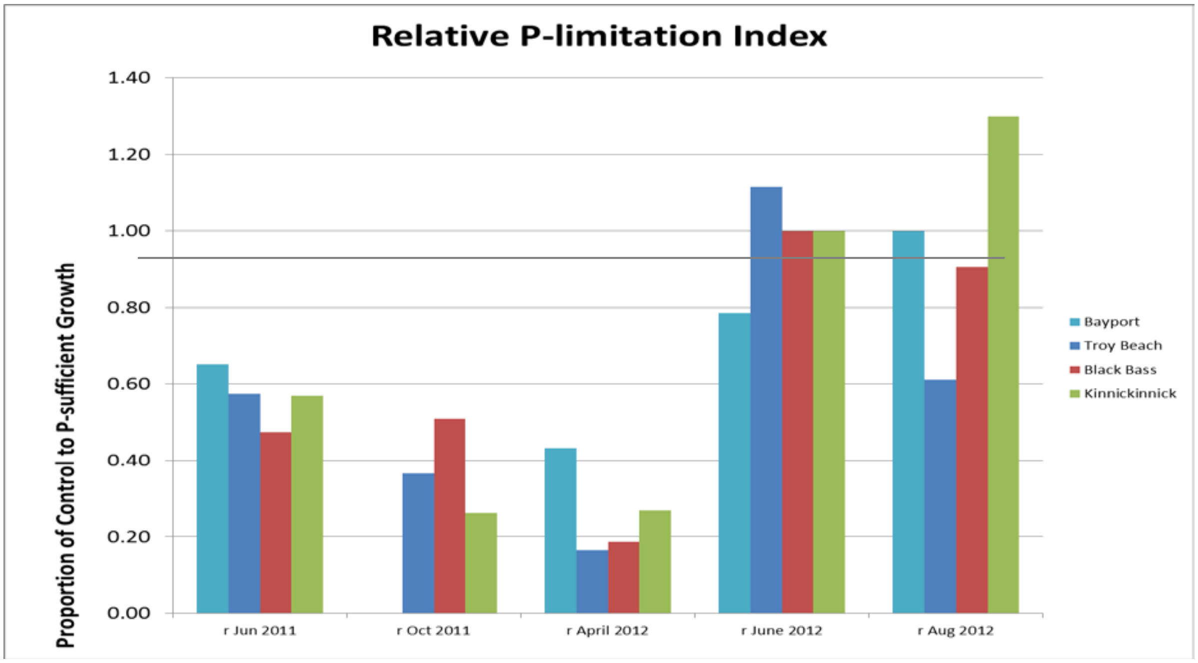


Figure 39. Index of P limitation for all treatment dates and locations. Values equal to or greater than 1.00 indicate that the ambient growth rates were not limited by the availability of P.

Deep-hole Samples of Zooplankton

Zooplankton grazing rates were estimated by measuring phytoplankton community growth rates as a function of zooplankton density. Experimental zooplankton density gradients were created by either diluting ambient bioassay water with particle-free lake water to lower zooplankton density or by adding zooplankton to bioassay water to achieve elevated zooplankton densities. The end result was a density gradient of zooplankton grazers against which phytoplankton community growth rates were measured.

In most cases, a log-linear relationship develops between phytoplankton community growth rates and zooplankton density when grazing losses are significant. When this relationship is statistically significant, the slope of represents the density-dependent zooplankton grazing rate. Figure 40 presents the results from the June 2012 grazing experiment conducted on bioassay water samples from all four pools of Lake St. Croix. Results from all four pools are very similar (Table 5) suggesting a common phytoplankton community response to grazing pressure. Pool 4 had the most negative slope and therefore had the highest grazing rate. Pool 2 had the least negative slope, suggesting the lowest grazing rate. However, the relationship for Pool 2 was not statistically significant. Taken together, these results point to a broad functional relationship between zooplankton density and phytoplankton community growth rates in Lake St. Croix. Therefore, algal biomass accumulation in the mixed layer of Pools 1-4 will be affected by zooplankton densities, suggesting that zooplankton grazing must be considered when characterizing the seasonal succession of phytoplankton species in Lake St. Croix.

