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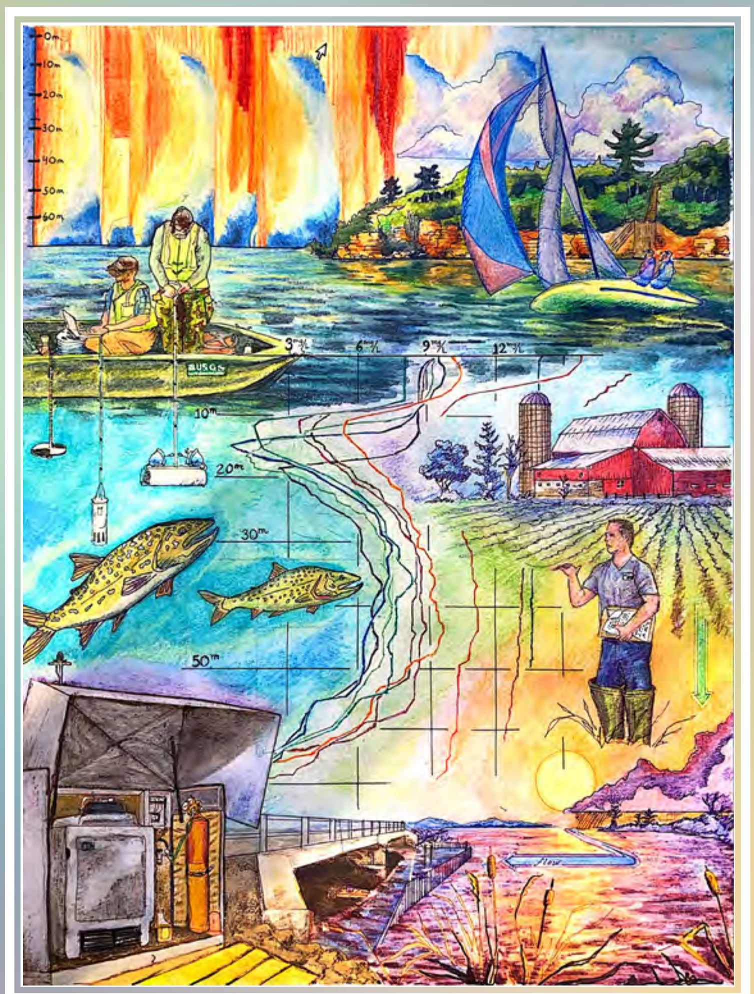
Response of Green Lake, Wisconsin, to Changes in Phosphorus Loading, With Special Emphasis on Near-Surface Total Phosphorus Concentrations and Metalimnetic Dissolved Oxygen Minima



Scientific Investigations Report 2022–5003

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

Cover. Illustration created by Ben Siebers, U.S. Geological Survey.

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By Dale M. Robertson, Benjamin J. Siebers, Robert Ladwig, David P. Hamilton, Paul C. Reneau, Cory P. McDonald, Stephanie Prellwitz, and Richard C. Lathrop

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Conversion Factors

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square kilometer (km^2)	247.1	acre
square meter (m^2)	10.76	square foot (ft^2)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
milliliter (mL)	0.03381	ounce, fluid (fl. oz)
liter (L)	33.81	ounce, fluid (fl. oz)
liter (L)	0.2642	gallon (gal)
cubic meter (m^3)	264.2	gallon (gal)
cubic meter (m^3)	35.31	cubic foot (ft^3)
cubic meter (m^3)	0.0008107	acre-foot (acre-ft)
	Flow rate	
cubic meter per second (m^3/s)	70.07	acre-foot per day (acre-ft/d)
cubic meter per year (m^3/yr)	0.000811	acre-foot per year (acre-ft/yr)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
cubic meter per day (m^3/d)	35.31	cubic foot per day (ft^3/d)
cubic meter per day (m^3/d)	264.2	gallon per day (gal/d)
cubic meter per second (m^3/s)	22.83	million gallons per day (Mgal/d)
	Precipitation rate	
meter per year (m/yr)	39.37	inch per year (in/yr)

Multiply	By	To obtain
	Mass	
kilogram (kg)	2.205	pound avoirdupois (lb)
kilogram per year (kg/yr)	2.205	pound avoirdupois per year (lb/yr)
	Density	
kilogram per cubic meter (kg/m^3)	0.06242	pound per cubic foot (lb/ft^3)
	Deposition rate	
kilogram per hectare per day ([kg/ha]/d)	0.8921	pound per acre per year ([lb/acre]/d)
kilogram per hectare per month ([kg/ha]/month)	0.8921	pound per acre per year ([lb/acre]/month)
kilogram per hectare per year ([kg/ha]/yr)	0.8921	pound per acre per year ([lb/acre]/yr)
kilogram per square kilometer per year ([kg/km^2]/yr)	0.008921	pound per acre per year ([lb/acre]/yr)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F = (1.8 \times °C) + 32.$$

Datum

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

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

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

Supplemental Information

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm).

Concentrations of chemical or biological constituents in water are given in either milligrams per liter (mg/L), micrograms per liter (µg/L), or millimoles per cubic meter ($mmol/m^3$). Atmospheric deposition and sediment demand are given in millimoles per square meter per day ($mmol/m^2/d$). Water column flux rates are given in millimoles per cubic meter per day ($mmol/m^3/d$). Particle size is given in micrometers (µm).

Abbreviations

AED	Aquatic Ecodynamics modeling library
Chl-*a*	chlorophyll-*a*
DO	dissolved oxygen
DRP	dissolved reactive phosphorus
fDOM	fluorescent dissolved organic matter
GCLAS	Graphical Constituent Loading Analysis System
GIS	geographic information system
GLM	General Lake Model
GLM–AED	General Lake Model coupled to the Aquatic Ecodynamics modeling library
HWY	highway
K_W	light extinction
MOM	metalimnetic dissolved oxygen minimum concentration
NSE	Nash-Sutcliffe efficiency
NWIS	National Water Information System
P	phosphorus
r^2	coefficient of determination
RMSE	root mean square error
SD	Secchi disk depth
SWAT	Soil and Water Assessment Tool (model)
SWIMS	Surface Water Integrated Monitoring System (database of the Wisconsin Department of Natural Resources)
TMDL	total maximum daily load
TP	total phosphorus
TSI	trophic state index
USGS	U.S. Geological Survey
WDNR	Wisconsin Department of Natural Resources
WY	water year (a 12-month period from October 1 of one year to September 30 of the following year)

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By Dale M. Robertson,[1] Benjamin J. Siebers,[1] Robert Ladwig,[2] David P. Hamilton,[3] Paul C. Reneau,[1] Cory P. McDonald,[4] Stephanie Prellwitz,[5] and Richard C. Lathrop[6]

Abstract

Green Lake is the deepest natural inland lake in Wisconsin, with a maximum depth of about 72 meters. In the early 1900s, the lake was believed to have very good water quality (low nutrient concentrations and good water clarity) with low dissolved oxygen (DO) concentrations occurring in only the deepest part of the lake. Because of increased phosphorus (P) inputs from anthropogenic activities in its watershed, total phosphorus (TP) concentrations in the lake have increased; these changes have led to increased algal production and low DO concentrations not only in the deepest areas but also in the middle of the water column (metalimnion). The U.S. Geological Survey has routinely monitored the lake since 2004 and its tributaries since 1988. Results from this monitoring led the Wisconsin Department of Natural Resources (WDNR) to list the lake as impaired because of low DO concentrations in the metalimnion, and they identified elevated TP concentrations as the cause of impairment.

As part of this study by the U.S. Geological Survey, in cooperation with the Green Lake Sanitary District, the lake and its tributaries were comprehensively sampled in 2017–18 to augment ongoing monitoring that would further describe the low DO concentrations in the lake (especially in the metalimnion). Empirical and process-driven water-quality models were then used to determine the causes of the low DO concentrations and the magnitudes of P-load reductions needed to improve the water quality of the lake enough to meet multiple water-quality goals, including the WDNR's criteria for TP and DO.

Data from previous studies showed that DO concentrations in the metalimnion decreased slightly as summer progressed in the early 1900s but, since the late 1970s, have typically dropped below 5 milligrams per liter (mg/L), which is the WDNR criterion for impairment. During 2014–18 (the baseline period for this study), the near-surface geometric mean TP concentration during June–September in the east side of the lake was 0.020 mg/L and in the west side was 0.016 mg/L (both were above the 0.015-mg/L WDNR criterion for the lake), and the metalimnetic DO minimum concentrations (MOMs) measured in August ranged from 1.0 to 4.7 mg/L. The degradation in water quality was assumed to have been caused by excessive P inputs to the lake; therefore, the TP inputs to the lake were estimated. The mean annual external P load during 2014–18 was estimated to be 8,980 kilograms per year (kg/yr), of which monitored and unmonitored tributary inputs contributed 84 percent, atmospheric inputs contributed 8 percent, waterfowl contributed 7 percent, and septic systems contributed 1 percent. During fall turnover, internal sediment recycling contributed an additional 7,040 kilograms that increased TP concentrations in shallow areas of the lake by about 0.020 mg/L. The elevated TP concentrations then persisted until the following spring. On an annual basis, however, there was a net deposition of P to the bottom sediments.

Empirical models were used to describe how the near-surface water quality of Green Lake would be expected to respond to changes in external P loading. Predictions from the models showed a relatively linear response between P loading and TP and chlorophyll-*a* (Chl-*a*) concentrations in the lake, with the changes in TP and Chl-*a* concentrations being less on a percentage basis (50–60 percent for TP and 30–70 percent for Chl-*a*) than the changes in P loading. Mean summer water clarity, quantified by Secchi disk depths, had a greater response to decreases in P loading than to increases in P loading. Based on these relations, external P loading to the lake would need to be decreased from 8,980 kg/yr to about 5,460 kg/yr for the geometric mean June–September TP

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concentration in the east side of the lake, with higher TP concentrations than in the west side, to reach the WDNR criterion of 0.015 mg/L. This reduction of 3,520 kg/yr is equivalent to a 46-percent reduction in the potentially controllable external P sources (all external sources except for precipitation, atmospheric deposition, and waterfowl) from those measured during water years 2014–18. The total external P loading would need to decrease to 7,680 kg/yr (a 17-percent reduction in potentially controllable external P sources) for near-surface June–September TP concentrations in the west side of the lake to reach 0.015 mg/L. Total external P loading would need to decrease to 3,870–5,320 kg/yr for the lake to be classified as oligotrophic, with a near-surface June–September TP concentration of 0.012 mg/L.

Results from the hydrodynamic water-quality model GLM–AED (General Lake Model coupled to the Aquatic Ecodynamics modeling library) indicated that MOMs are driven by external P loading and internal sediment recycling that lead to high TP concentrations during spring and early summer, which in turn lead to high phytoplankton production, high metabolism and respiration, and ultimately DO consumption in the upper, warmer areas of the metalimnion. GLM–AED results indicated that settling of organic material during summer might be slowed by the colder, denser, and more viscous water in the metalimnion and thus increase DO consumption. Based on empirical evidence from a comparison of MOMs with various meteorological, hydrologic, water quality, and in-lake physical factors, MOMs were lower during summers, when metalimnetic water temperatures were warmer, near-surface Chl-*a* and TP concentrations were higher, and Secchi depths were lower. GLM–AED results indicated that the external P load would need to be reduced to about 4,060 kg/yr, a 57-percent reduction from that measured in 2014–18, to eliminate the occurrence of MOMs less than 5 mg/L during more than 75 percent of the years (the target provided by the WDNR).

Large reductions in external P loading are expected to have an immediate effect on the near-surface TP concentrations and metalimnetic DO concentrations in Green Lake; however, it may take several years for the full effects of the external-load reduction to be observed because internal sediment recycling is an important source of P for the following spring.

Introduction

Green Lake is the deepest natural inland lake in Wisconsin, with a maximum depth of 72 meters (m) (fig. 1). Green Lake is a dimictic lake that becomes thermally stratified during summer and under winter ice and has a two-story fishery (in other words, it supports both warm- and cold-water fish species). As in many lakes in agricultural and urban areas, the water quality in Green Lake has declined as a result of cultural eutrophication because of excessive phosphorus (P) input from its watershed (Litton and others, 1972; Stauffer, 1985a; Panuska, 1999; Sesing, 2015; Schindler and others, 2016). The high P input has resulted in high total phosphorus (TP) and algal concentrations (chlorophyll-*a* [Chl-*a*]) in the epilimnion of the lake and low dissolved oxygen (DO) concentrations in the metalimnion and hypolimnion.

Based on sediment core analyses, Green Lake is believed to have been oligotrophic with very good water quality in the 1800s (Garrison, 2002; Sesing, 2015). As a result of increased nutrient (P and nitrogen) inputs, the water quality in the lake began to degrade around 1930, as TP concentrations increased and water clarity decreased (Garrison, 2002). Continued high P inputs, especially from agricultural inputs from the basin and effluent from the Ripon wastewater treatment plant that discharges to Silver Creek, further increased TP concentrations in the lake (Stauffer, 1985a).

As a result of the cultural eutrophication, several studies and monitoring efforts were conducted on Green Lake. Litton and others (1972) documented the water quality of the lake in 1972 and estimated the inputs of external sources of P. They estimated the total P input to the lake around 1970 to be 11,100 kilograms per year (kg/yr), with wastewater treatment plant input representing about 50 percent of the external loading (inputs from the atmosphere, waterfowl, and internal sediment recycling were not included). Stauffer (1985a, b) documented the water quality of the lake in 1971, 1972, and 1978, estimated the external P loading to the lake, and developed a model to explain how lakes respond to P loading from their watersheds. Stauffer (1985a) estimated the external P loading to the lake to be about 16,500 kg/yr during 1971–72 and 4,150 kg/yr in 1978 (inputs from septic systems, waterfowl, and internal sediment recycling were not included). Between 1972 and 1978, the Ripon wastewater treatment plant was modified to reduce its exported P load. Stauffer (1985a) estimated that this improvement reduced P in its effluent by about 90 percent, which reduced his estimated P loading to the lake by almost 75 percent. Stauffer estimated that the P load from the wastewater treatment plant prior to 1976 represented about 80 percent of the external P loading to the lake. Even after reduced P input from the upstream wastewater treatment plant, TP concentrations in the lake remained relatively high. From 1981 through 2001, the Wisconsin Department of Natural Resources (WDNR) documented the water quality of the lake, initially in detail and then on approximately a quarterly basis as part of the WDNR's Long-Term Trends network. In 1997–98, Panuska (1999) expanded the WDNR sampling schedule to monthly during the open-water period, quantified the external P loading to the lake, and used the Canfield-Bachmann natural-lake model (Canfield and Bachmann, 1981) to estimate how changes in P loading to Green Lake would be expected to affect its water quality. Panuska estimated that the TP loading to the lake during 1997–98 was about 8,440 kg/yr (inputs from waterfowl and internal sediment recycling were not included). From 2004 to 2020, the U.S. Geological Survey (USGS) monitored the water quality of Green Lake on approximately a monthly basis during the open-water period

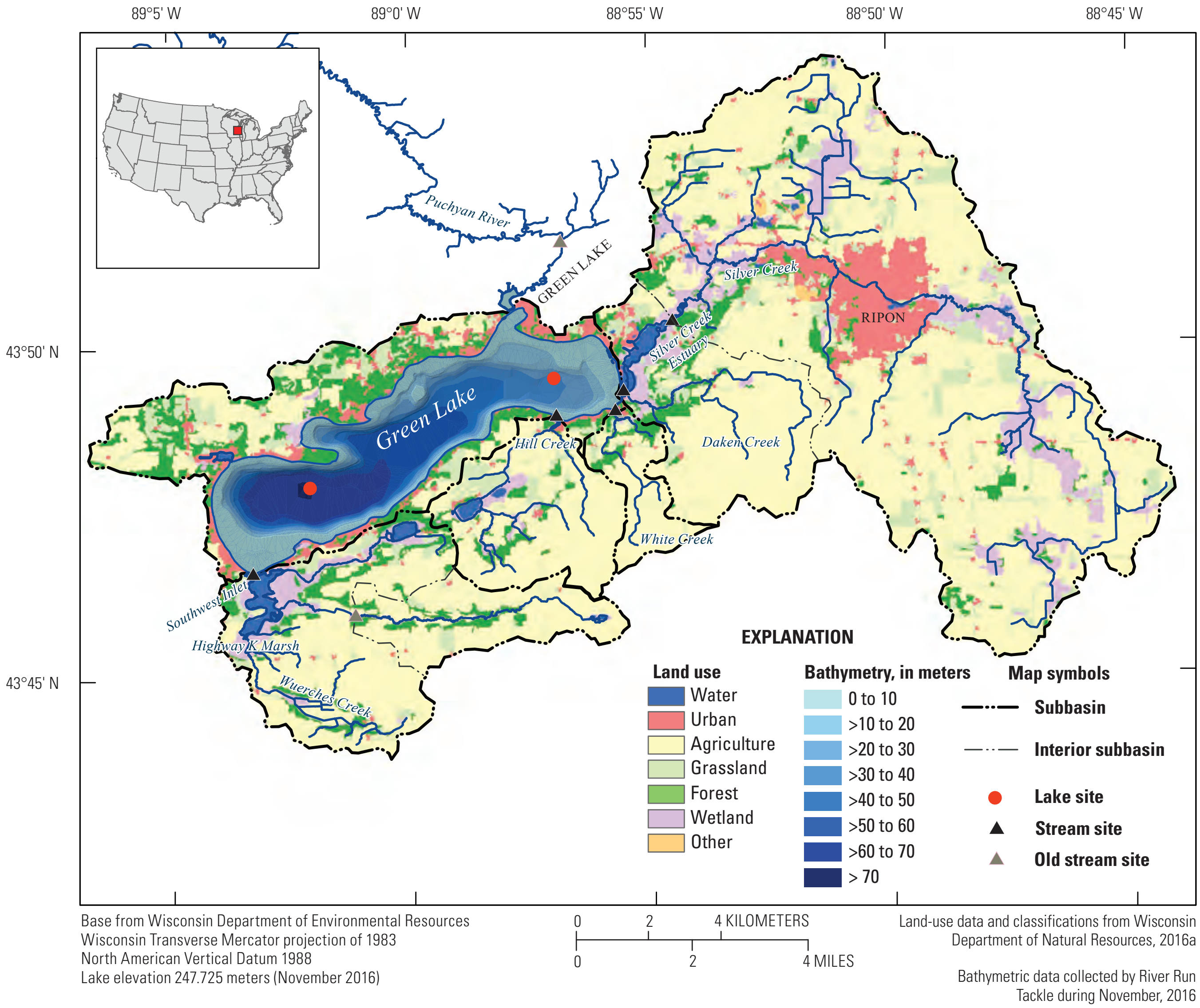


Figure 1. Lake bathymetry, sampling locations, and land use in Green Lake, Wisconsin, and the surrounding watershed. Land-use classifications were defined from Wiscland 2.0 data (Wisconsin Department of Natural Resources, 2016a). m, meter.

and monitored P and sediment loading from various tributaries to the lake (described in detail in the section “Data-Collection Methods and Sites”). Although Green Lake is a single-basin lake, because of its size, it was monitored at two locations on the east and west sides of the lake, similar to the monitoring done by the WDNR. The phosphorus removal from the Ripon wastewater treatment plant was again modified in 2005, which further reduced its P export by over 60 percent: from about 1,200 kilograms (kg), with a mean TP concentration of 0.62 milligram per liter (mg/L), during 1993–2005 to about 450 kg, with a mean TP concentration of 0.22 mg/L, during 2006–19 (Johnson, 2021). But even after this reduction in P in wastewater, TP concentrations in the lake remained high.

The WDNR has established a criterion for surface TP concentration in dimictic lakes with a two-story fishery: geometric mean near-surface TP concentrations should not exceed 0.015 mg/L for the period between June 1 and September 15 for the most recent 10-year period, but data from the most recent 5 years are given preference (Wisconsin Department of Natural Resources, 2019). The phosphorus loading capacity for a lake is defined as the annual loading of phosphorus a lake can experience and still maintain a near-surface TP concentration below the defined criterion. Because TP concentrations in Green Lake exceeded the TP criterion, it was included in the Upper Fox and Wolf Rivers Total Maximum Daily Load (TMDL) study (Cadmus Group, 2018; Wisconsin Department of Natural Resources, 2020a). In the TMDL, the Cadmus Group calculated the mean near-surface lakewide (mean of data from the west and east sides of the lake) surface TP concentration during June–September to be 0.017 mg/L and the mean annual external P loading to the lake during water years (WYs) 2009–13 to be 5,360 kg/yr (inputs from precipitation, atmospheric deposition, waterfowl, and internal sediment recycling were not included). The Cadmus Group used the Canfield-Bachmann natural-lake model (Canfield and Bachmann, 1981), similar to that used by Panuska (1999), to estimate the loading capacity for Green Lake to be 4,230 kg/yr, which was about a 21-percent decrease in external P loading from that measured during 2009–13. However, the Wisconsin Department of Natural Resources (2019) stated that when a lake includes multiple deep monitoring stations or basins, all areas should be used for assessment purposes. Therefore, for Green Lake, the loading capacity would need to be based on the P loading necessary to maintain TP concentrations in the east side of the lake (the station with higher TP concentrations) at less than 0.015 mg/L.

In addition to eutrophication causing increased algal concentrations and decreased water clarity, eutrophication can lead to a depletion in DO that fish, macroinvertebrates, and other aquatic organisms require to live (Wang and others, 1978) and that affects biogeochemical cycling of nutrients in aquatic systems (Rhodes and others, 2017; Terry and others, 2017). DO concentrations that are present in lakes represent a balance between oxygen introduced at the surface of the lake or produced in the lake by photosynthesis, oxygen added or removed with inflows and outflows, and oxygen consumed by respiration of biological organisms and by decomposition of organic materials in the water column and bottom sediments. Oxygen depletion often occurs in deep areas of stratified or ice-covered lakes when oxygen-consuming processes decrease the finite pool of oxygen that is isolated from atmospheric exchange and most oxygen-producing photosynthetic processes (Wetzel, 2001). Below the photic zone, DO is depleted by mineralization of organic matter in the water column and by sediment oxygen demand (Livingstone and Imboden, 1996). Oxygen-depletion rates depend on the magnitude of the organic matter pool (Müller and others, 2012, 2019), the trophic status of the lake (Rhodes and others, 2017; Rippey and McSorley, 2009), the area-to-volume relationship over depth (Livingstone and Imboden, 1996), and the chemical demand of the water column and sediments (Yin and others, 2011). Low DO conditions can be detrimental to fisheries and bottom living creatures, and fish kills occur in extreme cases (Wisconsin Department of Natural Resources, 2019). Typically, the most intense oxygen demand is at the sediment-water interface, so that a zone of hypoxic (less than 2 mg/L) or anoxic (less than 0.2 mg/L) water is present at the bottom of the hypolimnion or throughout the hypolimnion in productive stratified lakes (Wetzel, 2001). In some aquatic systems, however, a DO minimum can occur not only in the deeper zones of the hypolimnion but also in the metalimnion of the lake (referred to here as metalimnetic dissolved oxygen minimum, or simply MOM), such as found in Green Lake. Extreme MOMs (with DO concentrations less than 2 mg/L) are relatively uncommon in natural lakes; however, they are more common in reservoirs (Effler and others, 1998) and can cause fish kills (Rice and others, 2013). A MOM can also form a barrier for many organisms and affect the vertical distribution and production of invertebrates (Horppila and others, 2000) and fish (Liljendahl-Nurminen and others, 2008).

Hypolimnetic DO depletion and slight MOMs were documented in Green Lake in the early 1900s (Birge and Juday, 1911), although the MOMs were much less pronounced than those observed more recently and did not occur in all years. Stauffer (1985a) documented a MOM in Green Lake in 1978 with a lower DO concentration than was measured in the early 1900s, but Stauffer did not observe a MOM in 1971 or 1972. Minimum DO concentrations in the metalimnion continued to decrease, according to additional data collected by the WDNR from 1981 through 2000 (Wisconsin Department of Natural Resources, 2020b) and by the USGS from 2004 to 2020 (U.S. Geological Survey, 2020). Based on these data, the MOMs that occurred intermittently before 1978 occurred in all monitored years since 1978, and the minimum DO concentrations in the metalimnion since 1978 were lower than those measured in earlier years, at times measured below 1 mg/L. Because DO concentrations within the metalimnion have consistently dropped below 5 mg/L in midsummer to late summer (the DO criterion for warm-water fisheries), Green Lake was described as not meeting its water quality standards and listed as "impaired" by the WDNR in 2014, and high phosphorus input was listed as the likely cause (Wisconsin Department of Natural Resources, 2021).

There are several possible causes of a MOM developing, which can be divided into external and internal causes (fig. 2). The main potential external cause is the direct input of rivers with cool water and low DO concentrations or high oxygen demand, which is entrained midway in the water column (interflows). Interflow is the most common cause of MOMs in reservoirs and the ocean (Nix, 1981; LaBounty and Horn, 1997). Internal causes of MOMs may be subdivided into sediment and water-column causes. Morphological features (shelves) in the lake resulting in a high bottom area of sediment per unit volume of water at a specific depth can result in oxygen depletion if the sediment in that area has a high oxygen demand (Shapiro, 1960; Nix, 1981; Mi and others, 2020). This usually results in lower DO concentrations in the metalimnion near the shore than in the metalimnion in open waters. Oxygen-consuming processes can also occur within the water column, specifically the metalimnion, resulting from respiration and decomposition/oxidation of bacteria, phytoplankton, or zooplankton, collectively referred to as community respiration (Birge and Juday, 1911; Shapiro, 1960; Stauffer, 1985a; Schram and Marzolf, 1994; Effler and others, 1998; Müller and others, 2012; Wentzky and others, 2019; Mi and others, 2020). Often, concentrations of organic material are high in the metalimnion because of high concentrations of organic material being present when stratification develops or because the settling of seston is slowed by the colder, denser, and more viscous water in the metalimnion compared with the surface water. The depth of the thermocline with respect to the photic zone (the depth of 1 percent of the surface light and often referred to as the compensation point where production equals respiration) during the most highly

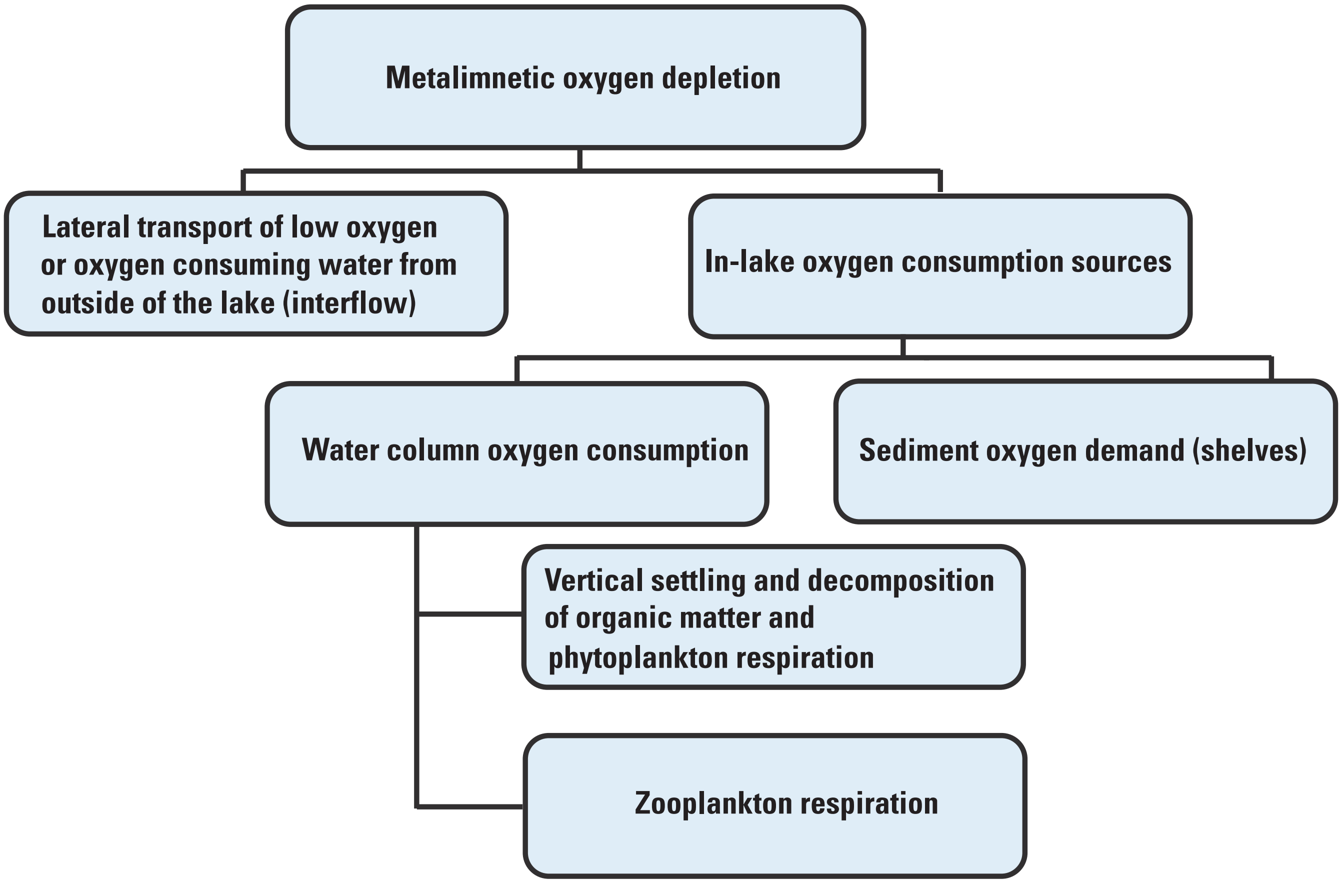


Figure 2. Potential causes of metalimnetic oxygen minima in lakes and reservoirs.

productive months can determine if the net result of photosynthesis and community respiration in the metalimnion results in an oxygen minimum or oxygen maximum (Boyd, 1980). Boyd hypothesized that if the lower limit of the photic zone was typically above the thermocline, a MOM may be expected; however, if the lower limit of the photic zone typically extended below the thermocline, a DO maximum may be expected. In most systems, more than one of these causes may contribute to the formation of a MOM (Shapiro, 1960; Wetzel, 2001; Effler and others, 1998; Mi and others, 2020).

Various approaches have been used to determine the causes of MOMs, from documenting the water temperature and chemical composition of the inflows (Nix, 1981; LaBounty and Horn, 1997), to quantifying zooplankton in the MOM and quantifying their respiration rates (Shapiro, 1960), to quantifying losses in particulate organic matter within the MOM and determining how much DO it would take to cause the reduction in particulate organic matter (Schram and Marzolf, 1994), to using coupled hydrodynamic-ecological models to simulate the changes in the water quality of the lake and explain the mechanisms of DO depletion (Mi and others, 2020). Mi and others (2020) used the two-dimensional hydrodynamic water-quality model CE–QUAL–W2 (Cole and Wells, 2006) to simulate the hydrodynamics and biogeochemical processes causing MOMs in Rappbode Reservoir in Germany. Mi and others (2020) found that the MOMs were primarily driven by benthic processes; however, water-column processes were also important.

Watras (2014) presented evidence that the MOM in Green Lake was caused by water-column processes (community respiration). In that study, high-vertical-resolution profiles of temperature, DO, pH, light, and chlorophyll fluorescence were collected at several locations in the lake during midsummer of 2014. More light attenuation, peaks in chlorophyll fluorescence, and lower pH values were all found at the depths of the MOMs and other depths with relatively low DO in the hypolimnion than at other depths. Low pH values in these areas with lower DO concentrations suggested that carbon dioxide was produced as a result of water-column (community) respiration that consumed oxygen, resulting in a DO minimum.

Naziri Saeed (2020) also examined the causes of MOMs in Green Lake. In that study, water temperature and DO data were collected by using continuous recording sensors spaced throughout the epilimnion, metalimnion, and upper hypolimnion placed on the east and west sides of the lake.

Naziri Saeed (2020) used water temperature data to calibrate the hydrodynamic model Simstrat (Goudsmit and others, 2002) for Green Lake; the model was used to describe vertical mixing in the lake. Vertical mixing information and DO concentration data were then used to construct a coupled hydrodynamic-metabolism model to explain the changes in DO concentrations in Green Lake by using terms describing productivity, respiration, sediment oxygen demand, and surface oxygen fluxes. This model presented evidence that the MOMs in Green Lake were caused primarily by water-column processes (community respiration) and that hypolimnetic hypoxia was caused by sediment oxygen demand. The model did not directly include the effects of nutrients on water-column productivity; therefore, the model was unable to be used to directly estimate the nutrient reductions necessary to reduce water-column productivity and increase DO concentrations in the metalimnion to acceptable levels.

Study Goals

The primary goals of this study by the USGS, in cooperation with the Green Lake Sanitary District, were to

1. describe the water quality in Green Lake since intermittent data were first collected from 1905 to 2020, primarily the constituents for which the lake is impaired, including near-surface TP concentrations and metalimnetic DO concentrations;
2. quantify the water and P inputs to the lake that are likely causing the impairments;
3. describe the factors that have caused low DO concentrations in the metalimnion of Green Lake;
4. quantify how changes in P loading are expected to affect near-surface TP and Chl-*a* concentrations, water clarity, and the minimum DO concentrations in the metalimnion of the lake;
5. estimate the reductions in P loading needed for Green Lake to no longer be listed as impaired for high TP concentrations and low DO concentrations (in other words, estimate its P-loading capacity); and
6. compare the loading capacity estimated from this study with that previously estimated as part of the Upper Fox and Wolf Rivers TMDL study (Cadmus Group, 2018).

This report describes the methods of data collection and compilation, flow and loading estimation, and eutrophication modeling and presents the study results. Most of the material in this scientific investigations report was previously published as an appendix (Robertson and others, 2021) to "Diagnostic and Feasibility Findings: Water quality Improvements for Green Lake" by the Green Lake Association (Prellwitz, 2021) but is republished in this report with permission by the WDNR for visibility purposes. The data collected and compiled for this study are available in the USGS National Water Information System (NWIS) database (U.S. Geological Survey, 2020), WDNR Surface Water Integrated Monitoring System (SWIMS) database (Wisconsin Department of Natural Resources, 2020b), and a data release by Robertson and Kennedy (2021).

General Approach

To achieve these goals, the following approach was used:

1. assemble all available water-quality data from the various monitoring studies conducted on Green Lake describing TP concentrations, water clarity as defined by Secchi disk depths (SD), and DO concentrations (because most data describing low DO concentrations in the metalimnion were collected during August, only August DO profile data were included in this study);
2. quantify "current" water quality of Green Lake, defined as the 2014–18 base period;
3. quantify mean annual water and P inputs to the lake for the base period by estimating inputs from all major P sources;
4. use the empirical Canfield-Bachmann natural-lake model (Canfield and Bachmann, 1981), the same model as that used by Panuska (1999) and the Cadmus Group (2018), to estimate the reduction in external P loading needed for the mean near-surface summer TP concentration in the lake for the base period to be reduced to 0.015 mg/L (a similar approach to that used by the Cadmus Group [2018] but using refined data for an updated period);
5. use empirical regression techniques applied to the assembled DO (MOM) data and the General Lake Model coupled to the Aquatic Ecodynamics modeling library (GLM–AED) hydrodynamic water-quality model (Hipsey and others, 2017, 2019a) to describe the factors that cause MOMs in Green Lake; and
6. use the GLM–AED hydrodynamic water-quality model to simulate daily changes in water quality of Green Lake and estimate the reduction in external P loading needed for DO concentrations in the metalimnion to only decrease below 5 mg/L in less than 25 percent of the years (in this study, this would be in no more than 1 out of the 5 years being simulated).

Green Lake and Its Watershed

Green Lake is a drainage lake in Green Lake County, west of Ripon, Wis., in the Upper Fox River watershed (fig. 1). Green Lake is a calcareous, hard-water lake with high calcium and sulfate concentrations, which results in low dissolved iron concentrations, including during turnover events (Stauffer, 1985b). In this type of deep lake, most of the P from internal sediment recycling that is released to the surface waters during fall turnover remains in the water column until spring (Stauffer, 1985b).

There are several tributaries to Green Lake (fig. 1). Silver Creek enters the eastern side of the lake through Silver Creek estuary, which includes inputs from Dakin Creek; Dakin Creek flows into the Silver Creek estuary prior to Silver Creek discharging to Green Lake. Effluent from the city of Ripon is discharged into Silver Creek. The Southwest Inlet, which includes inputs from Roy Creek, Wuerches Creek, and Spring Creek (not labeled in fig. 1), enters the southwestern side of the lake through the Highway (HWY) K marsh. Other important tributaries include Hill Creek and White Creek on the southeastern side of the lake. Water is discharged from the lake through the Puchyan River on the north side of the lake. The Puchyan River then flows into the Upper Fox River, which flows into Lake Winnebago, and eventually to Green Bay, Lake Michigan (not shown in fig. 1).

Green Lake is a large, single-basin lake. The area and volume of Green Lake were originally estimated to be 2,973 hectares (ha) and 9.38×10^8 cubic meters (m^3), respectively, based on the 1974 WDNR lake contour map (Wisconsin Department of Natural Resources, 2020c). A graph of depth in relation to area for the lake, based on the 1974 map, showed a decrease in area (in other words, a shelf) between 12-m and 13-m depths (fig. 3); other shelves appeared at depths of about 38 m, 53 m, and 67 m. As part of this study, the shape of the lake was reevaluated to better define these shelves based on detailed data collected in the lake by River Run Tackle in November 2016 (R. Valley, Navico Inc., written commun., 2018[1]). The depth contours in figure 1 were delineated from using a geographic information system (GIS). The average water level of Green Lake in November 2016 was 247.724 m above the North American Vertical Datum of 1988. Surface-water features around the lake were not well defined in the updated data; therefore, an aerial image obtained from the 2005 National Agricultural Imagery Program (U.S. Department of Agriculture, 2006) was used to define these features and estimate the total surface area of the lake. The lake stage during the open-water period in 2005 was within 0.2 m of that during November 2016. The more extensive data collected in 2016 did not show a shelf between depths of 12 m and 13 m or the shelves at depths of 38 m and 53 m (fig. 3). The original shelves were probably an artifact of limited data collected in 1974 and the interpolation method used to draw the contour lines. Based on the updated information and use of GIS techniques, Green Lake has a surface area of 2,948 ha, volume of 9.89×10^8 m^3, mean depth of 33.6 m, and fetch of 12.2 kilometers (km). The morphometry (areas and volumes at various depths, and fetch) estimated in this study was used for all computations in this report.

The watershed of Green Lake is 235 square kilometers (km^2) and consists of moderate hill slopes and pervious fertile soils derived from loess and calcareous glacial drift (Martin 1965). Green Lake is incised into Cambrian sandstone that crops out at present-day lake levels (Stauffer, 1985b). Based on data from Wiscland 2.0 (Wisconsin Department of Natural Resources, 2016a), land use in the watershed is a mixture of agriculture (61.7 percent), grasslands (10.9 percent), forest (9.8 percent), urban and barren areas (7.2 percent), and wetland and open water (10.3 percent) (table 1). Therefore, agriculture and urban (including wastewater treatment effluent) inputs were expected to be the largest external sources of nutrients to the lake. Silver Creek, the main tributary to the lake, has a drainage area of 142.5 km^2 and primarily drains a mixture of agriculture (64.4 percent), grasslands (10.8 percent), wetlands (10.6 percent), and urban (7.9 percent) areas. The second largest tributary to the lake is the Southwest Inlet, which has a drainage area of 38.9 km^2 and primarily drains agriculture (66.8 percent) and forested (11.5 percent) land.

