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Interannual and long-term changes in the trophic state of a multi-basin lake: effects of morphology, climate, winter aeration, and beaver activity

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Abstract

Little St. Germain Lake (LSG), a relatively pristine multi-basin lake in Wisconsin, USA, was examined to determine how morphologic (internal), climatic (external), anthropogenic (winter aeration), and natural (beaver activity) factors affect the trophic state (phosphorus, P; chlorophyll, CHL; and Secchi depth, SD) of each of its basins. Basins intercepting the main flow and external P sources had highest P and CHL and shallowest SD. Internal loading in shallow, polymictic basins caused P and CHL to increase and SD to decrease as summer progresses. Management actions: winter aeration used to eliminate winterkill increased summer internal P loading and decreased water quality, while reductions in upstream beaver impoundments had little effect on water quality. Variations in air temperature and precipitation affected each basin differently. Warmer air temperatures increased productivity throughout the lake, and decreased clarity in less eutrophic basins. Increased precipitation, increased P in the basins intercepting the main flow, but had little effect on the isolated deep West Bay. These relations are used to project effects of future climatic changes on LSG and other temperate lakes.

Introduction

Little St. Germain Lake (LSG), Wisconsin, USA, like many lakes, has experienced a recent decline in water quality (enhanced eutrophication). However, interannual variability in its water quality makes it difficult to quantify the exact magnitude and timing of any long-term changes. Because interannual variability in water quality results from the interaction of many potential factors, it is difficult to readily quantify the importance of anthropogenic factors, such as modifications made to the lake (e.g., winter aeration) and its watershed (e.g., removal of beaver impoundments), relative to natural factors (e.g., interannual variations in meteorology and longer-term changes in climate). Prior to taking actions to try to improve the water quality in LSG, it is important to understand how each of the various factors affects the short-term variability and long-term changes in eutrophication.

Many factors affect the interannual variability and long-term changes in the trophic state of lakes. One factor affecting lake productivity is the variability in the amount of nutrients input from its watershed (external loading). The effects of nutrient loading (for many lakes this primarily refers to phosphorus, P; Schindler 1975) are sufficiently understood that empirical eutrophication models have been developed to predict in-lake total P concentrations (TP), chlorophyll *a* concentrations (CHL), and water clarity (Secchi depth, SD) from lake morphometry and external water and TP loading (Cooke et al. 1993; Panuska and Kreider 2003). Interannual changes in external nutrient loading primarily occur as a function of hydrologic variability (more nutrients are usually delivered during wet years than during dry years). Long-term changes in external P loading are often caused by changes in anthropogenic factors in the watershed, such as changes in land use or contributions from point sources; however, other alterations in the watershed can also affect external loading, such as anthropogenic or natural

73 impoundments. One example of a natural impoundment is the construction of upstream beaver
74 ponds. Impoundments by beavers have been shown to either increase or decrease P
75 concentrations in streams (Klotz 1988, 1998; Muskopf 2007; and Bledzki et al. 2011).

76 External P loading drives inflake P concentrations and lake productivity (plant and algal
77 growth) and then P is either exported out the outlet or deposited in the lake sediments. Not all P
78 that is deposited in the bottom sediments, however, remains there; some of it is released back
79 into the water column, referred to as “internal P loading”. Mortimer (1941) demonstrated that P
80 release rates increase dramatically when dissolved oxygen near the sediment interface is
81 completely depleted (anoxia). Various factors may affect the annual magnitude of this source,
82 with internal P loading increasing as: the length of thermal stratification increases (Kling et al.
83 2003), water temperatures increase promoting microbial decomposition (James and Barko 2004;
84 Jensen et al. 2006), and as the amount of readily available P in the sediment increases (Rydin and
85 Welch 1999).

86 Other factors, such as interannual variability in weather and long-term changes in climate
87 (primarily air temperatures) may also affect lake productivity. Lake productivity has been shown
88 to increase as water temperatures increase (Regier et al. 1990; Mooij et al. 2005; Jeppesen et al.
89 2014) and as the length of the ice-free period increases (Fee et al. 1992). Increased air
90 temperatures have been found to cause increased CHL and decreased clarity in both field (Mooij
91 et al. 2005) and modeling studies (Mooij et al. 2007). Regier et al. (1990) found that plankton
92 biomass and fishery yields increased with increased water temperatures. Increased water
93 temperatures are also expected to lead to the increase occurrence of algal blooms and toxin-
94 producing cyanobacteria (Jeppesen et al. 2009; Kosten et al. 2012). However, in some lakes,
95 increases in air temperature can decrease lake productivity. Bates et al. (2008), Brooks and

96 Zastrow (2002), and Verburg and Hecky (2003) found warmer air temperatures increase the
97 length and intensity of stratification in deep dimictic lakes, which results in a decline in the
98 nutrients available from spring mixing and internal loading, which can decrease late-summer
99 productivity (i.e., running out of nutrients). However, even when surface nutrients decline, some
100 cyanobacteria can proliferate in late summer. Therefore, the effects of climate change (increased
101 air temperatures) may resemble cultural eutrophication (Jeppesen et al. 2014).

102 Productivity of shallow lakes may, however, respond differently than deep lakes because of
103 how the factors just described behave, especially internal loading. Shallow lakes, defined as
104 lakes having a maximum depth $< 6\text{m}$ (Osgood et al., 2002), typically experience frequent periods
105 of mixing throughout summer, and are therefore referred to as polymictic lakes. Typically, in
106 deep dimictic lakes, P released from the deep sediments is retained in the hypolimnion and
107 primarily released to the epilimnetic water during spring and fall turnover. Therefore, near-
108 surface TP in dimictic lakes usually remains stable or decreases as summer progresses (Welch
109 and Cooke 1995). In shallow polymictic lakes, microzones of anaerobic activity can develop at
110 the sediment-water interface during short periods of stratification, especially during summer
111 when biological activity is high. These frequent mixing events result in near-surface TP and
112 productivity increasing throughout summer in polymictic lakes (Welch and Cooke 1995). Mixing
113 throughout summer typically results in near-bottom temperatures in polymictic lakes being
114 higher than in dimictic lakes, which results in the rate of sediment P release being higher than in
115 deep lakes. The importance of internal P loading may vary through time and among years
116 because of variations in water temperatures and length of the summer period, and may increase
117 with climatic warming (Magnuson et al. 1997; Kling et al. 2003; Jeppesen et al. 2014).

Depletion of oxygen also occurs under the ice during winter. In productive shallow lakes, including LSG, dissolved oxygen concentrations throughout the water column can drop below critical concentrations during winter and cause fish kills (referred to as “winterkill”), increased internal P loading, and odor problems where water leaves the lake (Wetzel 1983; Hudson and Kirschner 1997). To try to eliminate these issues, aeration systems have been installed and operated throughout winter. Winter aeration has been shown to have dramatic effects on winter water quality in LSG (Robertson et al. 2005) and other lakes (Hudson and Kirschner 1997) and has eliminated the winterkills in LSG (S. Gilbert, WDNR, pers. commun., 2014).

The main goals of this paper are to: 1) quantify the variability and long-term changes in the water quality of the various basins in LSG from 1991 to 2013; 2) quantify the relative importance of the various factors affecting the water quality (trophic state) in each of the basins in LSG (i.e., the relative importance of morphometry and hydrodynamics, internal and external loading, and meteorological/climatic variability); 3) determine if past anthropogenic changes in the lake (winter aeration) and the watershed (reduction in beaver impoundments) have affected LSG’s water quality; and 4) use the understanding of how the various factors have affected the water quality in basins in LSG, including the effects of interannual variability in meteorology, to project how these factors may affect LSG and other temperate lakes in the future, including how future climate change may affect water quality.