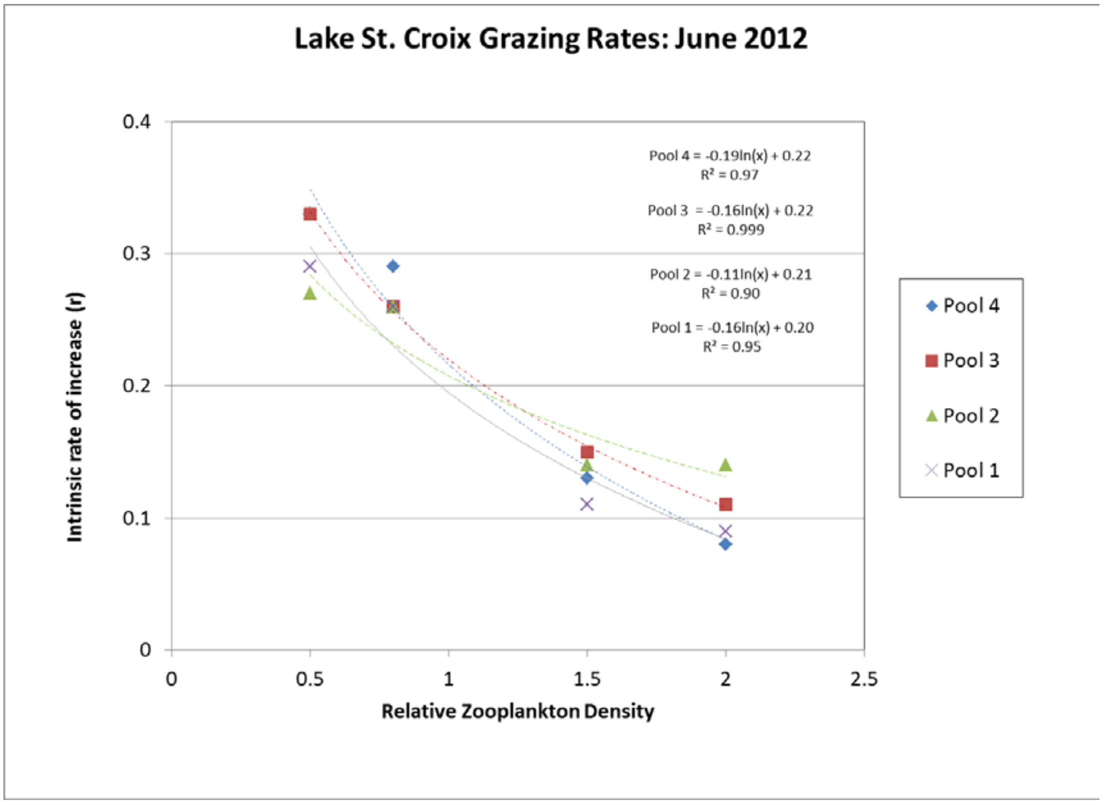


Figure 40. Phytoplankton community growth rates (r) as a function of relative zooplankton density for Lake St. Croix Pools 1-4. Slopes of the logarithmic functions fit to each pool estimate the density-dependent zooplankton grazing rate on the ambient phytoplankton community.

Table 5. Phytoplankton community growth rates (r) as a function of relative zooplankton density for Lake St. Croix Pools 1-4, June 2012.

Lake St. Croix Phytoplankton Community Population Growth Rate (r) June 12				
Relative Zooplankton Density	**Pool 1**	**Pool 2**	**Pool 3**	**Pool 4**
0.5	0.29	0.27	0.33	0.33
0.8	0.26	0.26	0.26	0.29
1.5	0.11	0.14	0.15	0.13
2.0	0.09	0.14	0.11	0.08

Deep-hole Samples of Lake Bed Sediments

The most recent water quality modeling study of Lake St. Croix (Robertson and Lenz 2002) used estimates of sediment nutrient release to calculate internal nutrient loading to the lake. Robertson and Lenz (2002) combined preliminary estimates of oxic and anoxic nutrient rates with estimates of the spatial extent of hypoxic bottom waters to calculate an internal load. For the current study, we repeated this calculation using direct measurements of sediment nutrient release coupled with dissolved oxygen profile data collected through the summer.

During April and August 2012, sediment cores with an intact sediment-water interface were collected from the four deep pool sites in the lake and incubated in the laboratory under ambient temperature and dissolved oxygen conditions. Initial and final dissolved nutrient concentrations in the overlying water were used to estimate the daily rate of phosphorus release under the oxic conditions that occurred during April 2012 and the anoxic conditions that were common during August 2012 (Figure 41 and Table 6).