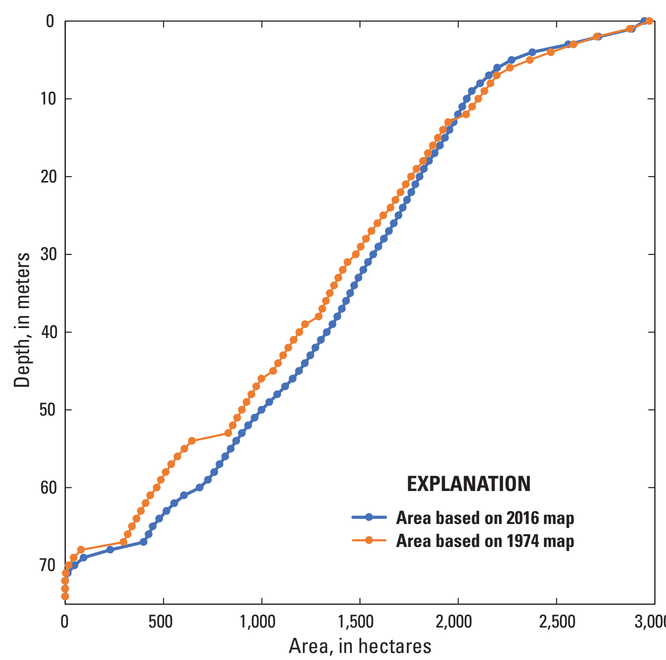


Figure 3. Depth and area for Green Lake, Wisconsin, based on 1974 contour map (Wisconsin Department of Natural Resources, 2020c) and updated bathymetry shown in figure 1. The average water level of Green Lake when the 2016 data were collected was 247.724 meters above the North American Vertical Datum of 1988.

[1]Data have limited availability owing to restrictions of River Run Tackle. Contact River Run Tackle, Mosinee, Wisconsin, for more information.

Table 1. Land use in the Green Lake, Wisconsin, watershed, and that of individual subbasins.

[Land-use data and classifications from Wisconsin Department of Natural Resources (2016a). km^2, square kilometer]

Basin	Percent of area, by specific land use							Total area (km^2)
	Urban	Agriculture	Grassland	Forest	Open water	Wetland	Other	
Silver Creek	7.9	64.4	10.8	5.1	0.9	10.6	0.3	142.5
Upstream of Spaulding Road	9.0	63.2	11.4	4.4	0.4	11.3	0.4	118.2
Downstream	2.6	69.9	8.1	8.4	3.7	7.4	0.0	24.3
White Creek	1.5	77.8	7.3	12.3	0.0	1.0	0.0	6.7
Hill Creek	1.4	73.0	10.8	7.5	1.7	5.5	0.0	18.4
Southwest Inlet	1.0	66.8	7.9	11.5	3.5	9.4	0.0	38.9
Roy Creek	0.4	81.4	5.8	10.1	0.4	2.0	0.0	12.3
Downstream	1.2	60.0	8.9	12.1	5.0	12.8	0.0	26.6
All ungaged	8.5	14.8	8.2	15.6	51.9	0.9	0.0	58.0
Ungaged, not including Green Lake	17.3	30.1	16.7	31.8	2.3	1.9	0.0	28.5
Total watershed (not including Green Lake)	7.2	61.7	10.9	9.8	1.6	8.7	0.2	235.1

Methods of Data Collection, Flow and Load Estimation, and Eutrophication Modeling

This section of the report describes the methods used for data collection and compilation of lake stage, lake water quality, cross-sectional transects, internal sediment recycling, streamflow, stream water quality, loads in streams, meteorological data, and atmospheric sources of P and nitrogen. Methods used to determine water and P inputs from unmonitored areas around the lake are described. Statistical and numerical eutrophication models used to describe the response of the lake to changes in P loading also are described.

Data-Collection Methods and Sites

USGS personnel began collecting flow and water-quality data in White Creek in 1981, but consistent monitoring in Silver Creek did not begin until 1987, and monitoring in Green Lake did not begin until 2004. Data collection continued through 2021, but the locations and frequency of monitoring have varied. In addition to the data collected by the USGS, tributary and lake data were collected by students and staff from University of Wisconsin (Madison and Beloit), the Wisconsin Conservation Department, the WDNR, the Green Lake Association, and local observers. Because of concerns over the quality of the historical data, only specific types of the historical data were used in this study, primarily SDs, TP concentrations after 1970, and DO data. In most studies done on Green Lake, additional data were collected, but they are not discussed in this report.

Lake-Stage and Lake Water-Quality Monitoring

A lake-stage gage was installed and operated near Silver Creek at HWY A from 1993 through 2012 (USGS station 434928088553601). From 2013 through 2020, continuous (5-minute) lake stage was recorded with streamflow monitoring equipment at the Southwest Inlet (USGS station 040734605). The continuous data from 2013 through 2020 and the daily mean data for both sites are associated with USGS station number 434840089000001 (U.S. Geological Survey, 2020).

As part of USGS monitoring during 2004–20, Green Lake was typically sampled monthly from April through September. Although Green Lake is a single-basin lake, because of its size, it was monitored at two locations: west side (near the deep hole at approximately 68-m depth; USGS station 434756089020500) and east side (approximately 30-m depth; USGS station 434928088570000) of the lake. More frequent sampling, however, was conducted during 2017–20, which included monthly sampling from March through October, twice per month either in July and August or in August and September, and in February and December in 2019. During most samplings, SDs were measured with a 20-centimeter (cm) black and white disk, and profiles of water temperature, DO, specific conductance, and pH were measured with multiparameter instruments. During most samplings, near-surface and near-bottom water samples were collected with a Van Dorn sampler. Surface samples were analyzed for TP, dissolved reactive phosphorus (DRP), nitrogen species (nitrite plus nitrate, Kjeldahl nitrogen, and ammonia), and Chl-*a* concentrations. Bottom samples were analyzed for TP concentrations. During spring turnover, typically April or May in each year, surface samples were analyzed for many common ions including, but not limited to, calcium, sulfate, silica, and iron, and other characteristics such as color, turbidity, alkalinity, and total dissolved solids. In 2017, additional samples were collected on the west side of the lake once per month from May through September from the middle of the epilimnion, top of the metalimnion, middle of the metalimnion (depth with minimum DO concentration), bottom of the metalimnion, 50-m depth, and just above the bottom. These samples, including the surface sample, were analyzed for TP, DRP, nitrogen species (nitrite plus nitrate, Kjeldahl nitrogen, and ammonia), and total and dissolved carbon concentrations. In addition, the samples from the middle of the epilimnion and middle of the metalimnion were analyzed for biological oxygen demand. In September of 2018 and 2019, additional samples were collected in the hypolimnion on the west side of the lake at depths of 50 m, 63 m, 66 m, and 67 m and analyzed for TP concentrations. The additional TP data were used to estimate the change in the mass of P in the hypolimnion from April through September, which represents the accumulation of P from internal sediment recycling during summer. For further details on USGS sampling techniques, see Manteufel and others (2012). All chemical analyses were performed by the Wisconsin State Laboratory of Hygiene in accordance with standard analytical procedures (Wisconsin State Laboratory of Hygiene, 2020). All lake-stage, streamflow, and water-quality data (including loads) collected by the USGS are available from the NWIS database (U.S. Geological Survey, 2020) and can be accessed by using the applicable USGS station numbers provided in this report section.

In addition to water chemistry, water samples from the middle of the epilimnion, middle of the metalimnion, and bottom of the metalimnion were collected in May, June, July, August, and September of 2017 for phytoplankton and zooplankton identification. Each 700-milliliter (mL) phytoplankton sample was collected with a Van Dorn sampler, preserved with 10 mL of glutaraldehyde, and shipped to the Wisconsin State Laboratory of Hygiene for identification. During each of these sample collections, tow samples were also collected with a Wisconsin plankton net sampler (0.0105-square-meter opening and 153-micrometer [µm] mesh) for zooplankton identification from the surface to top of metalimnion, from the top of metalimnion to bottom of metalimnion, and from the bottom of metalimnion to 40 m; tow samples were preserved with an equal volume of 70-percent ethanol and shipped to PhycoTech, Inc., for enumeration.

SDs, water temperatures, and DO data were first collected in the lake by the University of Wisconsin-Madison. SDs, measured by using a 10-cm white disk during 1905–27, were obtained from the University of Wisconsin Archives (Robertson and Kennedy, 2021), and temperature and DO profiles, measured during 1905 to 1909, were obtained from Birge and Juday (1911). Lathrop (1992) developed conversion factors for adjusting data collected with various types of Secchi disks. The 0.956 conversion from a 10-cm white disk to a 20-cm black and white disk was used to adjust all early SD readings. Water temperatures were collected by using slow reversing thermometers, and DO concentrations were estimated by using either boiling methods or Winkler titration methods (Birge and Juday, 1911).

The Wisconsin Conservation Department, subsequently called the WDNR, collected water temperature and DO profiles in 1947 (Robertson and Kennedy, 2021) and SDs and TP concentrations in 1966 (Lueschow and others, 1970). All chemical analyses were performed by the Wisconsin Laboratory of Hygiene in accordance with standard methods (American Public Health Association, 1965), and SDs were measured with a standard 20-cm black and white disk. DO concentrations were estimated by using Winkler titration methods. The WDNR collected sporadic SD data during 1968–73; TP, SD, and temperature and DO profiles monthly throughout the entire year during 1981–83; and TP, SD, and temperature and DO profiles approximately monthly during spring and summer of 1988–2001. During each visit, the WDNR followed the protocols of its Long-Term Trend monitoring program (Wisconsin Department of Natural Resources, 2016b), and chemical analyses were performed by the Wisconsin State Laboratory of Hygiene in accordance with standard analytical procedures (Wisconsin State Laboratory of Hygiene, 2020). SDs were measured with a standard 20-cm black and white disk. DO concentrations were estimated by using Winkler titration methods until the early 1990s and multiparameter sondes thereafter. Data from the Wisconsin Conservation Department and the WDNR were obtained from either the SWIMS database (Wisconsin Department of Natural Resources, 2020b) or from previously unpublished WDNR files (Robertson and Kennedy, 2021).

SDs were collected by Ripon College during summer of 1972 (Litton and others, 1972). Stauffer (1985a) measured SDs, TP concentrations, and temperature and DO profiles during July–August 1971, March–October 1972, and March–September of 1978. Stauffer used a 25-cm black and white Secchi disk. A 1.043 conversion was used to adjust the SD data to the equivalent of data measured with a 20-cm black and white disk (Lathrop, 1992). Total phosphorus concentrations were determined with Menzel and Corwin (1965) digestion and the ascorbic acid colorimetric procedure (Murphy and Riley, 1962). Water temperature and DO profiles were collected with various types of meters (Stauffer, 1985a). Extensive SD data were collected by local citizens as part of the WDNR Citizen Lake Monitoring Program (Wisconsin Department of Natural Resources, 2020d) and by Green Lake Association volunteers. These data were obtained from SWIMS or supplied by the Green Lake Association.

All SD, TP concentration, DO concentration, and plankton data used in this report are available in a USGS data release associated with this report (Robertson and Kennedy, 2021). To reduce potential biases from nonuniform sampling through time, individual water-quality values (TP, Chl-*a*, and SD data) were averaged monthly, and then mean summer values and geometric mean values for June–September or July–September were computed from the monthly values. This resulted in a few concentrations and depths being used to represent July or September from slightly outside the WDNR evaluation periods for these constituents. Months with missing data were estimated by averaging monthly data from the preceding year and the following year.

Cross-Sectional Transects in the Lake

To further describe the spatial variability in water quality during midsummer from that measured during routine monitoring, a sensor sled that continuously measured various water-quality characteristics was towed behind a boat moving at about 8 kilometers per hour on August 22, 2017, and August 14, 2018. The sled was raised and lowered in the water column in a yo-yo pattern to describe the distribution in water quality throughout the lake. The horizontal position of the sled was tracked by a differential global positioning system, with an accuracy of ±1 m. Speed, total depth, direction of water movement, and backscatter (number of particles in the water) were measured by a 600-kilohertz acoustic Doppler current profiler. Depths of all measurements were made with a submersible pressure sensor mounted to the sled, with an accuracy of ±0.03 m. Sensors on the sled of particular interest in this study were water temperature, DO, fluorescent dissolved organic matter (fDOM), Chl-*a*, and turbidity. DO and water temperature were measured with a high-speed sensor (Rincko model 3, JFE Alec Co., LTD) capable of a 90-percent response in less than 1 second. This sensor had a precision of ±0.4 mg/L for DO and ±0.01 degrees Celsius (°C) for water temperature. Water temperature data were used to describe water density by use of a formula presented by the United Nations Educational, Scientific, and Cultural Organization (1981). Chl-*a*, fDOM, and turbidity were measured with Turner C7 probes (Turner Designs), which respond almost instantaneously. The Turner probes were blanked in milli-Q water before each cruise. Output from the Turner probes was in units of raw fluorescent unit blanked. Linear interpolation in the horizontal direction was used to interpolate between measurements collected with the sled to describe horizontal changes in water quality throughout the lake.

Internal Sediment Recycling

The amount of P from internal sediment recycling was estimated from the change in the mass of P in the hypolimnion from April through September in 2018 and 2019. Five additional TP concentrations were collected in September to better define the TP distribution in the hypolimnion (at 15, 50, 63, 66, and 67 m). TP concentrations at each depth below 15 m were computed by assuming a linear change in concentrations between these depths. The volume of water at each depth was computed from the bathymetric data for the lake shown in figure 1 by using GIS (Robertson and Kennedy, 2021) (fig. 3). Because of the lack of resolution in TP concentrations, this is a rather crude estimate of internal sediment recycling. This approach may overestimate the accumulation of P in the hypolimnion if TP concentrations increase exponentially with depth in the hypolimnion. It may also overestimate the amount of internal sediment recycling because part of the P accumulated in the hypolimnion may come from the decomposition of organic matter that is settling in the water column. However, this approach may underestimate the amount of internal sediment recycling because stratification (and sediment P release) typically continues through much of October.

Stream Monitoring

Six tributary and outlet sites (fig. 1; table 2) were instrumented by the USGS to monitor continuous flow (5- to 15-minute intervals) and estimate P and sediment loading from the watershed or out of the lake. At three sites (White Creek at Spring Grove Road—USGS gage number 04073462; Hill Creek near the mouth—040734615; and the Puchyan River—04073473), water levels were measured and used to estimate flow by using standard stage-discharge relations (Rantz and others, 1982). At three sites (Silver Creek at HWY A—04073468, Silver Creek at Spaulding Road—04073466, and the Southwest Inlet at HWY K—040734605), flow was estimated by using acoustic Doppler velocity meters and water-level sensors (Sauer, 2002) because backwater conditions did not enable flow to be estimated solely from changes in water elevation. Each site was in operation over different time periods (table 2), with Hill and White Creeks operated only in WY 2018 during the base period. From the 5- to 15-minute flow data, daily mean flows were computed for each site. Data collected after WY 2019 were not included in this study.

Samples for water-quality analyses for all stream sites, except Silver Creek at HWY A, were collected by using automated samplers and augmented with routine, manually collected base-flow samples. Only manually collected samples were collected at Silver Creek at HWY A. The mean numbers of samples per year collected for TP analysis were 21, 60, 64, 81, and 84 samples at Silver Creek at HWY A, Silver Creek at Spaulding Rd, Southwest Inlet, Hill Creek, and White Creek, respectively (table 2). About 20 percent of those samples were analyzed for dissolved P. In addition, during WYs 2017–19, about 25 samples (total from each site) from Silver Creek at HWY A and the Southwest Inlet were analyzed for nitrate plus nitrite, ammonia, organic nitrogen, and Kjeldahl nitrogen. Additionally, about 10 samples (total from each site) were analyzed for dissolved organic carbon, total organic carbon, and biological oxygen demand. Samples for nitrate plus nitrate and ammonia were filtered in the laboratory. All chemical analyses were done by the Wisconsin State Laboratory of Hygiene in accordance with standard analytical procedures (Wisconsin State Laboratory of Hygiene, 2020).

Daily P loads were computed for each site by using the Graphical Constituent Loading Analysis System (GCLAS; Koltun and others, 2006). GCLAS was developed to estimate constituent loads from continuous measurements of streamflow and instantaneous constituent concentrations.

Table 2. Summary of monitoring on tributaries to and outflow from Green Lake, Wisconsin.

[Volumetrically weighted mean (VWM) total phosphorus (TP) concentrations, in milligrams per liter (mg/L) for 2014–18, are given. USGS, U.S. Geological Survey; km^2, square kilometer; CT, County Trunk; nr, near; WI, Wisconsin; SW, Southwest; @, at]

Site name	Official U.S. Geological Survey name	USGS gage/station number	Years of gage operation	Years of load computation	Drainage area (km^2)	Mean number of TP samples per year		VWM TP concentration during 2014–18, mg/L	Median May–October TP concentration during 2014–18, mg/L
						Full period	2014–18		
Silver Creek at Highway A	Green Lake Inlet at CT Highway A nr Green Lake, WI	04073468	1987–2011	1988–2020	142.5	56	21	0.090	0.076
Silver Creek at Spaulding Road	Silver Creek at Spaulding Road near Green Lake, WI	04073466	2012–21	2012–20	118.2	60	60	0.117	0.133
Southwest Inlet at Highway K	Green Lake SW Inlet @ CT Highway K nr Green Lake, WI	040734605	2012–21	2012–20	38.9	63	64	0.159	0.212
White Creek at Spring Grove Road	White Creek at Spring Grove Road nr Green Lake, WI	04073462	1982–87; 1997–2011; 2018–20	1982–87; 1997–2011; 2018–20	6.7	72	[1]84	[1]0.143	[1]0.118
Hill Creek near the mouth	Hill Creek nr mouth near Green Lake, WI	040734615	2018–20	2018–20	18.4	81	[1]81	[1]0.287	[1]0.565
Puchyan River at Green Lake	Puchyan River at Green Lake, WI	[2]04073470	1997–2012	[3]1997–2017	264.6	17	0	[4]0.041	[4]0.040

[1]For 2018–19.

[2]Streamflow collected at gage number 04043473.

[3]Computed by Johnson (2021).

[4]For 1997–2012.

Generally, concentrations were linearly extrapolated between measurements except at the beginning and end of events with large changes in flow. Before the extrapolations, additional estimated concentrations were often added to the time series to better describe the concentrations just before and after an event or to describe events without measured concentrations. The filled-in concentrations at the beginning of an event were typically estimated from concentrations measured during previous low-flow periods. Concentrations at the end of the events were typically estimated from concentrations measured in that stream shortly after the end of an event. All flow, stream water quality, and load data can be accessed from the NWIS database (U.S. Geological Survey, 2020) by using the USGS gage numbers given in table 2.

One additional site upstream from the gage on the Southwest Inlet on Roy Creek (USGS gage number 0407345) was operated by the USGS during WYs 2013–17. Data from this site were not included in this study. Flow, nutrient and sediment concentrations, and loads for this site are available from the NWIS database (U.S. Geological Survey, 2020).

Meteorological Data

Daily meteorological data (air temperature, precipitation, short- and long-wave radiation, relative humidity, and wind speed) from 1979 to 2019 for Green Lake and its watershed were extracted from hourly data estimated for center of the lake (latitude: 42.81, longitude: −89.00) in the North American Land Data Assimilation System (Mitchell and others, 2004). Spatial subsets of the dataset were selected by using the geoknife package in R (R Development Team, 2013; Read and others 2016), which accesses processing capabilities from the USGS Geo Data Portal (Blodgett and others, 2011). Relative humidity was calculated as a function of specific humidity, air temperature, and surface pressure. For all inputs except wind, daily data were computed from the mean or total of the hourly data over the 24 hours. For wind, the daily average velocity was computed from the cube root of the mean of the cubed hourly data. If this adjustment was not made to wind velocity, the daily wind power would be reduced when going from hourly data to daily summaries (Engineering ToolBox, 2009).

Atmospheric Sources of Phosphorus and Nitrogen

Atmospheric inputs of P are not readily available for most areas of the United States. Therefore, atmospheric inputs of P on Green Lake were estimated from TP concentrations measured in wetfall (rain and snow) and dryfall (P deposition rates), both of which were estimated based on data collected during 2005–9 near Delavan Lake, Wisconsin (135 km southeast of Green Lake). Deposition rates and TP concentrations were measured with a 30-cm diameter stainless-steel collection bowl mounted on a table about 1 m above ground. Netting was placed over the bowl to prevent contamination by birds. This equipment was also used to estimate atmospheric P inputs to lakes in northwestern Wisconsin (Robertson and others, 2009). Delavan Lake Sanitary District personnel serviced the monitoring station and collected wet samples and dry samples. For wet samples, the bowl was deployed within a few hours of an imminent precipitation event, and the samples were collected as soon as possible after the event to reduce evaporation and other contamination. Dry samples were those collected after several days, usually 7–10 days, without precipitation. Dry samples were taken from the bowl by rinsing the bowl twice with 125 mL of distilled water. Both types of samples were acidified and chilled before they were sent to the Wisconsin State Laboratory of Hygiene for analysis of TP concentrations. The stainless-steel bowl was cleaned with phosphate-free soap before each deployment. Rates of phosphorus dryfall deposition were computed by determining the mass of P deposited (dry sample concentration × 0.25 liter [L]) per unit area per day of deployment. All wetfall concentrations were adjusted to remove the P from dryfall during its collection period by subtracting the mass of P deposited during the wetfall sample based on the monthly mean dryfall deposition rate, and then dividing the resulting mass of P in the sample by its volume. Samples that included insects and other nondepositional material were discarded. All P data collected for these analyses are available in Robertson and Kennedy (2021).

Atmospheric deposition of nitrogen was estimated from data collected during WYs 2014–18 at site ID WI31, Devils Lake, Wisconsin, by the National Atmospheric Deposition Program (2020). Site WI31 is approximately 66 km southwest of Green Lake. The mean wetfall total nitrogen concentration was 45.53 millimoles per cubic meter (mmol/m^3; 0.637 mg/L), and the mean dryfall rate of nitrogen was 0.497 millimole per square meter per day (mmol/m^2/d).

Unmonitored Water and Phosphorus Inputs and Outputs

This section of the report describes methods for estimating outflow from Green Lake and flows from unmonitored areas around the lake. Methods for estimating nutrient inputs for unmonitored years and from unmonitored areas also are described.

Outflow from Green Lake and Flows from Unmonitored Areas

Outflow from Green Lake and flows from unmonitored areas around the lake that were needed for GLM–AED modeling were estimated in an iterative manner by using available flows, water-level data for the lake, and preliminary hydrodynamic General Lake Model (GLM) model simulations run without incorporating the Aquatic Ecodynamics modeling library. Most of this report deals with flow and P inputs to Green Lake after 2012, when the Southwest Inlet was monitored but the Puchyan River (lake outlet) was not monitored. Therefore, flows from the lake outlet after 2012 needed to be estimated. Flows from the Puchyan River after 2012 were estimated from flows measured at Silver Creek at HWY A in a two-step process. First, flows from the lake were estimated by multiplying the monthly measured flows from Silver Creek at HWY A after 2012 by the mean monthly ratio between measured flows in the Puchyan River and monthly flows in Silver Creek at HWY A from 1997 to 2012. These monthly flows in the Puchyan River were later refined to a daily basis by constructing a mass balance for water for the lake (described below in this section).

Inputs of water from the watershed around Green Lake, not including those from Silver Creek and the Southwest Inlet, are referred to as flows from unmonitored areas, which represented about 23 percent of the watershed. Although flows from White and Hill Creeks (11 percent of the watershed) were monitored during WYs 2018–20, these data were not used to estimate total flows to the lake or flows from the unmonitored areas that were only used in GLM–AED modeling because only limited water-quality data were available from these sites. Daily flows from the unmonitored areas were estimated by using preliminary simulations of GLM (without AED). GLM, after being calibrated for 2012–18 with measured water-clarity and preliminary outflow data, was then run for the period May 18, 2004, to September 30, 2012, with measured precipitation, measured inflow from Silver Creek at HWY A, and measured outflow from the Puchyan River. Because GLM estimates evaporation, the only missing component in the water budget was additional inflows from all areas other than Silver Creek (including inputs from the Southwest Inlet, unmonitored surface-water input, and groundwater). From this iterative set of simulations, it was found that flow from all watershed areas not including Silver Creek needed to be set to 81 percent of the inputs from Silver Creek to make simulated and measured water levels match at the end of the 2012–18 simulation. In other words, Silver Creek contributed 55 percent of the total flow from the watershed, and all other areas contribute 45 percent of the flow.

Daily flows from the remaining unmonitored part of the watershed (not including inflow from Silver Creek and the Southwest Inlet) and outflow (refined) then needed to be estimated for 2012 to 2018. Daily inputs from the unmonitored part of the watershed were estimated by subtracting the measured daily flows from the Southwest Inlet from the measured daily flows from daily Silver Creek flows × 0.81 (described in the previous paragraph). If this estimate of flow was negative, the flows were set to 1 cubic meter per day (m^3/d) and the remaining negative flows were subtracted from adjacent days until their flows dropped below 1 m^3/d. By estimating inflows from unmonitored areas in this manner, the inflows also incorporated the net input of groundwater to the lake. Finally, refined daily outflows from the lake were estimated by constructing a daily mass balance (outflow = all total tributary inflows + precipitation − evaporation − change in the storage in Green Lake). Daily evaporation from 2012 to 2018 was computed within GLM. The change in storage was computed from daily changes in water levels, assuming the area of the lake remained constant (29.478 km^2). Estimated daily outflows were then smoothed using a 10-day moving mean. If the computed daily mean was still negative, then it was set to 10 m^3/d and the remaining negative flows were subtracted from adjacent days until these flows decreased below 10 m^3/d.

Nutrients in Unmonitored Years and Unmonitored Areas

Nutrients from unmonitored areas were estimated differently for eutrophication modeling and GLM–AED modeling because of the availability of different types of data. P loading from Hill Creek and White Creek were monitored only in WYs 2018–19. Therefore, to estimate the P loading to the lake from these tributaries during WYs 2014–17, as needed for eutrophication modeling with the Canfield-Bachmann model, the mean percentage difference in P loadings for Silver Creek at HWY A and the Southwest Inlet from those measured in WY 2018 were used to estimate P loads from Hill Creek and White Creek in each WY by assuming the loads in these tributaries varied in a similar manner among years. In other words, if P loading in Silver Creek was 12 percent higher than in WY 2018, and the P loading in the Southwest Inlet was 14 percent higher than in WY 2018, then it was assumed that P loading in Hill Creek and White Creek each went up 13 percent (the average percent change) from what was measured at those sites in WY 2018.

For the eutrophication modeling, the total P loading from the unmonitored areas (P loading from the watershed not including that from Silver Creek, Southwest Inlet, Hill Creek, and White Creek plus that from groundwater input) was estimated based on the Soil and Water Assessment Tool (SWAT) model (Arnold and others, 2012) developed for the Green Lake watershed (Baumgart, 2015). From SWAT results, the total P loading to Green Lake from all areas not including Silver Creek, White Creek, Hill Creek, and the Southwest Inlet was 630 kg/yr. It was assumed that the P loading for the unmonitored areas was 630 kg in WY 2018. Then to estimate the P loading to the lake from these unmonitored areas during WYs 2014–17, the mean percentage differences in P loadings for Silver Creek at HWY A and the Southwest Inlet from those measured in WY 2018 were again used to estimate the P loading from the unmonitored areas in each WY by assuming the loads from these areas varied in a similar manner among years.

For GLM–AED modeling, the total input of P and other constituents to Green Lake from Hill Creek, White Creek, other unmonitored surface water, and total groundwater input were combined into one tributary input. In other words, the monitored information from Hill Creek and White Creek were not used in GLM–AED modeling. This combined input was estimated by multiplying the flows from all areas other than Silver Creek and the Southwest Inlet (described in the section “Outflow from Green Lake and Flows from Unmonitored Areas”) by the constituent concentrations in Silver Creek.

Statistical and Numerical Eutrophication Models

This section of the report describes the overall modeling approach used to help describe the factors affecting the water quality of Green Lake and to estimate the lake's response to changes in P loading from its watershed. Methods for applying each of the empirical models and GLM–AED are then described.

Modeling Approach

A combination of empirical models, statistical techniques, and hydrodynamic water-quality models were used to help explain the factors affecting the water quality of Green Lake and to estimate the lake's response to changes in P loading from its watershed. For describing changes in near-surface water quality, the Canfield-Bachmann natural-lake model (Canfield and Bachmann, 1981) was first used to estimate how near-surface TP concentrations in Green Lake are expected to respond to changes in P loading. Then, the changes in TP concentrations were used to estimate the expected responses of Chl-*a* concentrations and SDs by using empirical relations developed by using data collected from the east and west sides of Green Lake, the trophic state index (TSI) relations by Carlson (1977), and the TP–Chl-*a* relation by Hickman (1980). Multiple approaches were used to provide a range in the possible water quality that may be expected following changes in external P loading to the lake. In all empirical model simulations, the east and west sides of the lake were each treated as an independent lake with full watershed loading and full lake morphometry. This approach assumes that circulation patterns in the lake and the relative differences between the east and west sides of the lake observed during the model-evaluation period will remain in the future.

Pearson correlations were used to determine which hydrologic and meteorological factors were most strongly correlated with interannual variability in near-surface water quality and the intensity of annual MOMs. The hydrodynamic water-quality model GLM–AED (Hipsey and others, 2017, 2019a) was used to explain how thermal stratification, vertical mixing, and water quality (including the distribution of DO concentrations and MOMs) in Green Lake responds to changes in meteorological boundary conditions and P loading to the lake.

For each of the empirical models and the GLM-AED hydrodynamic water-quality model, the following steps were taken:

1. apply the model to the lake for a defined base period (2014–18; GLM–AED was used for 2013–18, but only data from 2014 to 2018 were used to examine the response of water quality);
2. calibrate the models if necessary (for the empirical models, there are no true calibration parameters; therefore, to account for model biases, all model results were adjusted by the percentage of bias in the results found for the base-period [reference] scenario; calibration of GLM–AED is described later in this section in "GLM–AED Model");
3. use the calibrated models to simulate the 5-year base period (2014–18) and simulate those same years but with a range of changes in P loading to obtain response curves for TP and Chl-*a* concentrations, SD, and MOM (with the hydrodynamic water-quality model, changes in P loadings were simulated by only changing the inflow P concentrations);
4. estimate the P-load reduction needed for the water quality to meet the defined criteria from the response curves;
5. use the empirical models to simulate near-surface reference conditions using P-loading rates based on conditions with no agriculture or urban areas in the basin from the Cadmus Group (2018) and Singer and Rust (1975); and
6. estimate the P loading to the lake that is expected to result in a near-surface TP concentration of 0.012 mg/L (oligotrophic classification based on the TSI relation by Carlson, 1977) by iteratively changing P loading to the lake in the Canfield-Bachmann model and evaluating estimated TP concentrations.

Canfield-Bachmann Natural-Lake Total Phosphorus Model

Empirical models that relate P loading to a lake to specific water-quality characteristics can be used to estimate how changes in P loading to Green Lake are expected to affect its water quality. Most of these models were developed based on data from many different lake systems with specific measures describing lake water quality, such as near-surface mean summer TP, Chl-*a*, and SD. One of these models is the Canfield-Bachmann natural-lake model (Canfield and Bachmann, 1981) (eq. 1), which relates hydrology and external P loading to the near-surface summer mean TP concentration:

$$TP = \frac{L_A \div 1{,}000}{z \times \left(0.162 \times \left(L_A \div z\right)^{0.458} + 1 \div \tau\right)}, \qquad (1)$$

where

- TP is near-surface mean summer total phosphorus concentration, in milligrams per liter;
- L_A is the phosphorus-loading rate per unit area of the lake, in milligrams per square meter per year;
- z is the mean depth of the lake, in meters; and
- τ is the water residence time, in years.

Internal sediment recycling is not typically included as a source in many of these empirical models, including the Canfield-Bachmann model, because most of these models were developed without explicitly including this source (Dillon and Rigler, 1974; Canfield and Bachmann, 1981). Therefore, these models indirectly incorporate contributions from internal sediment recycling (Walker, 1976). Internal sediment recycling is typically only added as a source when a lake has abnormally high internal sediment recycling (Walker, 1976), such as Lake Winnebago, Wisconsin (Robertson and Diebel, 2020). Therefore, the Canfield-Bachmann model was applied to Green Lake with and without contributions from internal sediment recycling included in the annual P load to the lake to determine which loading estimate provided better results.

Empirical Chlorophyll-*a* and Secchi Disk Depth Models

Water-quality data collected in the east and west sides of Green Lake from 1986 to 2019 were used to develop linear regression relations between the measured annual near-surface mean TP concentrations from June through September and the measured mean Chl-*a* concentrations and SDs from July through September. The predicted TP concentrations from the Canfield-Bachmann model (eq. 1) were then used to estimate Chl-*a* concentrations and SDs by using the relations developed from Green Lake data, the TSI relations (eqs. 2–4) by Carlson (1977), and TP–Chl-*a* relations (eq. 5) by Hickman (1980), and to produce response curves for Chl-*a* concentrations and SDs to changes in P loading to the lake. For the TSI approach, equations 2–4 were reformatted and used to estimate the Chl-*a* concentrations and SDs that yielded a similar TSI value as that based on actual or predicted TP concentrations:

$$TSI_P = 4.15 + 14.42\ [\ln \text{TP concentration (in micrograms per liter)}] \quad (2)$$

$$TSI_{Chl} = 30.6 + 9.81\ [\ln \text{Chl-}a \text{ concentration (in micrograms per liter)}] \quad (3)$$

$$TSI_{SD} = 60.0 - 14.41\ [\ln \text{SD (in meters)}] \quad (4)$$

$$\log [\text{Chl-}a \text{ (in micrograms per liter)}] = 0.542 + 3.27 * \log [\log \text{TP concentration (in micrograms per liter)}] \quad (5)$$

Pearson-Correlation Analyses

Pearson-correlation analyses were used to determine which hydrologic, water quality, and meteorological factors were correlated with the interannual variability in near-surface water quality and the intensity of MOMs (lowest DO concentration measured in August). Meteorological factors included air temperature (means for June, May–June, June–August, and water year), precipitation (totals for February–August, June–August, and water year), wind velocity (means for June, July, August, and June–August), and variability in wind (standard deviations of daily velocities for June, July, August, and June–August). Hydrologic factors included ice-out date and P loading for specific periods (totals for January–April, May–August, current water year, previous water year, and the current and previous water years combined). Water-quality factors included near-surface TP concentrations for specific periods (means for June, June–August, July–August, and June–September), SDs (means for May, June, June–August, July–August, and July–September), and Chl-*a* concentrations (means for June, June–August, July–August, and July–September). In addition, five parameters describing physical features near the MOM were examined: water temperatures at the depth of the MOM, 12 m, and 20 m; depth of the MOM; and the relative depth of the thermocline. The relative thermocline depth was defined as the ranked depth where 15 °C was measured in August; the year when the shallowest 15 °C measurement (typical location of maximum temperature change) was given a value of 1 and the deepest 15 °C measurement was given a value of 24.

GLM–AED Model

Use of vertical one-dimensional hydrodynamic water-quality models, such as GLM–AED, is an established approach to evaluate how the hydrodynamics, chemistry, and biology in lakes respond to changes in external drivers, such as changes in weather conditions and P loading (Hipsey and others, 2017, 2019a). An advantage of using a hydrodynamic water-quality model (also referred to as a lake ecosystem model), calibrated to measured conditions in the lake, over an empirical model is that hydrodynamic water-quality models quantify and track processes relevant to nutrient and oxygen dynamics by using accepted physical, biogeochemical, and ecological interactions. Hydrodynamic water-quality models also produce results at finer temporal and spatial resolutions than observational data permit for most ecosystems (Stanley and others, 2019). Therefore, the use of hydrodynamic water-quality models, such as GLM–AED, enable the investigation of complex ecosystem dynamics, including the development of a MOM (Ward and others, 2020).

GLM–AED (version GLM_3.1.0a1 with version AED2; Hipsey and others, 2017, 2019a) was used to simulate the hydrodynamics, chemistry (especially changes in DO concentrations), and biology (especially changes in phytoplankton community structure) in Green Lake. The hydrodynamic part of the model, GLM, incorporates a flexible Lagrangian grid, in which each layer thickness dynamically changes in response to changes in water density (Hipsey and others, 2019a). Surface-mixing processes are computed by using an energy-balance approach that compares the available kinetic (turbulent) energy to the internal potential energy of the water column. The water-quality part of the model, the Aquatic Ecodynamics (AED) modeling library, was configured to simulate changes in DO; silica; inorganic carbon; organic matter (refractory, dissolved, and particulate); inorganic matter (refractory, particulate and dissolved carbon, nitrogen, and P);

three functional phytoplankton groups representing diatoms, cryptophytes and cyanobacteria; and three functional zooplankton groups representing copepods, small-bodied daphnids, and large-bodied daphnids. Sediment heat fluxes were eliminated from the model by setting heat conductivity of the sediment to 0.0. Input to GLM–AED includes lake morphometry (fetch, width, and areas at specific depths), daily meteorological data (described in the "Meteorological Data" section), daily atmospheric inputs of P and nitrogen (described in the "Atmospheric Sources of Phosphorus and Nitrogen" section), and daily inflow and outflow of water with selected constituent concentrations depending on what is being modeled (described in the "Sources of Phosphorus and Other Constituents" section). The model was run on an hourly time step, and output data were saved at a daily timestep at noon. Simulations run with hourly data output showed that the output time had little effect on the results other than water temperatures near the surface of the lake. The thickness of each layer in the model (set to a maximum of 750 layers) was allowed to vary between 0.1 and 1.5 m, with a minimum layer volume of 0.025 m^3. Thin layers enabled better vertical resolution than thick layers, especially in the metalimnion of the lake.

GLM–AED was calibrated in a sequential manner by a combination of automated and manual techniques. Initial conditions for calibration of the hydrodynamic part of the model (GLM) were set with temperature and conductivity (converted to salinity) profile data measured in the lake and daily light extinction (K_w) in the lake estimated throughout the 10-year calibration period from April 16, 2009, to September 30, 2018. Daily K_w values were estimated from daily interpolated SDs by the equation $K_w = 1.7/SD$ (Poole and Atkins, 1929). The model fit was estimated by using water temperatures measured during that period for comparison. By using measured water clarity in the lake during this initial calibration, the effects of biological changes in the lake to be later simulated with AED were incorporated into the model. During this first phase of calibration, optimal values for the wind factor, longwave radiation factor, sensible heat transfer factor, hypolimnetic mixing efficiency, and mixing efficiency for Kelvin-Helmholtz billowing were estimated by using the derivate-free, covariance matrix adaptation-evolution algorithm for strategy optimization (Hansen, 2016). This algorithm was used to minimize the root mean square error (RMSE) between measured and simulated water temperature data. Following this optimization, values describing snow albedo and the ice-on averaging period were obtained by using a manual, iterative technique to better simulate the ice on and ice off dates observed on the lake. After calibrating the physical part of the model (GLM), the model was used to estimate the flow from all areas of the watershed except Silver Creek from measured precipitation, measured inflows from Silver Creek, and measured outflows from 1994 through 2012 (as described in the "Unmonitored Water and Phosphorus Inputs and Outputs" section).

After calibrating values for several of the parameters used in GLM and estimating total inflows to the lake, values for several selected parameters used in AED were calibrated for May 7, 2013, to November 30, 2018, by using a manual, iterative calibration technique. A shorter period of time was used for this part of the calibration because of limited measured water-quality data prior to 2013. Automated calibration techniques were not used simply to obtain optimal predictions of DO concentrations in order that guidance from other physical, chemical, and biological measurements could be more easily applied (soft constraints; Yen and others, 2016; Mi and others, 2020). It was believed that this approach would provide a more robust model than calibrating the model based solely on DO concentrations and that the model would accurately simulate the processes affecting the changes in DO, such as the distribution of P and nitrogen (and their individual components) concentrations, succession of phytoplankton species, and particle settling. The model was initiated with water temperature, specific conductance converted to salinity, and water chemistry measured in the lake during spring overturn on May 7, 2013.