Methods

Study area

LSG is a relatively pristine, 396-ha multi-basin lake (Fig. 1). The lake has four main basins: East Bay (164 ha; mean depth 2.5 m), Upper East Bay (48 ha; mean depth 2.2 m), West Bay (86

ha; mean depth 7.1 m) and South Bay (98 ha; mean depth 2.8 m) (Table 1). The major tributary to LSG is Muskellunge Creek, which flows about 5 km from shallow, naturally eutrophic Muskellunge Lake into the north end of East Bay (Fig. 1). The general flow pattern through the lake is from Muskellunge Creek, through East and Upper East Bays, through South Bay, and into Little St. Germain Creek. East and Upper East Bays appear to be strongly connected, especially during summer, based on limited variations in water temperatures and water quality. The limited hydrologic connection between the East and West Bays, which are separated by a 1.5 m sill, result in West Bay being moderately isolated from the rest of the lake. The overall residence time in LSG, based only on inflow is ~2.2 years; however, it is only ~0.8 years in East and Upper East Bays, and much longer in West Bay. A previous study (Robertson et al. 2005) found that Muskellunge Creek contributed 43-53% of the total inflow to the lake and 53-62% of the TP; the remainder is primarily groundwater entering East and Upper East Bays and rainfall.

The watershed of LSG is ~25 km², and is predominantly forest (68%), wetland (17%), and water (24%), although low-density residential areas are present, especially immediately around the lake (Fig. 1; Robertson et al. 2003). LSG's relatively pristine watershed was expected to limit external loading and enable the effects of other typically less important factors (such as meteorological/climatic variability) to be determined.

Data collection

Water quality in LSG has been monitored throughout summer since 1991 by either the U.S. Geological Survey (USGS) or the LSG Lake Protection and Rehabilitation District (LSG Lake District) as part of the Wisconsin Department of Natural Resources (WDNRs) Citizen Lake Monitoring Program (WDNR 2008); however, only limited data were collected in 1995, 1996, and 1998. In each year, sampling was conducted four times (May, June, July, and August) near

the deepest locations in East, West, and South Bays (Fig. 1). Data were also collected in Upper East Bay during 2000–13. During all years, near-surface water samples were collected and analyzed for TP and CHL, and SDs were measured by both groups. During all USGS samplings, profiles of water temperature, dissolved oxygen, specific conductance, and pH were measured with multi-parameter instruments, and near-surface and near-bottom samples were collected with a Van Dorn sampler and analyzed for TP and CHL (CHL only for surface samples). USGS sampling and analytical protocols are described in Marsh et al. (2012). All Citizen Lake Monitoring was done in accordance with protocols described by Wickman and Herman (2005). During all open-water sampling, near-surface samples were collected at the same locations, at a similar sampling frequency, and using relatively similar sampling techniques. When computing summer-average (June–August) values, months with missing data were first estimated using regressions with adjacent months. Trophic state index values were then estimated using Carlson’s (1977) Trophic State Index (TSI) equations.

During 1991–2013, the USGS conducted four intense studies, which also examined winter water quality and tributary loading, as part of cooperative studies with the LSG Lake District. In a 1991–94 study, water quality throughout the lake was quantified. In a 1994–2000 study, extensive areas of winter anoxia were found in Upper East and South Bays and water and P budgets of the lake were quantified, which demonstrated that Muskellunge Creek was the dominant P source to the lake (Robertson and Rose 2000). The study also found that TP in Muskellunge Creek were much higher in 1997 than in 1999, especially in mid to late summer. The higher TP concentrations and resulting higher P loads were believed to be caused by release of P from upstream impoundments and marshy areas flooded by beaver activity (W. Egtvedt, Vilas County Forestry, written commun., 1994). The amount of impounded water was greatly

reduced just prior to 1999. As a result of the 1994–2000 study, aerators were placed near the centers of the Upper East Bay, South Bay, and in the northern end of the East Bay to reduce winter anoxia and prevent winterkill. Aerators were installed in Dec. 2001 in the East and Upper East Bay and in the South Bay in Dec. 2002 and operated each winter until the present, except the aerator in East Bay that was discontinued after ice out in 2004. In a 2000–04 study, the aerators in Upper East and South Bays were found to be effective at eliminating low dissolved oxygen concentrations during winter in their respective basins. The study also found that water quality throughout most of the lake was relatively stable during 1992–2001; however, after 2001, water quality appeared to degrade (Robertson et al. 2005). As a result of these studies, use of the aerators in Upper East and South Bays continued, and extensive efforts (primarily from 2006 to 2013) were taken to totally prevent beavers from impounding water on Muskellunge Creek. In a 2010–13 study, changes in stream water quality were further evaluated.

Inflow and P loading to LSG were determined for five separate years (1997, 1999, 2001, 2011, and 2012) from flow measurements and water samples collected at LSG Inlet (Fig. 1). During Sept. 1996–Oct. 97 and Dec. 1998–Dec. 2000, daily flows at the Inlet were estimated by linearly interpolating between monthly measurements. From March 2001–Oct. 2001 and Nov. 2010–Dec. 2012, continuous (15-minute intervals) water-elevation measurements were collected with recording gages at LSG Inlet and also at Muskellunge Lake Outlet (Fig. 1) to better describe changes occurring in Muskellunge Creek. The monthly streamflow measurements were used to develop stage-discharge relations, which were then used in conjunction with water-elevation data to compute continuous flow at both locations. During 1996–97, monthly water samples were collected at LSG Inlet and analyzed for TP. During 1998–99, monthly samples were collected at LSG Inlet and analyzed for TP and dissolved P (DP). During 1999–2003, monthly samples were

collected at LSG Inlet and Muskellunge Lake Outlet and analyzed for TP and DP. During 2010–12, water samples were collected monthly and during high-flow events at LSG Inlet and Muskellunge Lake Outlet and analyzed for TP and DP.

To estimate P loading in 1997, 1999, and 2001, daily TP concentrations were generated by linearly interpolating between measured samples. For load computation in 2011 and 2012, daily TP concentrations were generated by first linearly extrapolating the concentrations between samples during baseflow. Then when flows increased, the measured TP concentrations were extended in and out during the events at 15-min intervals for the entire event. Baseflow concentrations were used up to the start of the event and then continued after the event. Daily loads were then computed by multiplying the measured or estimated flows by the measured or estimated concentrations. Volumetrically weighted concentrations were obtained by dividing the total load by the total flow for a specified period.

All chemical analyses of lake and tributary samples were done by the Wisconsin State Laboratory of Hygiene (WSLH) in accordance with standard analytical procedures (WSLH 1993). All data collected and loads computed by the USGS were published in online USGS Annual Data Report series available at <http://wdr.water.usgs.gov/index.html> (accessed 1 May 2015) and are available in the National Water Information System database at <http://waterdata.usgs.gov/nwis> (accessed 1 May 2015). All lake water-quality data are also available in the WDNR SWIMS database at <http://dnr.wi.gov/topic/surfacewater/swims/> (accessed 1 May 2015).

Daily air temperature and precipitation data from 1991 to 2013 for LSG were extracted from a $1/8^\circ$ gridded dataset of meteorological observations (Maurer et al. 2002). Spatial subsetting of the dataset was performed using the geoknife R (R Development Core Team 2013) package

(usgs-r.github.io/geoknife; accessed 30 Dec. 2014), which accesses processing capabilities from the USGS Geo Data Portal (Blodgett et al. 2011).