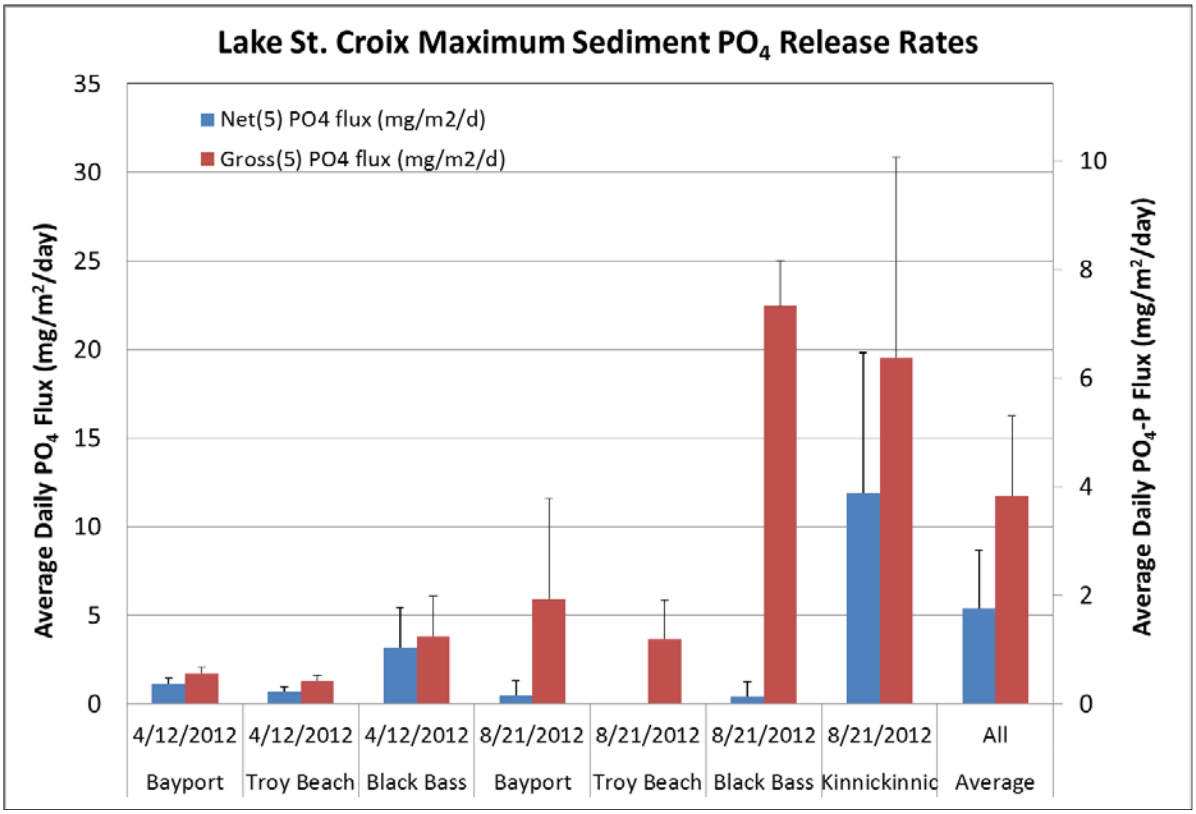


Figure 41. Seasonal estimates of phosphorus release from Lake St, Croix sediment cores. Cores were collected from the four main pools of the lake and incubated in a laboratory environmental chamber in the dark under ambient temperature.

The areal extent of anoxia in Lake St. Croix was estimated from dissolved oxygen profile data collected at fixed monitoring stations in each deep pool biweekly (Figure 42). Dissolved oxygen profiles were used to estimate the depth at which hypoxia developed and the duration of anoxic conditions. The depth and dissolved oxygen conditions were assumed to be uniform across each pool, and this information was combined with the Robertson and Lenz (2002) estimates of each pool's sediment area based on the hypsographic curves in Figure 43.

Table 6. Estimates of rates of phosphorus release from Lake St. Croix sediments, 2012. Sediments were collected from individual pools within the lake and incubated in a dark environmental chamber in the laboratory under ambient temperature.

Gross Flux of Phosphorus From Lake St. Croix Sediment Cores							
Lake St. Croix Pool	**Date**	**Gross PO_4 flux ($mg/m^2/d$)**	**N**	**SDEV**	**CI95**	**CV (%)**	**Gross PO_4-P flux ($mg/m^2/d$)**
Bayport Pool 1	4/12/2012	1.73	2	0.24	0.33	14	0.56
Troy Beach Pool 2	4/12/2012	1.31	2	0.20	0.28	15	0.43
Black Bass Pool 3	4/12/2012	3.81	2	1.62	2.25	43	1.24
Bayport Pool 1	8/21/2012	5.9	2	4.09	5.66	69	1.94
Troy Beach Pool 2	8/21/2012	3.7	2	1.56	2.16	42	1.20
Black Bass Pool 3	8/21/2012	22.4	2	1.87	2.59	8	7.32
Kinnickinnic Pool 4	8/21/2012	19.5	2	8.17	11.3	42	6.38
Average	All	11.7	14	8.67	4.54	74	3.83