Mean errors, mean absolute errors, RMSEs, and Nash-Sutcliffe efficiency (NSE) fit values for several constituents were computed to demonstrate how well GLM and GLM–AED simulated the physical and chemical changes in the lake. All simulations were run by using Windows version 1809, and data summaries were performed by using R version 3.6.3 in RStudio Version 1.3.959.

Lake Water Quality

This section of the report describes the physical (water temperature and water clarity), chemical (DO, nutrients, organic carbon, and major ions), and biological (Chl-*a* and plankton) characteristics of the lake at the two main sampling sites and throughout the lake based on data collected from a sensor sled towed behind a boat. Relations between near-surface TP concentrations and SD and Chl-*a* are then described.

Water Temperature

Temperature profiles collected in the west and east sides of Green Lake during 2014–18 showed that water temperatures typically were similar on the west and east sides of the lake at any defined depth, except for the effects of the wind-caused internal oscillations of water in the lake that are also referred to as seiches. Thermal stratification typically developed in May and lasted through late fall (November), and it also developed under the ice. Thermal stratification limited the extent of vertical mixing during these periods. Maximum temperatures in the upper layer of the lake (epilimnion) reached about 25–27 °C in late July to early August. The thermocline developed in May and reached an equilibrium depth of about 10–12 m in July and August, before deepening during September through November. During summer, near-bottom temperatures near the deep hole (west side) reached about 4.6 to 5.2 °C and stayed almost constant throughout most of the summer. The seasonal progression in water temperatures in 2017 is shown for the west side of the lake in figure 4*A*.

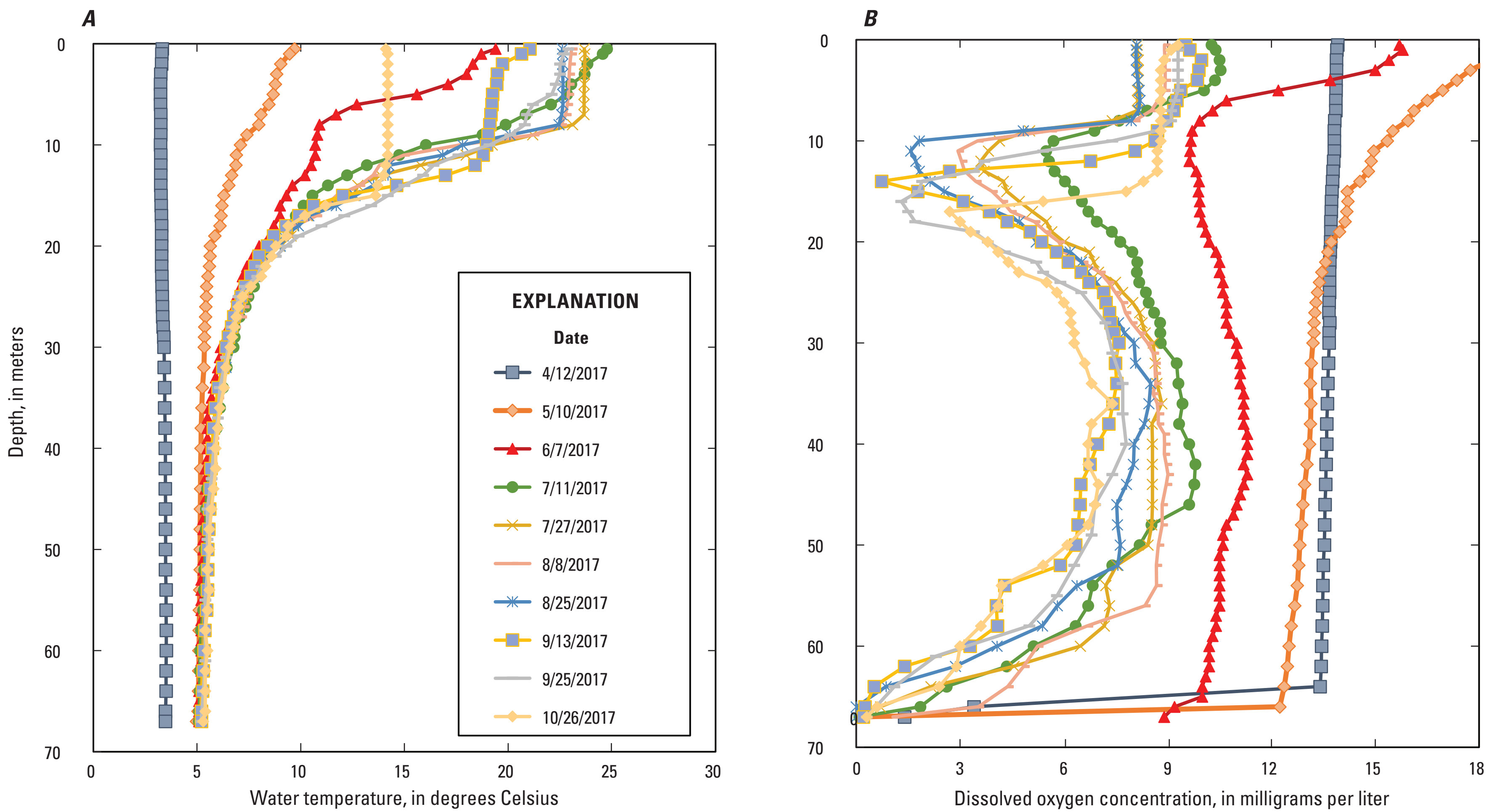


Figure 4. Changes in *A*, water temperature and *B*, dissolved oxygen in the west side of Green Lake, Wisconsin, in 2017.

Water Clarity

Water clarity data, quantified using SD, for the west and east sides of Green Lake during 2014–18, are shown in figure 5*A*. SDs were typically shallower on the east side of the lake than on the west side, but the difference was usually less than 0.5 m (fig. 5*A*). Water clarity had a relatively consistent seasonal pattern, in which clarity was best just after ice out (April, with a mean SD = 5.5 m), decreased in May (mean SD = 2.3 m), increased in June (mean SD = 6.8 m), then gradually decreased as summer progressed (SD = 4.6 m in September). The annual mean July–September SDs ranged from 4.2 m to 5.9 m on the west side of lake and from 3.5 m to 5.6 m on the east side of the lake. July through September is the period evaluated by the WDNR for SDs. The geometric mean July–September SDs for the west and east sides of the lake for the entire 2014–18 period were 4.9 m and 4.4 m, respectively (table 3). If it is assumed that the lower limit of the photic zone was 2.7 to 3 times the SD (Luhtala and Tolvanen, 2013), then the photic zone in Green Lake during summer typically extended below the thermocline depth. Boyd (1980) hypothesized that if the lower limit of the photic zone extends below the thermocline depth, then a maximum in DO concentration may be expected to occur, which is contrary to what was observed in Green Lake. The mean summer clarity during 2014–18 was slightly better than for the 10-year period 2009–18 or for the entire 2004–20 period. Based on these SD data, Green Lake would be classified as an oligotrophic lake (Carlson, 1977).

As part of this study, SDs were assembled from data collected on Green Lake beginning in 1906 (described in the "Lake-stage and Lake Water-quality Monitoring" section). The SDs were most commonly measured in July and August; therefore, historical data from these months were examined to determine if and how water clarity has changed during these months since 1906 (fig. 6*A*). SDs during July and August were about 4 m in the early 1900s but were variable among years. Clarity appeared to be better in July in the late 1960s and early 1970s, with SDs often 6 m or greater. Clarity then appeared to decline in the late 1970s, with SDs usually 3 to 5 m. From the 1980s until around 2000, SDs were relatively stable around 4 to 6 m. After around 2000, SDs increased until around 2014, possibly because of the introduction of *Dreissena polymorpha* (zebra mussels), a filter feeding invasive species that was first documented in the lake in October 2001 (fieldwork comments in the WDNR SWIMS database; Wisconsin Department of Natural Resources, 2020b). Mean July–September SDs for both sides of the lake from 1978 to 2020 demonstrate a significant increase ($p<0.05$) and the relatively consistent difference between the west and east sides of the lake (fig. 7*A* and table 4). It is difficult to draw water-quality conclusions from changes in SDs because of the confounding dynamics of the zebra mussels.

Dissolved Oxygen

DO concentrations throughout the lake were near saturation shortly after ice out, typically reaching more than 15 mg/L in early May. After the lake stratified, near-surface DO concentrations stayed near saturation during summer, typically 8–9 mg/L. However, once stratification developed, DO concentrations below the epilimnion decreased, especially in the metalimnion and deeper parts of the lake (hypolimnion). Near-bottom DO concentrations near the deep hole on the west side of the lake typically declined to 0 mg/L in late July to early August. DO concentrations in the metalimnion decreased to less than 5 mg/L (the WDNR criterion for DO for a warm-water fishery; Wisconsin Department of Natural Resources, 2021) and occasionally decreased below 1 mg/L in late August and September. The MOM usually occurred at depths between 10 and 13 m. The seasonal progression in DO concentrations throughout the water column is shown for 2017 for the west side of the lake in figure 4*B*, with a MOM of 0.7 mg/L measured in mid-September. DO concentrations in the metalimnion are expected to continue to decrease while stratification persists later in the fall; however, DO measurements were not typically collected after early September in this study.

As part of this study, DO profiles were assembled from monitoring efforts that began as early as 1905 (described in the "Lake-stage and Lake Water-quality Monitoring" section). Although DO concentrations in the metalimnion are lowest in September, measured profiles describing the MOM were most commonly measured in mid- to late August; therefore, data from mid- to late August were examined to determine if and how DO concentrations in the metalimnion and hypolimnion have changed since 1905 (fig. 8). Minimum DO concentrations in the metalimnion have decreased since 1905 (statistically significant at $p<0.001$ as determined by a regression *t*-test). In the early 1900s, MOMs were much less pronounced than what were measured after the late 1970s, and in the earlier years, a MOM did not develop in all years. In the late 1970s, DO concentrations in the metalimnion were typically lower than in the early 1900s, but a MOM still did not occur in every year. Since the early 1980s, DO concentrations in the metalimnion were lower than in earlier years and DO concentrations typically declined to less than 5 mg/L. From 2007 to 2020, DO concentrations in the metalimnion in mid- to late August declined to less than 5 mg/L in all years and ranged from 0.24 mg/L in 2012 to 4.7 mg/L in 2014.

DO concentrations in the deepest areas of the lake (60 and 65 m) during August may have slightly declined since 1905; however, the changes in DO concentrations at these depths in the hypolimnion were not statistically significant ($p>0.35$; fig. 8*B*). All assembled water temperature and DO data measured in August are available in Robertson and Kennedy (2021).

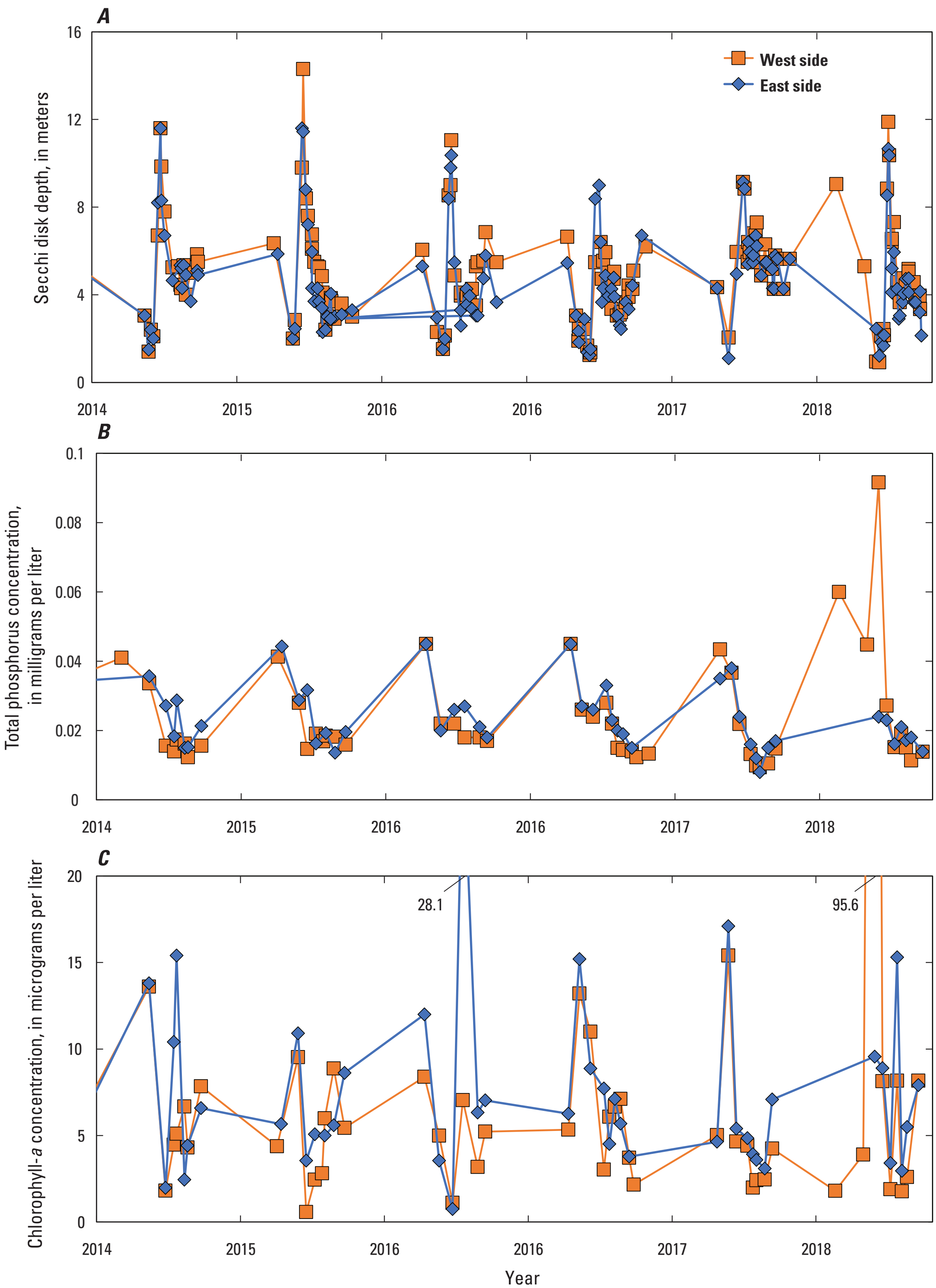


Figure 5. Changes in near-surface water quality in Green Lake, Wisconsin: *A*, Secchi disk depth, *B*, total phosphorus concentration, and *C*, chlorophyll-*a* concentration during 2014–18.

Table 3. Geometric mean summer total phosphorus concentrations in milligrams per liter, chlorophyll-*a* concentrations in micrograms per liter, and Secchi disk depths in meters for three different time periods in Green Lake, Wisconsin.

Location in lake	Total phosphorus, June–September	Chlorophyll-*a*, July–September	Secchi disk depth, July–Sept
Mean over 2004–20			
East side	0.019	5.6	4.0
West side	0.017	4.3	4.5
Average	0.018	5.0	4.2
Mean over the past 10 years (2009–18)			
East side	0.018	5.5	4.3
West side	0.016	4.2	4.8
Average	0.017	5.0	4.5
Mean over the past 5 years (2014–18)			
East side	0.020	6.4	4.4
West side	0.016	4.6	4.9
Average	0.019	5.8	4.7

Nutrients

Near-surface TP concentrations in both sides of the lake during 2014–18 had consistent seasonal patterns, with relatively high concentrations just after ice out, typically exceeding 0.040 mg/L, which then declined as summer progressed, typically decreasing to about 0.015–0.020 mg/L in summer (fig. 5*B*). Near-surface TP concentrations then increased in late fall and stayed high throughout winter according to the measurements made in February 2019, December 2019, and March 2020. The high TP concentrations in late fall and persisting through winter indicate that most of the P entering the water column from internal sediment recycling from the previous summer and P entering the lake from other external sources during fall and winter remains in the water column until spring. TP concentrations remaining high from fall overturn to the following spring is consistent with the seasonal pattern hypothesized by Stauffer (1985b) for calcareous lakes, like Green Lake.

Annual mean near-surface June–September TP concentrations during 2014–18 ranged from 0.017 mg/L to 0.019 mg/L on the west side of lake and from 0.017 mg/L to 0.023 mg/L on the east side of the lake. June through September is the period evaluated by the WDNR for near-surface TP concentrations. The geometric mean near-surface June–September TP concentrations for the west and east sides of the lake for the entire 2014–18 period were 0.016 mg/L and 0.020 mg/L, respectively (table 3). The geometric mean June–September TP concentrations for 2014–18 were slightly higher than those for 2009–18 and 2004–20 (table 3). The geometric mean near-surface June–September TP concentrations were above the 0.015-mg/L TP criterion for a lake with a two-story fishery for all three of these periods. Based on near-surface TP concentrations, Green Lake would be classified as a mesotrophic lake on the basis of the Carlson (1977) TSI.

As part of this study, TP concentration data were assembled from monitoring efforts on Green Lake beginning in 1971 (described in the "Lake-stage and Lake Water-quality Monitoring" section). Near-surface TP concentrations were most commonly measured in July and August; therefore, data from these months were examined to determine if and how near-surface TP concentrations have changed since 1971 (fig. 6*B*). From these data, no long-term trend was apparent. More consistent monitoring began around 1978 for the west side of the lake and began in 1988 for the east side of the lake. There was no statistically significant trend ($p>0.28$) in mean near-surface TP concentrations for June–September measured on either side of the lake from 1978 to 2020 (fig. 7*B*). TP concentrations may have been a little higher prior to 1986, but given the variability in the concentrations between 1978 and 1983, it was hard to define the mean concentration for that time period (fig. 7*B*; table 4). TP concentrations were occasionally higher on the east side of the lake than the west side, especially during 1995–2005.

Near-surface dissolved reactive phosphorus (DRP) concentrations also had a consistent seasonal pattern of high concentrations just after ice out, typically about 0.03 mg/L, which decreased to less than 0.01 mg/L in mid-May and remained near the detection limit (0.002 mg/L) until fall (U.S. Geological Survey, 2020). The low DRP concentrations throughout summer indicate that algal production in Green Lake is P limited. At fall overturn, near-surface DRP concentrations increased to 0.03 to 0.04 mg/L because of the P from internal sediment recycling being mixed throughout the lake, and then DRP concentrations remained high until spring.

Concentrations of TP and DRP were measured at various depths throughout the water column during May–September 2017. As summer progressed, TP and DRP concentrations in the epilimnion and metalimnion decreased. Concentrations of TP and DRP in the epilimnion above the MOM, at the depth of the MOM, and below the MOM in the lower part of the metalimnion were all similar. Near-bottom TP and DRP concentrations on the west side of the lake increased after the bottom water became anoxic. Near-bottom TP concentrations increased from about 0.05 mg/L in May to 0.178 mg/L in September. In late summer, most of the P near the bottom was in the dissolved form (DRP concentration of 0.154 mg/L). The increase in P concentrations above the bottom indicated that there was a release of P from the deep sediments (below about 40 m) of the lake (referred to as internal sediment recycling). Because of strong thermal stratification and the depth of the hypolimnion, P from internal sediment recycling was not likely released into the surface water until late fall. The only measurement taken in late winter (March, 2020) did not show anoxia in the deepest areas of the lake or a release of P from the sediments. TP and DRP concentrations remained about 0.05 mg/L from fall overturn to spring overturn.

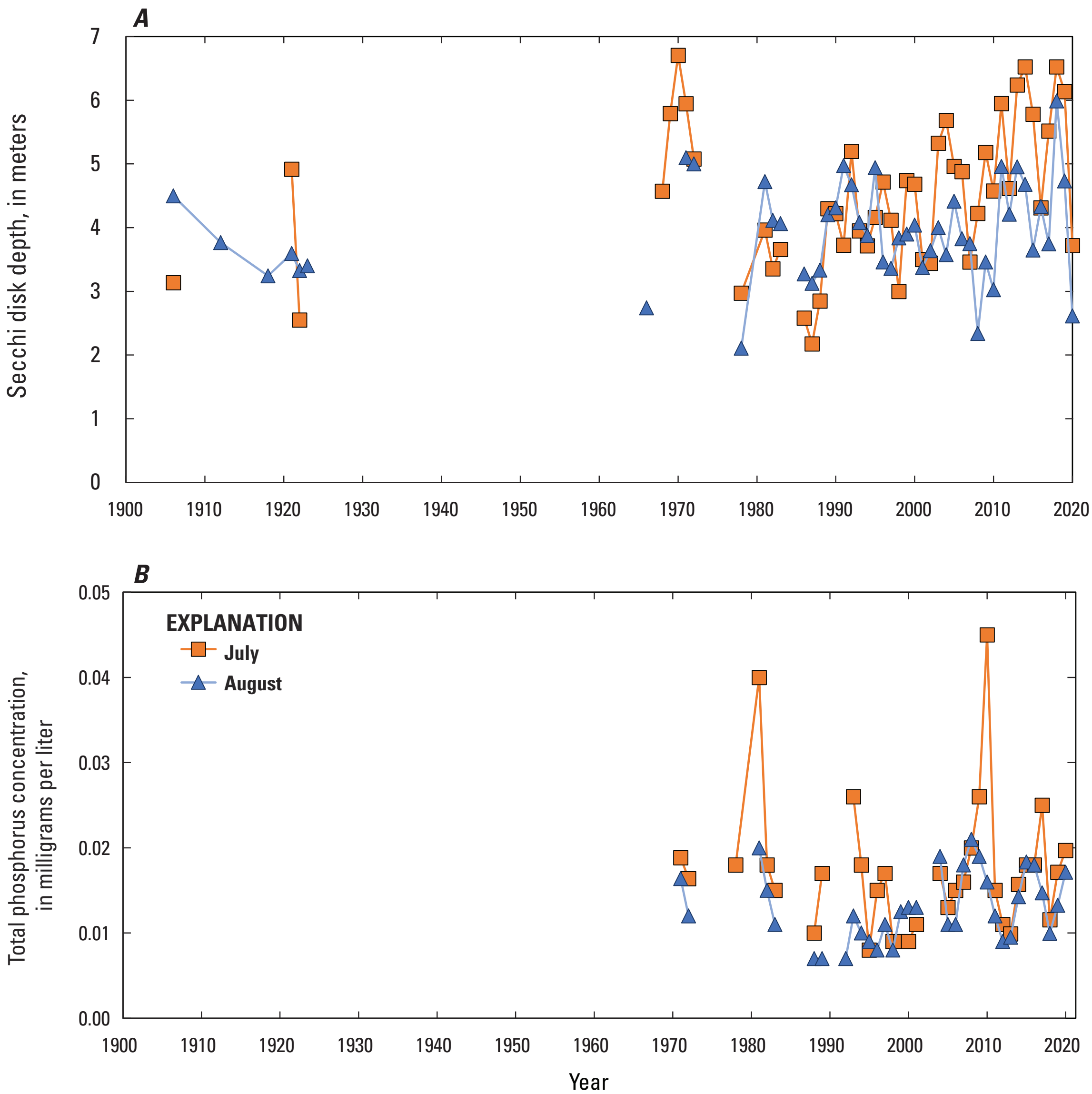


Figure 6. Long-term changes in mean July and August *A,* Secchi disk depths and *B,* near-surface total phosphorus concentrations in the west side of Green Lake, Wisconsin.

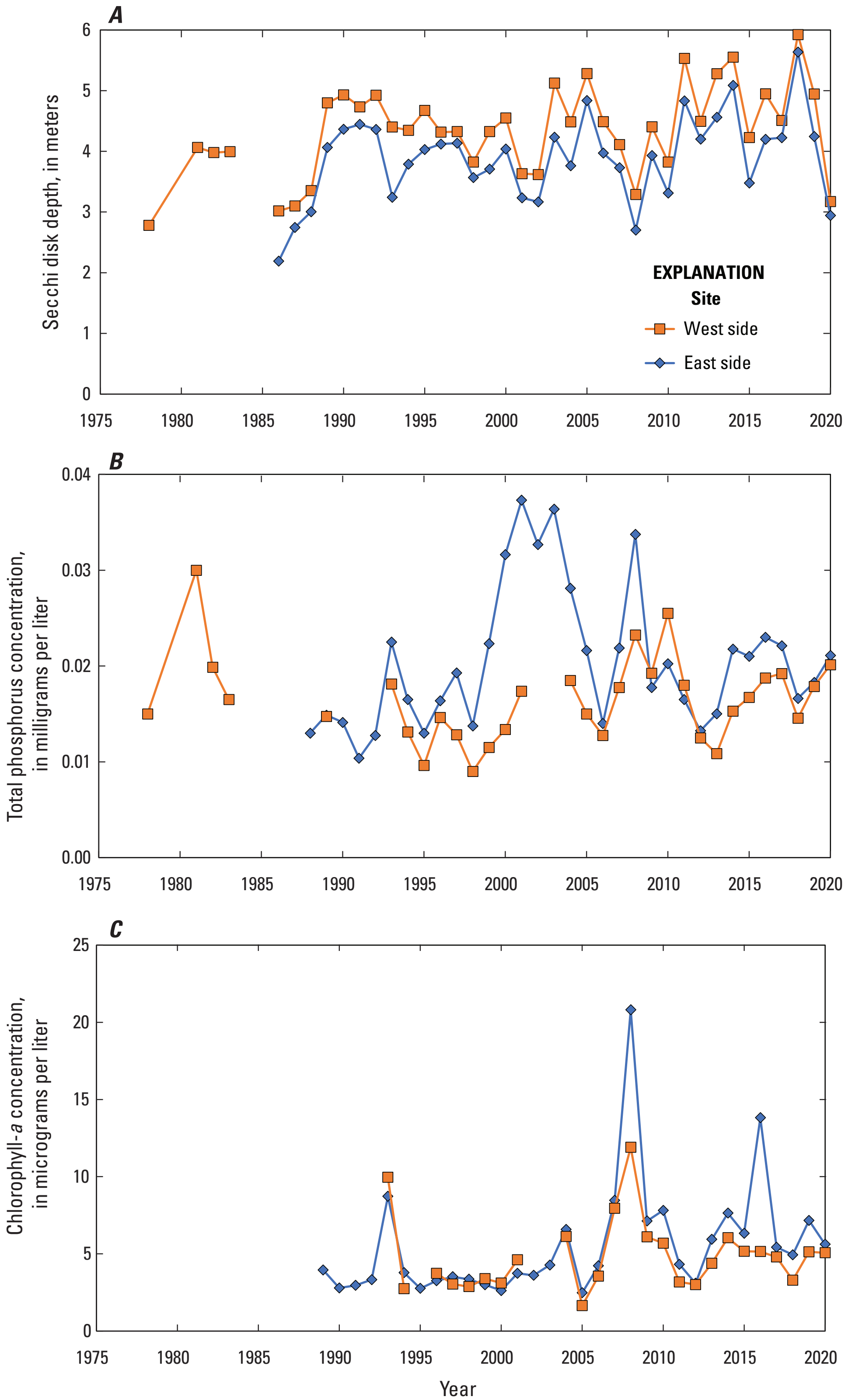


Figure 7. Changes in *A*, mean summer Secchi disk depth (July–September), *B*, near-surface total phosphorus (June–September), and *C*, chlorophyll-*a* concentrations (July–September) in west and east sides of Green Lake, Wisconsin, from 1978 to 2020.

Table 4. Near-surface, mean summer water quality in the west and east sides of Green Lake, Wisconsin.

[Secchi disk depth, in meters, is for July–September; total phosphorus, in milligrams per liter, is for June–September; and chlorophyll-*a*, in micrograms per liter, is for July–September. NA, not available]

Year	Secchi disk depth		Total phosphorus		Chlorophyll-a	
	West	East	West	East	West	East
1978	2.78	NA	0.015	NA	NA	NA
1981	4.06	NA	0.030	NA	NA	NA
1982	3.98	NA	0.020	NA	NA	NA
1983	4.00	NA	0.017	NA	NA	NA
1986	3.02	2.19	NA	NA	NA	NA
1987	3.10	2.75	NA	NA	NA	NA
1988	3.35	3.01	NA	0.013	NA	NA
1989	4.80	4.06	0.015	0.015	NA	4.0
1990	4.93	4.36	NA	0.014	NA	2.8
1991	4.73	4.45	NA	0.010	NA	3.0
1992	4.93	4.36	NA	0.013	NA	3.3
1993	4.40	3.24	0.018	0.023	10.0	8.7
1994	4.35	3.79	0.013	0.017	2.7	3.8
1995	4.68	4.03	0.010	0.013	NA	2.8
1996	4.32	4.12	0.015	0.016	3.7	3.3
1997	4.33	4.13	0.013	0.019	3.0	3.5
1998	3.82	3.57	0.009	0.014	2.9	3.4
1999	4.33	3.71	0.012	0.022	3.4	3.0
2000	4.55	4.04	0.013	0.032	3.1	2.6
2001	3.63	3.23	0.017	0.037	4.6	3.7
2002	3.62	3.17	NA	0.033	NA	3.6
2003	5.13	4.23	NA	0.036	NA	4.3
2004	4.49	3.76	0.019	0.028	6.1	6.6
2005	5.28	4.84	0.015	0.022	1.6	2.5
2006	4.49	3.97	0.013	0.014	3.6	4.2
2007	4.11	3.73	0.018	0.022	8.0	8.5
2008	3.29	2.70	0.023	0.034	11.9	20.8
2009	4.41	3.93	0.019	0.018	6.1	7.1
2010	3.82	3.31	0.026	0.020	5.7	7.8
2011	5.53	4.83	0.018	0.017	3.2	4.3
2012	4.50	4.20	0.013	0.013	3.0	3.1
2013	5.28	4.56	0.011	0.015	4.4	5.9
2014	5.55	5.09	0.015	0.022	6.0	7.6
2015	4.23	3.48	0.017	0.021	5.2	6.3
2016	4.95	4.20	0.019	0.023	5.1	13.8
2017	4.51	4.23	0.019	0.022	4.8	5.4
2018	5.92	5.64	0.015	0.017	3.3	4.9
2019	4.94	4.25	0.018	0.018	5.1	7.2
2020	3.17	2.94	0.020	0.021	5.1	5.6

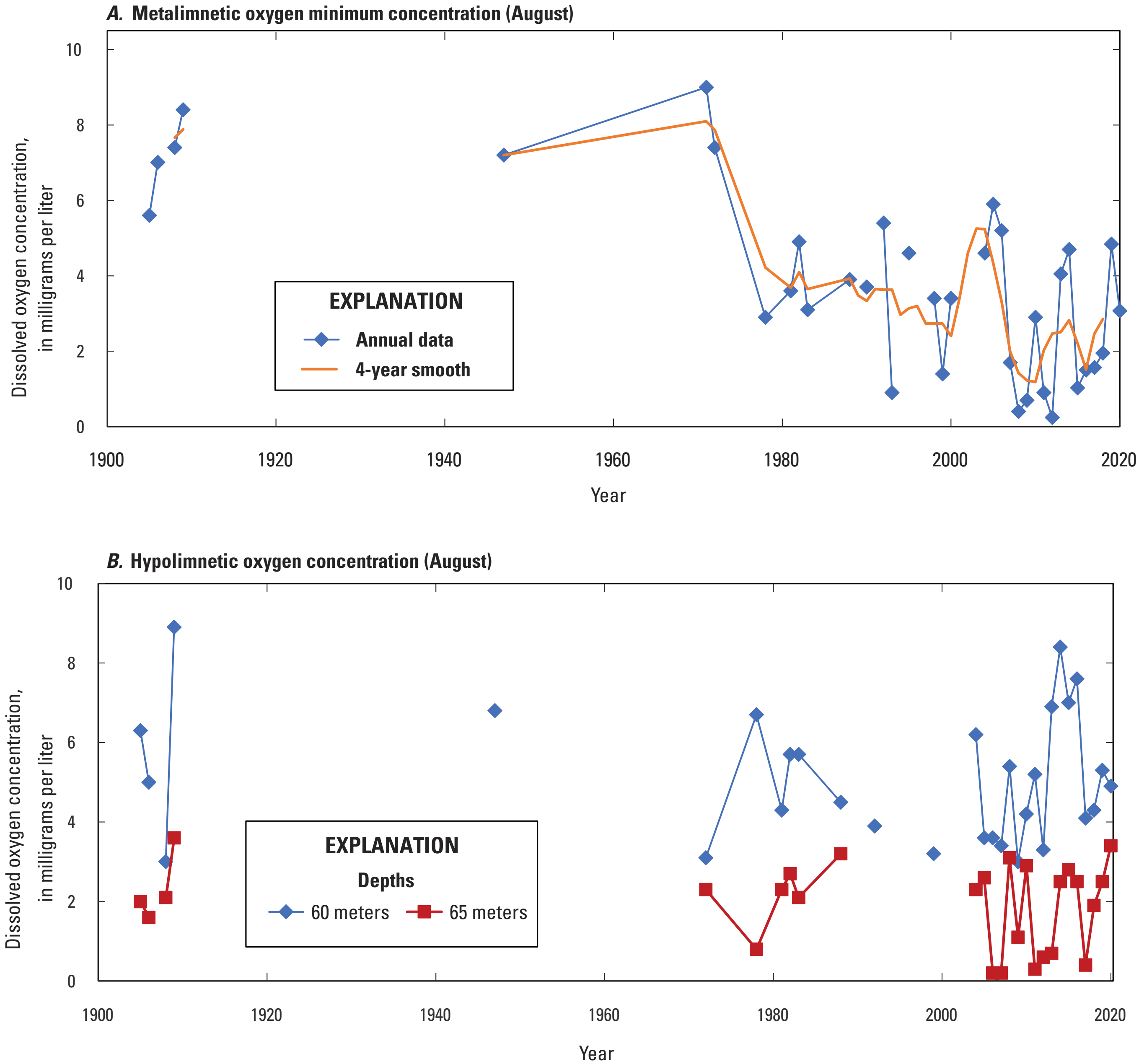


Figure 8. Minimum dissolved oxygen concentrations in the *A*, metalimnion—metalimnetic dissolved oxygen minimum—and *B*, two depths (60 and 65 meters [m]) in the hypolimnion of Green Lake, Wisconsin, measured in mid- to late August from 1905 to 2020.

Near-surface total nitrogen concentrations typically ranged from 0.5 to 1.0 mg/L, with little seasonal pattern other than that highest concentrations were typically measured in May (U.S. Geological Survey, 2020). Near-surface Kjeldahl nitrogen concentrations, similar to total nitrogen, did not consistently change throughout the year (typically around 0.5 mg/L), but highest concentrations were typically measured in May. Near-surface dissolved nitrite plus nitrate concentrations, however, had a consistent seasonal pattern of high concentrations just after ice out (typically 0.3 to 0.5 mg/L), which then declined to near the detection limit (0.019 mg/L) in June and remained low throughout summer. Near-surface dissolved ammonia concentrations were near the detection limit at all times of the year. The very low concentrations of dissolved forms of nitrogen throughout summer indicate that Green Lake may have been colimited by P and nitrogen.

The various forms of nitrogen were measured at several depths throughout the water column during May–September 2017. During May, total nitrogen concentrations were highest in the epilimnion and relatively uniform in the metalimnion and hypolimnion. From June through September, total nitrogen concentrations were lowest in the epilimnion, increased throughout the metalimnion, and were highest in the hypolimnion. The increase in total nitrogen concentrations with increasing depth was the result of losses of nitrogen from the epilimnion and upper part of the metalimnion. A slight increase in particulate nitrogen (Kjeldahl nitrogen) concentrations was often observed at the depth of the MOM. Near-bottom total nitrogen concentrations increased from about 0.7 mg/L in May to 1.2 mg/L in September, with the increase a result of increases in both particulate and dissolved forms. The increase in nitrogen concentrations just above the bottom indicates a possible release of nitrogen from the deep sediments of the lake (internal recycling of nitrogen from the sediments). Because of strong thermal stratification and the depth of the hypolimnion, nitrogen from internal sediment recycling was not likely released into the surface water until late fall.

Organic Carbon

Total and dissolved organic carbon concentrations were measured at various depths throughout the water column during May–September 2017 (U.S. Geological Survey, 2020). In general, total and dissolved organic carbon concentrations were similar, indicating that most of the organic carbon was in the dissolved form. In May, total and dissolved concentrations were relatively uniform at about 4.5 mg/L throughout the water column. During June through September, total and dissolved organic carbon concentrations increased to about 5.5 mg/L throughout the epilimnion, declined gradually through the metalimnion to about 4.5 mg/L, and remained unchanged from May in the lower metalimnion and throughout the hypolimnion.

Chlorophyll-*a*

Near-surface Chl-*a* concentrations in the west and east sides of Green Lake during 2014–18 are shown in figure 5*C*. Chl-*a* concentrations were typically higher in the east side of the lake than in the west side, but usually within 1 microgram per liter (µg/L), except at high concentrations. There was a consistent seasonal pattern in Chl-*a* concentrations, in which concentrations were typically high in May and lowest during summer; however, on occasion a relatively high concentration was measured during a summer algal bloom. July through September is the period evaluated by the WDNR for Chl-*a* concentrations. Annual mean July–September Chl-*a* concentrations ranged from 3.3 µg/L to 6.0 µg/L in the west side of the lake and from 4.9 µg/L to 13.8 µg/L in the east side of the lake. The geometric mean July–September Chl-*a* concentrations for the west and east sides of the lake for the entire 2014–18 period were 4.6 µg/L and 6.4 µg/L, respectively (table 3). From 1989 to 2020, there was a statistically significant (p<0.05, using a regression t-test) increase in mean July–September Chl-*a* concentrations in the east side of the lake but no significant (p=0.77) change in Chl-*a* concentrations in the west side of the lake (fig. 7). The highest July–September Chl-*a* concentrations were measured in 2008 in both sides of the lake and followed the largest rainfall event since the meteorological record started in 1979, which occurred in June 2008. The geometric mean July–September Chl-*a* concentrations for 2014–18 were a little higher than those for 2009–18 and 2004–20 (table 3). Based on near-surface Chl-*a* concentrations, Green Lake would be classified as a mesotrophic lake based on the Carlson (1977) TSI.

Major Ions

Major-ion data for Green Lake are available in U.S. Geological Survey (2020). Specific conductance, an indicator of the total major-ion concentration, was typically about 525 microsiemens per centimeter at 25 °C (µS/cm) throughout the lake in spring and in the hypolimnion when the lake was thermally stratified. However, when the lake stratified during summer, specific conductance in the epilimnion dropped to about 500 µS/cm. The pH in the lake was typically about 8 standard units throughout the lake in spring and in the hypolimnion when the lake was thermally stratified. During stratification, the pH in the epilimnion increased to about 8.5 to 8.8 standard units. Concentrations of several major ions were measured during spring turnover of each year. During 2014–18, mean concentrations during spring overturn were as follows: calcium—34 mg/L, sulfate—28 mg/L, magnesium—36 mg/L, chloride—40 mg/L, and dissolved iron—always less than 0.1 mg/L. The alkalinity or hardness of the water was 231 mg/L (as calcium carbonate), which classifies it as a very hard-water lake (U.S. Geological Survey, 2021). Lillie and Mason (1983) collected data from a random set of about 660 Wisconsin lakes, 44 of which were in central Wisconsin. For comparison, mean concentrations for the central Wisconsin Lakes in the Lillie and Mason (1983) study were as follows: calcium—24 mg/L, magnesium—20 mg/L, chloride—4 mg/L, and alkalinity (hardness)—122 mg/L. Green Lake had higher concentrations of most major ions, especially calcium, and had harder water than most nearby lakes and other lakes throughout Wisconsin. The water color in Green Lake was 1 platinum-cobalt unit, which was less color than in most central Wisconsin lakes.