Statistical analyses

Relations between the water quality and meteorological/climatic variables were examined using non-parametric Spearman correlations. Spearman correlations were used because of the small number of years in the analysis. Correlations were also examined after removing any long-term linear trends in each variable to remove the effects of possible co-linearity that may exist between variables; i.e., removing the effects of two variables independently changing over time.

Temporal changes in specific meteorological variables were examined using linear regression and Mann-Kendall trend tests that compensate for non-independent time series data. Temporal changes in TP concentrations in the lake were examined by first removing the effects of interannual differences in meteorology using simple regression relations between summer-average TP and the most significant meteorological variables (summer average air temperature and 2-year average precipitation). Changes in the resulting residuals were then examined using a simple 3-year moving average and using abrupt transition Auto Regressive Integrated Moving Average (ARIMA) intervention models (Box and Tiao 1975). The ARIMA models were used to determine if and how mean conditions had changed, i.e., abrupt change or gradual trends. The best ARIMA models to describe changes in mean conditions were determined by comparing the Akaike Information Criterion (AIC), which penalizes larger more complicated models, with equal fit. ARIMA analyses were performed with the SAS¹ statistical package (SAS Institute, Inc. 1989).

¹ Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Results

Mixing and water quality in LSG

Very little stratification was found in temperature and dissolved oxygen during summer in East and Upper East Bays (near-bottom temperatures in late summer often exceeded 22 C and near-bottom anoxia was seldom observed), whereas once stratification developed in West Bay in May it remained stratified through October (near bottom temperatures remained less than 7 C and near-bottom anoxia was observed in late summer every year). South Bay was between these two extremes with bottom temperatures reaching about 18 C and anoxia often occurring. Based on the extent of stratification of water temperature and dissolved oxygen, East and Upper East Bays were polymictic with frequent mixing events throughout summer, West Bay was dimictic, and South Bay was partially polymictic.

The general flow pattern through the lake and the extent of mixing resulted in consistent patterns in water quality (TSI variables) throughout the lake. East and Upper East Bays generally had the worst water quality (highest TP and CHL, and shallowest SD), South Bay had intermediate, and West Bay had best water quality (lowest TP and CHL, and deepest SD) (Fig. 2 and Table 2; only data for the East and West Bays are provided in Table 2). Overall average spring (May) surface TP ranged from 0.021 mg/L (West Bay), to 0.033 mg/L (South Bay), to 0.040 mg/L (East Bay), and overall summer-average (June-August) surface TP ranged from 0.009 mg/L (West Bay) to 0.080 mg/L (East Bay) (Table 2 and Fig. 2). Although there was considerable interannual variability in TP, a step increase in summer TP appeared to occur between 2001 and 2002 in East, Upper East, and South Bays, whereas TP in West Bay appeared to gradually increase from 1991 to 2013.

TP generally increased as summer progressed in East, Upper East, and South Bays, whereas TP changed very little or decreased slightly as summer progressed in West Bay (demonstrated for East and West Bays in Fig. 3). These seasonal patterns are consistent with that expected in polymictic (TP increases as summer progresses) and dimictic (TP remains constant or decreases slightly) systems. This pattern, in combination with West Bay starting at a lower TP, resulted in a large difference in TP in West Bay from the rest of the lake in late summer. The difference in TP between West and East Bays was 0.019 mg/L in spring, but increased to 0.042 mg/L for the summer average for the entire 1991–2013 period.

Overall summer-average surface CHL ranged from 6.1 µg/L (West Bay) to 47.2 µg/L (East Bay) (Fig. 2 and Table 2). CHL in Upper East Bay were generally close to those in East Bay, and those in South Bay were between East and West Bays. There was considerable interannual variability in CHL; however, once again there appears to be a step increase between 2001 and 2002 in East, Upper East, and South Bays and a gradual increase in West Bay. CHL in West Bay increased from 2–4 µg/L in the early 1990s to about 10 µg/L in the early 2010s. Similar to TP, CHL increased as summer progressed in East, Upper East, and South Bays, whereas CHL changed very little or increased slightly as summer progressed in West Bay (Fig. 3). This resulted in a dramatic difference in CHL in West Bay from the rest of the lake in late summer.

Overall summer-average SDs ranged from 1.0 m (East Bay) to 3.0 m (West Bay) (Fig. 2 and Table 2). SDs in Upper East Bay were similar to those in East Bay. SDs in South Bay were between those in East and West Bays, but became more similar to those in East and Upper East Bays after 2002. There was considerable interannual variability in SDs; however, once again there appears to be a step change between 2001 and 2002, but for SDs, it was most noticeable in South Bay. SDs in West Bay decreased from about 3.5 m in the early 1990s to about 2.5–3.0 m in

the early 2010s. SDs decreased as summer progressed in East, Upper East, and South Bays, whereas SDs changed very little as summer progressed in West Bay (Fig. 3), again resulting in a dramatic difference in SDs in West Bay from the rest of the lake in late summer.

TP, CHL, and SD data were used to compute monthly and summer-average TSIs to determine if and how the trophic state of LSG has changed. In general, TSI values based on TP, CHL, and SD provided very comparable results and therefore, only TSI values for SD are described here (Fig. 4). All three indices indicated that East and Upper East Bays are eutrophic (TSI's between 50 and 60) in June but became hypereutrophic (TSI's > 60) by August. The trophic state of South Bay not only changes seasonally but also changed from 1991 to 2013. Prior to 2002, South Bay changed from being mesotrophic (TSI between 40 and 50) in June to borderline meso-eutrophic in August. Post 2002 (2002–13), South Bay changed from being borderline meso-eutrophic in June to being borderline eutrophic-hypereutrophic in August. The trophic state of West Bay changes very little seasonally, but has gradually changed from being oligo-mesotrophic in the 1990s to being meso-eutrophic in the 2000s.

External loading to LSG

To determine how external P loading, which is affected by interannual differences in streamflow and possibly impoundments associated with beaver activity, affects the water quality of LSG, inflow, P concentrations and P loading were determined for 1997, 1999, 2001, 2011, and 2012 at the LSG Inlet and at Muskellunge Lake Outlet for 2001, 2011, and 2012 (Table 3; Fig. 5). Less detailed concentration data were also collected at both sites in 1999, 2000, 2002, and 2003. There was distinct seasonality in TP concentrations at both sites, with lower TP in fall and winter (about 0.030 mg/L at Muskellunge Lake Outlet and 0.040 mg/L at the LSG Inlet) and

higher TP during spring and summer (about 0.040 mg/L at Muskellunge Lake Outlet and 0.060 to 0.108 mg/L at the Inlet) (Fig. 5). About 45% of the P was in dissolved forms at Muskellunge Lake Outlet compared with about 55% at LSG Inlet. Highest TP concentrations were measured in summer of 1997 (0.108 mg/L), when streamflow was highest and beaver activity in the watershed was thought to be highest and lowest concentrations were measured from 2000 to 2011 (0.06 mg/L). The second highest concentrations occurred in 2012 (0.089 mg/L), when beaver activity and streamflow were believed to be lowest. Average-annual and average-summer streamflow at the Inlet ranged from about 0.3 m³/s (highest) in 1997 to 0.15 m³/s (lowest) in 2012 (Table 3). This resulted in total annual TP loading at the Inlet ranging from 668 kg in 1997 to 315-353 kg in all other years, and total May-Sept. loading ranging from 363 kg in 1997 to 139-155 kg in all other years (Table 3). Approximately 20-30% of the TP load reaching LSG originated from upstream Muskellunge Lake, the remainder came from sources in the area between the lakes (primarily groundwater and instream sources).