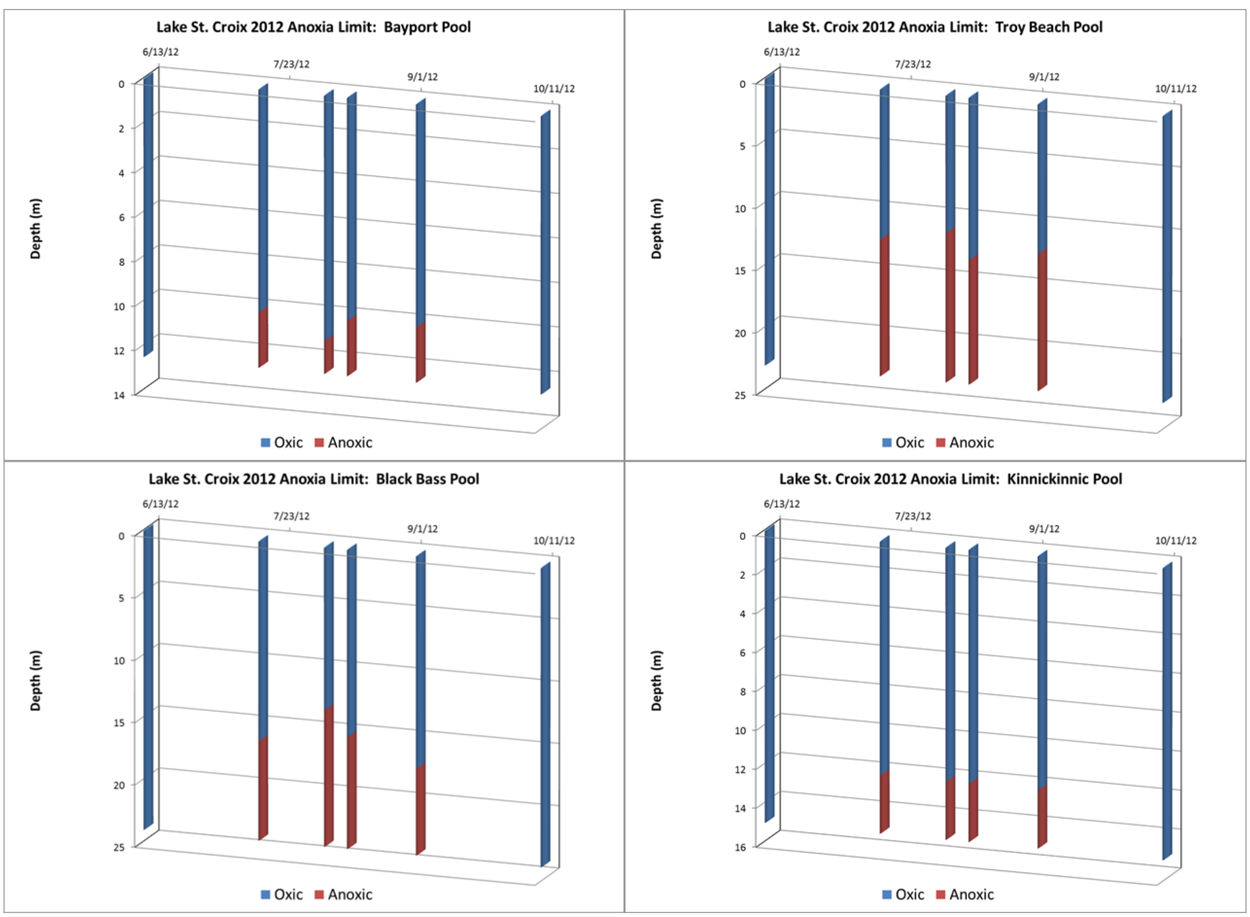


Figure 42. Estimated zone of anoxia for the four main pools in Lake St. Croix, 2012. Zones were identified from dissolved oxygen profile data collected by Harper Consulting and by the U.S. Geological Survey.