Plankton

Samples collected from the middle of the epilimnion, middle of the metalimnion, and bottom of the metalimnion in May, June, July, August, and September of 2017 demonstrated that two types of phytoplankton were dominant during May–September in the upper water of Green Lake: cryptophytes dominated in May and cyanophytes (blue-green algae) dominated in July–September (fig. 9). Phytoplankton populations were highest in the epilimnion, moderate in the middle of the metalimnion, and lowest in the bottom of the metalimnion. Although diatoms were rarely found in Green Lake, the spring algal community is commonly dominated by diatoms in temperate lakes (Sommer and others, 1986); therefore, diatoms may have been present and dominated the community earlier in spring. Samples collected from these three depth zones showed that the zooplankton community was dominated by calanoid copepods in May and June and by calanoid copepods and various sizes of *Daphnia* during July–September (Robertson and Kennedy, 2021). Zooplankton populations were highest in the epilimnion, moderate in the middle of the metalimnion, and lowest in the bottom of the metalimnion and upper hypolimnion. High concentrations of phytoplankton or zooplankton were not observed in the middle of the metalimnion at the depth of the MOM.

Cross-Sectional Transects in the Lake

To further describe horizontal and vertical variability in water quality (water temperature and density, particulates with backscatter, algae with Chl-*a*, dissolved material with fDOM, and DO concentrations) in Green Lake at a finer resolution in midsummer, a sensor sled was towed behind a boat as it moved through the lake during August of 2017 and 2018. These cross sections were made when light winds were expected and the effects of internal seiches were expected to be minimal. In 2017, however, a breeze caused the thermocline to deepen on the east side of the lake, making it difficult to examine horizontal differences in the lake. In 2018, data were collected during calmer conditions, but the thermocline was still depressed slightly on the east side of the lake (fig. 10). The movement of the sensor sled, shown in figure 10*A* with a yellow line, was concentrated in the metalimnion near the MOM to better describe the changes in this zone. A dashed horizontal line at a depth of 10 m is shown on each plot to allow a better horizontal comparison among constituents. Data shown in figure 10 and data collected on north-to-south transects demonstrated that, in the absence of internal seiches, water quality throughout the lake was relatively uniform horizontally, with no changes propagating out from any tributary or shoreline. Therefore, the MOM is likely not the result of nearshore interflow or DO consumption of sediment at the depth of the MOM. These plots show strong thermal stratification beginning at about 8 m (fig. 10*A*), a layer of high Chl-*a* fluorescence peaking at about 4.5 m (fig. 10*B*), strong backscatter from the surface down to about 9 m (fig. 10*C*), a layer of fDOM peaking at 9.5 m (fig 10*D*), and the lowest DO concentrations in the metalimnion at about 10 m (fig. 10*E*). The peaks shown in figure 10 are exaggerated because of the small range in the scales used for each parameter. Although there was a peak in Chl-*a* at about 4.5 m, it is likely that Chl-*a* concentrations were high throughout most of the epilimnion, but nonphotochemical quenching resulted in lower fluorescence values near the surface. With exposure of phytoplankton to high light energy, such as in shallow parts of the epilimnion, nonphotochemical quenching can cause a decrease in fluorescence quantum yield per unit algal pigment, which complicates the direct use of in vivo fluorescence measurements as a proxy for pigment concentrations (Fennel and Boss 2003; Hamilton and others, 2010). Highest fDOM concentrations (fig. 10*D*) coincided with the MOM at about 10 m. Generally, fDOM concentrations show similar patterns to measured dissolved organic carbon concentrations; however, measured dissolved organic concentrations were high throughout the epilimnion and upper part of the metalimnion (see the "Lake Water Quality—Organic Carbon" section). The lower fDOM concentrations (fig. 10*D*) in the shallow parts of the epilimnion may have been caused by slightly higher backscatter in the epilimnion (fig. 10*C*) or ultraviolet interference in shallow water. Therefore, it is believed that high fDOM concentrations may extend from the surface down to the depth of the MOM. The high fDOM and high dissolved organic carbon concentrations in the epilimnion and upper metalimnion suggest that the low DO concentrations in the metalimnion near the bottom of the high fDOM and high dissolved organic carbon may partially result from phytoplankton respiration at this depth and the decomposition of settling organic matter (mostly dead algae) that have reached a depth of relatively similar density.

Although water quality throughout the lake was relatively uniform horizontally when internal seiches were not present, strings of continuous recording temperature and DO sensors placed in the west and east sides of Green Lake in 2017 and 2018 showed that winds frequently cause stratification to deepen on the downwind side of the lake and become shallower on the upwind side of the lake (Naziri Saeed, 2020). The effects of the wind are accentuated in Green Lake when winds are from the east or west because of the longer fetch in those directions. When wind velocities decrease, the water initially returns to its stable horizontal condition but typically overshoots the stable condition and, thus, initiates a rocking motion. Therefore, water-quality measurements at the same

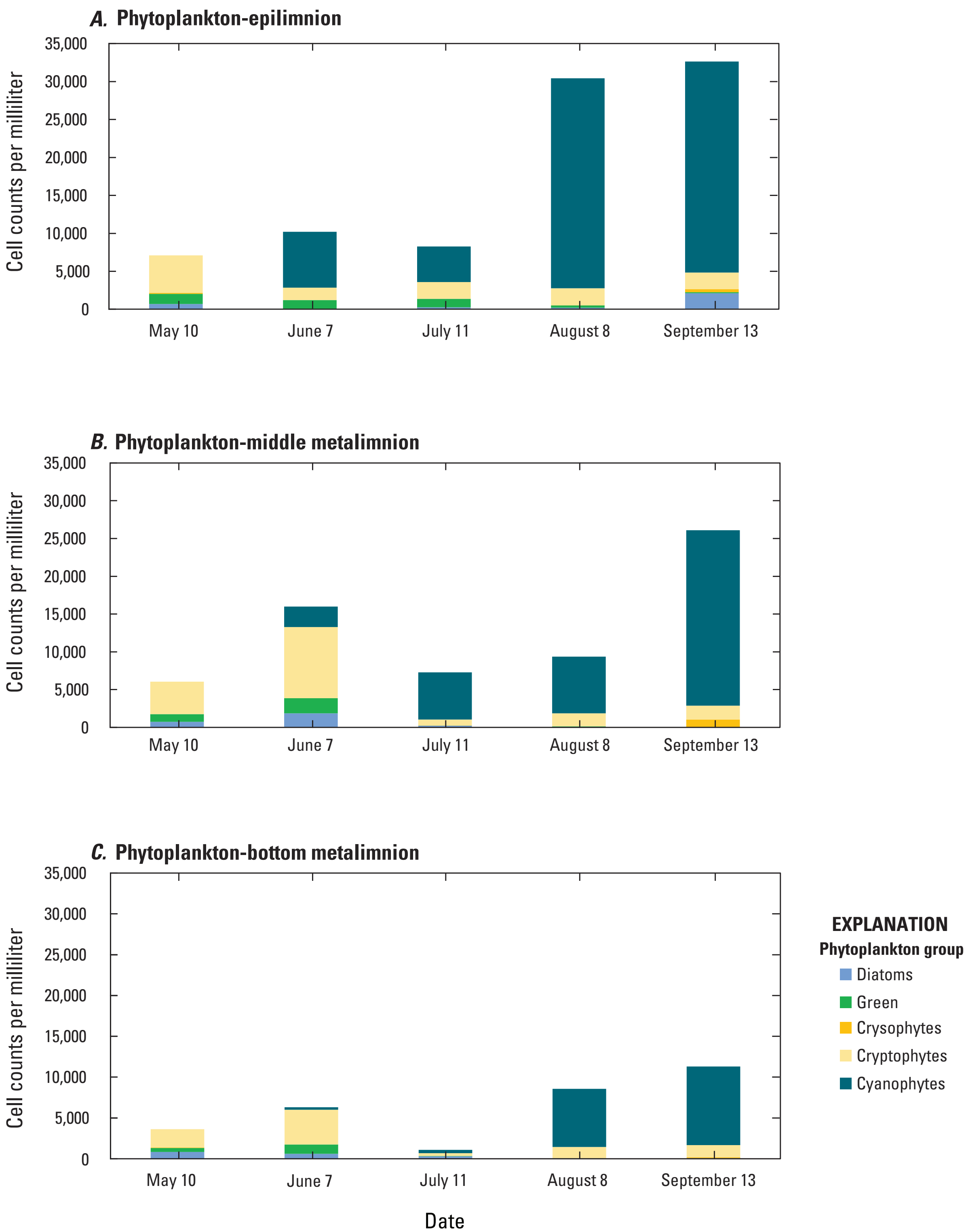


Figure 9. Phytoplankton groups measured in the *A*, epilimnion, *B*, middle of the metalimnion, and *C*, bottom of the metalimnion in Green Lake, Wisconsin, during 2017.

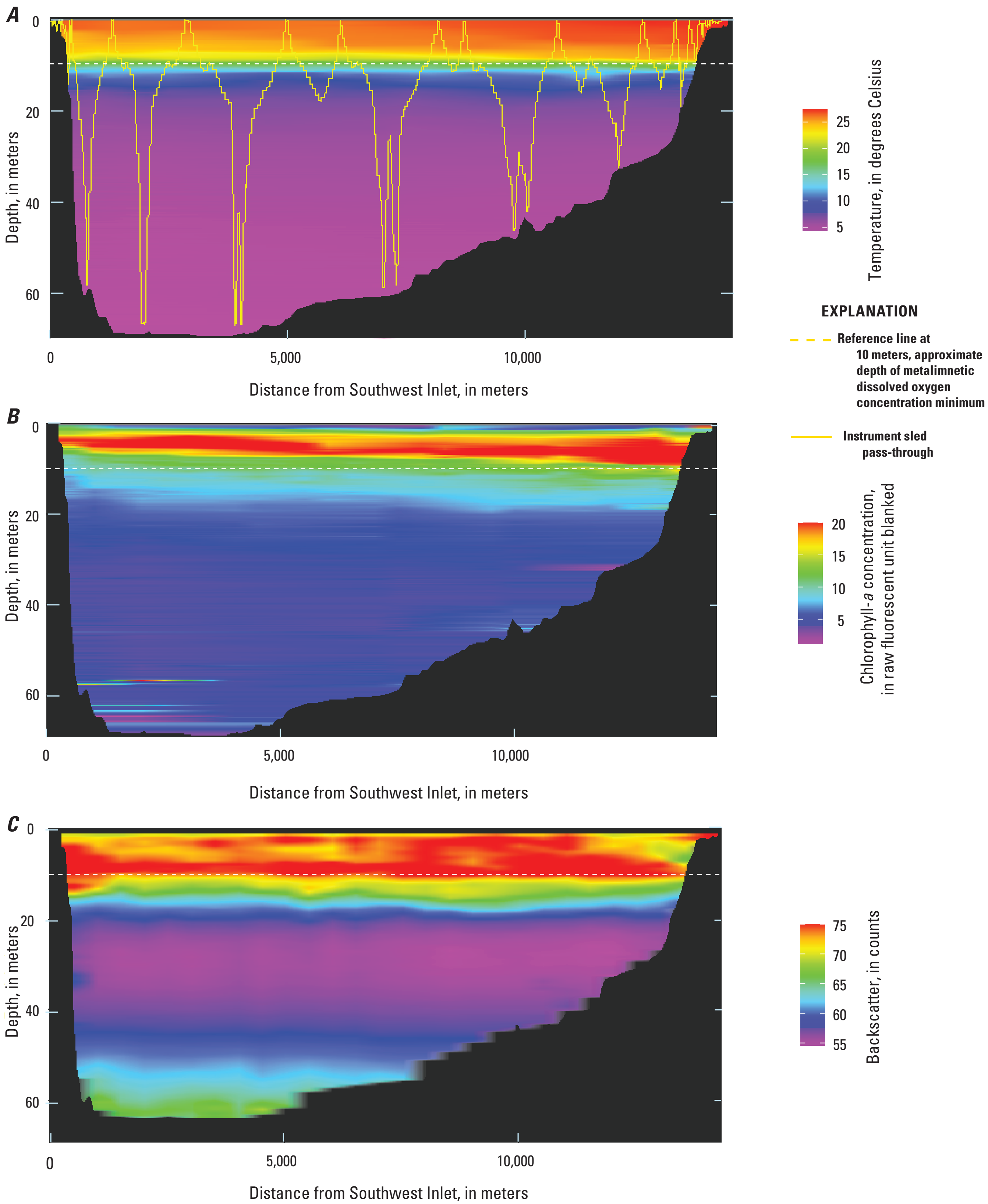


Figure 10. Cross sections of *A*, water temperature, *B*, chlorophyll-*a*, *C*, backscatter, *D*, fluorescent dissolved organic matter, and *E*, dissolved oxygen through Green Lake, Wisconsin, from the Southwest Inlet to the Silver Creek Inlet on August 14, 2018. The yellow line in *A* represents the movement of the sensor sled. Dashed lines represent a 10-meter depth reference.

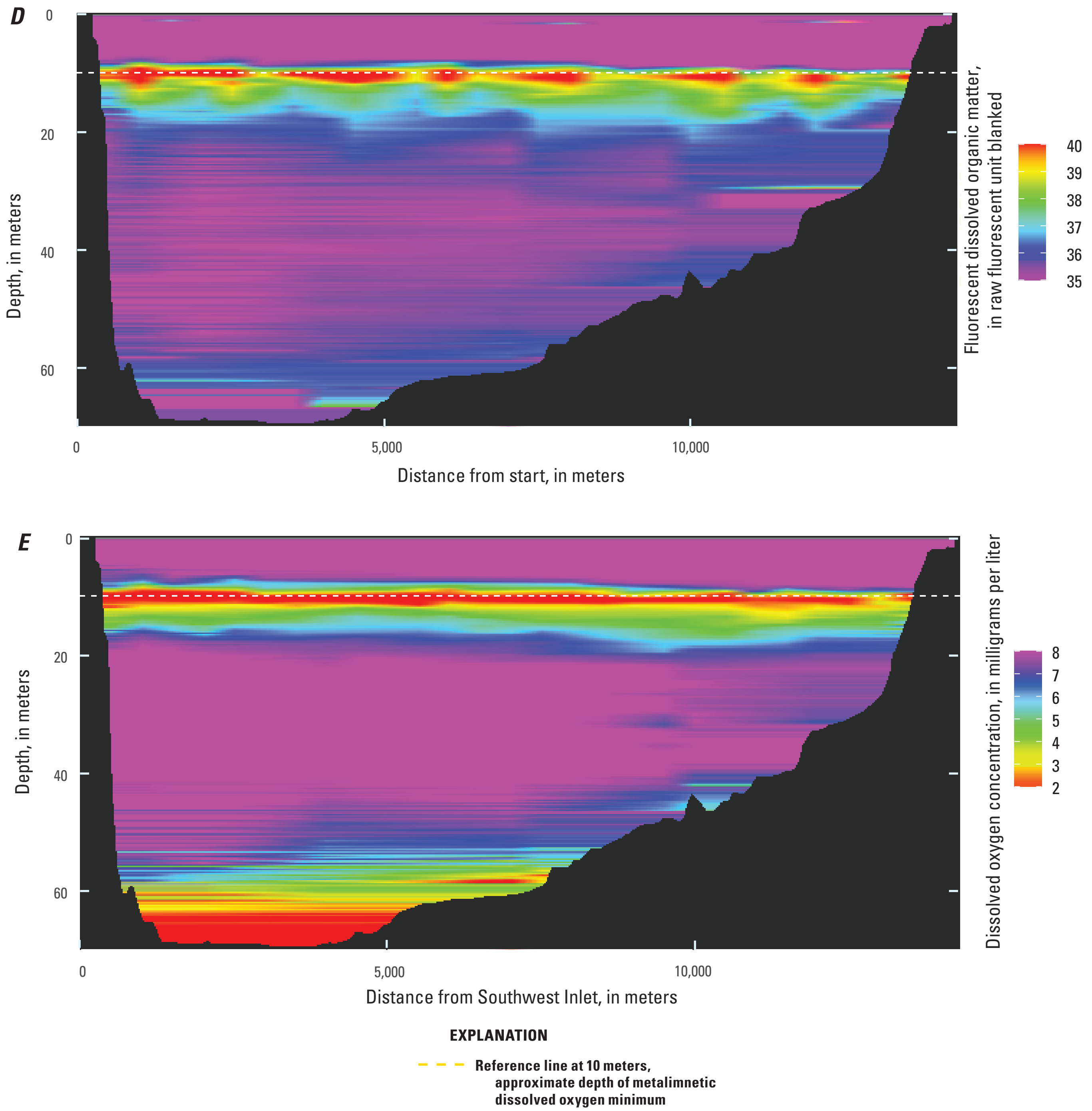


Figure 10.—Continued

depth in the west and east sides of the lake often differed on any given day, especially in the metalimnion where there was a rapid change in temperature and DO concentrations with depth. Internal seiches and the resulting turbulence may entrain metalimnetic water with relatively high dissolved P and nitrogen into the epilimnion, and this water may promote additional algal growth.

Relations Between Total Phosphorus Concentrations and Secchi Disk Depths and with Chlorophyll-*a* Concentrations

Several published empirical relations have been developed to estimate summer mean Chl-*a* concentrations and SDs from TP concentrations in a lake; however, there is often much uncertainty in these relations because of variability in conditions among lakes. Therefore, site-specific empirical relations were developed based on the annual measured data from the east and west sides of Green Lake from 1986 to 2019 (fig. 11). Near-surface mean June–September TP concentrations and mean July–September SDs from table 4 are shown for each year in figure 11 (*A* and *B*). Different integration periods were used because the time periods for the WDNR evaluations are different. Mean July–September SDs can be estimated from measured June–September TP concentrations by using equations 6 and 7, although the relation for the west side of the lake was only significant at p=0.20. Near-surface mean June–September TP concentrations and mean July–September Chl-*a* concentrations are shown in figure 11 (*C* and *D*). Mean July–September Chl-*a* concentrations can be estimated from measured June–September TP concentrations by using equations 8 and 9, although the relation for the east side of the lake was only significant at p=0.06:

$$\text{SD (in meters)} = 4.64 - 30.90 * \text{TP (in milligrams per liter)} - \text{east side} \quad (6)$$

$$\text{SD (in meters)} = 5.20 - 40.89 * \text{TP (in milligrams per liter)} - \text{west side} \quad (7)$$

$$\text{Chl-}a \text{ (in micrograms per liter)} = 1.98 + 172.82 * \text{TP (in milligrams per liter)} - \text{east side} \quad (8)$$

$$\text{Chl-}a \text{ (in micrograms per liter)} = -1.51 + 395.93 * \text{TP (in milligrams per liter)} - \text{west side} \quad (9)$$

Hydrology and Water Budget

The hydrology of Green Lake can be described in terms of components of its water budget. Water enters the lake as precipitation (PPT), surface-water inflow (SW_{In}), and groundwater inflow (GW_{In}). Surface-water inflow (SW_{In}) is the sum of monitored (gaged) surface-water inflows ($SW_{GagedIn}$) from Silver Creek and the SW Inlet and unmonitored (ungaged) surface-water inflows ($SW_{UngagedIn}$). Inflows from White Creek and Hill Creek were included in $SW_{UngagedIn}$. Water leaves the lake through evaporation (E), surface-water outflow (SW_{Out}), and groundwater outflow (GW_{Out}). The change in the volume of water stored in the lake (ΔS) is then equal to the sum of the volumes of water entering the lake minus the sum of the volumes leaving the lake. The water budget for a lake for a period of interest may be represented by equation 10:

$$\Delta S = (PPT + SW_{In} + GW_{In}) - (E + SW_{Out} + GW_{Out}). \quad (10)$$

From the beginning of WY 2014 to end of WY 2018, the water level of Green Lake changed little ($\Delta S = 0$), and groundwater was not quantified; therefore, if net groundwater input is assumed to be incorporated into $SW_{UngagedIn}$, equation 10 can be simplified to equation 11:

$$PPT + SW_{GagedIn} + SW_{UngagedIn} = E + SW_{Out}. \quad (11)$$

The methods used to compute daily estimates for each term in equation 11 are described in the section “Data-Collection Methods and Sites,” except E, which was computed by using the calibrated GLM model described in the “GLM-AED Calibration” section.

During the 5-year study period, mean precipitation on the lake surface was 0.944 meters per year (m/yr); annual precipitation ranged from 0.780 m in WY 2015 to 1.068 m in WY 2018 (fig. 12). The most recent, long-term 30-year mean annual precipitation (1990–2019) was 0.871 m/yr, which was 7.6 percent less than during this study period, and the mean annual precipitation during the Upper Fox and Wolf Rivers TMDL study (Wisconsin Department of Natural Resources, 2020a) was 0.848 m, which was 10.1 percent less than this study period.

Daily values for each term of the water budget were measured or estimated for WYs 2014–18 and summarized annually in table 5 and for entire period in figure 13. Total annual evaporation ranged from 0.764 m in WY 2017 to 1.005 m in WY 2014. Total annual inflow ranged from 66.1 million m^3 in WY 2015 to 129.4 million m^3 in WY 2017. During WYs 2014–18, Silver Creek was the largest contributor of water to Green Lake (39.9 percent), followed by precipitation (27.8 percent), unmonitored areas including net groundwater input (19.4 percent), and the Southwest Inlet (12.9 percent). The largest loss of water from Green Lake was to outflow in the Puchyan River (74.8 percent), followed by evaporation (25.2 percent). It was assumed that there were no water losses to groundwater.

The residence time of water in Green Lake, which is also referred to as the number of years to replace the volume of water in the lake, was computed by dividing the volume of Green Lake by its mean annual inflow volume and its mean annual outflow volume. Typically, when using inflow to estimate residence times, only the volume of precipitation minus evaporation is used in the computations (Panuska and Kreider, 2003). Based on the total water input to Green Lake

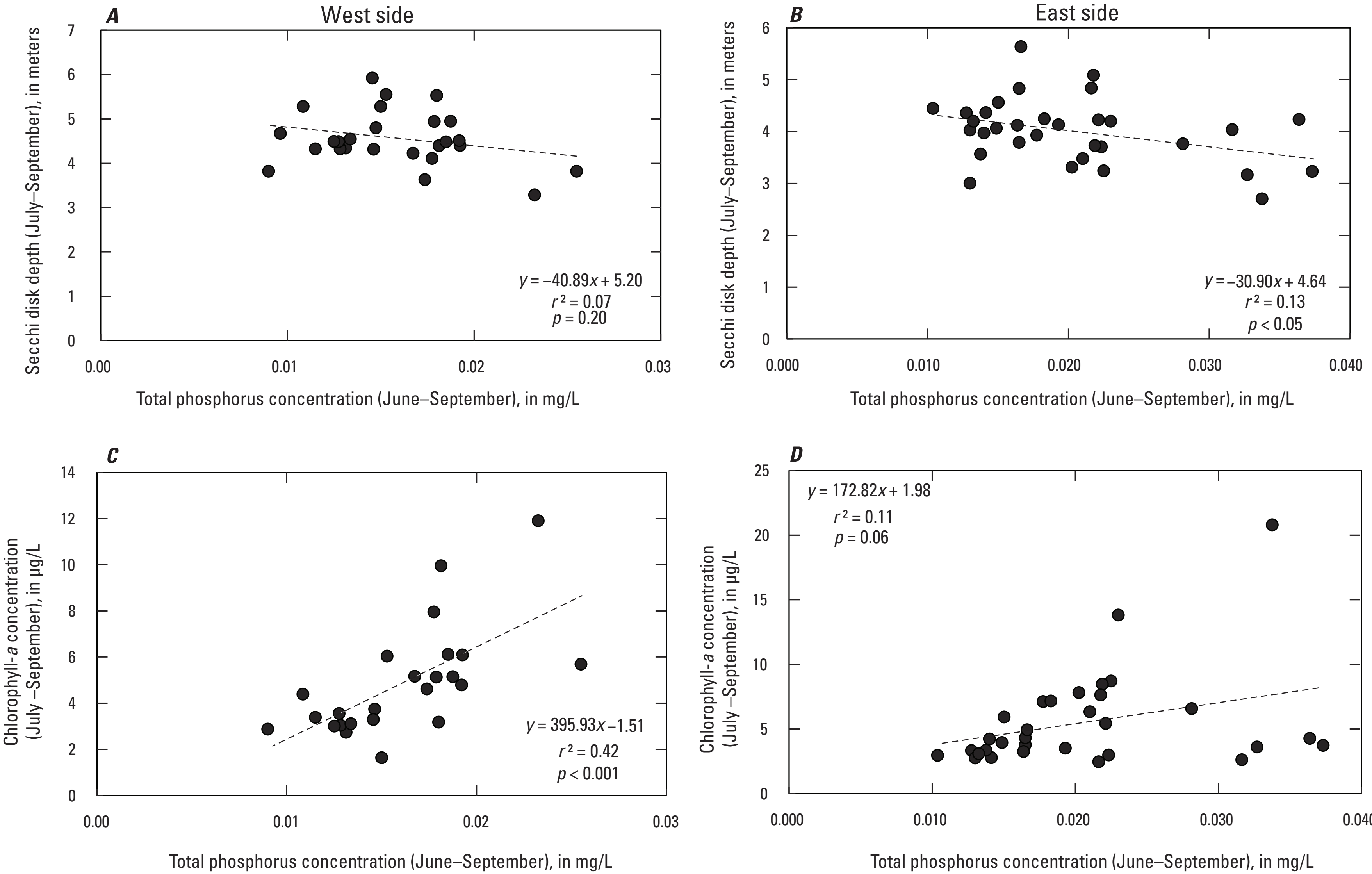


Figure 11. Near-surface mean June–September total phosphorus concentrations versus mean July–September Secchi disk depth in the *A*, west side and *B*, east side of Green Lake, Wisconsin, and near-surface mean June–September total phosphorus concentrations versus near-surface mean July–September chlorophyll-*a* concentrations in the *C*, west side and *D*, east side of Green Lake, Wisconsin, for 1986 to 2019. Linear regression lines are displayed on each graph. mg/L, milligram per liter. µg/L, microgram per liter.

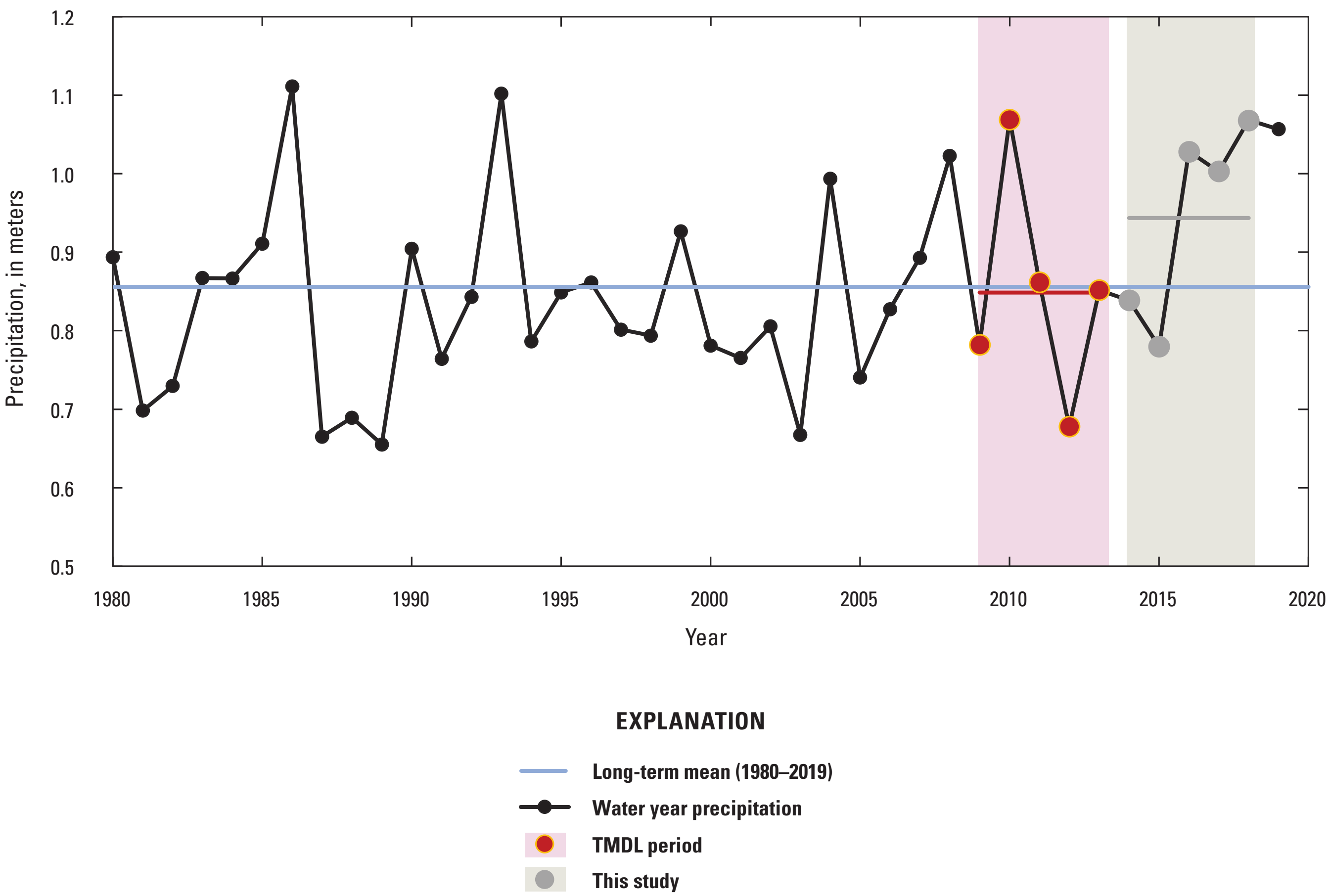


Figure 12. Annual precipitation on Green Lake, Wisconsin, during water years 1980–2019, with means provided for this study and that of the Upper Fox and Wolf Rivers Total Maximum Daily Load (TMDL) study (Wisconsin Department of Natural Resources, 2020a).

Table 5. Water budget for Green Lake, Wisconsin, for water years 2014–18.

[All inflows and outflows are in millions of cubic meters per year. Unmonitored areas include inputs from Hill and White Creeks and net groundwater input. Residence time based on inflows were computed using total inputs minus evaporation. HWY, Highway; WY, water year]

Water year	Silver Creek at HWY A	Southwest Inlet	Unmonitored area	Precipitation	Total inputs	Evaporation	Outflow	Total inputs less evapo-ration	Residence time based on inflows, years	Residence time based on outflow, years
2014	36.5	7.3	22.2	24.7	90.7	29.6	59.6	61.1	16.2	16.6
2015	23.8	7.3	12.0	23.0	66.1	24.6	39.1	41.5	23.8	25.3
2016	43.2	15.1	19.9	30.3	108.6	24.4	83.4	84.2	11.7	11.9
2017	55.2	18.4	26.3	29.6	129.4	22.5	109.5	106.9	9.3	9.0
2018	42.6	17.2	17.3	31.5	108.6	25.1	82.3	83.6	11.8	12.0
Mean of WYs 2014–18	40.3	13.1	19.6	27.8	100.7	25.2	74.8	75.4	13.1	13.2

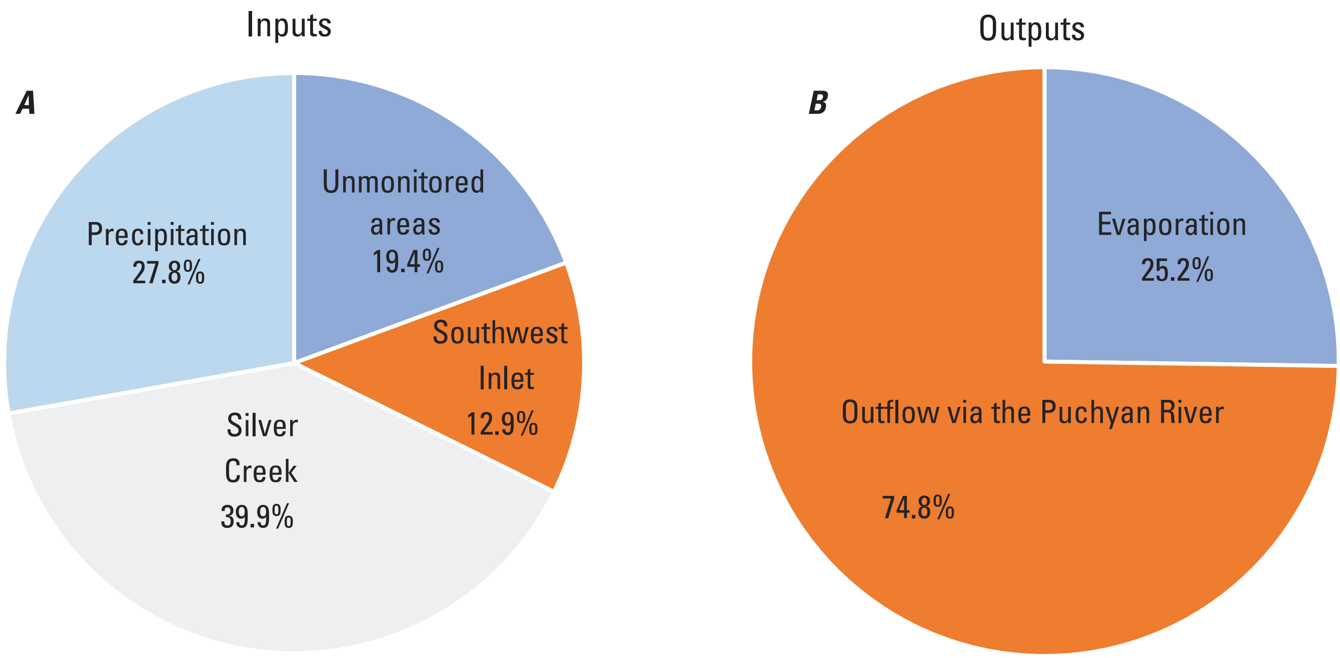


Figure 13. Water budget (total inputs and outputs) for Green Lake, Wisconsin, for water years 2014–18. *A*, Inputs. *B*, Outputs.

(precipitation, streamflow from Silver Creek, Southwest Inlet, and unmonitored areas) minus evaporation, the residence time varied from 9.3 years in WY 2017 to 23.8 years in WY 2015. For the entire WYs 2014–18 period, the water residence time was 13.1 years based on inflows and 13.2 years based on outflows.

Sources of Phosphorus and Other Constituents

Total P inputs to Green Lake were estimated daily for WYs 2014–18 to explain where its P originates and how much of that P is from controllable sources that may be reduced with management actions. Estimated external sources of P to the lake included inputs from atmospheric sources (precipitation and dryfall), surface-water inflow (from Silver Creek, Southwest Inlet, Hill Creek, White Creek, and unmonitored areas), septic systems, and waterfowl. There may be other minor external sources that were not included in this estimate. In addition to external sources, P can be released from the bottom sediments of the lake during certain times of the year, which is considered an internal seasonal source of P. Phosphorus is primarily removed from the water column of Green Lake into the Puchyan River or deposited into the lake sediments.

Atmospheric Sources

Atmospheric input of P to Green Lake was estimated from dryfall deposition and TP concentrations measured in wetfall (rain and snow) at a station installed near Delavan Lake, Wisconsin, 135 km southeast of Green Lake (Robertson and Kennedy, 2021). Phosphorus dryfall deposition rates were measured for 16 periods without precipitation during 2005–9. The dryfall deposition rates had distinct seasonality, with lowest deposition rates during December through February (usually less than 0.0002 kilograms per hectare per day [(kg/ha)/d]) and highest deposition rates in spring and late fall (usually more than 0.0004 [kg/ha]/d) (fig. 14). The individual deposition rates were used to estimate mean monthly dryfall deposition rates and a total annual dryfall deposition rate (0.117 [kg/ha]/yr) (table 6). Total mean annual dryfall deposition of P on Green Lake during WYs 2014–18 was 345 kg/yr (table 7).

Phosphorus concentrations, each adjusted for dryfall during the wetfall-collection period, measured in 40 precipitation (wetfall) events during 2005–9 were averaged by month. Phosphorus concentrations in precipitation were generally lower in winter and spring (about 0.010 mg/L) and higher in early and late summer (about 0.025 mg/L) (fig. 14). A mean annual wetfall concentration (0.0147 mg/L) was computed by weighting the monthly mean TP concentrations by the long-term (1983–2019) mean monthly precipitation near Delavan (measured at Geneva and Clinton, Wisconsin; table 6). Total annual input of P from precipitation (wetfall) on Green Lake was then computed by multiplying the total

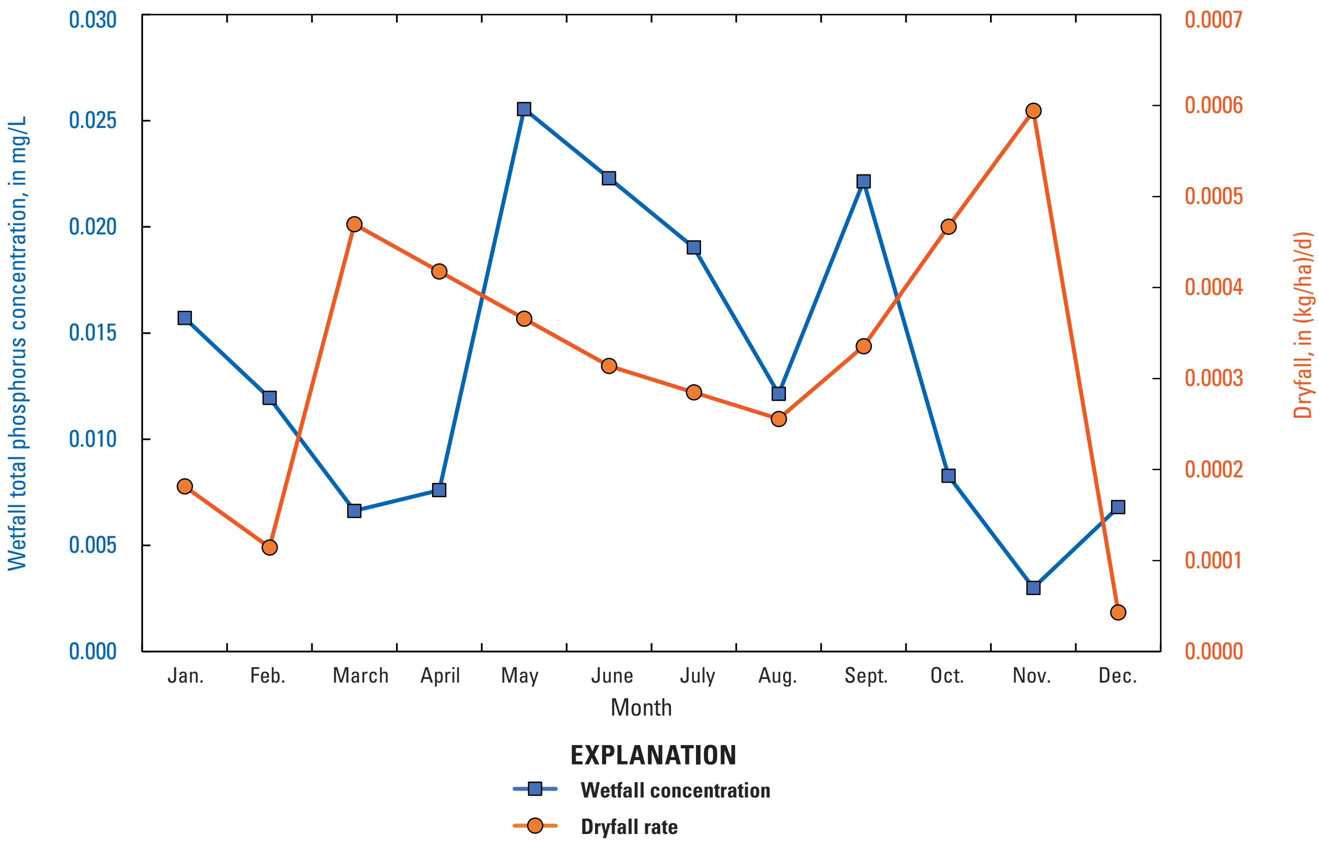


Figure 14. Mean monthly total phosphorus concentrations in wetfall and phosphorus deposition rates in dryfall near Delavan Lake, Wisconsin, based on data collected from 2005 to 2009. Wetfall concentrations were adjusted to remove dryfall deposition during its collection period. (kg/ha)/d, kilogram per hectare per day; mg/L, milligram per liter.