To determine if differences in sampling strategies among years affected average-annual and average-summer flows, concentrations, and loads, the streamflow and water-quality data from 2011–12 were subsampled similar to how the data were collected in previous years and the summary values were again determined. Reducing the streamflow and sampling frequency to monthly, resulted in only small changes in annual-average and summer-average: flows of 0.5–6.5% and -5.5–12.7%, TP loads of -1.7–7.1% and -5.5–16.2%, and volumetrically weighted TP concentrations of -2.2–3.3%, respectively. The difference in sampling strategies had only a small effect because about 50% of the flow in the stream came from Muskellunge Lake, and the intervening area between lakes was primarily forested. These resulted in flow being non-flashy

and TP concentrations having almost no relation to discharge ($R^2=0.03$). Therefore, the original differences among years were not caused by changes in sampling frequency.

The decrease in TP concentrations in 1999 from those in 1997 in Muskellunge Creek, which was believed to be primarily caused by reducing the amount of impounded water caused by beaver activity (Robertson et al. 2005), appeared to cause a decrease in TP throughout the lake (Fig. 2 and Table 2). Summer-average TP in East Bay decreased from 0.061 mg/L in 1997 to 0.038 mg/L in 1999; however, CHL concentrations increased and SDs decreased in 1999 from those in 1997. Continued efforts to completely eliminate impounded water in Muskellunge Creek, primarily started in 2006, resulted in relatively low TP concentrations in Muskellunge Creek (remaining below 0.09 mg/L) and low TP loading to LSG (Table 2); however, it is difficult to determine if maintaining low beaver activity has had a beneficial effect on the lake itself because of changes in the other factors affecting water quality. Only the initial reduction in impounded water caused by beaver activity (around 1998) appears to have had a beneficial impact on LSG water quality.

Winter aeration (internal loading)

The single largest long-term change in LSG's water quality during summer appears to be a rather abrupt change occurring between 2001 and 2002 (Fig. 2), which is coincident with the beginning of winter aeration. Although there is considerable interannual variability before and after this time, there appears to be a regime shift occurring throughout most of the lake. This change appears as an increase in summer-average TP (by about 0.010–0.015 mg/L) and CHL (by about 20 µg/L) in East, Upper East, and South Bays (Fig. 2). In addition, SDs decreased by 0.5–1.0 m in South Bay at this time. Although part of this change occurred in early summer, the

majority occurred in mid to late summer (Fig. 3). In August, TP increased by 0.020–0.030 mg/L and CHL increased by 30–40 µg/L in these three basins, and SDs in South Bay decreased by about 0.75 m (all statically significant at $p < 0.005$ student T tests with equal or unequal variance). Although other factors may have caused part of this change in water quality, none of them dramatically changed in 2002. The effect of winter aeration is further evaluated below after compensating for interannual variability in meteorological conditions among years.

Meteorological variability

Average annual (Oct-Sept) and summer (June-Aug) air temperatures and total annual (Oct-Sept) and summer (May-Aug) precipitation near LSG for 1991-2013 are shown in Fig. 6. Annual- and summer-average air temperatures, which ranged from 2.4 to 7.5°C and 15.2 to 19.9°C, respectively, increased by 0.10°C/year during this period ($p < 0.05$ using both linear regression and Mann-Kendall tests). Although much of northern Wisconsin experienced a drought around 2007–09 (Watras et al. 2013), only a small overall insignificant downward trend in annual and summer precipitation occurred near LSG ($p = 0.14$, $p = 0.40$, respectively). The changes in precipitation had little effect on water level in LSG because water levels are controlled by a dam at the LSG Outlet. As a result of the increases in air temperatures, ice-out dates for LSG may have become earlier. Ice-out dates for nearby Anvil Lake (~30 km east of LSG) are shown in Fig. 6. There was a slight downward trend in ice out dates (0.3 days/year); however, because of the considerable variability it was not statistically significant ($p = 0.41$).

Although the overall changes in air temperatures, precipitation, and ice-out dates from 1991 to 2013 are small, there are two things to note: 1) no abrupt change in any of these factors occurred in 2002 that should have caused the abrupt change in water quality, and 2) considerable

interannual variability is present in these factors that may affect water quality. Therefore, by comparing the annual water quality data with the annual meteorological data, it may be possible to determine how each of these meteorological/climatic factors affects water quality in LSG in the past, and how long-term changes in these factors as a result of climate change may affect water quality in the future. Variability in water quality of the Upper East Bay was not statistically evaluated because of the limited number of years it was sampled.

In general for the shallow East Bay, increases in air temperature (especially summer-average temperature) were significantly correlated with increased summer-average CHL and TP, and weakly correlated with decreases in SD (Table 4 and Fig. 7). Increases in TP and CHL were also weakly correlated with increases in precipitation and earlier ice-out dates. To further evaluate whether these relations were driven by interannual variability in meteorology and not simply co-linearity that may exist in these variables, i.e., correlations stemming from the fact that two variables are independently changing over time, the spearman correlations were also examined after all of the variables were adjusted to remove any long-term linear trends, i.e. detrended data were examined. After adjusting for the long-term trends, increases in summer air temperatures were still strongly correlated ($p < 0.05$) with increases in CHL and less strongly correlated with increases in TP ($p < 0.18$), and increases in precipitation, especially the last two years, were strongly correlated with increases in TP (Table 4).

Relations in South Bay were relatively similar to those found in East Bay; however, detrending the data had a slightly more dramatic effect. Therefore, part of the apparent relation between air temperature and TP in these two basins may have been caused by other factors affecting long-term changes in water quality, such as winter aeration.

Increases in air temperatures were also correlated with increases summer-average CHL and TP and decreases in SD in the dimictic West Bay (Table 4 and Fig. 7). The stronger relations with SD in West Bay may have resulted from more interannual variability in SD. After adjusting for the long-term trends, increases in summer air temperatures were still strongly correlated ($p < 0.05$) with increased CHL and decreased SD, but not correlated with changes in TP. Increases in precipitation, especially the last 2 years, was strongly correlated with increased TP ($p < 0.05$) and increased CHL ($p < 0.10$), but not correlated with SD.

The p values for many of these relations are relatively weak because of the few years that were available for this comparison, and because of the interactions of the driving factors.

Discussion

Factors affecting water quality and their relative importance

Hydrodynamics and polymixis—In general, the hydrodynamics, i.e., flow pattern through the lake and extent of vertical mixing, had the largest effect on the variability in water quality throughout LSG. East and Upper East Bays have the worst water quality (highest TP, CHL, and poorest water clarity, especially in late summer) and West Bay has the best water quality. Much of this difference is caused by East and Upper East Bays having a strong connection with the major external source of nutrients, Muskellunge Creek, and West Basin being partially isolated from the rest of the lake by a 1.5 m sill between East and West Bays. It wasn't described here, but the East and Upper East Bays also have the highest input of groundwater with relatively high P concentrations (Robertson et al. 2005). This flow pattern resulted in average spring TP decreasing from 0.040 mg/L in East Bay, to 0.033 mg/L in South Bay, to 0.021 mg/L in West Bay.