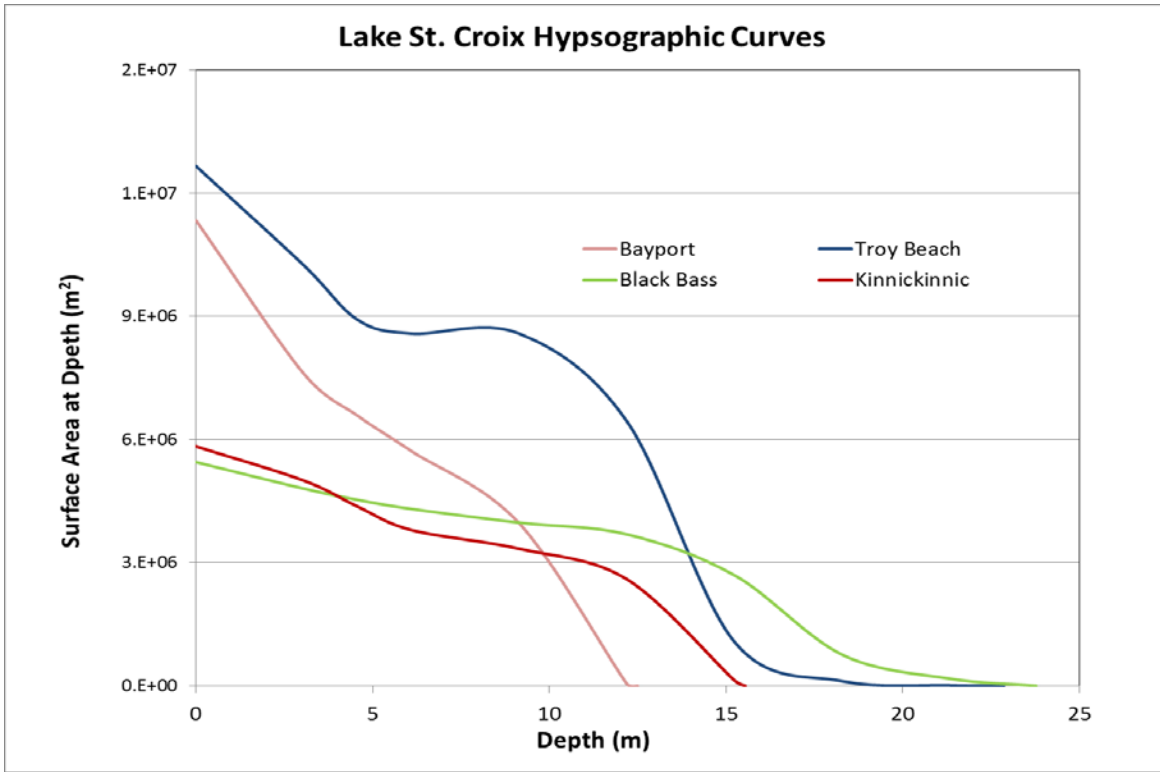


Figure 43. Lake St. Croix hypsographic curves for the four majors pools. Data were obtained from Robertson et al. (2002).

Nutrient release rates (Figure 41; Table 6) were combined with estimates of the areal extent of anoxic sediments throughout the summer of 2012 (Figure 42) to calculate the mass load of phosphorus from the sediment to the overlying water column (Table 7). The methodology consisted of estimating the areal extent of anoxic sediments for each pool for each sampling period and multiplying the area times the nutrient release rate from the sediment incubations and the duration of anoxia in days. The end result was an estimate of total nutrient mass transfer to the water column for each pool. Estimates for summer (June-September) 2012 internal load were calculated by summing the results from the four deep pools (Table 8). This calculation assumes that released nutrients do not re-enter the sediments but become mixed into the lake profile. To fully understand the fate and transport of the released nutrient would require a dynamic bio-physical water quality model.

Table 7. Internal phosphorus (PO4-P) load estimated released from anoxic sediments for summer 2012. Loads were based on the areal extent of anoxic sediment. Sediment areas were estimated using previous published hypsographic curves for the four pools (Robertson and Lenz 2002). Sediment flux rates were measured twice during 2012.

Internal Phosphorus Load Estimate from Lake St. Croix Sediment Cores				
Lake St. CroixPool	**Date**	**Anoxia Isopleth (m)**	**Anoxic Sediment Area (m^2)***	**Estimated Load (kg PO_4-P)**
Bayport Pool 1	4/12/2012	na	0	0
Troy Beach Pool 2	4/12/2012	na	0	0
Black Bass Pool 3	4/12/2012	na	0	0
Bayport Pool 1	8/21/2012	10	3961063	376
Troy Beach Pool 2	8/21/2012	12	6445833	378
Black Bass Pool 3	8/21/2012	15	2692374	966
Kinnickinnic Pool 4	8/21/2012	12	2596463	811
Total			15695733	2532

Comparison of Sediment Loading Estimates

The sediment nutrient release rates observed for 2012 were not in complete agreement with previous estimates. For example, the 2012 release rates for anoxic sediments in the Bayport and Troy Beach pools were up to 67% less than those measured by W. James as reported by Robertson and Lenz (2002). Part of this difference is likely a result of different methods of incubation. For example, the core incubations in 2012 were not continuously bubbled with nitrogen gas to maintain an anoxic environment. However, overlying water in the cores remained anoxic and continued to release H_2S gas during the incubations.