Table 6. Mean monthly wetfall total phosphorus concentrations and total monthly and annual phosphorus deposition rates for southern Wisconsin.

[Data were collected near Delavan Lake, Wisconsin. Wetfall concentrations were adjusted to remove dryfall during its collection period. Monthly precipitation was measured at Geneva and Clinton, Wisconsin. mg/L, milligram per liter; mm, millimeter; (kg/ha)/month, kilogram per hectare per month; (kg/ha)/yr, kilogram per hectare per year]

Month	Mean monthly wetfall concentration (mg/L)	Mean monthly precipitation (mm)	Total monthly dryfall ([kg/ha]/month)
January	0.016	48.0	0.006
February	0.012	46.5	0.003
March	0.007	59.9	0.015
April	0.008	90.8	0.013
May	0.026	101.0	0.011
June	0.022	124.8	0.009
July	0.019	92.2	0.009
August	0.012	99.7	0.008
September	0.022	96.8	0.010
October	0.008	74.0	0.014
November	0.003	68.6	0.018
December	0.007	56.0	0.001
Annual weighted concentration (mg/L) or total (mm or [kg/ha]/yr)	0.0147	958.4	0.117

Table 7. Total annual phosphorus load to Green Lake, Wisconsin, by source, in kilograms, and overall yield, in kilograms per square kilometer, for water years 2014 to 2018.

[Yields represent the load per unit area: for atmospheric inputs, septic systems, and waterfowl, the area is the surface of the lake. WY, water year; Ck., Creek; HWY, Highway; Rd, Road]

Source, area	WY 2014	WY 2015	WY 2016	WY 2017	WY 2018	WYs 2014–18	Overall yield
Atmosphere							
Wetfall	363	338	446	434	463	409	13.9
Dryfall	345	345	345	345	345	345	11.7
Tributaries							
Silver Ck. at HWY A	3,730	1,990	3,010	4,540	4,840	3,620	25.4
Silver Ck. at Spaulding Rd	3,920	2,620	3,850	5,420	6,300	4,350	36.8
Southwest Inlet	1,010	1,050	1,760	3,360	3,680	2,170	55.8
Hill Ck.	773	515	814	1,370	1,480	990	53.8
White Ck.	262	174	276	464	501	335	49.9
Unmonitored area	329	219	347	583	630	422	14.8
All tributary input	6,110	3,950	6,210	10,300	11,100	7,540	32.1
Other							
Septic systems	103	103	103	103	103	103	3.5
Waterfowl	580	580	580	580	580	580	19.7
Total external loading							
All external sources	7,500	5,310	7,680	11,800	12,600	8,980	38.2
All controllable sources	6,200	4,050	6,310	10,400	11,200	7,640	32.5

annual precipitation on the lake by 0.0147 mg/L and the area of the lake. The mean annual input of P from precipitation during WYs 2014–18 was 409 kg/yr (table 7). Therefore, the mean annual atmospheric input of P during WYs 2014–18 was 754 kg/yr (0.256 [kg/ha]/yr).

Surface-Water Inflow

Five sites were monitored to estimate flow and P loading from the Green Lake watershed: Silver Creek at HWY A and at Spaulding Road, Southwest Inlet, White Creek, and Hill Creek (fig. 1; table 2). Total annual P loads at each site were estimated with GCLAS (Koltun and others, 2006). The annual loads were then divided by their total annual flows to estimate the mean annual volumetrically weighted TP concentration at each site. During WYs 2014–18, volumetrically weighted TP concentrations ranged from 0.090 to 0.117 mg/L in Silver Creek at HWY A and Spaulding Rd, respectively, to 0.143 and 0.159 mg/L in White Creek and the Southwest Inlet, respectively, to 0.287 mg/L in Hill Creek (table 2). The WDNR has set a TP concentration impairment criterion for wadeable streams in Wisconsin of 0.075 mg/L based on the median concentration between May and October (Wisconsin Department of Natural Resources, 2019). The median of the monthly mean TP concentrations for May–October (most similar value for comparison with the criterion) for these sites ranged from 0.076 mg/L in Silver Creek at HWY A, to 0.118 and 0.133 mg/L in White Creek and Silver Creek at Spaulding Rd, respectively, to 0.212 and 0.565 mg/L in the Southwest Inlet and Hill Creek, respectively (table 2). All of the median May–October concentrations, except possibly that of Silver Creek at HWY A, were above the 0.075 mg/L TP criterion.

During WYs 2014–18, mean annual P delivered from Silver Creek at HWY A was 3,620 kg/yr, resulting in a mean yield of 25.4 kg/km^2/yr from its watershed, and mean annual P delivered from Silver Creek at Spaulding Road, upstream of HWY A, was 4,350 kg/yr (table 7). Therefore, over 725 kg/yr of P was deposited in the HWY A marsh. Losses in the marsh, however, were variable among years ranging from 194 kg/yr in 2014 to 1,460 kg/yr in 2018, and in 2019 this area was a source of P. During 2014–18, the amount P deposited in the marsh was actually much more than 725 kg/yr because P from Dakin Creek (fig. 1) enters the marsh prior to Silver Creek reaching HWY A.

During WYs 2014–18, mean annual P delivered from Southwest Inlet was 2,170 kg/yr, resulting in a mean yield of 55.8 kg/km^2/yr from its watershed (table 7). Phosphorus loads in Hill Creek and White Creek were only measured in WYs 2018 and 2019; therefore, their loads in WYs 2014–17 were estimated from the measured annual P loads in 2018 at these sites (1,480 kg and 501 kg, respectively) and how P loadings from Silver Creek at HWY A and the Southwest Inlet changed from those measured in WY 2018. The mean annual P load estimated for WYs 2014–18 from Hill Creek was 990 kg/yr, resulting in a mean yield of 53.8 kg/km^2/yr. The mean annual P load from White Creek was 335 kg/yr, resulting in a mean yield of 49.9 kg/km^2/yr from its watershed.

Loading results from a SWAT model developed for the Green Lake watershed (Baumgart, 2015) were used to estimate loadings from the remaining unmonitored areas around Green Lake by assuming that output from the SWAT model represented the P loadings in 2018 (630 kg/yr). It was assumed that the annual changes in P loading from the unmonitored areas were similar to the changes measured at Silver Creek at HWY A and the Southwest Inlet; this approach was similar to that used to estimate P loads in Hill and White Creeks in unmonitored years. The resulting mean annual P load from the unmonitored areas during WYs 2014–18 was 422 kg/yr, resulting in a mean yield of 14.8 kg/km^2/yr (table 7).

Streamflow and annual P loads from Silver Creek at HWY A during WYs 1988–2019 were further examined to determine if there have been trends in flow or annual P loads from this part of the watershed since monitoring began at this site (fig. 15). From WY 1988 to WY 2019, there was a significant increase (at $p<0.05$ using a regression *t*-test) in mean annual flows (fig. 15*A*); however, most of this increase has been since 2008. From WY1988 to WY 2019, P loads in Silver Creek appeared to decline (fig. 15*B*); however, this trend was not statistically significant ($p>0.10$) partially because of large variability in loads and partially because of the increase in streamflow. Although there was no significant trend in P loads, there was a statistically significant decrease ($p<0.001$ using a regression *t*-test) in volumetrically weighted TP concentrations, which appeared to exponentially decrease from about 0.25 mg/L to about 0.10 mg/L (fig. 15*C*, dark blue line). The decrease in volumetrically weighted TP concentrations in Silver Creek was partly caused by a statistically significant decrease ($p<0.001$ using a regression *t*-test) in point-source contributions from the Ripon wastewater treatment plant upstream in the basin. Effluent data from the Ripon wastewater treatment plant (discharge and TP concentrations) from 1988 to 2019 were obtained from the WDNR System for Wastewater Applications, Monitoring, and Permits database (E. Lorenzen, Wisconsin Department of Natural Resources, written commun., 2020[2]). To compute effluent P loads, the mean monthly wastewater treatment plant discharges and mean monthly TP concentrations were first computed from the available data and then multiplied together to obtain monthly P loads, which were then summed to obtain annual effluent loads (fig. 15*B*). Phosphorus loads from the plant decreased from about 1,300 kg/yr in 1993 to about 300 kg/yr in 2019. To explain how the decrease in point-source inputs into Silver Creek could have affected the TP concentrations, the effluent loads were subtracted from the TP loads measured in Silver Creek, which assumed 100-percent transport downstream, and the remaining loads were used to compute adjusted volumetrically weighted TP concentrations (fig. 15*C*). There was a statistically significant ($p<0.01$ using a regression *t*-test) decrease in the adjusted volumetrically weighted TP concentrations. Based on the smoothed changes in volumetrically weighted TP concentrations (fig. 15*C*, gray line), improvements to the Ripon wastewater treatment plant may explain about 40 percent of the reduction in TP concentrations at HWY A (a decrease from about 0.25 mg/L to about 0.09 mg/L), which is consistent with that found by Fuller (2019). The remaining changes in volumetrically weighted TP concentrations may have been caused by changes in the marsh upstream from HWY A, which became much more vegetated and trapped more sediment and phosphorus (Fuller, 2019), and by changes in land-use practices upstream from HWY A that were made to reduce sediment and P losses from the watershed. P inputs from the effluent from the Ripon wastewater treatment plant for the years from 1976 to 1993 are expected to have been relatively similar to those in 1993; however, prior to 1976, the P loading from the plant may have been much higher because upgrades were made to the plant in 1976. Stauffer (1985a) estimated that P export from the plant prior to 1976 was about 14,000 kg/yr, compared to about 300 kg/yr in 2019.

In addition to the measurement of TP concentrations in Silver Creek at HWY A and the Southwest Inlet, about 20 percent of the samples collected at these sites were analyzed for dissolved P. From data collected after 2012, dissolved P represented about 49 percent and 29 percent of the TP at HWY A and the Southwest Inlet, respectively. During 2017–19, a subset of the samples at these sites was also analyzed for concentrations of nitrate plus nitrite, ammonia, organic nitrogen, Kjeldahl nitrogen, and dissolved and total organic carbon. From these data (augmented with data collected for these constituents in prior years mainly at HWY A), monthly mean concentrations for each of these constituents were computed (table 8).

For modeling with GLM–AED, daily time series of concentrations of various constituents were needed for all inflows (Silver Creek at HWY A, Southwest Inlet, and all remaining inflows). Daily TP concentrations in Silver Creek and the Southwest Inlet were estimated by dividing their daily TP loads computed from GCLAS by their corresponding daily flows. The various forms of P for these two sites were then estimated from daily TP concentrations and the mean fraction of P in dissolved forms. All of the dissolved P was assumed to be DRP: 49 percent and 29 percent of the TP at HWY A and the Southwest Inlet, respectively. Dissolved organic P was assumed to be 10 percent of the TP, adsorbed P was assumed to be constant at 0.010 mg/L, and particulate organic P was assumed to be the remaining P (about 40 percent and 60 percent for HWY A and the Southwest Inlet, respectively). Daily concentrations of nitrate (assumed to be equal to nitrite plus nitrate concentrations), ammonia, refractory dissolved organic nitrogen (assumed to be represented by Kjeldahl nitrogen), and dissolved organic carbon during each month were

[2]Data have limited availability owing to restrictions of the Wisconsin Department of Natural Resources. Contact the Wisconsin Department of Natural Resources Wastewater Program for more information.

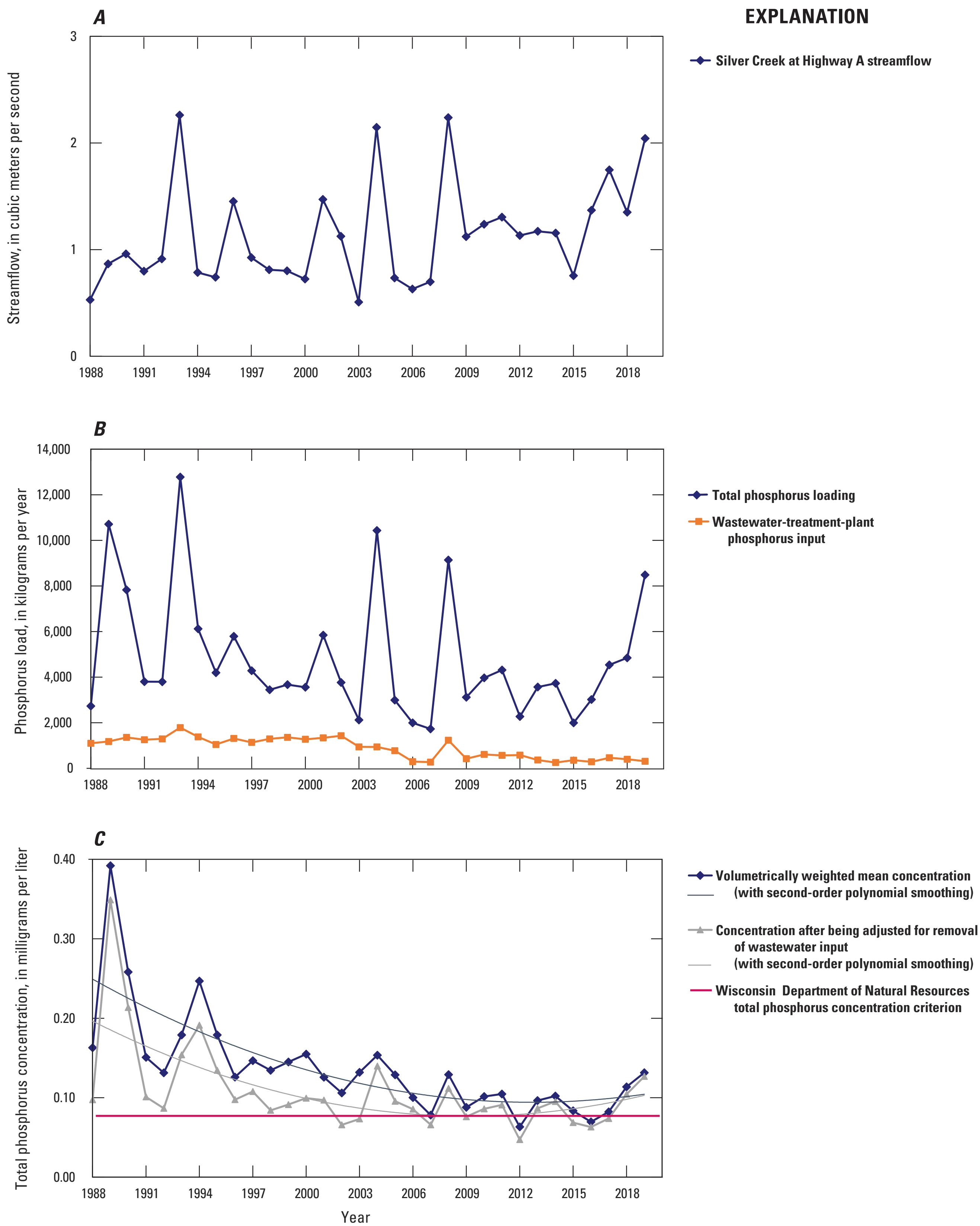


Figure 15. *A*, Mean annual streamflow, *B*, total phosphorus loads (total loading and that from the Ripon wastewater treatment plant), and *C*, volumetrically weighted total phosphorus concentrations in Silver Creek at Highway A from water year 1988 to water year 2019. Smoothed changes in volumetrically weighted total phosphorus concentrations are shown in *C*.

Table 8. Monthly mean concentrations, in milligrams per liter, measured in Silver Creek at Highway A and the Southwest Inlet at Highway K.

[HWY A, Silver Creek at Highway A; SW Inlet, Southwest Inlet at Highway K; NC, not collected]

Month	Total phosphorus		Dissolved phosphorus		Total nitrogen		Nitrite plus nitrate		Organic nitrogen		Kjeldahl nitrogen		Ammonia		Total organic carbon	
	HWY A	SW Inlet	HWY A	SW Inlet	HWY A	SW Inlet	HWY A	SW Inlet	HWY A	SW Inlet	HWY A	SW Inlet	HWY A	SW Inlet	HWY A	SW Inlet
January	0.086	0.069	0.031	0.025	3.585	1.790	2.860	1.145	0.640	0.470	0.725	0.645	0.071	0.104	[1]8.0	[1]6.0
February	0.133	0.191	0.032	0.070	4.205	2.895	3.580	1.750	0.500	1.100	0.625	1.145	0.250	0.400	[1]8.0	[1]6.0
March	0.167	0.173	0.121	0.187	2.818	4.292	0.994	1.942	1.800	1.200	1.823	2.350	0.700	0.459	[1]8.0	[1]6.0
April	0.136	0.165	0.047	0.039	2.368	2.945	1.544	1.336	0.824	1.323	0.824	1.609	0.025	0.100	[1]8.0	[1]6.0
May	0.171	0.258	0.039	0.026	1.724	3.028	0.751	0.908	0.760	2.050	0.973	2.120	0.037	0.022	9.2	6.4
June	0.195	0.328	0.083	0.030	2.717	3.272	1.197	0.083	1.225	1.205	1.520	3.189	0.391	0.068	9.2	7.5
July	0.203	0.222	0.066	0.025	1.738	1.514	0.683	0.018	0.962	1.667	1.054	1.496	0.102	0.015	[1]9.4	6.4
August	0.194	0.238	0.050	0.075	1.688	2.577	0.592	0.027	0.888	2.675	1.096	2.550	0.071	0.182	[1]7.1	8.4
September	0.149	0.140	0.061	0.021	1.963	3.702	0.695	2.476	1.258	0.985	1.268	1.226	0.129	0.150	[1]6.5	[1]6.8
October	0.097	0.111	0.090	0.017	3.255	3.050	2.250	1.350	0.980	1.700	1.005	1.700	0.070	0.122	[1]6.5	[1]6.0
November	0.102	0.092	0.011	NC	2.000	2.000	1.000	1.000	1.000	1.000	1.000	1.000	0.070	0.110	[1]7.0	[1]6.0
December	0.056	0.084	0.009	NC	2.000	2.000	1.000	1.000	1.000	1.000	1.000	1.000	0.070	0.110	[1]7.0	[1]6.0

[1]Extrapolated from measured data.

assumed to be equal to their mean monthly concentrations measured at these two sites. Daily concentrations of dissolved inorganic carbon were assumed to be constant at 60 mg/L, and pH was assumed to be constant at 7.7. Silica concentrations were assumed to be constant at 0.94 mg/L, which is similar to the mean concentrations measured in the lake during spring. All constituent concentrations in unmonitored inflows (all inflows except those from Silver Creek and the Southwest Inlet) were assumed to be the same as those in Silver Creek at HWY A.

Nearshore Septic Systems

Phosphorus loading from near-lake septic systems was estimated from the amount of P input into the septic systems that potentially drain to Green Lake by applying a theoretical soil retention value (S_R) (Reckhow and others 1980; Garn and others, 1996). The total amount of P input into the septic systems was computed by multiplying the number of people using the septic systems annually (capita years) by the input of P per capita per year (E_s). Input of P from septic systems (M) was estimated from equation 12:

$$M = E_s * (\text{capita years}) * (1 - S_R). \qquad (12)$$

Garn and others (1996) estimated E_s to be 0.68 kg/yr. Site-specific soil information was used to estimate S_R. An overall S_R coefficient of 0.70 was used: 0.45 for the fraction of P removed by the soil, plus 0.25 for the fraction of P removed by storage of solids in the septic tanks. In 2017, there was estimated to be about 900 full-time and seasonal residences with lake frontage; about 60 percent of those residences were sewered, and about 9 percent of the remaining residences had holding tanks (L. Reas, Green Lake Sanitary District, written commun., 2020). Therefore, there were about 354 properties directly adjacent to the lakes that were not sewered and without holding tanks. It was assumed that seasonal residences in these areas used septic systems 3 months of the year and that full-time and seasonal residences each had three occupants. The number of capita years was then computed by multiplying the number of residences by the number of persons per household by the fraction of the year occupied. In 2017, there were 117 full-time residences and 272 seasonal residences using near-lake septic systems, which equated to a total of 504 capita years. Therefore, the total P load contributed by septic inputs was 103 kg/yr, which was assumed to be constant for all years (table 7).

Waterfowl

Input of P from waterfowl was assumed to be similar to that estimated by Brooks and others (1980). Based on 3 years of data, they estimated that the Canada goose population contributed 580 kg/yr of P (0.45 g per goose per day) to Green Lake. The daily P loading per goose per day estimated by Brooks and others was similar to that found by Manny and others (1994; 0.49 g per goose per day). It was assumed that waterfowl contributed 580 kg/yr of P in all years (table 7). Given that the length of ice cover on Green Lake has decreased since 1980, that populations of geese may have increased in recent years, and that the estimate by Brooks and others (1980) was only for the Canada goose population and do not include that from other waterfowl, this may underestimate P loading from waterfowl in WYs 2014–18.

Internal Sediment Recycling

During spring overturn, TP concentrations throughout the water column in Green Lake were relatively uniform at about 0.041 to 0.045 mg/L, based on data from 2015 to 2019 (U.S. Geological Survey, 2020), resulting in a total mass of P that ranged from about 41,700 kg to 44,500 kg. TP concentrations throughout the entire water column then decreased during June and early July on the basis of detailed data in 2017, which resulted in a decrease of about 10,000 kg of P in the water column. Then, coinciding with anoxia in the deepest areas of the lake, TP concentrations in the deep water increased. TP concentrations in the hypolimnion increased from April to September in 2018 and 2019 from about 0.045 mg/L to 0.056–0.074 mg/L at about 50 m, from about 0.045 mg/L to 0.098–0.106 mg/L at 63 m, from about 0.045 mg/L to 0.123–0.135 mg/L at 66 m, and from about 0.045 mg/L to 0.142–0.155 mg/L at 67 m. Based on these increases in TP concentrations, the mass of P in the hypolimnion below 35–40 m increased by about 7,040 kg/yr from internal sediment recycling. The depth at which the mass of P increased from spring to fall varied slightly among years. It is believed that the increase in P in the hypolimnion was from sediment release and not from organic matter settling and decomposing in the water column because the TP concentrations in June and early July decreased throughout the entire water column prior to the occurrence of anoxia. The high TP concentrations in the lake after fall overturn persisted throughout the water column until spring. If the total mass of P in the lake in September were completely mixed throughout the lake, it would increase the TP concentrations in the shallow regions of the lake by about 0.022 mg/L, or from 0.014–0.017 mg/L to 0.036–0.039 mg/L. This is a rather crude estimate of the amount of P released from internal sediment recycling and does not include the net annual deposition of P to the sediments that occurs during other times of the year. On an annual basis, P deposited in the lake sediments exceeds that released from sediment recycling because the P loads leaving the lake in the Puchyan River (approximately 4,000 kg/yr; Johnson, 2021) were much less than the total P entering the lake.

Summary of Phosphorus Inputs to Green Lake

The mean annual external input of P to Green Lake during WYs 2014–18 was 8,980 kg/yr (table 7; fig. 16). Annual external inputs ranged from 5,310 kg/yr to 12,600 kg/yr, which is a range of about ±40 percent of the 5-year mean. Tributary inputs, including inputs from unmonitored areas, were the dominant source of P into the water column of Green Lake (7,540 kg/yr: 84 percent of total external inputs). About 48 percent of the tributary loading came from Silver Creek, 29 percent came from Southwest Inlet, 13 percent came from Hill Creek, 4 percent came from White Creek, and 6 percent came from other unmonitored areas. The four monitored tributaries represented about 88 percent of the watershed, and the P loading from these tributaries represented over 94 percent of the total estimated tributary loading to the lake. Atmospheric input (combined wetfall and dryfall) contributed 754 kg, about 8 percent of the external P loading. Inputs from waterfowl and septic systems represented about 7 percent and 1 percent of the total external P loading to the lake. In addition to the external inputs, internal sediment recycling may contribute an additional 7,040 kg/yr of P at fall overturn. But on an annual basis, there is a net deposition of P to the sediments.

Comparison with Phosphorus Summaries from Other Studies

To determine how the hydrologic conditions during the WYs 2014–18 base period, which were wetter than normal, may have affected the estimated mean water and P loading to Green Lake, the precipitation on Green Lake and the streamflow and annual P loads from Silver Creek at HWY A during the base period were compared with those measured during the long-term 1990–2019 period and other periods during which P loading to the lake was estimated (table 9). During WYs 2014–18, mean annual precipitation was 0.944 m/yr, whereas the long-term mean precipitation was 0.871 m/yr (the long-term mean was 7.6 percent lower than that during the study, WYs 2014–18). During WYs 2014–18, mean flow in Silver Creek was 1.28 cubic meters per second (m^3/s), whereas the long-term mean flow was 1.17 m^3/s (8.2 percent lower than during WYs 2014–18). Therefore, the TP loading to the lake during the WYs 2014–18 base period can be expected to be about 8 percent higher than during a period with precipitation more typical of the 1990–2019 period. However, the increase in precipitation and runoff during this base period may

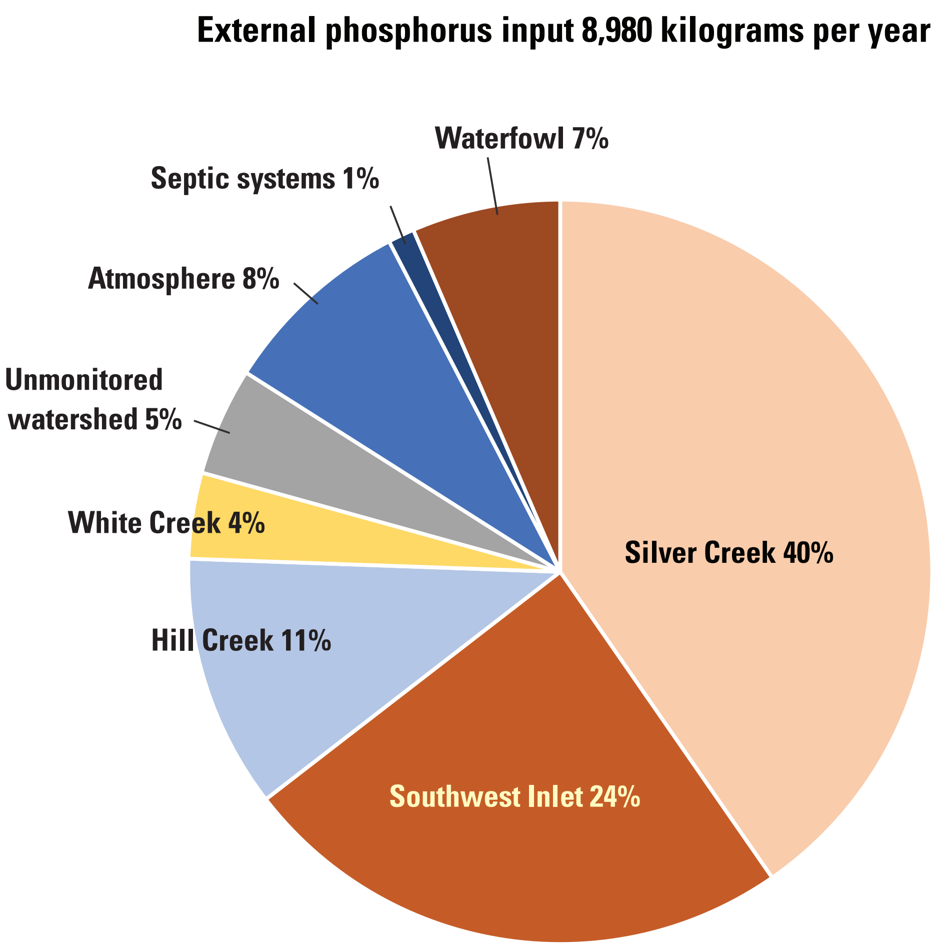


Figure 16. Mean annual input of phosphorus to Green Lake, Wisconsin, by source, for water years 2014–18.

Table 9. Comparison of precipitation, flow measured at Silver Creek, and phosphorus loads measured in Silver Creek from this study (water years 2014–18) with those estimated for 1997–98 (Panuska, 1999), those estimated for 2009–13 (Cadmus Group, 2018), and the long-term mean (1990–2019).

[The phosphorus load reductions needed for summer total phosphorus concentration (TP) in Green Lake, Wisconsin, to decline to 0.015 milligram per liter (in other words, phosphorus loading capacity) are provided. m, meter; m^3/s, cubic meter per second; kg, kilogram]

Period	Precipitation (m)	Precipitation (percent difference from this study)	Silver Creek flow (m^3/s)	Silver Creek flow (percent difference from this study)	Silver Creek phosphorus load (kg)	Silver Creek phosphorus load (percent difference from this study)	Estimated total tributary phosphorus loading (kg)	Total tributary phosphorus loading (percent difference from this study)	Estimated total external loading (kg)	Total external loading (percent difference from this study)	Estimated external phosphorus load needed to keep TP at or below 0.015 mg/L (kg)
1997–98	0.798	−15.5	1.24	−2.8	3,370	−7.0	7,650	1.5	8,440	−6.0	NA
2009–13	0.848	−10.1	1.19	−6.4	3,450	−4.9	5,330	−31.4	5,360	−40.3	4,230
2014–18	0.944	0.0	1.28	0.0	3,620	0.0	7,540	0.0	8,980	0.0	5,460
1990–2019	0.871	−7.6	1.17	−8.2	NA	NA	NA	NA	NA	NA	NA

represent new hydrologic conditions associated with climatic changes expected in the future for the Green Lake area (Katt-Reinders and Pomplun, 2011).

Phosphorus input to Green Lake was measured in several previous studies. However, previous studies that estimated the P input to Green Lake did not estimate inputs from all the sources considered in this study. Therefore, the best way to compare results from this study with those of previous studies was to compare the estimates of total tributary (watershed) and total external loading: total tributary loading in this study was estimated it to be 7,540 kg/yr, and total external loading was estimated to be 8,980 kg/yr for WYs 2014–18 (table 9). Earlier studies estimated total tributary P loading prior to 1976 to be 10,900 kg/yr (Litton and others 1972) and 15,400 kg/yr (Stauffer, 1985a), around 1978 to be 3,290 kg/yr (Stauffer, 1985a), in 1997–98 to be 7,650 kg/yr (Panuska, 1999), and in 2009–13 to be about 5,330 kg/yr (Cadmus Group, 2018). The higher P loading prior to 1976 was primarily caused by the large inputs from the Ripon wastewater treatment plant; however, Stauffer's estimated tributary loadings were low in his study of pre-1976 and 1978 loadings because most of his estimates were extrapolated from tributary data collected during low-flow conditions. Tributary P loading estimated by Panuska for 1997–98 (7,650 kg/yr) was close to that estimated in this study (7,540 kg/yr). The similarity in loading with that estimated in 1997–98 may have been the result of TP concentrations in Silver Creek being higher during 1997–98 (fig. 15) and the precipitation and streamflow during 1997–98 being about 15.5 and 2.8 percent less, respectively, than during 2014–18 (table 9). Panuska (1999) estimated the total annual external P loading to the lake to be 8,440 kg/yr, but the only additional source that was included besides tributary loading was atmospheric input (input of P from waterfowl was not included). Mean annual tributary P loading estimated in this study (7,540 kg/yr) was about 31 percent more than that estimated by the Cadmus Group for WYs 2009–13 (5,330 kg/yr) and used as part of the TMDL study (Cadmus Group, 2018). During WYs 2009–13, precipitation was about 10 percent less than in this study, which resulted in 6.4 percent less runoff and 4.9 percent lower P loading from Silver Creek than in this study (table 9). Therefore, part of the difference in loading may be due to the hydrologic differences among periods. TP concentrations in Silver Creek were similar in WYs 2009–13 and WYs 2014–18 (fig. 15*C*); therefore, differences in TP concentrations would not have caused a difference in loading. Therefore, the difference in tributary P input between this study and that estimated by the Cadmus Group (2018) appears to be a result of the SWAT modeling conducted by the Cadmus Group, who had modeled the entire Upper Fox and Wolf Rivers watershed. Also, the Cadmus Group estimated additional external P inputs only from septic systems, which resulted in a total annual external P loading to the lake of 5,360 kg/yr, which is about 40 percent less than the estimate in this study (8,980 kg/yr).

Response of Near-Surface Water Quality to Changes in Phosphorus Loading

To estimate how near-surface water quality (TP and Chl-*a* concentrations, and SDs) in Green Lake is expected to respond to changes in P loading, 15 scenarios were simulated with the Canfield-Bachmann TP model (Canfield and Bachmann, 1981; eq. 1), empirical relations based on measured water quality in the east and west sides of the lake (eqs. 6–9), Carlson (1977) TSI relations (eqs. 2–4), and the Hickman (1980) Chl-*a* relation (eq. 5). Scenario 1 simulated the mean conditions for WYs 2014–18, and the results were used to determine if and how results from all scenarios could be adjusted to better estimate conditions in Green Lake. Individual year-by-year simulations were not performed. Model results were adjusted (in other words calibrated) on the basis of the percentage of difference between the results found for scenario 1 (without adjustment) and water quality measured in the lake during WYs 2014–18. Eleven simulations (scenarios 2–12) were used to estimate the general response of the water quality of Green Lake to basinwide changes in potentially controllable P sources (all external sources except inputs from precipitation, atmospheric deposition, and waterfowl): decreases in the potentially controllable P sources by 95, 75, 50, 46, 25, and 17 percent and increases in controllable P sources by 25, 50, 75, 100, and 200 percent. In all scenarios, it was assumed that the amount of P from internal sediment recycling changed at a percentage similar to the change in controllable external P sources. This change in internal sediment recycling is not expected to occur immediately, but internal sediment recycling is expected to eventually come to equilibrium with the new external P loading with a change directly proportional to the percentage change in all external sources (Robertson and others, 2018; Robertson and Diebel, 2020). Then, to estimate loading prior to the urban and agricultural changes in the watershed, scenarios 13–14 simulated what reference or background conditions would have been expected in the lake on the basis of export rates found from data on natural land uses (forest or natural prairie) in other studies. Export rates from natural areas were obtained from two studies (5 kg/km^2/yr—Cadmus Group, 2018; and 9 kg/km^2/yr—Singer and Rust, 1975). Finally, scenario 15 was simulated to determine the load reduction needed for near-surface TP concentrations in the east and west sides of the lake to be 0.012 mg/L and the lake to be classified as oligotrophic on the basis of the Carlson (1977) TSI TP relation (eq. 2).

Four types of data were required as input into the empirical models: measured water-quality data (table 3), morphometric data (fig. 3), hydrologic data (table 5), and P-loading data (table 7). Although annual P-loading data are typically used as input into the Canfield-Bachmann model, the model simulates TP concentrations during the summer growing season; therefore, the seasonal water-quality data in table 3 were used to validate and calibrate the models. All input data used in these models and results from the models are available in Robertson and Kennedy (2021).

Verification and Calibration (Adjustments for Model Biases)

To determine how well the Canfield-Bachmann model (eq. 1) simulates changes in TP concentrations in Green Lake, it was used to simulate near-surface TP concentrations by using mean-annual P loading for the WYs 2014–18 base period. Because the model was developed without including internal P sources (Canfield and Bachmann, 1981; Dillon and Rigler, 1974), the model should indirectly incorporate contributions from internal sources (Walker, 1976). Therefore, the model was evaluated with and without seasonal contributions from internal sediment recycling. With P loading that included the seasonal inputs from internal sediment recycling, the simulated TP concentration was 0.025 mg/L, which is higher than the geometric mean concentrations measured from the east side (0.020 mg/L) and west side (0.016 mg/L) of the lake (table 3). This discrepancy may be due to the Canfield-Bachmann model already indirectly including the effects of internal sediment recycling when the model was developed. The model was then used to simulate near-surface TP concentrations by using the total mean annual external P loading, which did not include internal sediment recycling. The simulated TP concentration was 0.0174 mg/L (table 10), which is close to the geometric mean of the concentrations measured from the west and east sides lake (0.019 mg/L) but slightly lower than that measured on the east side of the lake (0.020 mg/L) and slightly higher than that measured on the west side of the lake (0.016 mg/L; table 3). This similarity between model results and measurements in the lake confirms that the model is expected to accurately simulate changes in Green Lake in response to changes in external P loading after minor adjustments are made to the simulated TP concentrations. In all simulations, the east and west sides of the lake were each treated as independent lakes with full watershed loading and full lake morphometry. All simulated TP concentrations were increased by 15.8 percent for estimating changes for the east side of the lake and were decreased by 5.5 percent for estimating changes for the west side of the lake, and each side of the lake was evaluated independently.

The coefficients of determination (r^2) and p values of each of the empirical relations developed on the basis of measured seasonal data from the east and west sides of Green Lake were examined to determine how well the relations in figure 11 are expected to simulate changes in Chl-*a* concentrations and SDs. The r^2 values for predicting Chl-*a* from TP were 0.11 (p<0.10) and 0.42 (p<0.001) for east and west sides, respectively. Based on these values, predictions for the west side may be more reliable than for the east side. Equations 8–9 were then used to estimate Chl-*a* concentrations from the mean TP concentrations from WYs 2014–18. The estimated mean Chl-*a* concentration for WYs 2014–18 for the east side was 5.5 µg/L (compared to the 6.4-µg/L geometric mean measured concentration) and for the west side was 5.0 µg/L (compared to the 4.6-µg/L geometric mean measured concentration); therefore, all predicted values with Green Lake relations were increased by 17.3 percent for the east side and decreased by 9.1 percent for the west side of the lake. The r^2 and p values for predicting SD (fig. 11) from TP concentrations were 0.13 and p<0.05 for the east side and 0.07 and p=0.20 for the west side of the lake. Therefore, predictions for both sides had much uncertainty, especially for the west side. The predicted SD from mean TP concentrations measured in WYs 2014–18 for the east side was 4.0 m (compared to the 4.4-m geometric measured concentration) and for the west side was 4.5 m (compared to 4.9-m geometric measured concentration). Therefore, all predicted SD values using the Green Lake relations were increased by 9.9 percent for the west side and 8.6 percent for the east side of the lake.

To determine how well the Carlson TSI relations simulated Chl-*a* concentrations and SDs in the east and west sides of Green Lake, the TSI values computed on the basis of the measured TP concentrations for WYs 2014–18 were used to compute Chl-*a* concentrations and SDs with similar TSI values. Given a TSI value of 47.5 for the east side of the lake (based on measured TP concentration of 0.020 mg/L), the predicted Chl-*a* concentration was 5.6 µg/L and predicted SD was 2.4 m for this TSI value, whereas measured geometric mean values were 6.4 µg/L and 4.4 m, respectively. Given a TSI value of 44.6 for the west side (based on measured TP concentration of 0.016 mg/L), the predicted Chl-*a* concentration was 4.2 µg/L and predicted SD was 2.9 m, whereas measured geometric mean values were 4.6 µg/L and 4.9 m, respectively, for this TSI value. Therefore, the TSI relations appear to predict Chl-*a* concentrations well but do not predict SDs well. To remove the biases in use of the TSI relations, all Chl-*a* concentrations predicted for the east side of the lake were increased by 12.9 percent and all predicted SDs were increased by 46 percent; all Chl-*a* concentrations predicted for the west side of the lake were increased by 9.0 percent, and all predicted SDs were increased by 41 percent.