Another factor causing differences among basins is the difference in vertical mixing: East and Upper East Bays are polymictic and the other bays are more dimictic. TP concentrations in East and Upper East Bays increased as summer progressed. Welch and Cooke (1995) found that frequent mixing of near-bottom water in polymictic lakes, which have high P caused by microzones of anaerobic activity developing during short periods of stratification, results in near-surface TP increasing throughout summer. In comparison, West Bay is a dimictic system. Similar to other deeper, dimictic lakes, most P released from its sediments is retained in the hypolimnion during summer resulting in near-surface P concentrations remaining relatively stable as summer progresses. The combination of West Bay starting at a lower TP and having less P released to surface waters from internal loading, resulted in it having lower summer average TP than the rest of the lake in late summer. The difference in TP between West and East Bays was 0.019 mg/L in spring, but increased to 0.042 mg/L for the summer average. The elevated P concentrations in East and Upper East Bays then cause CHL to increase and SD to decrease as summer progresses.

South Bay had better water quality than East and Upper East Bays because it receives only part of the nutrients delivered to East and Upper East Bays. Although West Bay is partially isolated from the rest of the lake, the horizontal mixing that does occur with East Bay has caused the water quality of West Bay to gradually degrade. From 1991 to 2013, the West Bay has gone from being borderline oligo-mesotrophic to borderline meso-eutrophic (Fig. 4).

Meteorologic and climatic factors—Increases in summer air temperatures were strongly related to increases in CHL throughout the lake, decreases in SD in West Bay, and less strongly related to increases in TP in the East and South Bays (Table 4). Increases in the past few years of

precipitation were strongly related to increases in TP throughout the lake after removing the effects of long-term trends. The interannual differences in the meteorologic factors explain some of the interannual variability in water quality, especially during periods when there were no complicating factors such as after the implementation of winter aeration. For example, after implementing winter aeration in Dec. 2001, water quality was best in most of the lake (East, Upper East, and South Bays) in 2004 and 2008: lower than normal CHL and TP (Fig. 2). Both of these years had unusually cool summer air temperatures, late ice out, and little summer precipitation (Table 3). In 2006, the water quality was worse than almost all other years since 2001: high CHL and TP in East, Upper East, and South Bays, and worse water clarity in West Bay (Fig. 2). 2006 was a warm summer with high annual and summer precipitation (Table 3).

Variability in air temperatures had a strong effect on interannual variations in water quality. During warmer years, there were increases in CHL throughout the lake and decreases in SD, primarily in West Bay. This is consistent with Regier et al. (1990) and Jeppesen et al. (2014) who found that increases in air temperatures caused increases in water temperature and an increase in primary productivity.

The increases in TP in the polymictic basins of the lake during warm years that primarily occurred in the latter part of the record may be the result of three factors: 1) warmer air temperatures causing warmer water temperatures, which in turn cause higher P release rate from the sediments (James and Barko 2004; Jensen et al. 2006); 2) warmer air temperature causing earlier ice out and a longer period for frequent stratification/mixing events. Frequent mixing in polymictic lakes has been shown to result in near-surface TP increasing throughout summer (Welch and Cooke 1995); and 3) winter aeration primarily affecting the years after 2001. The possible long-term effects of winter aeration, removed by detrending the data, may be the reason

that the spearman correlation decreased from 0.55 to 0.31 for the East Bay and from 0.35 to 0.29 in the South Bay (Table 4). Higher TP in the polymictic basins and other polymictic systems, in turn should lead to increased productivity (higher CHL) and worse water clarity (shallower SD).

Variability in TP in deeper dimictic West Bay was only weakly correlated with changes in air temperatures (Table 4), especially after correcting for long-term trends. Lack of increase in near-surface TP during warm years may be the result of the relatively permanent stratification in West Bay inhibiting the transfer of P generated from internal P loading in this basin.

The effect of changes in precipitation is discussed with external loading.

Winter aeration—It appeared that the single largest long-term change in LSG summer water quality was the rather abrupt degradation occurring between 2001 and 2002, coincident with the beginning of winter aeration; none of the other factors examined explained this rather abrupt change. To better evaluate the effects of the anthropogenic factors (winter aeration and removal of beaver impoundments), summer-average TPs in the East, South, and West Bays were adjusted to remove the effects of the interannual differences in the most significant meteorological factors (summer air temperature and 2-yr average precipitation) using: $TP_{Residual} = TP_{Measured} - TP_{Predicted}$, where $TP_{Predicted} = \alpha (\text{summer air temperature}) + \beta (\text{2-yr average precipitation})$. Prior to removing the meteorological effects, average TP in the East Bay increased by 0.016 mg/L (Fig. 2) after 2001; however, there was considerable variability in concentrations. After removing the effects of meteorological variability, the rather abrupt change in water quality ($TP_{Residual}$) after 2001 in the East Bay is even more apparent (Fig. 8), especially when the data are smoothed with a 3-year moving average. After adjustments, the average $TP_{Residual}$ only increased by 0.011 mg/L after 2001, but the variability in the values decreased resulting in a statistically significant

increase ($p < 0.01$, based on a student t-test). To further evaluate changes in TP, the residuals were then examined using several abrupt transition Auto Regressive Integrated Moving Average (ARIMA) intervention models (Box and Tiao 1975). ARIMA models were used because there was significant ($p < 0.05$) lag-1 correlation in the data. Based on AIC values for various models (linear change, step change with and without linear changes in the before and after aeration periods), the best description of the time series (by at least 2 AIC units) is a constant $TP_{Residual}$ value up to 2001, followed by a step change, followed by a constant value after 2001. Therefore, there was significant degradation in water quality after 2001 that appears to be partly due to increases in air temperature, but mainly due to implementation of winter aeration. Changes in TP in South Bay (not shown) were also better explained with a step change after 2001 than a linear change over the entire period. TP in West Bay (Fig. 8) gradually changed from 1991 to 2013, and was better explained by a linear change over the entire period than a step change after 2001.

There are at least two possible explanations for deteriorated water quality following winter aeration (or reduction in winterkill): 1) changes in the food web, and/or 2) changes in bottom sediment composition. Winterkill can dramatically affect a lake's food web and water clarity (Haertel and Jongsma 1982). Haertel and Jongsma found that water clarity following winterkill improved because of increased zooplankton grazing of algae in the absence of or reduction in fish predation. Therefore, if winter aeration eliminates winterkill, then it is expected that following winter aeration, water clarity may get worse – similar to that observed in LSG. Following the use of winter aeration in LSG, the lack of winterkill has changed the fish community structure (increase in predatory fish and a decrease in bullheads, *Ameiurus natalis*) (S. Gilbert, WDNR, pers. comm. 2014), similar to that found by Haertel and Jongsma, but plankton data were not available for LSG to further evaluate this explanation. Haertel and

Jongsma, however, also showed that following winterkill, TP stayed relatively constant or increased. TP increased in LSG following winter aeration (and lack of winterkill), the opposite of that found by Haertel and Jongsma; therefore, this does not seem to be the likely or most important cause of deteriorated water quality.

The more likely reason for slightly degraded water quality following winter aeration is that it eliminated anoxia in the Upper East and South Bays, which reduced P release from the bottom sediments of these basins during winter (Robertson et al. 2005). Prior to aeration, some of the P released from the sediments during winter should have been transported out of the lake as water moves from the Upper East and East Bays, through the South Bay, and to the Outlet. However, by aeration eliminating P release during winter, the P content of the bottom sediment in the aerated basins should remain high, which would result in an increased summer P release rate (Rydin and Welch 1999). Increased P would then drive increased productivity (CHL) and decreased water clarity. This mechanism would explain the changes observed in East, Upper East, and South Bays (shown for East Bay in Fig. 3). The increased TP and CHL and decreased SDs in the East Bay during August post-2002 were all statistically significant ($p < 0.005$; student T test with equal or unequal variance).