Robertson and Lenz (2002) used a general release rate of 2 mg/m^2/day for the Bayport Pool and 1 mg/m^2/day for sediments in the three remaining pools in order to calibrate their BATHTUB model. These rates are low compared to the 2012 observed rates for anoxic conditions and less than half the average rate observed for all oxic sites in 2012. Despite these differences, the calculated internal load necessary to calibrate the BATHTUB model was three times the calculated release rates in 2012 (Table 8). If an integrated or merged estimate of internal loading is constructed using the W. James release rates for anoxic sediments in Bayport and Troy Beach pools and the 2012 anoxic release rates for Black Bass and Kinnickinnic pools, annual internal loading is greater in 2012 than in 1999 simulations (Table 8). Phosphorus release rates from sediments estimated by Robertson and Lenz (2002) for the Bayport and Troy Beach pools were combined with measured release rates from Black Bass and Kinnickinnic pools to develop a merged estimate of internal loading for 2012. The merged internal load estimate of 9117 kg is greater than the 2012 estimate of 2532 kg and greater than the reported value of 7927 kg from Robertson and Lenz (2002).

Table 8. Comparison of summer (May-September) internal phosphorus load released from sediments observed in 2012 with those reported by Robertson and Lenz (2002). Merged release rates use the 2002 release rates (from Robertson and Lenz 2002) for the Bayport and Troy Beach pools (Pools 1 and 2), and the 2012 release rates (from this study) for the Black Bass and Kinnickinnic pools (Pools 3 and 4).

Comparison of Lake St. Croix Loading using 2012 Sediment Release Data (kg/summer)						
	1999		Dry Year (1988)		Wet Year (1996)	
	Kilograms	Percent of total	Kilograms	Percent of total	Kilograms	Percent of total
2002 Total P Load (Robertson and Lenz, 2002)	127,352	100	42,173	100	155,369	100
Total Tributary Load	113,845	89.4	28,667	68	141,862	91.3
Point Sources in Contributing Basins	6,938	5.4	6,938	16.5	6,938	4.5
Direct Point Sources to Lake St. Croix	5,125	4	5,125	12.2	5,125	3.3
Atmospheric Load	455	0.4	455	1.1	455	0.3
2002 Internal Sediment Load	7,927	6.2	7,927	18.8	7,927	5.1
2012 Internal Sediment Load	2,532	2.0	2,532	6.0	2,532	1.6
Merged Internal Sediment Load	9,117	7.2	9,117	21.6	9,117	5.9

SYNTHESIS

We developed a conceptual model of Lake St. Croix that incorporates our observations of the spatial and temporal variabilities in Lake St. Croix. Accordingly, the conceptual model includes both spatial and temporal dimensions and constraints, aiding in the clarification of the driving mechanisms in Lake St. Croix ecological functioning.

Driving Mechanisms of Ecological Functioning

Seasonality

Seasonality is driven by the annual cycles in air temperature, precipitation, and the resulting plant growth. The warming air temperatures heat the surface of the Lake St. Croix pools, making them more conducive to algal growth. During the study period, surface water temperatures were consistent across the pools for any given month; the surface layer warms uniformly across this flowing water body. Succession occurs among the algal groups, as the phytoplankton community adjusts to the seasonal warming. Zooplankton grazing serves as a feedback mechanism on phytoplankton succession. Chlorophyll-a concentrations increase throughout the summer, peaking in August and September before cooling temperatures and autumn turnover. Laboratory analyses of nutrient-enriched bioassays indicate that maximum nutrient-sufficient (i.e., NP treatment) phytoplankton growth rates occur in summer and autumn. All of the pools were P-limited in the spring of 2012 and moved towards N-limitation or co-limitation by late summer in August of 2012.

The depth profile measurements reveal both temporal variability and spatial variability in the vertical dimension. All four pools of Lake St. Croix, between June and September every year, undergo lake stratification into a warm oxic epilimnion overlying a cold hypoxic hypolimnion. The two zones are separated by a thermocline, the metalimnion, which inhibits vertical mixing between the zones. In essence, this creates two separate populations that partially diverge for the duration of the summer months: 1) a warm oxygenated river continuously sliding over 2) a series of four isolated cold hypoxic pools. During the summer months of the study period, the deepest part of all four pools was anoxic, yet the narrows between Pool 1 and Pool 2 at SC-3 was fully oxygenated. These observations, repeated every June-September, support the conceptual model of a warm river sliding over four isolated, cold pools.