To determine how well the Hickman (1980) Chl-*a* relation simulated the mean Chl-*a* concentration in the east side of Green Lake, the measured mean summer TP concentration for WYs 2014–18 was used to estimate the mean summer Chl-*a* concentration. Given a measured TP concentration of 0.020 mg/L, the estimated Chl-*a* concentration was 8.82 µg/L, whereas the measured geometric mean concentration was 6.42 ug/L. Therefore, this relation appeared to overestimate Chl-*a* concentrations. To remove the potential bias in this relation, all Chl-*a* concentrations predicted with this relation were decreased by 22.9 percent.

Table 10. Response of near-surface total phosphorus and chlorophyll-a concentrations and Secchi disk depths in the east and west sides of Green Lake, Wisconsin, in response to various phosphorus-loading scenarios based on the Canfield and Bachmann natural-lake model (Canfield and Bachmann, 1981), Green Lake derived equations, Carlson (1977) trophic-state-index equations, and Hickman (1980) relation.

[Total phosphorus loading does not include inputs from internal sediment recycling. All changes in loading are based on percentage changes in controllable phosphorus (P) loading (all but atmospheric input). Scenarios: export rates from natural areas, 5 kilograms per square kilometer per year (kg/km^2/yr) (Cadmus Group, 2018); and 9 kg/km^2/yr (Singer and Rust, 1975); and oligotrophic conditions with a near-surface total phosphorus concentration of 0.012 milligrams per liter (mg/L). Surface area was 2,948 hectares, volume was 989,090,000 cubic meters, mean depth was 33.55 meters (m), and residence time was 13.0 years. kg, kilogram; mg/L; Chl-a, chlorophyll–a; µg/L, microgram per liter; TSI, trophic state index; SD, Secchi disk depth; %, percent]

Scenarios/ descriptiotn	Sce-nario number	Annual P load from adjustable external sources (kg)	Total annual P load from external sources (kg)	Percent change in total external sources from base loading	Canfield & Bach-mann TP (mg/L)	East side predicted TP (adjusted Canfield & Bachmann; 15.8% adjustment) (mg/L)	West side predicted TP (adjusted Canfield & Bachmann; –5.5% adjustment) (mg/L)	East side predicted Chl–*a* (adjusted Green Lake model; 17.3% adjustment) (µg/L)	East side predicted Chl–*a* (adjusted TSI model; 12.9% adjustment) (µg/L)	East side predicted Chl–*a* (adjusted Hickman; –22.9% adjustment) (µg/L)	West side predicted Chl–*a* (adjusted Green Lake model; –9.1% adjustment) (µg/L)	East side predicted SD (adjusted Green Lake model; 9.9% adjustment) (m)	East side predicted SD (adjusted TSI model; –46.0% adjustment) (m)	West side predicted SD (adjusted Green Lake model; 8.6% adjustment) (m)
General response, by percent change in controllable P sources														
−95%	2	382	1,720	−80.9	0.006	0.007	0.006	3.8	1.4	1.6	0.7	4.9	12.6	5.4
−75%	3	1,830	3,170	−64.7	0.009	0.011	0.009	4.5	2.5	2.9	1.8	4.7	8.4	5.3
−50%	4	3,820	5,160	−42.6	0.012	0.014	0.012	5.2	3.9	4.3	2.8	4.6	6.2	5.1
−46%	5	**4,130**	**5,460**	−39.2	0.013	0.015	0.012	5.3	4.1	4.5	3.0	4.6	6.0	5.1
−25%	6	5,730	7,070	−21.3	0.015	0.017	0.014	5.9	5.2	5.5	3.8	4.5	5.1	5.0
−17%	7	**6,340**	**7,680**	−14.5	0.016	0.018	0.015	6.0	5.6	5.8	4.0	4.5	4.9	5.0
0%−Base	**1**	**7,640**	**8,980**	**0**	**0.0174**	**0.020**	**0.016**	**6.4**	**6.4**	**6.4**	**4.6**	**4.4**	**4.4**	**4.9**
+25%	8	9,560	10,900	21.3	0.020	0.023	0.019	6.9	7.6	7.3	5.3	4.3	3.9	4.8
+50%	9	11,500	12,800	42.6	0.022	0.025	0.020	7.4	8.8	8.0	6.0	4.2	3.6	4.7
+75%	10	13,400	14,700	63.9	0.023	0.027	0.022	7.8	9.9	8.7	6.6	4.2	3.3	4.7
+100%	11	15,300	16,600	85.1	0.025	0.029	0.024	8.3	11.1	9.4	7.2	4.1	3.0	4.6
+200%	12	22,900	24,300	170.3	0.032	0.037	0.030	9.7	15.4	11.6	9.4	3.9	2.4	4.3
Specific scenarios														
Natural (5 kg/km^2/yr)	13	1,180	2,510	−72.0	0.008	0.009	0.007	4.2	2.0	2.3	1.3	4.8	9.8	5.3
Natural (9 kg/km^2/yr)	14	2,120	3,450	−61.6	0.010	0.011	0.009	4.6	2.7	3.1	1.9	4.7	8.0	5.2
Oligotrophic: east side −67% reduction; west side −48% reduction with a total P load of 5,320 kg	15	2,540	3,870	−56.9	0.010	0.012	0.012	4.8	3.0	3.4	2.9	4.7	7.4	5.1

Response of Water Quality to Basinwide Changes in Phosphorus Loading

In this section of the report, the response of near-surface TP and Chl-*a* concentrations and SDs to changes in external P loading from the watershed is described.

Total Phosphorus

On the basis of simulations representing a range of basin-wide changes to potentially controllable external P sources (scenarios 2–12) with the Canfield-Bachmann model, mean June–September TP concentrations on both sides of the lake are expected to be smaller, on a percentage basis, than the changes in external P loadings, and the response to reductions in P loading is expected to be larger than similar increases in loading (table 10; figure 17*A*). Changes to in-lake TP concentrations were about 50–60 percent of the changes to external P loading. For example, a 50-percent decrease to the controllable external P sources, which equates to a 42.6-percent decrease in total external P loading, is expected to result in a 29-percent decrease to in-lake TP concentrations on both sides of the lake, whereas a 50-percent increase to controllable P sources is expected to result in a 24-percent increase to in-lake TP concentrations. Based on these response curves, external P loading to the lake would need to be decreased from 8,980 kg/yr to 5,460 kg/yr for the mean summer TP concentration in the east side of the lake to decrease to 0.015 mg/L during June–September (WDNR TP criterion for the lake), which is a 46-percent reduction from the controllable external P sources and a 39.2-percent reduction in all external loading from that measured in WYs 2014–18 (scenario 5). However, the external P loading would only need to be decreased from 8,980 kg/yr to 7,680 kg/yr for the mean summer TP concentration in the west side of the lake to decrease to 0.015 mg/L, which is a 14.5-percent reduction from the controllable P sources measured in WYs 2014–18 (scenario 7).

In developing the response curves in figure 17, it was assumed that the amount of P from internal sediment recycling changed with the change in external P loading. This change in internal sediment recycling is not expected to occur immediately, but internal sediment recycling is expected to eventually come to equilibrium with the new external P loading with a change directly proportional to the percentage change in all external sources (Robertson and others, 2018; Robertson and Diebel, 2020). One way to determine how near-surface TP concentrations may change shortly after external P reductions would be to determine the percentage change in overall P loading (external P plus P from internal sediment recycling) from the 2014–18 base period rather than the change in absolute loading in kilograms, and then determine the resulting TP concentration from figure 17 and table 10 on the basis of the overall percentage change in external P loading and seasonal P contributions from internal sediment recycling.

Chlorophyll-*a*

Three different approaches were used to determine how near-surface mean July–September Chl-*a* concentrations in the east side of Green Lake are expected to respond to changes in P loading and the resulting changes to in-lake TP concentrations estimated with the Canfield-Bachmann model. The relation based on Green Lake data was the only approach used for the west side of the lake. Based on the relations developed from measured Green Lake data, Chl-*a* concentrations are expected to have a relatively linear response to changes in controllable external P sources (fig. 17*B*); however, the changes, on a percentage basis, are expected to be larger for the west side of the lake than the east side, and the changes for both sides of the lake are expected to be smaller than the changes in controllable P sources. For example, a 50-percent change in the controllable external P sources, which equates to 42.6-percent in total external P loading, is expected to cause 18-percent and 38-percent changes in Chl-*a* concentrations in the east and west sides of the lake, respectively. Based on these response curves, the 46-percent reduction in external P sources that is expected to be needed for TP concentration in the east side of the lake to meet the 0.015-mg/L criterion is expected to cause mean near-surface July–September Chl-*a* concentrations to decrease from 6.4 µg/L to 5.3 µg/L in the east side and from 4.6 µg/L to 3.0 µg/L in the west side of the lake.

The response of Chl-*a* concentrations in the east side of the lake was also estimated with the Carlson (1977) TSI relation. Chl-*a* concentrations again had a linear response to changes in controllable external P loading, similar to the percentage change in total external loading. This response of Chl-*a* concentrations was larger than that based on the relation derived from Green Lake data (fig. 17*B*; table 10). The 46-percent reduction in controllable sources is expected to cause the mean July–September Chl-*a* concentrations in the east side of the lake to decrease from 6.4 µg/L to 4.1 µg/L.

The response of near-surface mean July–September Chl-*a* concentrations in the east side of the lake was also estimated with the Hickman Chl-*a* relation (eq. 5). Mean Chl-*a* concentrations, on a percentage basis, had a larger response to decreases in P loading than to increases in loading, and the response was between those estimated with the relation based on Green Lake data and estimated TSI values (fig. 17*B*; table 10). A 50-percent decrease in controllable external P sources is expected to cause a 33-percent decrease in the mean Chl-*a* concentration, whereas a 50-percent increase in P loading is expected to cause a 25-percent increase in concentration. Based on the Hickman relation, a 46-percent reduction in external P loading is expected to cause the mean July–September Chl-*a* concentration in the east side of the lake to decrease from 6.4 µg/L to 4.5 µg/L. Therefore, based on the three different approaches, the 46-percent reduction in the controllable external P sources (reduction in total external loading from 8,990 kg/yr to 5,480 kg/yr) is expected to cause mean summer Chl-*a* concentrations on the east side of the lake to decrease from 6.4 µg/L to 4.1–5.3 µg/L.

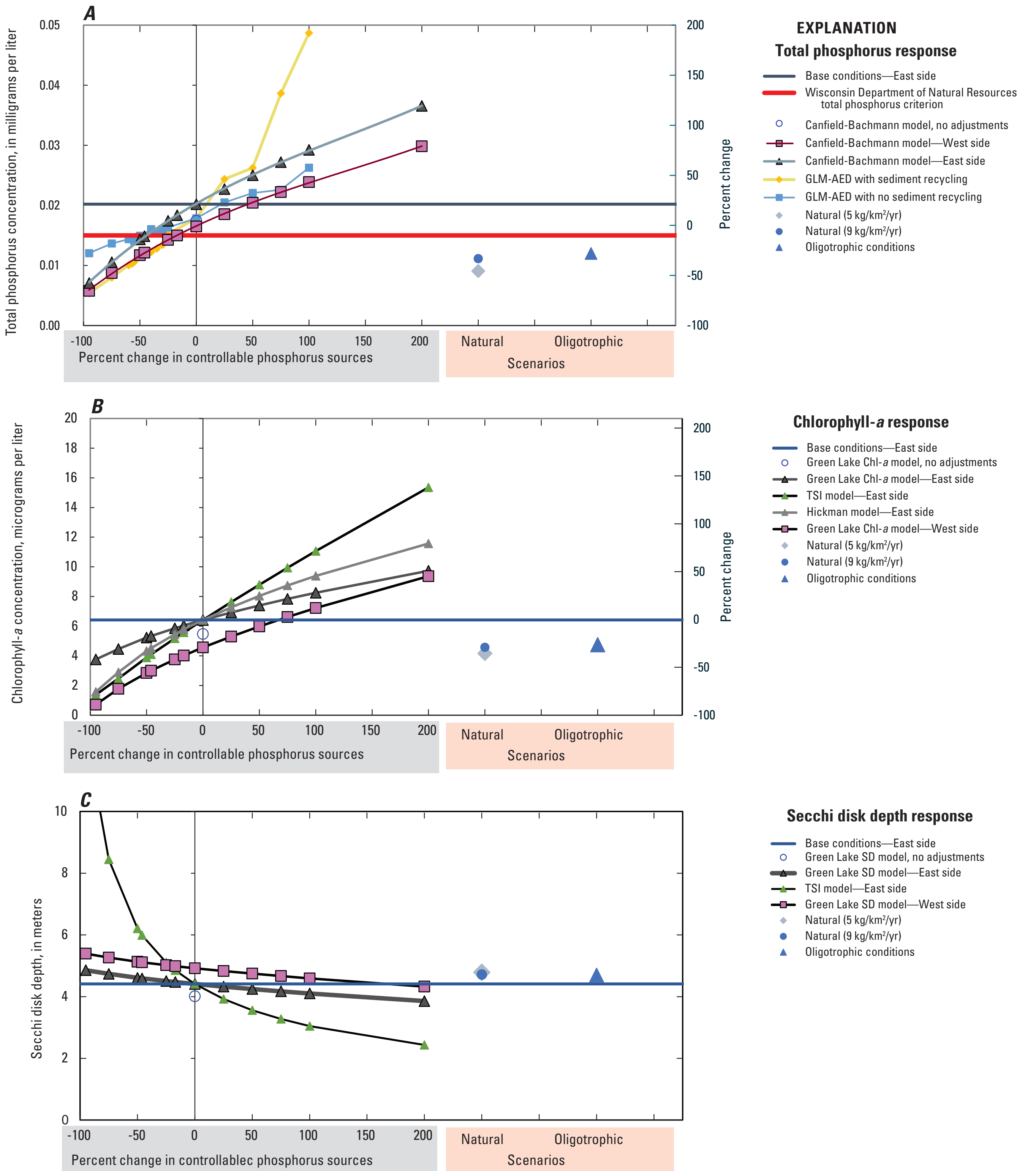


Figure 17. Changes in mean summer water quality in Green Lake, Wisconsin, in response to various phosphorus-loading scenarios simulated by using the Canfield-Bachmann natural-lake model (Canfield and Bachmann, 1981), empirical relations based on measured water quality from the lake, Carlson (1977) trophic state index (TSI) relations, the Hickman (1980) chlorophyll-*a* (Chl-*a*) relation, and the General Lake Model coupled to the Aquatic Ecodynamics (GLM–AED) modeling library for *A*, total phosphorus, June 1–September 15, *B*, chlorophyll-*a*, July 15–September 15, and *C*, Secchi disk depth (SD), July 15–September 15. Scenarios include export rates from natural areas of 5 kilograms per square kilometer per year (kg/km²/yr) (Cadmus Group, 2018) and 9 kg/km²/yr (Singer and Rust, 1975), and oligotrophic conditions with a near-surface total phosphorus concentration of 0.012 milligrams per liter.

Secchi Disk Depth

Two approaches were used to estimate how the mean July–September SD in the east side of Green Lake is expected to respond to changes in P loading and the resulting changes to in-lake TP concentrations estimated with the Canfield-Bachmann model. The relation derived from Green Lake data was the only approach used for the west side of the lake. Based on the relations developed from measured Green Lake data, SDs are expected to have a linear response to changes in the controllable external P sources (fig. 17*C*), and the changes, on a percentage basis, are expected to be smaller than the changes in controllable P sources. For example, a 50-percent change in the controllable external P sources is expected to cause only a 4–5-percent change in SDs. Based on these response curves, the 46-percent reduction in the controllable P sources is expected to cause the mean July–September SD to increase from 4.4 m to 4.6 m on the east side of the lake and from 4.9 m to 5.1 m on the west side of the lake.

The response of SDs on the east side of the lake to changes in P loading was also estimated with the Carlson (1977) TSI relation. Based on the TSI relation, SDs are expected to have a nonlinear response to changes in controllable external P sources (fig. 17*C*; table 10). For example, a 50-percent decrease in the controllable external P sources is expected to cause a 41-percent increase in SDs, whereas a 50-percent increase in the controllable sources is expected to cause a 19-percent decrease is SDs. The 46-percent reduction in controllable P sources is expected to cause the mean July–September SDs in the east side of the lake to increase from 4.4 m to 6.0 m. Therefore, based the two approaches, the 46-percent reduction in controllable external P sources is expected to result in mean SDs on the east side of the lake to increase from 4.4 m to 4.6–6.0 m.

Estimated Natural Water Quality in Green Lake

Scenarios 13–14 simulated what reference or background conditions may have been in Green Lake prior to agricultural and urban activities in the watershed. Based on two different export rates from natural areas, 5 $kg/km^2/yr$ (Cadmus Group, 2018) and 9 $kg/km^2/yr$ (Singer and Rust, 1975), natural external TP loading to Green Lake may have been 2,510 kg/yr or 3,450 kg/yr (table 10). With these external P loadings, mean June–September near-surface TP concentrations are expected to have been between 0.007 and 0.011 mg/L, and mean July–September Chl-*a* concentrations are expected to have been between 2.0 and 4.6 µg/L on the east side of the lake and between 1.3 and 1.9 µg/L on the west side. With these loadings, SDs are expected to be greater than during WYs 2014–18, but there was a large range in what they may have been expected to be (4.8–9.8 m).

Load Reductions Needed for Green Lake to be Classified as Oligotrophic

Scenario 15 estimated that a reduction in the controllable external P sources of 67 percent, which equates to total external P loading of 3,870 kg/yr, is expected to be needed for the east side of the lake to be classified as oligotrophic on the basis of the Carlson TSI relation for TP (TP concentration of 0.012 mg/L; table 10). A 48-percent reduction in the controllable P sources, which equates to total external P loading of 5,320 kg/yr, is expected to be needed for the west side to be classified as oligotrophic on the basis of mean summer TP concentrations. A 67-percent reduction in the controllable P sources is expected to result in a near-surface mean July–September Chl-*a* concentration of about 2.9–4.8 µg/L and a mean SD of 4.7–7.4 m (table 10).

Comparison with Results of Other Studies

Based on the results of this study, external P loading to the lake would need to be decreased from 8,980 kg/yr measured in WYs 2014–18 to 5,460 kg/yr for the geometric mean June–September near-surface TP concentration from the east side of the lake to decrease to the 0.015 mg/L TP criterion for the lake. This is a 3,520 kg/yr reduction in external P loading from that measured in WYs 2014–18, which equates to a 46-percent reduction in the controllable P sources and about a 39-percent reduction in total external P loading. It is assumed that there will be a similar percentage reduction in P from internal sediment recycling following the reduction in external loading as the lake comes to a new equilibrium. The only previous study that estimated the P-load reduction needed to decrease the June–September near-surface TP concentrations in Green Lake to 0.015 mg/L was that by the Cadmus Group (2018), which was done as part the of Upper Fox and Wolf Rivers TMDL study (Wisconsin Department of Natural Resources, 2020a). Panuska (1999) also developed TP response curves, but only for the TP concentration during spring turnover, not during June–September. The Cadmus Group estimated that external P loading to the lake needed to be reduced from 5,360 kg/yr to 4,230 kg/yr (a 1,130-kg/yr reduction, or 21 percent, from that measured in 2009–13) for the mean June–September near-surface TP concentration to decrease to 0.015 mg/L.

There are several reasons for the differences in the estimated P-load reductions needed for Green Lake to meet the 0.015-mg/L criterion. First, our study was based on determining the P-load reduction needed for the mean TP concentration from east side of the lake (which was 0.020 mg/L as measured during WYs 2014–18) to reach 0.015 mg/L, rather than for the lakewide (average of east and west sides of the lake) mean TP concentration (which was 0.017 mg/L as measured during 2009–13) to reach 0.015 mg/L. The Wisconsin Department of Natural Resources (2019) has stated that when there are multiple deep monitoring locations in a lake, the site with a higher

TP concentration should be used for assessment purposes. Therefore, load reductions for Green Lake should be based on the P loading needed for the higher TP concentrations in the east side of the lake to meet the criterion. Second, external P loading in our study was based on a more complete inventory of P inputs, which included atmospheric and waterfowl inputs, than included by the Cadmus Group (2018). Third, P loading in our study was based on actual monitored tributary loading that represented about 88 percent of the watershed and over 90 percent of the total tributary P loading to the lake (table 7; fig. 16), whereas the Cadmus Group simulated P loading from the watershed with a SWAT model developed for the entire Upper Fox and Wolf Rivers watershed and that appeared to underestimate the watershed loading to Green Lake. Finally, the Cadmus Group estimated P loading during years with less rainfall and less runoff than during this study (fig. 12; table 9). During our study, precipitation and runoff were about 8 percent higher than the long-term 1990–2019 mean, whereas during 2009–13 precipitation and runoff were near the long-term mean. The difference in hydrology between study periods may explain about 6–10 percent of the difference in watershed loading, but the differences in hydrology do not explain the 31.4-percent difference in tributary loading and the 40.4-percent difference in total external loading used to calibrate the models and develop response curves (fig. 17). The increase in precipitation and runoff during our study may represent new hydrologic conditions associated with climatic changes expected for the Green Lake area (Katt-Reinders and Pomplun, 2011).

Empirical Evidence of Factors Affecting Metalimnetic Dissolved Oxygen Minima and Near-Surface Water Quality

To provide empirical evidence of which meteorological, hydrologic, water-quality, and in-lake physical factors were correlated with the interannual variability in metalimnetic DO concentrations in August and which meteorological and hydrological factors were correlated with near-surface summer water quality, Pearson correlations were examined for data between 1988 and 2019, which included 24 years with measured August MOMs and 32 years with near-surface water-quality data. Parameters with an absolute value of the correlation coefficient greater than 0.2 were included in table 11, as were parameters that were originally considered important, such as total annual P loading, total annual precipitation, and the date of ice out. The only statistically significant relations at $p<0.05$ were between MOMs with relatively low DO concentrations and relatively high summer Chl-*a* concentrations (July–August Pearson correlation coefficient $r = -0.42$, and July–September $r = -0.40$) and relatively high metalimnetic water temperatures at 20-m depth ($r = -0.41$; at $p<0.10$); relatively low DO concentrations in the metalimnion were also significantly correlated with relatively high spring and summer precipitation (February–August) and deep June SDs. Variability in the water temperatures at 20 m is believed to reflect variability in the metalimnetic temperatures that occurred at the onset of stratification and persisted throughout summer. Variability in MOM intensity was not significantly correlated with variability in P loading for that year or the timing of ice out. Based on these correlations, DO concentrations in the metalimnion were low during years with poor summer water quality (high Chl-*a* and TP concentrations and low mid- to late summer SDs), which may partially be a result of more spring and summer precipitation. Metalimnetic DO concentrations were especially low during years with warm metalimnetic water temperatures, which may have been the result of strong and variable winds (higher standard deviation in daily wind velocities than in other years). Warmer metalimnetic water temperatures may have increased the rate of community metabolism and respiration, resulting in lower DO concentrations. To demonstrate possible interactions between the variables that were significantly correlated with MOM intensity and the other variables, their Pearson correlation coefficient values are also presented in table 11.

The intensity of MOMs had an inverse relation with years, which suggests that there was a tendency toward low DO concentrations in the metalimnion from 1988 to 2019, although this was not a statistically significant relation ($r = -0.28$). The intensity of MOMs was significantly correlated with the temperature of the metalimnion when stratification was established (water temperature at 20-m depth) but had little relation with any other physical feature, such as water temperature at the depth of the MOM, depth of the MOM, or the depth of the thermocline. From 1988 to 2019, there were statistically significant trends ($p<0.05$) toward more precipitation, higher summer wind velocities, more variable summer winds, clearer water quality, shallower depths of the August thermocline and MOM, and warmer temperatures at the depth of the MOM.

In general, years with relatively poor near-surface mean summer water quality (relatively high TP and Chl-*a* concentrations and relatively shallow mid- to late summer SDs) had relatively high P loading, especially in years with high P loading during May through August (significant at $p<0.05$ for TP and Chl-*a* concentrations and $p<0.10$ for SD). Chl-*a* concentrations were also relatively high in years with more rainfall ($p<0.05$) and stronger winds during summer ($p<0.10$). In general, years with relatively high TP concentrations had better water clarity in June and worse clarity and higher Chl-*a* concentrations during midsummer to late summer than did other years. From 1988 to 2019, there was a significant increase in summer Chl-*a* concentrations at $p<0.05$, significant increase in summer SDs at $p<0.10$, and nonsignificant increase in summer TP concentrations.

Table 11. Pearson correlations between the intensity of metalimnetic dissolved oxygen minima in August and various meteorological, hydrological, water quality, and physical characteristics for Green Lake, Wisconsin, during 1988–2019 (24 years), and Pearson correlations for June–September total phosphorous concentrations and July–September Secchi disk depth and chlorophyll-a concentrations and various meteorological, hydrological, water quality, and physical characteristics, during 1988–2019 (32 years).

[For June–September total phosphorus (TP) concentrations and July–September Secchi disk depth and chlorophyll-a (Chl-a) concentrations, correlation values less than −0.34 or greater than +0.34 are statistically significant at p<0.05, and values less than −0.28 or greater than +0.28 are significant at p<0.10). For all other correlations, values less than −0.39 or greater than +0.39 are statistically significant at p<0.05, and values less than −0.33 or greater than +0.33 are significant at p<0.10). MOM, metalimnetic dissolved oxygen minimum concentration; P, Phosphorus; NC, not computed]

Factor	MOM	Year	Precipitation (water year)	Precipitation (Feb.–Aug.)	Wind velocity (June)	Wind velocity (June–Aug.)	Secchi disk depth (June)	Chl-*a* (June–Aug.)	Chl-*a* (July–Aug.)	TP (June–Sept.)	Secchi disk depth (July–Sept.)	Chl-*a* (July–Sept.)
MOM												
MOM	1.00	−0.28	−0.19	[2]−0.33	−0.30	−0.30	[2]−0.34	[2]−0.38	[1]−0.42	NC	NC	NC
Trend												
Year	−0.28	1.00	[1]0.40	[2]0.35	0.22	[1]0.40	[1]0.51	[2]0.34	[1]0.40	0.20	[2]0.31	[1]0.39
Meteorological												
Air temperature (May–June)	−0.12	0.29	−0.09	−0.02	0.22	0.36	0.06	−0.09	−0.02	−0.23	0.19	−0.02
Precipitation (Water year)	−0.19	[1]0.40	1.00	[1]0.82	0.24	0.04	0.16	[1]0.55	[1]0.51	0.09	−0.01	[1]0.53
Precipitation (Feb.–Aug.)	[2]−0.33	[2]0.35	[1]0.82	1.00	0.22	0.00	0.21	[2]0.33	[2]0.35	0.18	0.00	[1]0.38
Precipitation (June–Aug.)	−0.29	0.10	[1]0.72	[1]0.85	0.18	−0.06	0.18	[2]0.34	0.30	0.12	−0.10	[1]0.35
Wind velocity (June)	−0.30	0.22	0.24	0.22	1.00	[1]0.64	[2]0.34	0.30	0.29	0.15	−0.01	[2]0.32
Wind velocity (June–Aug.)	−0.30	[1]0.40	0.04	0.00	[1]0.64	1.00	[2]0.33	0.22	0.27	−0.05	0.04	[2]0.32
Wind variability (June)	−0.21	[2]0.36	[1]0.39	[1]0.50	0.06	0.22	0.17	0.08	0.11	0.05	−0.09	0.22
Wind variability (Aug.)	−0.28	[1]0.44	0.07	−0.06	0.00	[1]0.40	[2]0.33	0.03	0.00	0.07	0.04	0.04
Ice-out date	0.09	0.05	0.14	0.00	−0.11	0.13	0.09	0.17	0.13	−0.06	0.22	0.20
Hydrological												
P load (previous year)	0.21	−0.30	−0.25	−0.14	0.00	0.04	−0.06	[2]−0.36	[1]−0.40	−0.26	0.22	[1]−0.40
P load (Jan.–Apr.)	0.21	−0.26	0.20	0.12	−0.03	−0.31	−0.15	0.21	0.00	−0.19	−0.01	−0.01
P load (May-Aug.)	−0.20	−0.19	[1]0.40	[1]0.48	0.18	−0.15	0.08	0.32	0.30	[1]0.30	[2]−0.30	[1]0.38
P load (Water year)	−0.04	−0.26	[1]0.42	[1]0.40	0.09	−0.27	−0.02	[2]0.33	0.21	0.12	−0.20	0.26
Water quality												
TP (June–Sept.)	−0.25	0.20	0.09	0.18	0.15	−0.05	[1]0.46	0.17	0.31	1.00	[1]−0.36	[2]0.33
TP (June–Aug.)	−0.20	0.20	0.06	0.14	0.16	−0.05	[1]0.45	0.16	0.29	[1]0.97	[2]−0.33	[2]0.30
TP (July–Aug.)	−0.23	0.20	0.01	0.10	0.00	−0.15	[2]0.35	0.06	0.19	[1]0.92	[2]−0.28	0.15
Secchi (July–Sept.)	0.27	0.31	−0.01	0.00	−0.01	0.04	[2]0.06	−0.22	−0.25	[1]−0.36	1.00	[1]−0.35
Secchi (June)	[2]−0.34	[1]0.51	0.16	0.21	[2]0.34	0.33	[2]0.34	[2]0.38	[1]0.38	[1]0.46	0.06	[1]0.34
Chl-*a* (July–Sept.)	[1]−0.40	[2]0.39	[1]0.53	[2]0.38	0.32	0.32	[2]0.38	0.88	[1]0.94	0.33	−0.35	[1]1.00
Chl-*a* (June–Aug.)	[2]−0.38	[2]0.34	[1]0.55	[2]0.33	0.30	0.22	[2]0.38	1.00	[1]0.94	0.17	−0.22	[1]0.88
Chl-*a* (July–Aug.)	[1]−0.42	[1]0.40	[1]0.51	[2]0.35	0.29	0.27	0.32	[1]0.94	1.00	[2]0.31	−0.25	[1]0.94

Table 11. Pearson correlations between the intensity of metalimnetic dissolved oxygen minima in August and various meteorological, hydrological, water quality, and physical characteristics for Green Lake, Wisconsin, during 1988–2019 (24 years), and Pearson correlations for June–September total phosphorous concentrations and July–September Secchi disk depth and chlorophyll-a concentrations and various meteorological, hydrological, water quality, and physical characteristics, during 1988–2019 (32 years).—Continued

[For June–September total phosphorus (TP) concentrations and July–September Secchi disk depth and chlorophyll-a (Chl-a) concentrations, correlation values less than −0.34 or greater than +0.34 are statistically significant at p<0.05, and values less than −0.28 or greater than +0.28 are significant at p<0.10). For all other correlations, values less than −0.39 or greater than +0.39 are statistically significant at p<0.05, and values less than −0.33 or greater than +0.33 are significant at p<0.10). MOM, metalimnetic dissolved oxygen minimum concentration; P, Phosphorus; NC, not computed]

Factor	MOM	Year	Precipitation (water year)	Precipitation (Feb.–Aug.)	Wind velocity (June)	Wind velocity (June–Aug.)	Secchi disk depth (June)	Chl-*a* (June–Aug.)	Chl-*a* (July–Aug.)	TP (June–Sept.)	Secchi disk depth (July–Sept.)	Chl-*a* (July–Sept.)
Physical features												
MOM temperature	−0.08	[1]0.57	0.21	0.21	[1]0.46	0.23	−0.09	0.24	0.22	NC	NC	NC
Water temperature (12-m depth)	−0.22	−0.29	−0.09	−0.11	0.54	0.23	0.14	0.10	0.04	NC	NC	NC
Water temperature (20-m depth)	[1]−0.41	−0.21	−0.05	0.19	0.45	0.02	0.06	−0.10	−0.10	NC	NC	NC
MOM depth	−0.10	[1]−0.55	−0.16	−0.12	0.19	−0.05	0.02	−0.13	−0.15	NC	NC	NC
Thermocline depth	0.03	[1]−0.44	−0.03	−0.12	0.11	−0.01	0.14	−0.01	−0.08	NC	NC	NC

[1]Values are statistically significant ($p<0.05$).

[2]Values are statistically significant ($p<0.10$).

Simulating Daily Changes in Water Quality and Metalimnetic Dissolved Oxygen Minima in Green Lake

The hydrodynamic water-quality model GLM–AED (Hipsey and others, 2017, 2019a) was calibrated for Green Lake and then used to describe the factors affecting water quality in the lake, especially DO concentrations and the intensity of the MOM. Scenarios were then run to estimate how P loading to the lake would need to be changed for the metalimnetic DO concentrations to be below 5 mg/L in less than 75 percent of the years (the WDNR target).

GLM–AED Calibration

Several parameters in the hydrodynamic part of the GLM–AED model (GLM, not using the Aquatic Ecodynamics modeling library part of the model) were first calibrated for the 10-year period from April 16, 2009, to September 30, 2018, with input data including initial water temperature and salinity profiles measured in the lake, daily light extinctions estimated from daily interpolated SDs, total inflow estimated from Silver Creek at HWY A, total outflow (described in the "Unmonitored Water and Phosphorus Inputs and Outputs" section), and daily meteorological data. By using daily varying water clarity during this initial calibration, the biological effects on water clarity and light extinction (to be simulated with AED) were incorporated into the model. During this first phase of calibration, optimal values for the wind factor, longwave radiation factor, sensible heat transfer factor, hypolimnetic mixing efficiency, and mixing efficiency for Kelvin-Helmholtz billowing were estimated by using the derivate-free, covariance matrix adaptation-evolution algorithm for strategy optimization (Hansen, 2016) to minimize the RMSE between the measured and simulated water temperature data (table 12). These parameters were found to be sensitive in affecting GLM results (Ladwig and others, 2021). After calibration of these physical model parameters, values for parameters describing ice evolution were manually fit by an iterative calibration technique to best match the measured and simulated ice on and ice off dates for Green Lake (table 12). The snow albedo factor used in the model (0.4) was low, possibly because of wind blowing most of the snow off of the lake and because ice off on Green Lake was gradual compared to other lakes, which is typically more abrupt. As a result of this calibration, the RMSEs for this 10-year period were 1.28 °C for all water temperatures, 1.38 °C for surface temperatures, 1.62 °C for temperatures at 20-m depth, 0.66 °C for near-bottom temperatures, 1.13 m for the June–August thermocline depths (defined as the center of buoyancy; Imberger and Patterson, 1989), 11.4 days for ice on dates, and 13.3 days for ice off dates (table 13).

After GLM was calibrated, this initial model was used to estimate daily evaporation from the lake, which was then used in a mass balance to estimate daily flows from the remaining unmonitored part of the watershed (not including areas draining into Silver Creek and the Southwest Inlet) and to refine the outflow estimates for the lake (described in the "Unmonitored Water and Phosphorus Inputs and Outputs" section). From this

Table 12. Values for selected parameters used in the General Lake Model coupled to the Aquatic Ecodynamics modeling library for simulating the hydrodynamics in Green Lake, Wisconsin.

[See University of Western Australia (2019) for descriptions of all variables and default (original) values. m^3, cubic meter; m, meter]

Parameter description	Parameter in model	Units	Original value (example)	Value used
Morphometry				
Maximum layers	max_layers	Integer	500	750
Minimum layer volume	min_layer_vol	m^3	0.025	0.025
Minimum layer thickness	min_layer_thick	m	0.15	0.1
Maximum layer thickness	max_layer_thick	m	1.5	1.5
Hydrodynamics—coefficients				
Wind factor	Wind_factor	Real (none)	1	0.9463
Longwave factor	Lw_factor	Real (none)	1	0.95315
Sensible hear transfer factor	C_H	Real (none)	1	0.001455
Hypolimnetic mixing efficiency	Coef_mix_hyp	Real (none)	0.5	0.47999
Kelvin-Helmholtz mixing efficiency	Coef_mix_KH	Real (none)	0.3	0.069724
Ice cover				
Snow albedo	snow_albedo_factor	Real (none)	1	0.4
Ice-on averaging period (fraction of a day)	dt_iceon_avg	Real (none)	0.25	0.16

Table 13. Summary statistics for the initial calibration of the General Lake Model (2009–18) and the full General Lake Model coupled to the coupled to the Aquatic Ecodynamics modeling library (2013–18) for water temperature, ice cover, dissolved oxygen, phosphorus, and nitrogen at selected depths in meters.

[GLM, General Lake Model; GLM–AED, General Lake Model coupled to the Aquatic Ecodynamics modeling library; m, meter; NC, not computed; DRP, dissolved reactive phosphorus]

Parameter	Number of observations	Mean of observations	Mean of model	Mean error	Mean absolute error	Root mean square error	Nash-Sutcliffe efficiency
Initial calibration of GLM—water temperature							
All temperatures	4,025	8.20	8.20	0.00	0.92	1.28	0.95
Surface	66	19.15	18.58	−0.57	1.18	1.38	0.96
20 m	66	6.84	7.35	0.52	1.23	1.62	0.06
65 m	66	4.66	4.49	−0.17	0.53	0.66	−0.05
Thermocline depth	40	10.42	10.69	0.27	0.93	1.13	0.38
Initial calibration of GLM —ice							
Ice on day number	9	13.9	10.1	−3.8	8.9	11.4	NC
Ice off day number	9	87.4	84.9	−2.6	10.1	13.3	NC
GLM–AED calibration—water temperature							
All temperatures	2,656	8.21	8.48	−0.27	1.14	1.65	0.93
Surface	46	19.18	18.52	−0.66	1.13	1.39	0.96
20 m	46	6.57	7.80	1.23	1.27	1.84	−0.33
65 m	46	4.52	4.51	−0.01	0.59	0.65	−0.53
Thermocline depth	28	10.29	10.31	0.02	1.16	1.58	−0.49
GLM–AED calibration—ice							
Ice on day number	5	11.20	14.40	3.20	6.00	7.2	NC
Ice off day number	5	85.40	90.40	6.00	7.20	13.5	NC
GLM–AED calibration—dissolved oxygen							
West side—all	2,651	9.18	8.62	0.56	1.30	1.77	0.66
West side—5 m	45	9.96	10.03	−0.06	0.76	1.05	0.74
West side—12 m	46	6.98	7.79	−0.81	1.75	2.09	0.72
West side—13 m	44	6.93	7.88	−0.95	1.45	1.73	0.79
West side—40 m	43	10.30	9.46	0.84	0.94	1.19	0.48
West side—60 m	41	8.33	7.14	1.19	1.95	2.40	0.34
East side—5 m	45	10.12	10.03	0.10	0.89	1.27	0.69
East side—12 m	46	7.45	7.80	−0.34	2.12	2.67	0.49
East side—13 m	44	7.10	7.88	−0.78	1.84	2.26	0.64
GLM–AED calibration—phosphorus							
Total—surface	45	0.020	0.021	−0.001	0.005	0.006	0.68
DRP—surface	33	0.006	0.007	−0.001	0.004	0.006	0.47
GLM–AED calibration—nitrogen							
Total—surface	34	0.624	0.643	−0.019	0.091	0.114	0.06
Nitrate—surface	34	0.069	0.083	−0.014	0.049	0.067	0.65

iterative set of simulations, it was estimated that all areas other than Silver Creek contributed 45 percent of the total flow from the watershed (Silver Creek flow × 0.81).

With measured inflows from Silver Creek and Southwest Inlet and refined flow estimates from the unmonitored areas and their respective measured or estimated water quality (described in the "Unmonitored Water and Phosphorus Inputs and Outputs" section), parameters for various water-quality and biological components in AED (Hipsey and others, 2019b) were calibrated for the 6-year period from May 7, 2013, to December 30, 2018, by using a manual, iterative calibration technique. Automated calibration techniques were not used in order that other guidance from empirical observations could be more easily applied. In other words, many soft constraints were considered during calibration (Yen and others, 2016; Mi and others, 2020). As much observational information as possible was examined during the final calibration, including the extent of thermal stratification, timing of ice on and ice off, distribution of total and dissolved P and nitrogen concentrations, succession of phytoplankton species, particle settling, and most importantly the vertical changes in DO concentrations. The model was initiated with water temperature, salinity, and water chemistry measured in the lake during spring overturn on May 7, 2013.