The general water movement through the lake appeared to increase the spatial effectiveness of the aerators and affect the water quality through most of the lake. For example, aerators were not operated in South Bay in winter 2002, but water with increased dissolved oxygen concentrations caused by aeration in East and Upper East Bays slightly increased the dissolved oxygen concentrations in South Bay (Robertson et al. 2005). During summer 2002, the increased TP and CHL observed in East and Upper East Bays was transported to South Bay causing an increase in concentrations; however, the concentrations in South Bay were lower than the

upstream basins. Another example of the effects of aeration propagating downstream can be seen in the years following the discontinuation of the aerator in the East Bay in 2004. Following 2004, the degraded water quality in Upper East Bay appeared to continue to affect the water quality in East Bay.

Aerators operated during winter have been shown to have dramatic effects on the winter water quality in LSG (Robertson et al. 2005) and other lakes (Hudson and Kirschner 1997), and operated during summer have also been shown to have dramatic effects on summer water quality (Cooke et al. 1993), but few studies other than this one have examined how aerators operated during only the winter may affect summer water quality.

Beaver activity and external loading—Reduction in TP concentrations in Muskellunge Creek from 1997 to 1999 and continued lower concentrations after 1999 (Table 3) is believed to be caused by initially reducing the extent of the beaver impoundments (large effect) and thereafter maintaining low beaver activity (limited additional effect). Previous studies have shown that beaver activity can have a range of effects on TP concentrations in streams. Beaver activity has been found to decrease TP (Muskopf 2007), especially in streams with high suspended sediment and sediment bound P (Maret et al. 1987); however, suspended sediment is quite low in Muskellunge Creek. Klotz (1988, 1998) found that beaver ponds can increase TP in streams because of microbial decomposition at the sediment water interface leading to anoxic conditions, much like that described in the polymictic basins of LSG. When beaver impoundments are removed, TP can initially increase because of erosion of the sediments behind beaver dams, but later when sediments become aerobic, they absorb P (Klotz 1998). This may be why TP decreased in Muskellunge Creek in 1999. Recent efforts to keep beavers out of the watershed

have kept TP low in Muskellunge Creek; however, if beaver activity increases in the future, impounded water could result in increased TP concentrations in the stream.

In most lakes, variations in external P loading drive interannual and long-term changes in water quality and was shown to be significantly related to TP concentrations in LSG (Table 4). But, the question remains: Has the removal of beaver impoundments affected LSG water quality? Decreases in TP concentrations and loads in Muskellunge Creek from 1997 to 1999 appeared to decrease TP in LSG (summer-average TP in East Bay decreased from 0.061 mg/L in 1997 to 0.038 mg/L in 1999; and the residuals after compensating for differences in air temperature and precipitation decreased by 0.017 mg/L in Fig. 8). But, it appears that variability in external loading caused by differences in beaver activity has only a weak consistent effect on LSG's water quality (although external loading to LSG was only documented for five years). All internal and other climatic factors being equal, given the measured external flow and P loading in those years, water quality in LSG in 2001, 2011, and 2012 would have been expected to be similar to 1999. But, the beneficial effects of reduced loads were not observed in LSG after 2002 possibly because of other factors affecting water quality (such as winter aeration). Prior to 1997, it is difficult to quantify the effects of beaver activity, but it is believed to be similar to 1997. If TP concentrations in Muskellunge Creek prior to 1997 were similar to those in 1997, then beaver activity, which appears to have affected stream water quality, appears to have had little effect on LSG's water quality, which was quite variable (Figs. 3 and 8).

In most lakes, it may be expected that variations in precipitation and external loading would have a larger effect on downstream water quality than observed here. Part of the reason for the limited interannual variability in external loading to LSG and limited effects on lake water quality may be the buffering effect of the upstream lake and the relatively pristine area between

the lakes that is dominated by groundwater inputs and results in P concentrations in the inlet stream being about the same as in East Bay (Fig. 5). Effects of external loading are expected to be more dramatic in lakes with non-pristine watersheds (e.g., agricultural or urban watersheds) or watersheds of streams not dominated by groundwater contributions.

Future changes in the water quality of LSG and other shallow lakes

The relations found between interannual variability in LSG's weather and water quality enable us to project how LSG and other temperate lakes should behave in the future if air temperatures increase as predicted by various General Circulation Models (IPCC 2007). If air temperatures increase we can expect: earlier ice-out dates (Robertson et al. 1992; Magnuson et al. 2000), warmer water temperatures (Robertson and Ragotzkie 1990; Winslow et al. 2014), and a longer period of thermal stratification or period of polymixis (Robertson et al. 1992; IPCC 2007; WICCI 2011). These climate-induced changes should result in increases in TP and productivity (CHL), and a slight reduction in water clarity (SD) in East, Upper East, and South Bays of LSG and in other shallow polymictic lakes. Although the slopes in Figure 7 may be an indication of how these variables would be expected to change with increases in air temperature, part of these relations was caused by the implementation of winter aeration and therefore may overestimate what may be expected. These climatic changes should also result in increases in productivity (CHL) and reduction in water clarity (SD) in West Bay of LSG and other dimictic lakes. Degradation in water quality with increased warming is consistent with that found in other field and modeling studies on shallow lakes (Mooji et al. 2005; Mooji et al. 2007; Jeppesen et al. 2014).

Climate models predict that Wisconsin's summer air temperatures may increase 1.7 to 4.4°C by the middle of the 21st century (WICCI 2011). This type of climate change could dramatically increase TP and CHL and decrease water clarity in LSG and other temperate lakes.

Importance of long-term datasets

Various approaches are used to estimate how specific factors, such as winter aeration or climate change, affect the water quality of lakes, including field experiments, numerical models, space for time substitutions, and time series analyses (Jeppesen et al. 2014). Each of these approaches has its strengths and weaknesses. Field experiments often suffer from being too simple, typically only examining specific variables, and therefore not being realistic (Jeppesen et al. 2014). Complex dynamic models usually require comprehensive knowledge of numerous processes and therefore require calibration of a large number of unknown parameters (Jorgenson and Mitsch 1983). Because of the relatively sparse data that are typically available, complex models are often overfit and can lead to improper conclusions (McDonald et al. 2010). Causal relationships from space for time studies are often difficult to identify because of the spatial variability in the relationships and other differences occurring among the systems (Jeppesen et al. 2014). Time series studies examining an entire lake ecosystem, such as that used here for LSG, overcome some of the weaknesses of the other approaches. However, few ecosystem studies are conducted over a period as long as evaluated here; therefore, it is difficult to observe the complete effects of the factor(s) being examined (Carpenter 1998). Time series studies also often suffer from having multiple factors possibly leading to the observed change or lack of change.

In LSG, various short-term “monitoring” studies were conducted in the process of trying to better manage the lake. This has resulted in a relatively consistent long-term dataset, which has

enabled us to conduct indepth time-series analyses to determine the relative importance of the various factors affecting water quality, including the effects of management actions such as winter aeration, and also to suggest what types of changes in water quality may occur with climate change. This resulting dataset has been termed by Magnuson (1990), as the “invisible present” from long-term studies. Results from the investigation of the LSG dataset may be argued to be site specific, however, only through replicate ecosystem experiments (Carpenter 1998) can we be more certain that management actions, such as winter aeration, consistently affect water quality in other lakes. Additional studies using a range of the approaches described above also will better enable the full range in potential effects of climate change to be estimated.