Spatiality (flow routing)

Pool morphology, the shape of the lake and its sub-basins, sets up spatial variability in the downstream longitudinal sense. When the mainstem St. Croix River enters Lake St. Croix above Stillwater, channel widening distributes flow across a wider cross-section, decreases stream energies and drops the heavier sediments. The same mechanism occurs below each of the narrows that separates each of the pools, and the river progressively drops its load of suspended solids, sorting the bed sediments from coarse-grained at the head of the lake to finer-grained sediments farther downstream. In addition, Pool 1 at the head of the lake receives the majority of inputs to Lake St. Croix, via the mainstem river, two suburban streams, two major point sources, and a large tributary draining its own reservoir. Flow depths are shallow due to sedimentation in Pool1 and the upstream end of Pool 2, which decreases water residence time in those pools. Due to greater average depths in Pools 3 and 4, water residence times tend to be longer, which affect the ecological characteristics of the lower pools. Thus, the conceptual model of Lake St. Croix includes a downstream spatial component: Bayport Pool 1 tends to ecologically function like a turbid river, while Kinnickinnic Pool 4 tends to ecologically function like a clear lake. The transition between these ecological zones is located somewhere between Pool 1 and Pool 4, usually within Pool 2. Low clarity and shallow pools contribute to the turbid river zone, while high clarity and deeper pools enable the clear lake zone.

Exceedances of water quality standards for clarity, phosphorus and chlorophyll are more likely to occur in Pools 1 and 2. Secchi depth clarity showed a downstream trend of increasing clarity during the 2008-2012 study period, perhaps due to particulates settling out of suspension toward the downstream end of the lake. Perhaps for the same reason, total phosphorus concentrations in the water column showed a downstream trend of decreasing concentrations for most years except 2008. Longitudinal coarse-to-fine sorting of bed sediments may have played a role in the downstream trend of increasing potential for internal loading; fine-grained sediments are associated with higher phosphorus release rates from anoxic sediments, as was observed in the two lower pools, compared to release rates in the two upper pools. Chlorophyll-a showed a downstream trend of decreasing concentrations through Pool 1, then increased again in Pool 2 (at SC-4), before continuing its downstream trend of decreasing chlorophyll concentrations. Follow-up work will test a couple hypotheses: 1) the wide, long shallow areas at the upstream ends of Pool 1 and Pool 2 preferentially encourage abundant algal growth, or 2) upstream reservoirs serve as sources of allochthonous chlorophyll that are delivered to Pool 1 by the mainstem St. Croix River and to Pool 2 by the Willow River.

Spatial variability in the autumn depth profiles generated another set of competing hypotheses. Low-to-moderate flows during and after turnover in the autumn months showed a consistent downstream trend of decreasing DO concentrations leading to a hypoxic profile in the lower pools. Two hypotheses are 1) a process of isolated whole-pool mixing occurs where each pool mixes as the metalimnion degrades under seasonal cooling such that the lower pools have to incorporate deeper profiles containing a larger mass of low-oxygen waters, or 2) downstream mixing of the hypoxic waters in all pools progresses such that the lower pools become fully hypoxic, top-to-bottom. The 5-6 °C thermal differentiation of Bayport Pool 1 from the other pools observed in October 2009 points to the first hypothesis. The answer will depend upon a greater understanding of the flow-dependent residence time of each pool compared to the rate of downstream advection.

Flow variability

High discharge conditions were observed to cause a deeper mixing profile in Lake St. Croix, such that shallow, previously-stratified sites could become unstratified after high discharges in mid-summer. High spring peak discharges that were sustained into early summer could reduce the degree of stratification for the remainder of the summer season, pushing deeper mixing such that the lake surface was not as highly oxygenated when compared to low-flow summers. These interactions between flow variability and lake profiles have suggested a possible component in the ecological functioning of Lake St. Croix.

With the onset of lake stratification in May and June, the hypolimnion becomes increasingly anoxic, creating a reducing environment and enabling the release of phosphorus from its attachment to sediment particles. Thus, vertical stratification in Lake St. Croix allows its bed sediments to become a source of phosphorus, which is unusual for most river systems. If Lake St. Croix was not a dynamic, flow-through system, the phosphorus released from sediments would simply cycle in place. The monthly profile measurements give us a clue as to a potential mechanism for transfer of sediment-released phosphorus into the epilimnion. In response to sustained high flows, a massive pulse of surface flow appears to erode the metalimnion, driving the bottom of the epilimnion deeper into the lake profile and mixing upward some of the low-oxygen waters from the hypolimnion. In addition to hypoxic waters, this upward mixing could also transfer sediment-released phosphorus into overlying waters.