To calibrate the chemical and biological components of AED, parameter values from a GLM–AED model developed for Lake Mendota, Wisconsin, were used as a starting point (Snortheim and others, 2017). We then concentrated on calibrating the values for the parameters found by Ladwig and others (2021) to be most sensitive to AED results for simulating DO concentrations, especially values for the parameters in the DO, P, organic matter, and phytoplankton modules of the model. Most of the values for parameters used to simulate carbon, nitrogen, silica, and zooplankton were not adjusted. The final values for most parameters specifically adjusted for the calibrated GLM–AED model for Green Lake are provided in table 14. All inputs, outputs, and the final GLM–AED model are available in Robertson and Kennedy (2021).

After full GLM–AED calibration, with no changes in the coefficients of the parameters in GLM found during the initial calibration, the model simulated water temperature and ice cover about as well as in the initial calibration of GLM (shown for water temperature in fig. 18). The RMSEs for this 6-year period (2013–18) were 1.65 °C for all water temperatures (Nash-Sutcliffe efficiency, NSE, of 0.93), 1.39 °C for near-surface temperatures, 1.84 °C for temperatures at 20-m depth, 0.65 °C for near-bottom temperatures, 1.58 m for the June–August thermocline depth, 7.2 days for ice on dates, and 13.5 days for ice off dates (table 13).

GLM–AED simulated the changes in DO concentrations throughout the water column in Green Lake by incorporating oxygen inputs from the atmosphere and photosynthesis by phytoplankton, oxygen gains or losses back to the atmosphere as surface-water temperature change, respiration/metabolic losses of oxygen by phytoplankton and zooplankton, mineralization of organic matter, sediment oxygen demand, and vertical oxygen transport by mixing and diffusion. The growth rates and respiration/metabolic losses for each functional plankton group were estimated during the calibration process (table 14). Three parameters in the model explained oxygen consumption at the sediment-water interface by using Michaelis-Menten type limitation with a base sediment oxygen demand (35 millimoles per square meter per day [mmol/m^2/d]), the Arrhenius temperature multiplier for the oxygen sediment flux that results in more oxygen consumption at warmer water temperatures than at colder temperatures (1.08), and the half-saturation concentration describing the rate of the oxygen sediment flux (6.8 mmol/m^3). These values for the sediment oxygen demand parameters were used for the entire bottom of the lake.

GLM–AED simulated the changes in DO that were measured throughout the water column well, as shown at 5 m, 12 m (typical depth of the MOM in August), 40 m, and 60 m for the west side of the lake in figure 19. The RMSEs for DO concentrations for the 6-year period were 1.77 mg/L for all depths (NSE=0.66), 1.05 mg/L for 5 m, 2.09 mg/L (NSE=0.73) for 12 m, 1.19 mg/L for 40 m, and 2.40 mg/L for 60 m. The RMSE for DO concentrations at 12 m from the east side of the lake was slightly higher than that for the west side of the lake: 2.67 mg/L (NSE=0.49). The RMSE was a generally little higher and the NSE was little lower at 12 m than at other depths partially because of a strong seiche of the thermocline in Green Lake that resulted in large short-term changes in DO concentrations. The model simulated the development and intensity of the MOM well in all years (fig. 19*B*).

Rather than calibrating GLM–AED on the basis only of how well it simulated the DO concentrations in Green Lake, the following biological and chemical observations provided guidance for the specification of the model: (1) three functional groups of phytoplankton with a progression from diatoms to cryptophytes and cyanophytes as summer progressed; (2) settling of organic matter; (3) changes in nutrient concentrations, with high TP and DRP concentrations in winter and early spring and almost complete depletion of DRP and nitrate in the epilimnion as summer progressed, and (4) intense DO depletion in the metalimnion by late summer (figs. 4 and 8).

GLM–AED simulated the three functional groups of phytoplankton with a progression from diatoms, to cryptophytes, to cyanophytes (fig. 20). The model simulated phytoplankton primarily in the epilimnion and metalimnion of the lake, similar to the measurements collected during 2017 (fig. 9). Diatoms were the dominant species in spring, cryptophytes were the dominant species in early summer, and cyanophytes were the dominant species in midsummer to late summer. Given limited phytoplankton data available for Green Lake, the goal of the model was to simply simulate the seasonal progression of the three functional groups, which the model simulated.

GLM–AED simulated the transformation of phytoplankton into particulate organic carbon, nitrogen, and P and ultimately into their dissolved forms. The model simulated the settling of the particulate matter by using Stokes Law,

Table 14. Values for selected parameters used in General Lake Model coupled to the Aquatic Ecodynamics modeling library for simulating the water quality and biology in Green Lake, Wisconsin.

[See University of Western Australia (2017) and Hipsey and others (2019b) for descriptions of all variables and default (original) values. dom [DOM], dissolved organic matter; pop, particulate organic phosphorus; pon, particulate organic nitrogen; poc, particulate organic carbon; pom, particulate organic matter; mmol m^{-2} day^{-1},millimole per square meter per day; mmol m^{-3}, millimole per cubic meter; d, day; m, meter; kg m^{-3}, kilogram per cubic meter; C, degree Celsius]

Parameter description	Parameter in model	Units	Original value (example)	Value used
		Oxygen module		
Sediment oxygen demand	Fsed_oxy	mmol m^{-2} day^{-1}	−40	−35
Half-saturation constant for sediment oxygen flux	Ksed_oxy	mmol m^{-3}	50	6.8
Arrhenius temperature multiplier	theta_sed_oxy	No units	1.08	1.08
		Phosphorus module		
Sediment phosphorus flux	Fsed_frp	mmol m^{-2} day^{-1}	0.08	0.8
Half-saturation constant for sediment phosphorus flux	Ksed_frp	mmol m^{-3}	80	40
Arrhenius temperature multiplier for phosphorus	theta_sed_frp	No units	1.08	1.07
		Nitrogen module		
Sediment nitrate flux	Fsed_nit	mmol m^{-2} day^{-1}	−5	−1.755
Half-saturation constant for sediment nitrate flux	Ksed_nit	mmol m^{-3}	100	142
Sediment ammonia flux	Fsed_amm	mmol m^{-2} day^{-1}	4	55
Half-saturation constant for sediment ammonia flux	Ksed_amm	mmol m^{-3}	25	45
Maximum nitrification rate	Rnitrif	mmol m^{-2} day^{-1}	0.1	0.4
Half-saturation constant for nitrification	Knitrif	mmol m^{-3}	8	20
Maximum denitrification rate	Rdenit	mmol m^{-2} day^{-1}	0.3	1.5
Half-saturation constant for denitrification	Kdenit	mmol m^{-3}	2	2
Arrhenius temperature multiplier for all nitrogen forms in water and sediments	theta_	No units	1.08	1.08
		Silica module		
Sediment silica flux	Fsed_rsi	mmol m^{-2} day^{-1}	0	25
Half-saturation constant for sediment silica flux	Ksed_rsi	mmol m^{-3}	50	50
Arrhenius temperature multiplier for silica	theta_sed_rsi	No units	1.08	1.08
		Carbon module		
Sediment dissolved inorganic carbon flux	Fsed_dic	mmol m^{-2} day^{-1}	4	4
Half-saturation constant sediment dissolved inorganic flux	Ksed_dic	mmol m^{-3}	30	30
Arrhenius temperature multiplier for dissolved organic carbon and methane	theta_sed_	No units	1.08	1.08
		Organic matter		
Maximum rate of DOM mineralization	Rdom_minerl	d^{-1}	0.5	0.15
Half-saturation constant for dom mineralization	Kdom_minerl	mmol m^{-3}	31.25	31.25
Half-saturation constant for dom hydrolysis	Kdom_hydrol	mmol m^{-3}	31.25	31.25
Maximum rate of decomposition of pop	Rpop_hydrol	d^{-1}	0.05	0.04
Maximum rate of decomposition of pon	Rpon_hydrol	d^{-1}	0.05	0.03
Maximum rate of decomposition of poc	Rpoc_hydrol	d^{-1}	0.05	0.06
Maximum rate of mineralization of labile dom	Rdomr_minerl	d^{-1}	0.0001	0.0005
Maximum rate of course pom breakdown	Rcpom_bdown	d^{-1}	0.0001	0.001
Radius of pom	d_pom	m	0.000002	0.000009
Density of pom	rho_pom	kg m^{-3}	1,120	1,050

Table 14. Values for selected parameters used in General Lake Model coupled to the Aquatic Ecodynamics modeling library for simulating the water quality and biology in Green Lake, Wisconsin.—Continued

[See University of Western Australia (2017) and Hipsey and others (2019b) for descriptions of all variables and default (original) values. dom [DOM], dissolved organic matter; pop, particulate organic phosphorus; pon, particulate organic nitrogen; poc, particulate organic carbon; pom, particulate organic matter; mmol m^{-2} day^{-1},millimole per square meter per day; mmol m^{-3}, millimole per cubic meter; d, day; m, meter; kg m^{-3}, kilogram per cubic meter; C, degree Celsius]

Parameter description	Parameter in model	Units	Original value (example)	Value used
		Organic matter—Continued		
Radius of course pom	d_cpom	m	0.00001	0.00001
Density of course pom	rho_cpom	kg m^{-3}	1,300	1,400
		Phytoplankton diatoms		
Growth rate	pd%R_growth	d^{-1}	2.8	2.51
Optimum temperature	pd%T_opt	C	20	15.0
Respiration/metabolic loss rate	pd%R_resp	d^{-1}	0.12	0.027
		Phytoplankton cryptophytes		
Growth rate	pd%R_growth	d^{-1}	0.7–1.1	2.15
Optimum temperature	pd%T_opt	C	20–28	22.6
Respiration/metabolic loss rate	pd%R_resp	d^{-1}	0.012–0.12	0.12
		Phytoplankton cyanophytes		
Growth rate	pd%R_growth	d^{-1}	0.53–1.1	2.5
Optimum temperature	pd%T_opt	C	28–29	30
Respiration/metabolic loss rate	pd%R_resp	d^{-1}	0.08	0.13

which simulates settling of the particulates as a function of their density (1,050 kilograms per cubic meter) and radius (9 µm), and resulted in the accumulation of particulate matter in the metalimnion (shown for particulate organic carbon in figure 21*A*). The decomposition of particulate organic matter that accumulated in the metalimnion may partially explain the oxygen depletion at this depth. Although particulate organic carbon, nitrogen, and P accumulated in the metalimnion, simulated dissolved organic carbon concentrations (as well as nitrogen and P) were high throughout the epilimnion and upper metalimnion (shown for dissolved organic matter in figure 21*B*), which is consistent with the measured dissolved organic carbon concentrations.

GLM–AED simulated the changes in P and nitrogen concentrations measured in Green Lake well (fig. 22). The model simulated the increase in TP and DRP concentrations during fall turnover from internal sediment recycling, the sustained high TP and DRP concentrations throughout winter, the rapid depletion in DRP and nitrate concentrations in early summer, and the gradual decrease in TP concentrations during summer. The model underestimated early spring TP concentrations in a few years, but the model simulated the changes in DRP and nitrate concentrations that drive and ultimately limit algal production during spring and summer well. The model also simulated the uptake of DRP and nitrate by phytoplankton that resulted in the decrease in TP concentrations lagging the decrease in DRP concentrations. The overall RMSEs (and NSEs) for near-surface TP and DRP were 0.006 mg/L (NSE=0.68) and 0.006 mg/L (NSE=0.47), respectively. The overall RMSEs (and NSEs) for near-surface total nitrogen and nitrate were 0.114 mg/L (NSE=0.060) and 0.067 mg/L (NSE=0.65), respectively.

Evidence of Factors Affecting Metalimnetic Dissolved Oxygen Minima Based on GLM-AED

The GLM–AED model accurately simulated the development and intensity of the MOM, as well as changes in other chemical (nutrients; table 13) and biological (phytoplankton and organic matter) constituents measured in the lake. Therefore, it is believed that the model accurately simulated the relative importance of the processes causing the MOM to develop. To further examine which factors and processes were most important in affecting the development and intensity of the MOM (fig. 2), specific processes included in GLM–AED were systematically turned off, one at a time, and the DO concentrations in the lake were compared with the concentration simulated during the 2013–18 base simulation, which was a similar approach to that used by Mi and others (2020).

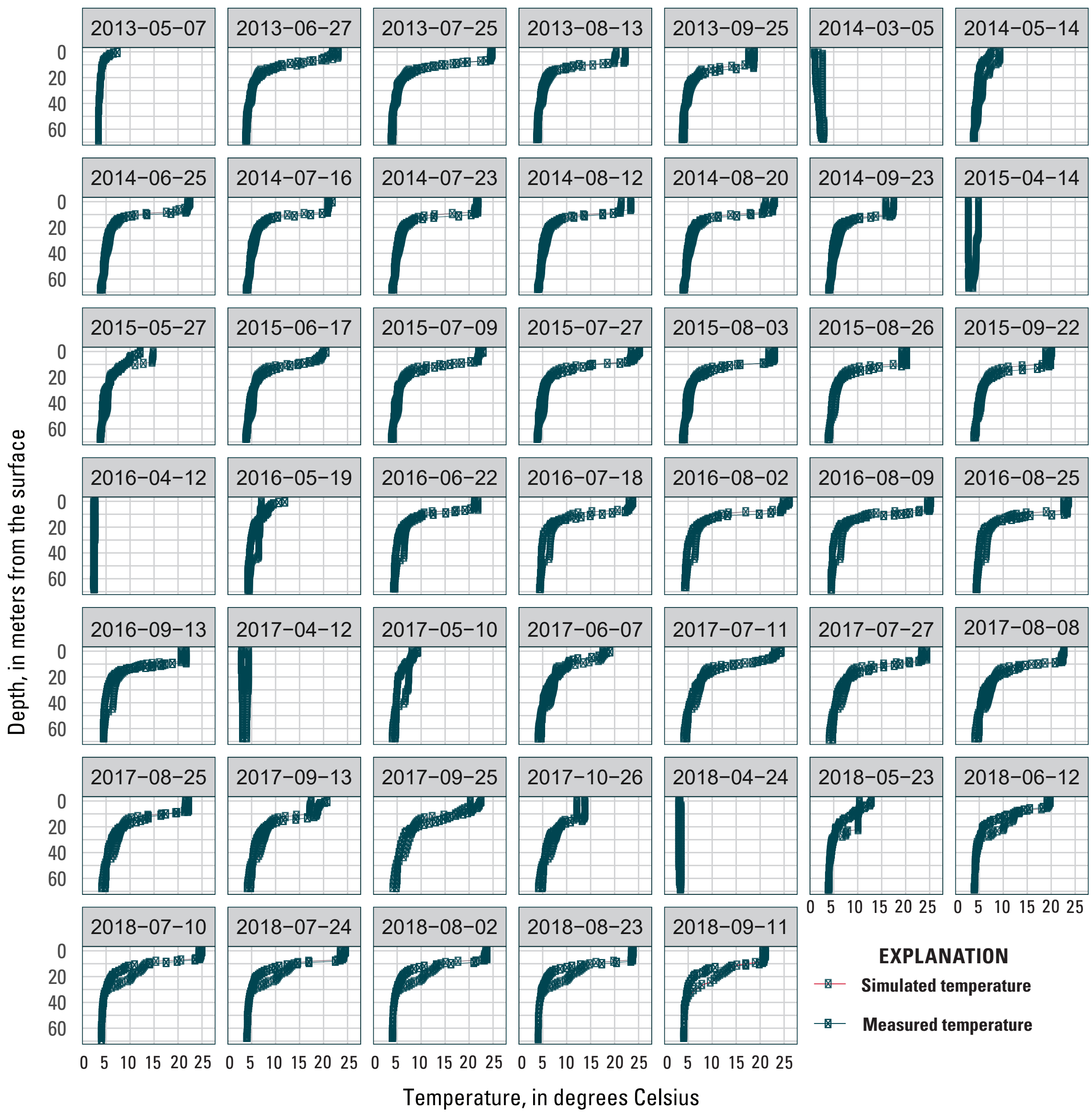


Figure 18. Comparison between water temperatures measured and simulated by the General Lake Model coupled to the Aquatic Ecodynamics modeling library during May 2013 to September 2018 for the west side of Green Lake, Wisconsin. Date format is YYYY–MM–DD.

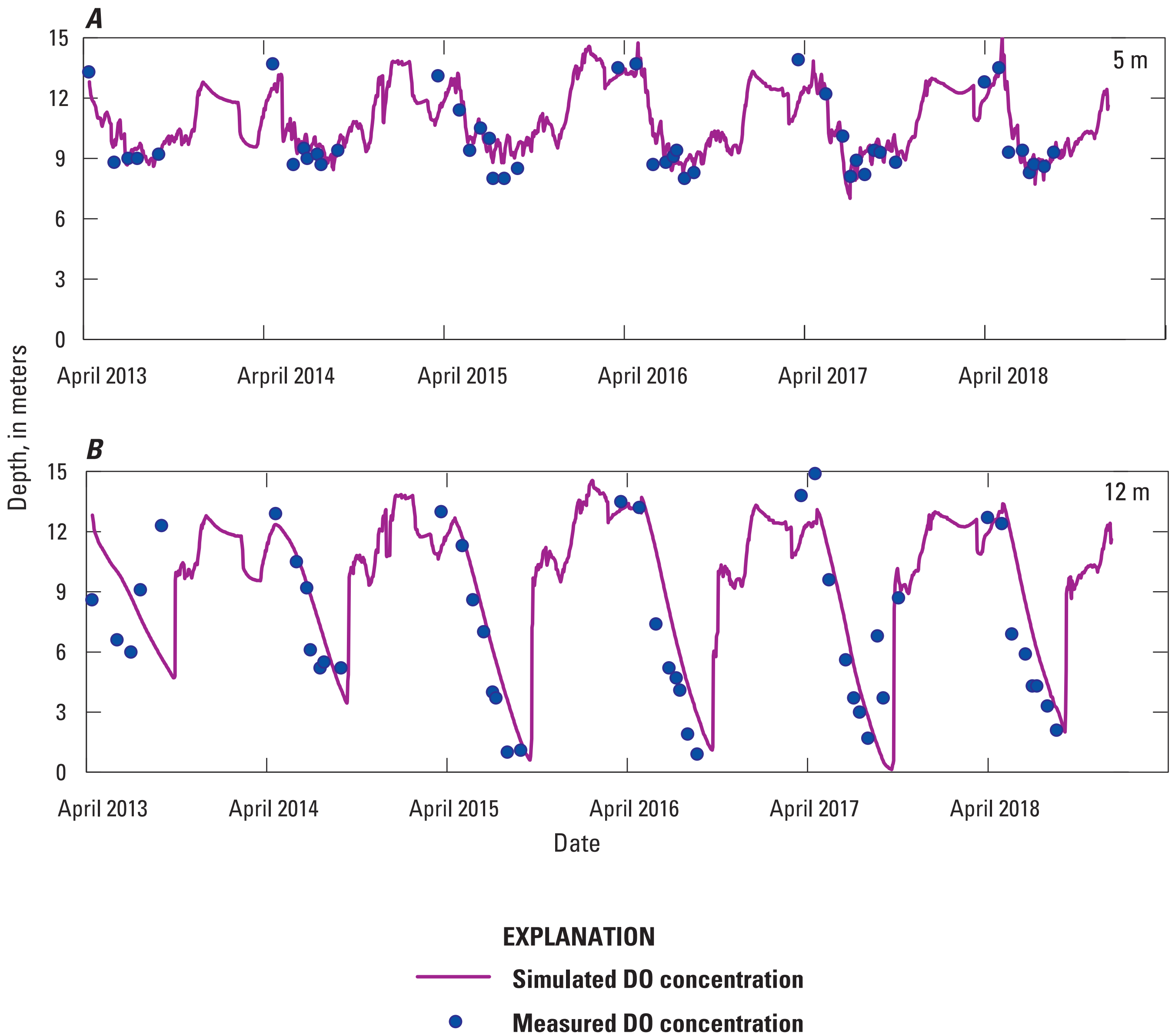
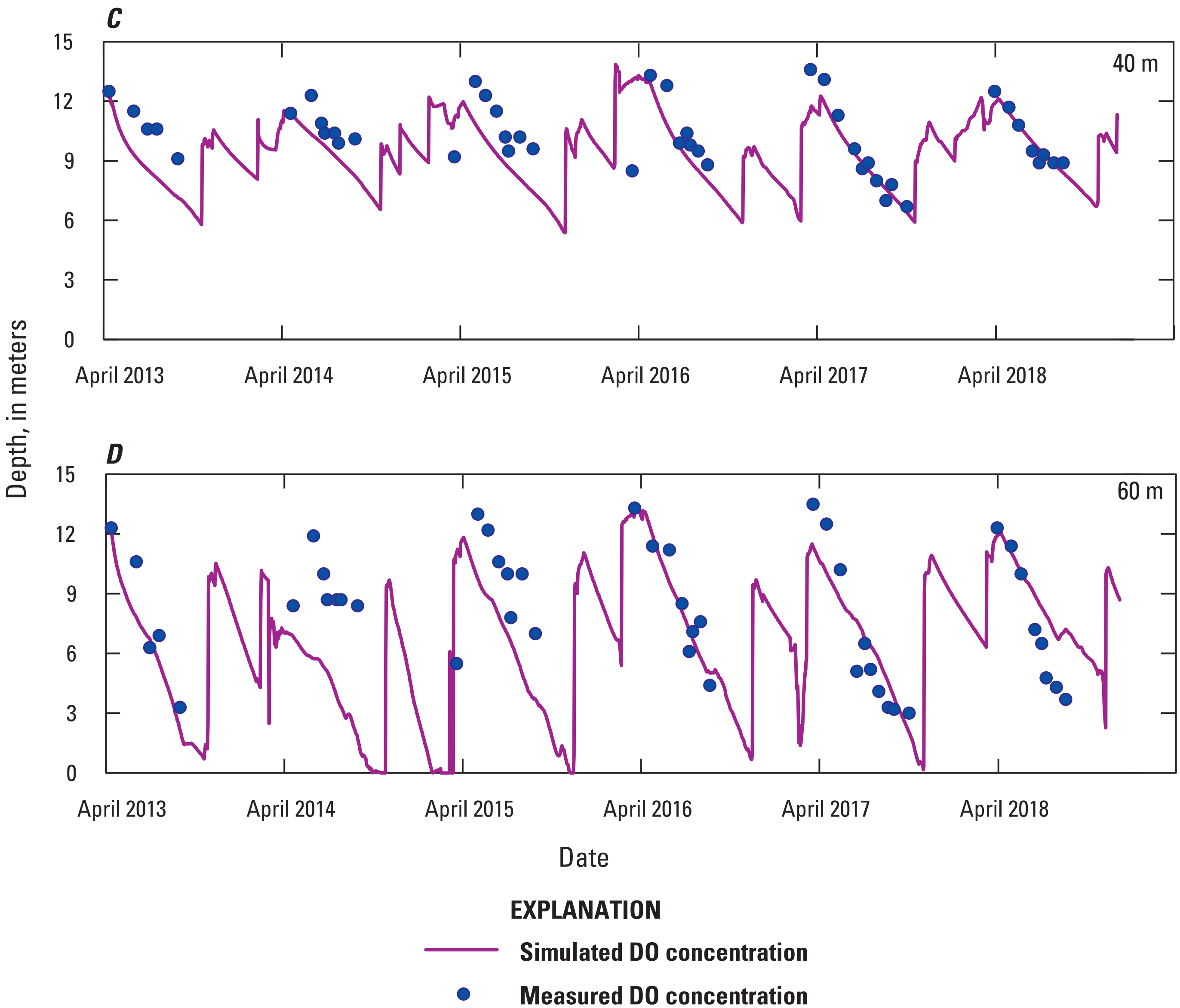


Figure 19. Comparison between dissolved oxygen (DO) concentrations measured and simulated by the General Lake Model coupled to the Aquatic Ecodynamics modeling library for the west side of Green Lake, Wisconsin, during May 2013 to December 2018 at *A*, 5 meters (m), *B*, 12 meters, *C*, 40 meters, and *D*, 60 meters below the surface of the lake.



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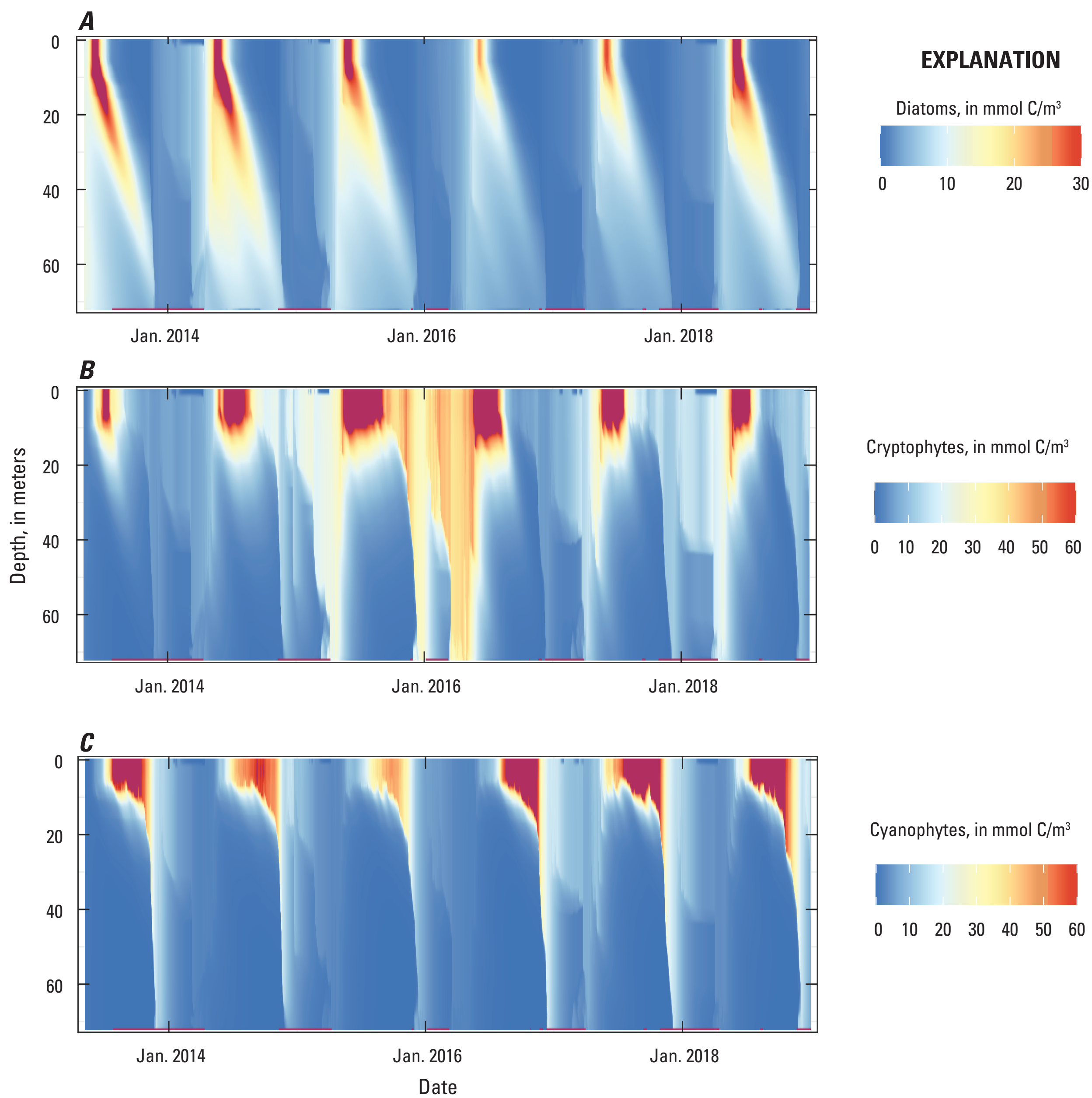

Figure 20. Distributions of *A*, diatoms, *B*, cryptophytes, and *C*, cyanophytes, in millimoles of carbon per cubic meter (mmol C/m^3), in Green Lake, Wisconsin, simulated with the General Lake Model coupled to the Aquatic Ecodynamics modeling library, from May 2013 to December 2018.

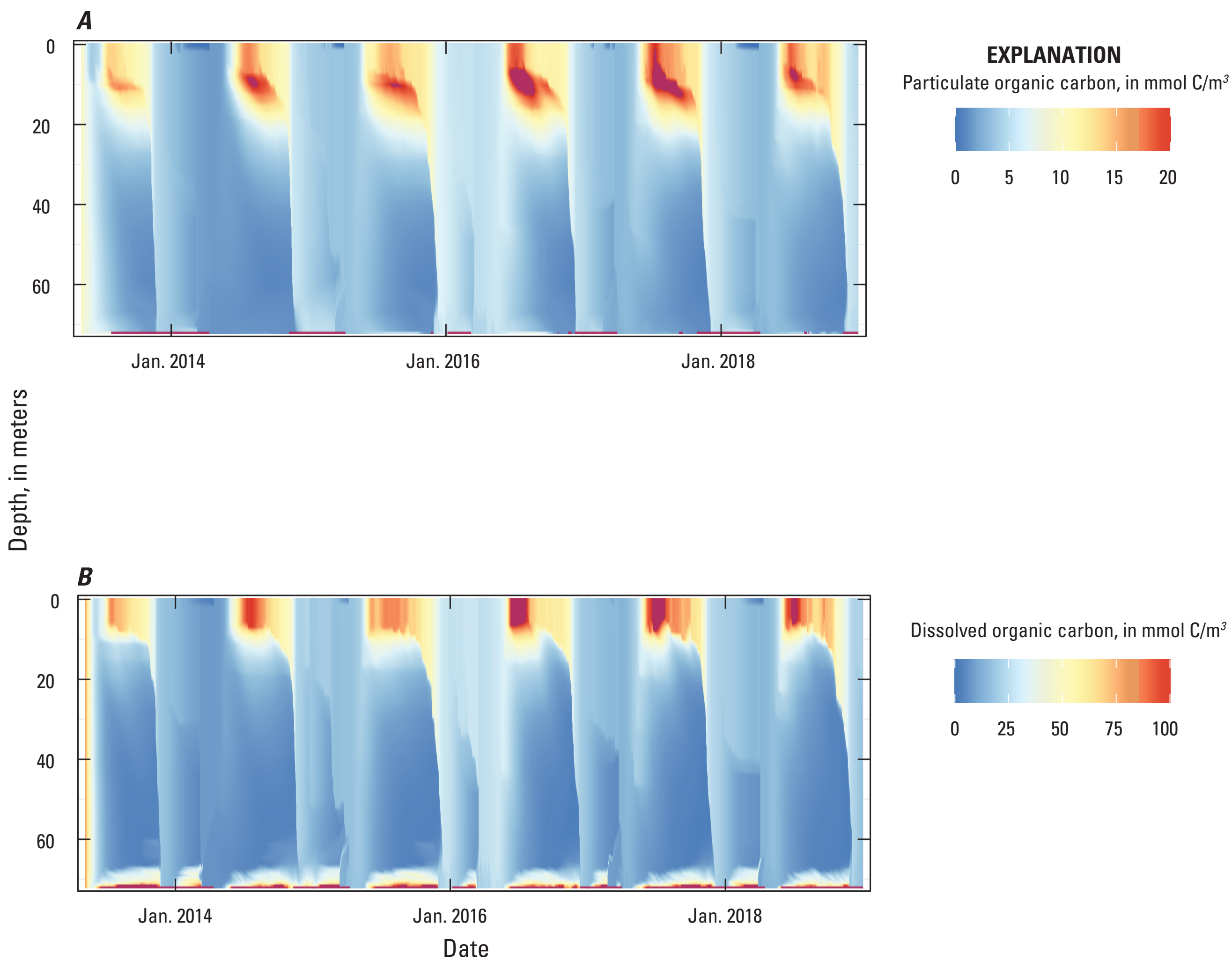

Figure 21. Distributions of *A*, particulate and *B*, dissolved organic carbon, in millimoles of carbon per cubic meter (mmol C/m^3), in Green Lake, Wisconsin, simulated with the General Lake Model coupled to the Aquatic Ecodynamics modeling library, from May 2013 to December 2018.

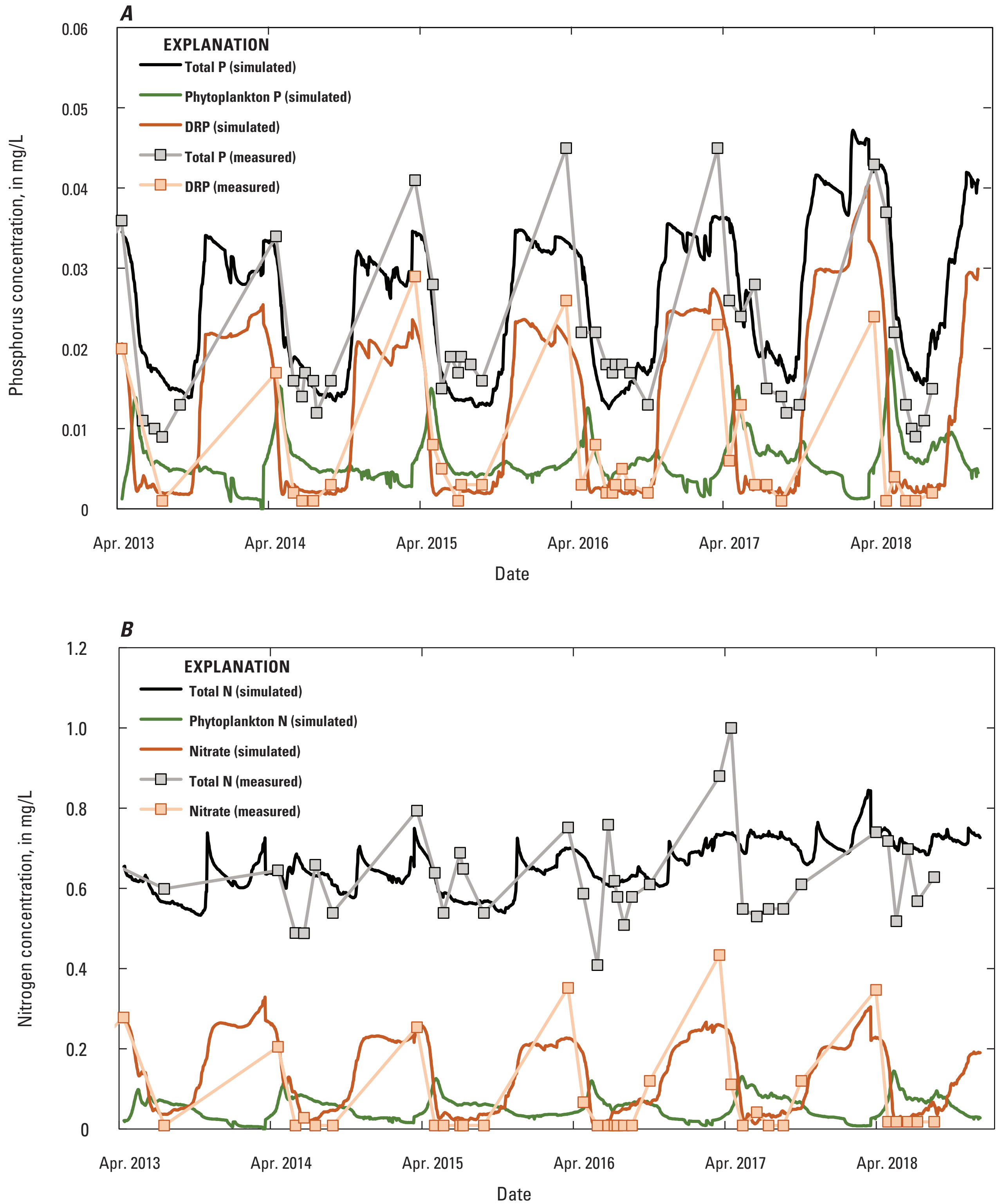

Figure 22. Comparison between concentrations measured and simulated by the General Lake Model coupled to the Aquatic Ecodynamics modeling library of various forms of *A*, phosphorus (P) and *B*, nitrogen (N) near the surface in Green Lake, Wisconsin, during May 2013 to December 2018. DRP, dissolved reactive phosphorus; mg/L, milligram per liter.

Seven scenarios were simulated to estimate how various processes affect the DO distribution in Green Lake:

1. scenario *A*, base simulation, a full simulation with all processes applied;
2. scenario *B*, elimination of interflow by setting water temperatures of the daily inflows to the simulated near-surface water temperatures in the base simulation;
3. scenario *C*, elimination of P in all inflows by reducing the inflow P concentrations by 95 percent;
4. scenario *D*, elimination of internal sediment recycling of P by setting the sediment P flux to 0.01 millimole per cubic meter per day;
5. scenario *E*, elimination of sediment oxygen consumption by setting the oxygen demand to 1.0 mmol/m^3;
6. scenario *F*, elimination of phytoplankton oxygen production and consumption by setting the growth rates of all three phytoplankton groups to 0.01 per day; and
7. scenario *G*, elimination of the effects of zooplankton by turning off the zooplankton module in GLM–AED.

Changes in DO concentrations for each scenario are shown in figure 23, and the minimum daily DO concentrations between 10 and 20 m (typical depth range of the MOM) are shown in figure 24.

Eliminating the possibility of interflow (scenario b) had little effect on DO concentrations (figs. 23 and 24), which suggests that interflow does not typically occur. Almost entirely eliminating P input from all external inflows (95-percent reduction in P concentrations of all inflows—scenario c) almost eliminated the MOM (metalimnetic DO concentrations only dropped below 5 mg/L in 1 year) and greatly reduced the amount of phytoplankton, but it had little effect on changes in DO concentrations in the deepest areas of the lake. This suggests that external P loading is an important P source to the lake and is important for algal productivity. Eliminating P release from the sediment, which partially leads to high P concentrations in spring, (scenario d) strongly affected the MOM (metalimnetic DO concentrations only dropped below 5 mg/L in 1 year) and reduced phytoplankton populations. This suggests that internal sediment recycling affects the P concentrations in the lake and ultimately the algal productivity. Reductions in external and internal sediment recycling had similar effects on the intensity of the MOM, which suggests these sources were of similar magnitude and that the magnitude was similar to that found when constructing an inventory of P inputs to the lake (see the “Summary of Phosphorus Inputs to Green Lake” section). Eliminating sediment oxygen consumption (scenario e) resulted in slightly increased DO concentrations throughout the hypolimnion and slightly less severe MOMs in most years. This suggests that DO consumption in the metalimnion is primarily driven by water column processes rather than sediment processes. Eliminating phytoplankton (scenario f) eliminated the MOM in all years (metalimnetic DO concentrations never declined below 6.5 mg/L), and DRP concentrations increased dramatically. This suggests that phytoplankton growth, phytoplankton death, and ultimately the mineralization of organic matter in the metalimnion is the primary driver of the MOM rather than sediment processes. Eliminating the effects of zooplankton (scenario g) had little effect on DO concentrations in all years. Results from these scenarios suggest that the MOMs in Green Lake are primarily caused by high P inputs to the lake (primarily external and seasonal inputs from internal sediment recycling) driving high phytoplankton production and ultimately high phytoplankton metabolism/respiration and decomposition within the water column, which ultimately consume much of the DO within the metalimnion. This conclusion agrees with the empirical evidence that the MOM intensity was correlated with the amount of Chl-*a* in the lake. These results also agree with results from a DO metabolism model constructed by using high-temporal-resolution DO data in Green Lake that showed the MOM was driven by water-column metabolism (Naziri Saeed, 2020). Therefore, the P loading to the lake that drives phytoplankton production would need to be reduced to reduce the intensity of the MOM in Green Lake.