This study demonstrates that many factors affect the spatial patterns and interannual variability in water quality in LSG. Much of the spatial variability in the water quality of LSG was caused by the general flow pattern through the lake isolating West Bay from most external loading, and the variable extent of vertical mixing caused by variations in basin morphometry. Water quality in the shallow, polymictic parts of the lake (East, Upper East, and South Bays) deteriorated (increasing TP and CHL and decreasing SDs) as summer progresses, compared to the deeper, dimictic parts of the lake. From 1991 to 2013, LSG had several factors that may have improved and degraded its water quality. Initial removal of beaver impoundments around 1998 may have decreased the external P loading, resulting in possibly a short-term improvement in water quality. Winter aeration started in the winter of 2002 eliminated winterkills and improved the fishery in LSG, but appears to have increased internal P loading during summer resulting in higher TP and CHL. Interannnnual variations in weather (air temperature and precipitation) have

resulted in considerable interannual variability in water quality. During warmer and wetter years, TP and CHL are higher and SDs are lower.

If air temperatures increase in the future, warmer water temperatures and earlier ice out are expected in all temperate lakes. These changes in turn should result in: increases in nutrient concentrations (TP) and productivity (CHL), and slight reductions in water clarity (SD) in the East, Upper East, and South Bays of LSG and in other polymictic lakes; and increases in lake productivity and reductions in water clarity in West Bay of LSG and in other dimictic lakes.

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Table 1. Morphometry of Little St. Germain Lake and its basins,
[NA, not applicable]

Basin/Total	Maximum Depth m	Mean Depth m	Area ha	Volume km ³
Upper East Bay	4.6	2.2	48.1	1.1
East Bay	3.7	2.5	163.8	4.1
West Bay	16.2	7.1	86.4	6.3
South Bay	6.4	2.8	97.5	2.7
Entire Lake	16.2	3.5	395.8	14.2

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Table 2. Summer-average (June-August) and August water quality in the East and West Bays of Little St. Germain Lake. [TP, total phosphorus, mg/L; CHL, chlorophyll a, ug/L; SD, Secchi depth, m; -, not available]

Year	East Bay							West Bay						
	Spring		Summer Average			August		Spring		Summer Average			August	
	TP	TP	CHL	SD	TP	CHL	SD	TP	TP	CHL	SD	TP	CHL	SD
1991	^e 0.067	0.047	46.0	1.0	0.023	24.0	0.8	^e 0.014	0.010	3.0	2.6	0.008	2.0	3.2
1992	0.033	0.050	36.7	1.1	0.070	55.0	0.8	0.012	0.011	4.3	3.4	0.011	4.9	3.2
1993	0.031	0.038	48.3	1.0	0.056	98.0	0.7	0.012	0.009	3.0	3.9	0.006	2.2	4.5
1994	0.033	0.049	35.3	1.0	0.058	28.0	0.9	0.016	0.010	2.1	3.7	0.012	0.7	3.2
1995	-	-	-	-	-	-	-	-	-	-	3.6	-	-	3.7
1996	-	-	-	-	0.030	35.4	1.0	-	-	-	3.2	0.007	3.8	2.9
1997	0.062	0.061	22.8	1.7	0.071	44.0	1.0	0.025	0.026	2.5	4.0	0.016	2.9	3.6
1998	-	-	-	-	-	-	-	-	-	-	-	-	-	-
1999 ^{ad}	0.030	0.038	33.5	1.0	0.052	60.6	0.7	0.015	0.012	4.8	2.8	0.013	7.3	2.6
2000	0.024	0.053	22.7	1.0	0.087	30.0	0.6	0.012	0.010	2.7	3.1	0.009	2.0	2.8
2001	0.033	0.056	42.7	0.9	0.069	55.0	0.8	0.015	0.021	7.1	3.6	0.023	3.2	4.1
2002	0.046	0.075	58.5	0.9	0.116	94.5	0.6	0.019	0.016	8.0	2.6	0.020	16.0	1.6
2003	0.049	0.072	67.8	0.9	0.102	105.0	0.6	0.021	0.015	6.0	2.8	0.014	8.3	2.5
2004	0.037	0.043	45.8	0.9	0.054	63.2	0.6	0.014	0.014	4.4	2.9	0.015	5.3	2.4
2005 ^{bd}	0.037	0.057	51.6	0.9	0.076	61.0	0.8	0.020	0.014	8.8	-	0.017	16.4	0.0
2006	0.038	0.080	69.4	0.8	0.116	89.1	0.6	0.024	0.022	9.4	2.1	0.018	16.2	1.8
2007 ^a	0.054	0.056	57.5	0.9	0.076	68.4	0.6	0.028	0.017	4.7	2.4	0.015	7.4	2.3
2008	0.041	0.046	39.4	1.0	0.068	73.3	0.6	0.022	0.016	5.5	2.6	0.014	5.4	2.1
2009 ^{ab}	0.040	0.068	46.0	0.9	0.092	65.1	0.6	0.033	0.017	5.5	3.0	0.015	7.3	2.6
2010	^e 0.031	0.071	64.1	1.1	0.103	103.0	0.7	^e 0.020	0.017	10.1	2.6	0.021	14.9	1.9
2011 ^c	0.028	0.076	46.9	1.0	0.128	86.3	0.7	0.029	0.020	9.6	2.4	0.019	8.7	2.3
2012	^e 0.042	0.071	56.1	0.9	0.086	44.6	0.6	^e 0.023	0.015	6.9	3.2	0.014	10.8	2.3
2013	0.044	0.062	52.0	0.8	0.079	64.8	0.7	0.043	0.024	13.9	2.7	0.018	6.1	2.8
Average	0.039	0.058	47.2	1.0	0.077	64.2	0.7	0.021	0.016	6.1	3.0	0.014	7.2	2.6
Average (2002-13)	0.041	0.065	54.6	0.9	0.091	76.5	0.6	0.025	0.017	7.7	2.7	0.017	10.2	2.0

^aNot sampled for TP and CHL in August; ^bNot sampled for TP and CHL in June; ^cNot sampled for TP and CHL in July: monthly values were estimated from regressions with nearby months prior to summer averages being computed.

^dSD not measured in August 1999 and June 2005: monthly values were estimated from regressions with nearby months prior to summer averages being computed.

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^eSpring TP not measured and estimated from June concentration.