Implications

Value of long-term monitoring

Ongoing monitoring efforts of cooperating agencies have generated a rich and informative dataset about the physical, chemical, and biological components of the Lake St. Croix riverine system. Although the funding cycle for the current project was three years, the previous efforts of cooperating agencies, especially Metropolitan Council Environmental Services, made it possible to extend the study period of the project to five years (2008-2013). The longer five-year period provided a much more informative dataset that encompassed a fuller range of low-flow to high-flow conditions, thus deepening our understanding of Lake St. Croix ecological functioning. In addition, this study would not be possible without the commitment of the USGS to long-term discharge monitoring within the St. Croix Basin.

Next steps: Hydrodynamic model of Lake St. Croix

This riverine system requires a hydrodynamic model to assess whole-lake processes. The capabilities of the CE-Qual-W2 model will make it possible to develop a more accurate representation of the physical, chemical, and biological components of Lake St. Croix:

- variations in coarseness of bed sediments that affect the potential for internal loading
- spatial and temporal variability of anoxia due to lake stratification can be incorporated into calculations of internal loading
- model can account for proportions of nutrients that are labile or refractory
- model can account for nutrient requirements and stoichiometric constraints of different algal groups, including N-fixers
- can model the seasonal succession of algal groups
- can assign a sink-rate or buoyancy-rate to an algal group to model its vertical transport rate through the water column
- can model the timing of as many as four algal bloom events
- can model the seasonal storage or sequestration of nutrients in deep-pool sites

Once the CE-Qual-W2 model is calibrated and validated, it will aid in clarification of the primary components of ecological functioning of Lake St. Croix:

- Accounting for the lake as an interconnected water body, but each pool has its own trajectory in terms of stratification and mixing characteristics.
- Calculation of how residence time for each pool scales with variations in flow, and develop an understanding of whether the residence time of each pool or the rate of downstream advection is a controlling mechanism in the hypoxia in the lower pools during autumn turnover
- Improvement on the current estimation of internal loading potential for the pools in Lake St. Croix that assumes phosphorus released from anoxic sediments become mixed into the lake profile and do not re-enter the sediments.
- Assessment of whether the different transport mechanisms for N (primarily dissolved) and P (primarily suspended) affect the timing of input loading and availability of the primary nutrients to the lake system.
- Assessment of the relative importance of temporal variabilities (e.g., percent of long-term annual load versus short-term seasonal abundance).
- Identification of the location and characteristics of the mid-lake transition between the turbid river zone in the upper pools and the clear lake zone in the lower pools.

CONCLUSION

This study has documented the spatial and temporal variability of ecological characteristics of the riverine Lake St. Croix. These efforts have led to an improved understanding of the complex ecological interactions that occur within the lake. Following the 2007 discovery of vertical stratification in Lake St. Croix, this 2008-2012 study was able to document the extent and stability of lake stratification in this riverine system. Initial estimates of this study indicate that the duration of hypolimnetic anoxia cause greater internal loading within the lake than previously calculated, and an upcoming hydrodynamic model of Lake St. Croix will quantify whole-lake nutrient processing more accurately than before. During this study, the first-ever documentation of conditions conducive to N-limitation in algal growth has pointed to the need for greater understanding of the dynamics of both primary nutrients (N and P) within this riverine system. Thanks to the USGS, flow conditions within the lake are now better understood, and when discharge analyses are finalized, a thorough loading analysis will be conducted, assessing progress toward the phosphorus and eutrophication TMDL reduction goal. Already, we were able to document that exceedances of clarity, phosphorus, and chlorophyll were most likely to occur in the shallow upper portions of Bayport Pool 1 and Troy Beach Pool 2.

Stratification within a flowing riverine system has implications for how that system processes nutrient loads. If there is more internal loading than anticipated, then there may be more phosphorus available for algal growth than originally calculated. This is one driving mechanism that could explain the decoupling of phosphorus reductions from ongoing annual algal blooms. Currently, Lake St. Croix is a mesotrophic-to-eutrophic riverine system that is dominated by blue-green algae (cyanobacteria) in the summer months, with implications for recreational health and safety. If the primary goal of the Lake St. Croix TMDL is to meet clarity, total phosphorus, and chlorophyll water quality standards, then a secondary goal is to reduce algal bloom frequencies and move algal specie abundance away from planktic species toward benthic dominance.

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