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Table 3. Factors affecting the water quality in Little St. Germain Lake. [TP, total phosphorus; Load in kg; Conc, concentration in mg/L; -, not available or applicable]

External Loading												Internal		Climate				Ice-out Date Anvil Lake Day #	
Muskellunge Lake Outlet				Little St. Germain Inlet						Beaver Activity ^a (1-3)	Winter Aeration ^b	Water Level	Precipitation		Air temperature				
Year	Annual		Summer (May-Aug)		Flow m3/s	Annual		Summer (May-Aug)			Winter (N-Y)	Summer m	Annual mm	Summer mm (May-Aug)	Annual C	Summer C (June-Aug)			
	TP	TP	TP	TP		Flow	TP	TP	Flow								TP		TP
	Load	Conc	Load	Conc		Load	Conc	Load	Conc								Load		Conc
1991	-	-	-	-	-	-	-	-	-	-	3	N	491.86	1011	458	5.4	18.6	106	
1992	-	-	-	-	-	-	-	-	-	-	3	N	491.85	823	326	4.0	15.2	117	
1993	-	-	-	-	-	-	-	-	-	-	3	N	491.88	787	390	4.0	16.9	104	
1994	-	-	-	-	-	-	-	-	-	-	3	N	491.83	777	358	3.8	17.2	111	
1995	-	-	-	-	-	-	-	-	-	-	3	N	491.87	765	442	5.3	19.2	110	
1996	-	-	-	-	-	-	-	-	-	-	3	N	491.88	934	404	2.4	16.6	136	
1997	-	-	-	-	0.301	668	0.070	0.316	363	0.108	3	N	491.89	897	412	3.7	17.3	117	
1998	-	-	-	-	-	-	-	-	-	-	2	N	491.80	651	291	6.8	18.5	98	
1999	-	-	-	-	0.171	315	0.058	0.195	155	0.075	2	N	491.86	862	535	6.1	18.5	104	
2000	-	0.034	-	0.034	-	-	0.049	-	-	0.061	2	N	491.82	780	428	6.0	17.3	93	
2001	94	0.034	41	0.042	0.198	320	0.051	0.200	147	0.069	2	Y - 3	491.83	696	362	5.4	18.7	111	
2002	-	0.045	-	0.052	-	-	0.052	-	-	0.064	2	Y - 3	491.87	990	409	6.3	19.3	108	
2003	-	0.040	-	0.047	-	-	0.050	-	-	0.060	2	Y - 3	491.84	699	305	4.5	18.2	110	
2004	-	-	-	-	-	-	-	-	-	-	2	Y - 2	491.83	740	286	5.1	16.2	111	
2005	-	-	-	-	-	-	-	-	-	-	2	Y - 2	491.67	726	306	6.3	19.9	102	
2006	-	-	-	-	-	-	-	-	-	-	2	Y - 2	491.75	869	417	6.6	19.3	103	
2007	-	-	-	-	-	-	-	-	-	-	2	Y - 2	491.76	602	232	6.3	19.4	108	
2008	-	-	-	-	-	-	-	-	-	-	2	Y - 2	491.82	768	312	5.0	17.9	119	
2009	-	-	-	-	-	-	-	-	-	-	2	Y - 2	491.74	582	290	4.8	16.8	108	
2010	-	-	-	-	-	-	-	-	-	-	1	Y - 2	491.83	1005	535	6.8	19.9	88	
2011	114	0.048	41	0.047	0.184	353	0.061	0.197	141	0.067	1	Y - 2	491.84	695	337	5.8	19.6	118	
2012	67	0.034	23	0.036	0.152	329	0.069	0.147	139	0.089	1	Y - 2	491.71	687	378	7.5	19.9	80	
2013	-	-	-	-	-	-	-	-	-	-	1	Y - 2	491.85	829	408	5.8	18.8	126	
Average	-	-	-	-	-	-	-	-	-	-	-	-	491.82	790	375	5.4	18.2	108	
Average (2002-13)	-	-	-	-	-	-	-	-	-	-	-	-	491.79	766	351	5.9	18.8	107	

Beaver Activity^a - 3 indicates no control, 2 indicates first efforts at reducing impounded waters, 1 indicates attempts to totally eliminate beavers and impounded water.Winter Aeration^b - N indicates no aeration, Y - 3 indicates 3 aerators in operation, and Y - 2 indicates two aerators in operation.

Table 4. Spearman correlations between summer-average Total Phosphorus (TP), Chlorophyll a (CHL) concentrations, Secchi Depths (SD) and climatic factors. Total summer precipitation is for May-August, whereas summer-average air temperature is for June-August [Boxed, BOLD, and shaded, $p < 0.05$; Bold and shaded, $0.05 < p < 0.10$; Bold, $0.1 < p < 0.2$]

Factor	TP	CHL	SD	TP	CHL	SD	TP	CHL	SD
	East Bay			South Bay			West Bay		
Air Temp - Summer	0.55	0.61	-0.19	0.35	0.47	-0.38	0.40	0.71	-0.60
Air Temp - Annual	0.42	0.54	-0.29	0.30	0.61	-0.48	0.25	0.63	-0.58
PPT - Summer	0.03	-0.07	0.34	-0.12	-0.28	0.48	-0.02	0.04	-0.02
PPT - Annual	-0.29	-0.20	0.47	-0.03	-0.24	0.40	-0.36	-0.46	0.50
PPT - 2 Yr	0.16	-0.05	0.47	0.14	-0.31	0.33	-0.08	-0.22	-0.10
Ice-out Date	-0.09	-0.30	0.05	-0.02	-0.27	0.04	0.34	-0.04	0.04
	East Bay (Detrended)			South Bay (Detrended)			West Bay (Detrended)		
Air Temp - Summer	0.31	0.53	-0.19	0.29	0.35	-0.12	0.12	0.48	-0.44
Air Temp - Annual	0.08	0.31	-0.04	0.13	0.34	-0.19	-0.22	0.27	-0.42
PPT - Summer	0.18	0.02	0.10	-0.17	-0.23	0.43	0.08	0.44	-0.12
PPT - Annual	0.34	0.25	0.42	0.17	0.17	0.22	0.17	0.20	0.21
PPT - 2 Yr	0.66	0.25	0.29	0.41	0.11	0.02	0.46	0.39	-0.21
Ice-out Date	-0.07	-0.38	-0.04	0.07	-0.29	-0.04	0.59	0.06	0.10

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Figure 1. Drainage basin of Little St. Germain Lake, Wisconsin, with basins and monitoring locations identified.

Figure 2. Summer-average total phosphorus, chlorophyll a, and Secchi depths for the four basins in Little St. Germain Lake, Wisconsin from 1991 to 2013. The time when winter aeration was started is identified on the graphs as gray bars.

Figure 3. Monthly-average total phosphorus, chlorophyll *a*, and Secchi depths for 1991 to 2013, Pre 2002 (1991-2001, 8 years), and Post 2002 (2002-13, 12 years) for the East and West Bays of Little St. Germain Lake, Wisconsin.

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Figure 4. Trophic State Index values (Carlson, 1977) based on Secchi depth values for June and

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August for the four basins of Little St. Germain Lake from 1991 to 2013. The time when

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winter aeration was started is identified on the graphs as gray bars.

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Figure 5. Phosphorus concentrations and streamflow in Muskellunge Creek from 1996 to 2012.

Figure 6. Meteorologic data (precipitation, air temperature, and ice-out date (Julian Day, number of days since Jan. 1) for the area near Little St. Germain Lake, Wisconsin. Significance of each trend was determined using Mann-Kendall and linear regression statistics.

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920 Figure 7. Summer-average total phosphorus and chlorophyll *a* concentrations and Secchi depths
 921 for the East and West Bays of Little St. Germain Lake as a function of summer air temperature.
 922 The lines are labeled with their respective regression equations (the slope reflects the change per
 923 degree change in summer air temperatures). Relations (Spearman *r*) significant at $p < 0.05$ are
 924 shaded, and those significant at $p < 0.10$ have borders but not shaded.

Figure 8. Residual (observed – predicted) summer-average total phosphorus concentrations in the East and West Bays of Little St. Germain Lake. Total phosphorus concentrations were predicted using summer average air temperature and 2-yr average precipitation. A 3-year moving average, and overall average (East Bay), and overall trend (West Bay) are plotted for each period (before and after Dec. 2001).