Response of Water Quality and Metalimnetic Dissolved Oxygen Minima to Basinwide Changes in Phosphorus Loading

Fifteen scenarios (table 15) were simulated with GLM–AED to describe how the water quality (DO, TP, and Chl-*a*) in Green Lake is expected to respond to changes in P loading and to estimate how much the external P loading needs to be reduced for DO concentrations in the metalimnion to drop below 5 mg/L in only 1 out of 5 years (in other words, to stay above 5 mg/L in more than 75 percent of the years). In each scenario, GLM–AED was run for 6 years from May 7, 2013, to 30 December 30, 2018, but only the last 5 years of the results were examined to reduce the effects of defining the initial conditions in the lake.

Scenario GLM–1 simulated current (2014–18) or base conditions in Green Lake. Fourteen simulations (scenarios 2–15) were then used to estimate the response in the water quality of Green Lake to changes from −95 to +100 percent in the controllable P sources (all P sources except atmospheric inputs; inputs from waterfowl and septic systems that collectively represented less than 8 percent of the external P sources were not included in the GLM–AED model). For each scenario, two simulations were performed that treated internal sediment recycling of P differently. The first simulation of each pair represented what would be expected in the years immediately after an external load reduction. In these first simulations, external tributary P loading was changed by the specified percentage, but internal sediment recycling of P was unchanged. This was done by changing the concentrations of all forms of P in the inflows by the specified percentage. In addition, the initial concentration of each phytoplankton group was changed by a percentage similar to the expected

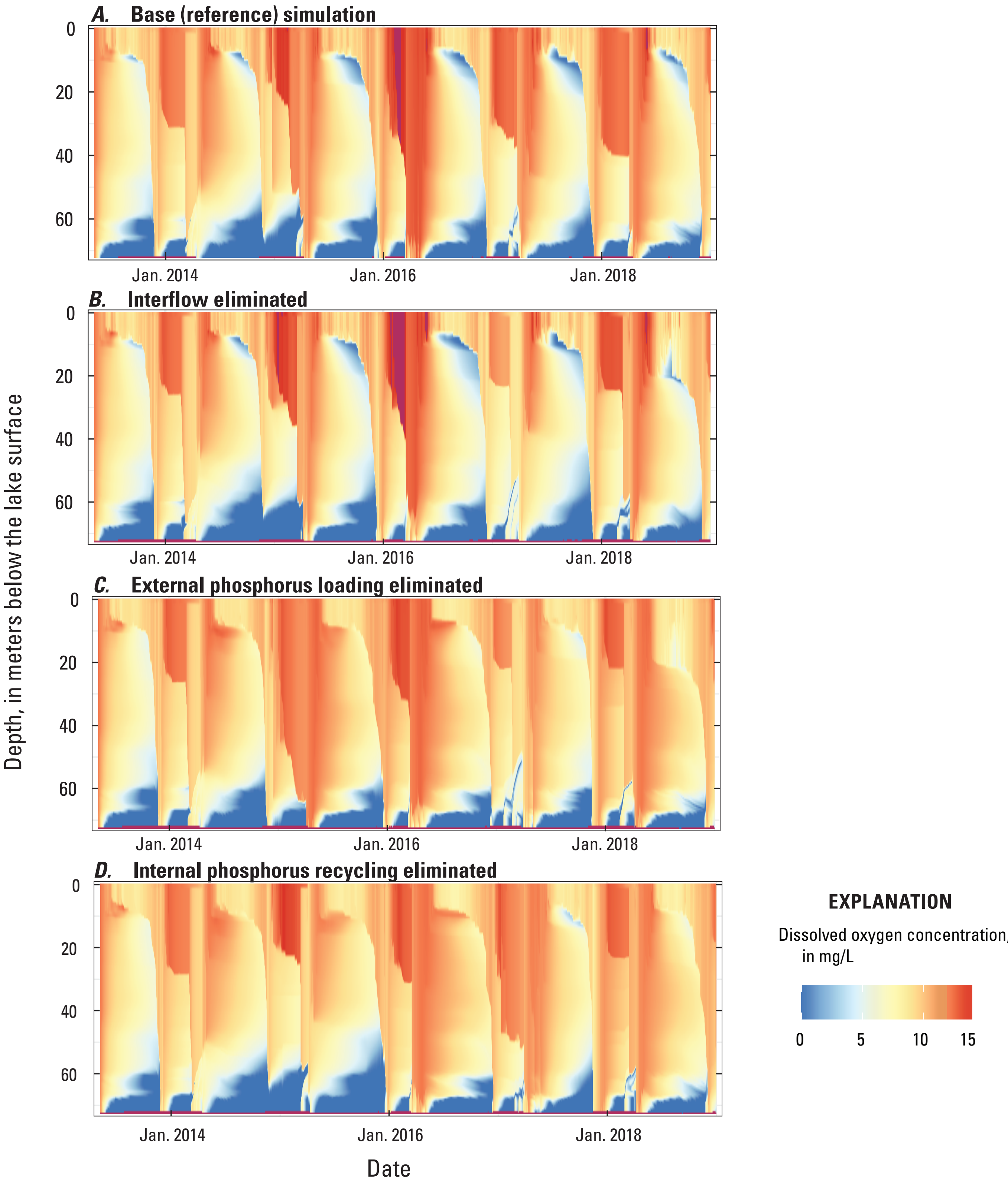

Figure 23. Distributions of dissolved oxygen concentrations in Green Lake, Wisconsin, simulated with the General Lake Model coupled to the Aquatic Ecodynamics modeling library, with individual processes or drivers in the model eliminated from the *A*, base (reference) simulation with the full model: *B*, interflow eliminated, *C*, external phosphorus loading eliminated, *D*, internal sediment recycling of phosphorus eliminated, *E*, sediment oxygen demand eliminated, *F*, all effects of phytoplankton eliminated, and *G*, all effects of zooplankton eliminated. mg/L, milligram per liter.

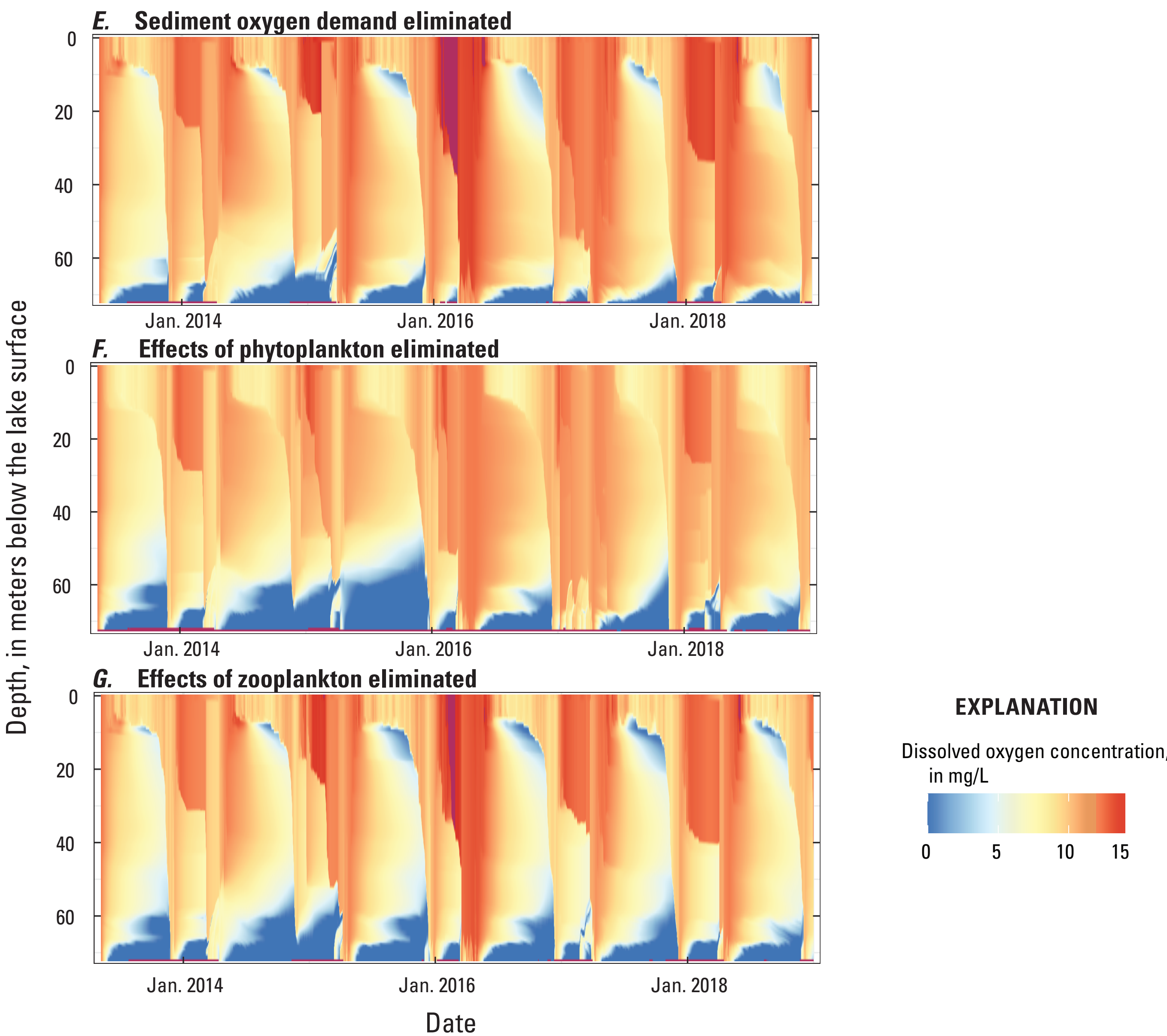

Figure 23.—Continued

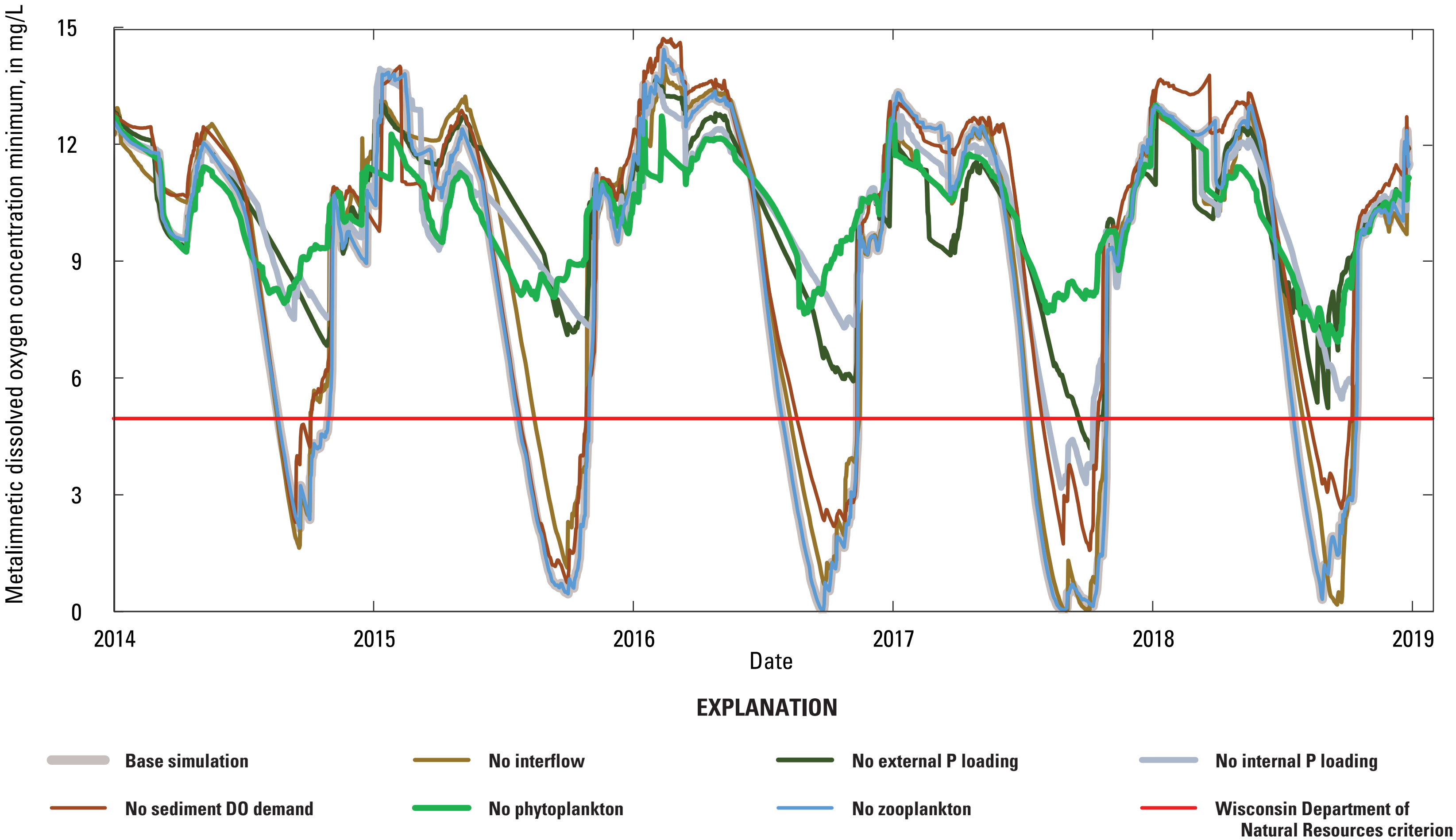

Figure 24. Minimum dissolved oxygen concentrations between 10- and 20-meter depths (metalimnion) in Green Lake, Wisconsin, for the seven scenarios shown in figure 23. DO, dissolved oxygen; P, phosphorus.

Table 15. Responses of minimum dissolved oxygen concentrations in the metalimnion and deep water (65 meters) during August 15 through September 30; spring (April 15 to May 1) and summer (June 1 to September 15) total phosphorus concentrations in milligrams per liter; and summer (July 15 to September 15) chlorophyll-a concentrations in micrograms per liter in Green Lake, Wisconsin, to various changes in controllable external phosphorus (P) sources based on the General Lake Model coupled to the Aquatic Ecodynamics modeling library simulations.

[All results are based on 6-year simulations, but loading and model results do not include the first year of the simulation. Initial concentrations of the forms of phosphorus (P) in the base simulation in millimoles per cubic meter: dissolved reactive phosphorus—0.746; particulate organic phosphorus—0.3; dissolved organic phosphorus—0.02. Initial concentrations of phytoplankton in the base simulation in millimoles of carbon per cubic meter: diatoms—12.1; cryptophytes—2.3; and cyanophytes—0.4. Initial internal P loading in the base simulation was 0.80 millimole of P per square meter per day. kg, kilogram; TP, total phosphorus; mg/L, milligram per liter; MOM, metalimnetic dissolved oxygen minimum; DO, dissolved oxygen; m, meter; Chl-a, chlorophyll-a; µg/L, microgram per liter]

Percent change in controllable P sources	Scenario number	Tributary loading (kg)	Total external P loading (kg)	Percent change in total external P loading	Percent change in west side TP[1]	Initial TP (mg/L)	Mean MOM[2] (mg/L)	Years with MOM less than 5 mg/L	Years with MOM less than 2 mg/L	Years with MOM less than 5 mg/L (no changes in internal loading)	Mean minimum DO at 65 m (mg/L)[2]	No change in internal loading	With change in internal loading		
												Mean summer TP (mg/L)	Mean spring TP (mg/L)	Mean summer TP (mg/L)	Mean summer Chl-*a* (µg/L)
−95	GLM–2	385	1,140	−87	−73.5	0.010	7.9	0	0	2	0.8	0.012	0.007	0.005	0.8
−75	GLM–3	1,920	2,680	−68	−51.9	0.017	7.3	0	0	2	0.7	0.014	0.011	0.008	1.7
−60	GLM–4	3,080	3,830	−55	−39.2	0.022	6.4	1	0	2	0.4	0.014	0.014	0.010	2.5
−57	GLM–5	3,310	4,060	−52	−36.8	0.023	6.5	1	0	4	0.7	0.014	0.016	0.010	2.6
−55	GLM–6	3,460	4,215	−50	−35.3	0.023	6.2	2	0	3	0.6	0.014	0.014	0.011	2.7
−50	GLM–7	3,850	4,600	−46	−31.6	0.025	5.3	2	0	5	0.3	0.015	0.017	0.012	3.2
−40	GLM–8	4,610	5,370	−36	−24.5	0.027	6.3	2	0	5	0.7	0.016	0.018	0.012	3.2
−35	GLM–9	5,000	5,750	−32	−21.2	0.028	5.4	2	0	5	0.1	0.015	0.020	0.013	3.5
−30	GLM–10	5,380	6,140	−27	−11.7	0.032	3.6	3	1	5	0.1	0.016	0.022	0.013	3.9
−25	GLM–11	5,770	6,520	−23	−14.8	0.031	3.3	5	1	5	0.2	0.016	0.025	0.015	4.6
0	**GLM–1 (Base)**	**7,690**	**8,440**	0	**0.0**	**0.036**	**0.6**	5	4	5	**0.1**	**0.018**	**0.036**	**0.018**	**6.1**
25	GLM–12	9,610	10,400	23	13.3	0.041	0.2	5	5	5	0.0	0.020	0.051	0.024	7.9
50	GLM–13	11,500	12,300	46	25.6	0.045	0.8	5	4	5	0.2	0.022	0.056	0.026	8.0
75	GLM–14	13,500	14,200	68	37.1	0.049	0.1	5	5	5	0.1	0.023	0.060	0.039	9.3
100	GLM–15	15,400	16,100	91	47.8	0.053	1.9	4	3	5	0.6	0.026	0.079	0.049	8.1

[1]Percent changes in near-surface total phosphorus concentrations in the west side of Green Lake found with the Canfield-Bachmann model for the specified change in external P loading.

[2]Mean minimum dissolved oxygen concentrations for the entire period from August 15 to September 30.

percent change in near-surface TP concentrations in the west side of the lake from the Canfield-Bachmann model (fig. 17). The second simulation of each pair represented what would be expected when the internal sediment recycling of P reached a new equilibrium with the change in external loading. In the second simulations, the full effects of the P-load changes were simulated by also changing the sediment P flux by the specified percentage (a similar percentage to the change in external P loading) and changing the initial concentrations of all forms of P in the lake by the percentage change in near-surface TP concentrations for the west side of the lake expected on the basis of results from the Canfield-Bachmann model (fig. 17). Initial TP concentrations and external P loading for each scenario are given in table 15. For all scenarios, it was assumed that the internal sediment recycling of P (for the second simulations) would change at a similar percentage as the change in external P loading, which is similar to how internal P recycling is dealt with in the Canfield-Bachmann model.

From each simulation, daily minimum DO concentrations between depths of 10 and 20 m (MOM) and DO concentrations at 65 m were obtained. Daily minimum DO concentrations between depths of 10 and 20 m for base conditions (scenario GLM–1) are shown in figure 25 (solid black line), which shows relatively high DO concentrations in fall, winter, and spring, followed by a rapid decline in DO concentrations to about 0–2 mg/L after the lake stratifies, and then a rapid increase in DO concentrations when the thermocline descends to 20-m depth in fall. The summer MOM decreased below 2.2 mg/L in all of the years simulated. The simulated mean June 1–September 15 near-surface TP concentration for the base simulation was 0.018 mg/L, which is similar to the geometric mean concentration measured in the lake during 2014–18 (table 3). For each simulation, the number of years with the MOM decreasing below 5 mg/L and below 2 mg/L at any time during August 15–September 15, near-surface mean spring (April 15 to May 1) and mean summer (June 1– September 15) TP concentrations, and mean summer (July 15–September 15) Chl-*a* concentrations were computed for the last 5 years of the simulation (table 15).

On the basis of results from GLM–AED simulations representing a range of changes in external P loading with coinciding changes in internal sediment recycling of P, the number of years with MOMs less than 5 mg/L is expected to decline with P-load reductions of controllable P sources greater than 25 percent from that estimated for 2014–18 (1,920 kg/yr reduction in external tributary loading: total external loading from 8,440 kg/yr to 6,520 kg/yr). With a 60 percent P-load reduction in controllable sources (4,610 kg/yr reduction in tributary loading: total external loading from 8,440 kg/yr to 3,830 kg/yr), MOMs less than 5 mg/L did not occur in any of the 5 years simulated; therefore, MOMs less than 5 mg/L are rarely expected (fig. 26). The number of years with a MOM less than 5 mg/L did not decline in a linear fashion because of the stepped annual response and possibly because of changes in the relative importance of the various functional phytoplankton groups. Rather than a linear change in phytoplankton concentrations, reductions in P loading initially caused a decline in populations of all functional phytoplankton groups. But with further P-load reductions, the cryptophyte/cyanophyte composition varied, and with even further P-load reductions, populations of all three groups declined. With decreases in P loading greater than 30 percent (2,310 kg/yr reduction in tributary loading: total external loading from 8,440 kg/yr to 6,140 kg/yr), MOMs less than 2 mg/L were rare. With all simulated increases in P loading, metalimnetic DO concentrations of less than 5 mg/L occurred in all years; however, the MOMs became more extreme, and DO concentrations less than 2 mg/L occurred in most years.

Based on the response curve in figure 26, total external P loading to the lake would need to be decreased from 8,440 kg/yr to 4,060 kg/yr (scenario GLM–5) for the MOMs to drop below 5 mg/L in only 1 out of 5 years (or stay above 5 mg/L in more than 75 percent of the years). This equates to a 4,380 kg/yr reduction in tributary loading (57-percent reduction) and a similar 57-percent reduction in internal sediment recycling of P from that measured during 2014–18 (52-percent reduction in total external P loading to the lake). With reductions in controllable P sources greater than 60 percent (4,610 kg reduction in tributary loading: total external loading from 8,440 kg/yr to 3,830 kg/yr, and equates to a 55-percent reduction in total external loading), MOMs less than 5 mg/L are expected to be rare. Daily changes in the MOMs are shown for various simulations in figure 25, which demonstrates that for reductions in controllable sources between 40 percent and 57 percent, the MOMs in 2017 and 2018 decreased to less 5 mg/L, but they only barely dropped below 5 mg/L in these years. During WYs 2017 and 2018, external P loadings to Green Lake were very high (31 and 41 percent above the WYs 2014–18 mean annual loading; table 7) as a result of very high precipitation (table 9).

Because internal sediment recycling of P is currently an important source of P for the following spring, it may take several years after a 57-percent reduction in controllable sources (total mean annual external P loading of 4,060 kg/yr) for DO concentrations in the metalimnion to sufficiently meet the WDNR criterion.

For each scenario, two simulations were performed that treated internal sediment recycling of P differently. In the first simulation of each pair, internal sediment recycling of P was assumed to remain unchanged. One way to estimate how DO concentrations in the metalimnion would be expected to change in the years shortly after external TP reductions would be to examine results from the first simulation of each scenario. If no changes were made to the internal sediment recycling of P, none of the simulated changes in external P loading alone would be sufficient to result in MOMs less than 5 mg/L in less than 2 of the 5 years. Even with a 95-percent reduction in controllable external P sources (reduction of 7,310 kg in tributary loading but no change in internal recycling: total external loading from 8,440 kg/yr to 1,140 kg/yr), MOMs less than 5 mg/L occurred in 2 of the 5 years.

Based on the results of simulations in which the chemical and ecological drivers were eliminated one at a time (fig. 23), high P loading to Green Lake from both external sources and

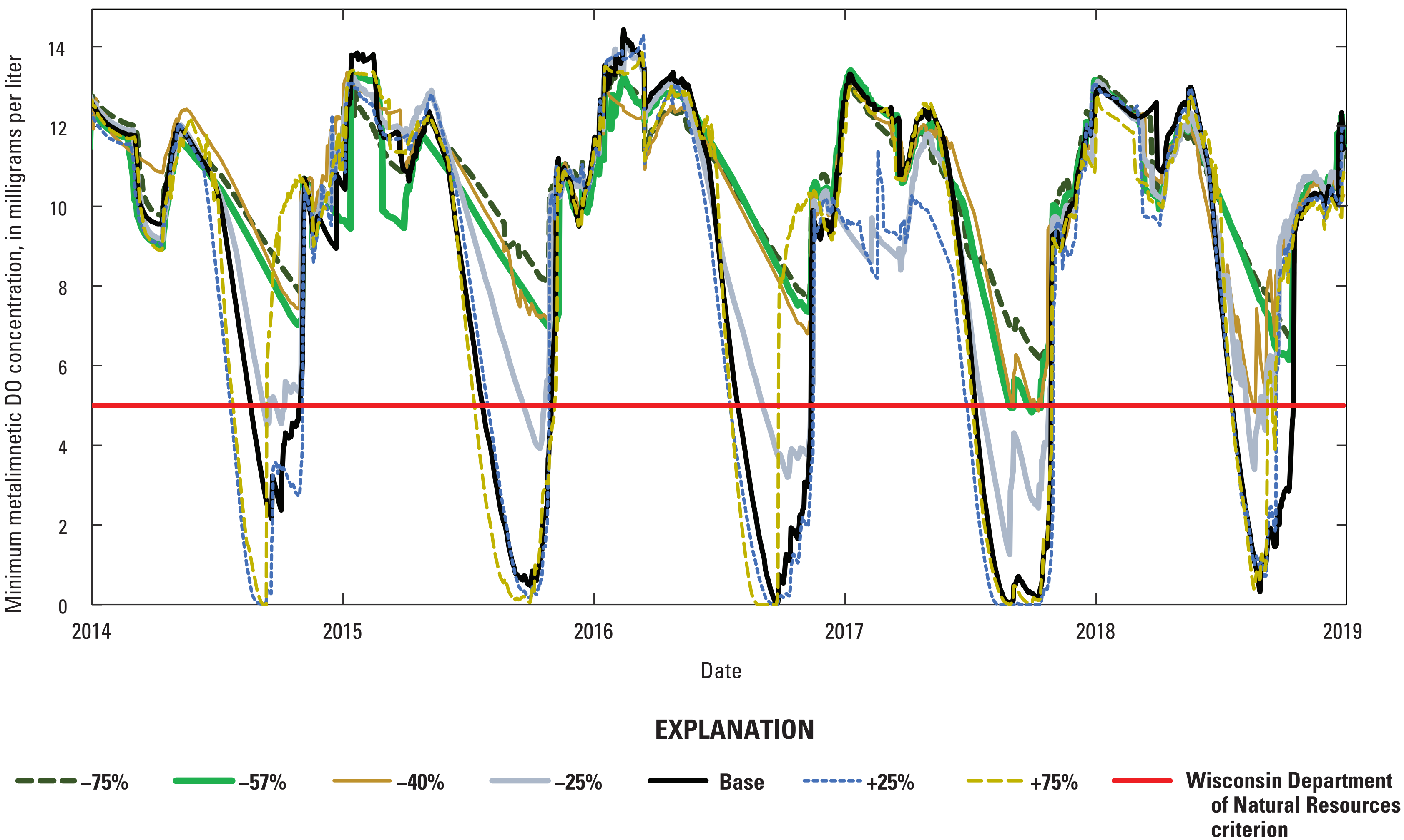

Figure 25. Changes in the minimum dissolved oxygen (DO) concentration between 10-meter and 20-meter depths (metalimnion) in Green Lake, Wisconsin, simulated with the General Lake Model coupled to the Aquatic Ecodynamics modeling library, in response to a subset of the various phosphorus-loading scenarios (changes in controllable phosphorus sources) described in table 15.

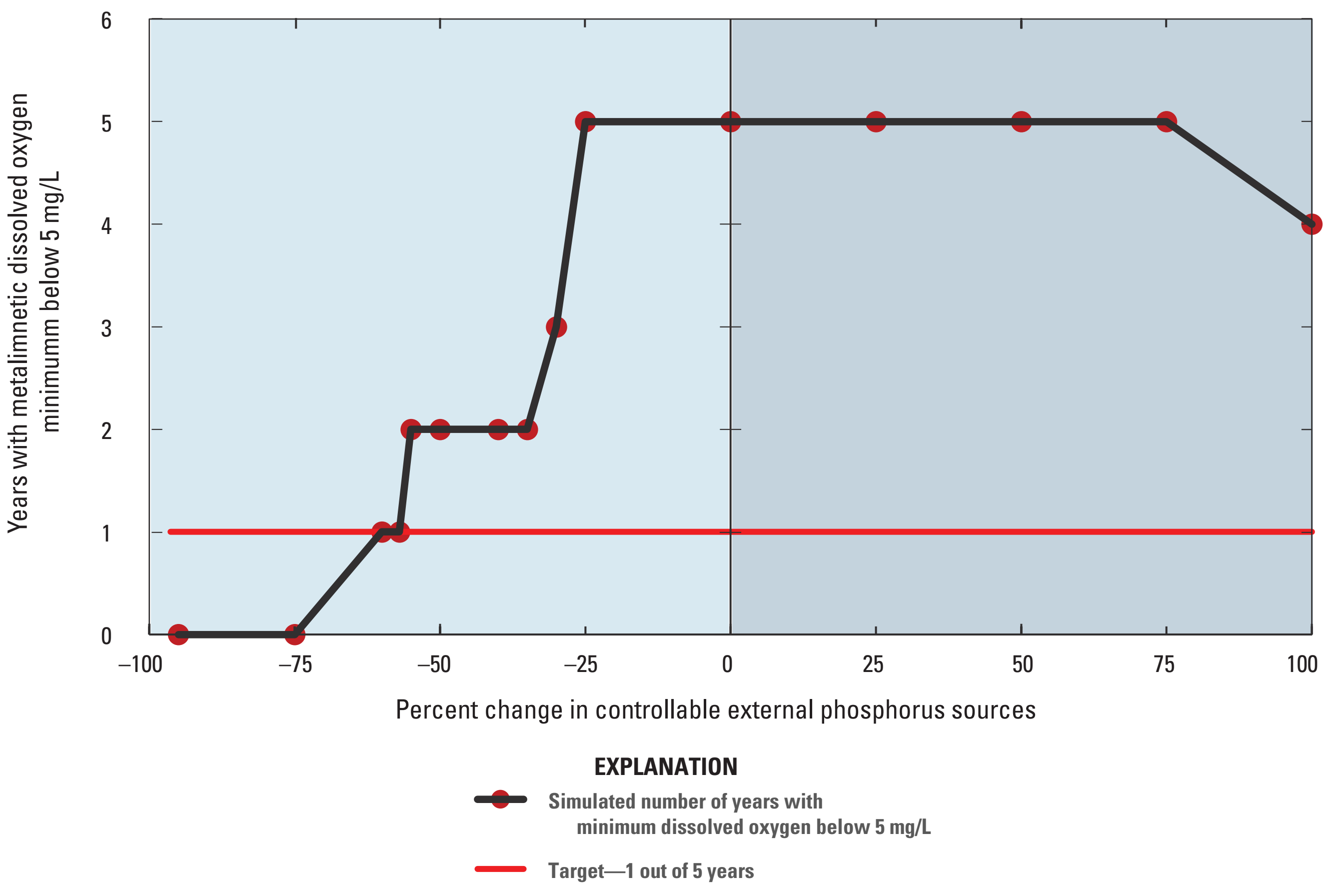

Figure 26. Number of years with metalimnetic dissolved oxygen minimum less than 5 milligrams per liter (mg/L) in the metalimnion of Green Lake, Wisconsin, in response to various changes in controllable external loading, simulated with the General Lake Model coupled to the Aquatic Ecodynamics modeling library.

contributions from internal sediment recycling led to high P concentrations in spring, increased phytoplankton production, increased metabolism and respiration, and ultimately MOMs of less than 5 mg/L. On the basis of the initial TP concentrations in the lake (table 15), spring TP concentrations needed to be reduced to about 0.020 mg/L to eliminate MOMs of less than 5 mg/L. A spring TP concentration of 0.020 mg/L is expected to be achieved only by greatly reducing both P from external sources of P and from internal sediment recycling. However, the only way to reduce P from internal sediment recycling without extensive in-lake management would be to reduce the external P loading that drives the internal sediment recycling.

The GLM–AED model was also used to examine the magnitude of the reduction in external P loading needed to reduce summer TP concentrations to the 0.015 mg/L criterion. Based on GLM–AED results (table 15; figure 17*A*), P loading from controllable P sources would need to be reduced by about 25 percent (1,920 kg/yr reduction in tributary loading: total external loading from 8,440 kg/yr to 6,520 kg/yr) to get mean June 1–September 15 near-surface TP concentrations down to 0.015 mg/L. This reduction is between that found for the east and west sides of the lake by using the empirical models (table 10). However, this 25-percent reduction would be for the mean near-surface TP concentration for the entire lake to meet the TP criterion, not the TP concentration on the east side of the lake.

The GLM–AED model was used to simulate how DO concentrations in Green Lake would be expected to respond to changes in P loading given existing climatic conditions. If air temperatures were to increase, it is expected that epilimnion temperatures would also increase (O'Reilly and others, 2015; Robertson and others, 2018), which could increase productivity and algal decay rates (Jeppesen and others, 2014; Conover and others, 2016), increase the rate of metalimnetic DO consumption, and worsen the MOM. Increases in air temperatures could also increase the length of the summer stratification period (Robertson and others, 2018; Radbourne and others, 2019; Shatwell and others, 2019), which would result in oxygen being consumed over a longer period, which in turn could result in lower DO concentrations in the metalimnion and deep areas of the lake and increase the amount of P released from the sediments of the lake (North and others, 2014). Therefore, in a future with a warmer climate, reductions in controllable P sources of more than the 57 percent (total mean external loading less than 4,060 kg/yr, which does not include inputs from septic systems and waterfowl) found in this study, may be needed to keep DO concentrations in the metalimnion of Green Lake from dropping below 5 mg/L in less than 25 percent of the years.

Conclusions

In the early 1900s, Green Lake in central Wisconsin was an oligotrophic lake based on its near-surface water quality, and dissolved oxygen (DO) concentrations were low only in the deepest part of its hypolimnion. As a result of increased phosphorus (P) inputs from agricultural and urban activities in its watershed, its water quality degraded, and low DO concentrations began to occur in its metalimnion and deep areas of the hypolimnion. Even with reductions in P export from the wastewater treatment plant in Ripon, relatively high near-surface total phosphorus (TP) concentrations and low metalimnetic DO concentrations occurred in the lake. Based on monitoring data, the Wisconsin Department of Natural Resources (WDNR) listed Green Lake as impaired for low metalimnetic DO concentrations because of high TP concentrations in the lake.

The goal of this study, by the U.S. Geological Survey in cooperation with the Green Lake Sanitary District, was to describe current and historical water quality in Green Lake (especially metalimnetic DO concentrations), to explain the causes of high TP concentrations and low metalimnetic DO concentrations, and to estimate the magnitude of P-load reductions needed to improve the water quality of the lake and thus remove it from Wisconsin's Impaired Waters List. Based on data from past monitoring, DO concentrations in the metalimnion in the early 1900s were lower than in most of the hypolimnion, but low metalimnetic DO concentrations have become more extreme; minimum DO concentrations measured in August were below 5 milligrams per liter (mg/L) (the WDNR criterion) in all years since 2008. The minimum August DO concentrations measured in the metalimnion during 2014–18 ranged from 1.3 mg/L to 4.7 mg/L. Based on sampling throughout the lake in 2017 and 2018, minimum metalimnetic DO concentrations were horizontally uniform, and they occurred in conjunction with a high concentration of dissolved organic matter, both of which occurred at the depth with a strong temperature/density gradient. Near-surface geometric mean June–September TP concentrations during 2014–18 on the west (0.016 mg/L) and east (0.020 mg/L) sides of the lake exceeded 0.015 mg/L (the WDNR criterion for TP concentrations).

Based on a detailed inventory of P inputs to Green Lake during water years (WYs) 2014–18, 8,980 kilograms per year (kg/yr) of P was input from external sources, of which tributary inputs contributed 84 percent, followed by precipitation and atmospheric deposition (8 percent), waterfowl (7 percent), and nearshore septic systems (1 percent). In addition, P from internal sediment recycling contributed about 7,040 kg/yr of P to the water column at fall turnover. This estimate of P from internal sediment recycling does not include the deposition of P to the sediments that occurs during other times of the year, which results in a net deposition of P to the sediment on an annual basis. Although internal sediment recycling is only a seasonal source of P, the mass of P that is introduced at fall overturn can increase TP concentrations in the shallow regions of the lake by about 0.022 mg/L, which then persists in the lake until the following spring.

Based on results from empirical models (Canfield-Bachmann TP model, empirical relations based on measured water quality from each side of the lake, Carlson trophic state index relations, and the Hickman chlorophyll-*a* [Chl-*a*] relation), near-surface TP and Chl-*a* concentrations are expected to respond to changes in external P inputs to the lake; changes in TP and Chl-*a* concentrations are expected to be smaller than the changes in P loading on a percentage basis. For any specified percent change in P loading, 50–60 percent of that change in loading is expected in TP concentrations and 30–70 percent of that change is expected for Chl-*a* concentrations. Water clarity, as measured by Secchi disk depth, responded more to decreases in P loading than to increases in loading; however, the estimated response varied greatly among models. Based on these relations, mean total external P loading to Green Lake would need to be decreased from 8,980 kg/yr (measured during WYs 2014–18) to 5,460 kg/yr for the geometric mean June–September TP concentration in the east side of the lake to reach the 0.015-mg/L TP criterion for the lake established by the WDNR. This is a 46-percent reduction in potentially controllable external P sources, which equates to a 3,520-kg/yr reduction in external P loading. Potentially controllable external P sources include P from all external sources except precipitation, atmospheric deposition, and waterfowl. For this improvement in water quality to occur, it was assumed that there will be a subsequent reduction in P from internal sediment recycling in the future as the lake sediments and water column come to a new equilibrium after the changes in external loading. The time for Green Lake to reach this new equilibrium was not examined in this study. Total external P loading was estimated to need to be decreased from 8,980 kg/yr to 7,680 kg/yr for near-surface TP concentrations in the west side of the lake to reach 0.015 mg/L, which equates to a 17-percent reduction in controllable external P sources. Total external P loading would need to be decreased to from 8,980 kg/yr to 3,870 kg/yr (west side)–5,320 kg/yr (east side) for the lake to be classified as oligotrophic, with a geometric mean June–September near-surface TP concentration of 0.012 mg/L. The required reduction in external P loading to Green Lake from 8,980 kg/yr (measured in WYs 2014–18) to 5,460 kg/yr (a 46-percent reduction in controllable external sources) for the mean June–September near-surface TP concentration to reach 0.015 mg/L is a larger reduction than that found in the Upper Fox and Wolf Rivers Total Maximum Daily Load (TMDL) study. In the TMDL study, the Cadmus Group estimated that external P loading to the lake would need to be reduced from 5,360 kg/yr to 4,230 kg/yr (a 1,130-kg/yr or 21-percent reduction from that measured in WYs 2009–13) for the mean near-surface June–September TP concentration to decrease to 0.015 mg/L. This difference in loading capacities was due to the Cadmus Group study estimating the P-load reduction needed for the mean lakewide (average of east and west sides of the lake) summer TP concentration to reach 0.015 mg/L rather than for the east side of the lake that had higher TP concentrations, estimating the total loading to the lake during drier years, and possibly underestimating the total P load to the lake.

After calibration, the General Lake Model coupled to the Aquatic Ecodynamics modeling library (GLM–AED) hydrodynamic water-quality model accurately reproduced the physical (water level and water temperature), biogeochemical (phosphorus, nitrogen, and DO concentrations), and ecological (algal community dynamics) conditions measured in the lake during 2014–18, particularly the spatial and temporal changes in metalimnetic DO concentrations. Results of model simulations that eliminated specific processes one at a time indicated that the metalimnetic DO minimum (MOM) was driven by P loading to the lake from both external sources and internal sediment recycling processes that cause high P concentrations in the lake during spring and early summer, which lead to high phytoplankton production (diatoms, cryptophytes, and cyanophytes) and high metabolism and respiration that consumes DO in the upper, warmer areas of the metalimnion.

GLM–AED model results also indicated that settling of particulate organic material during summer may be slowed by the relatively cold, dense, and viscous water in the metalimnion and increase oxygen consumption. Based on empirical evidence comparing metalimnetic DO minimum intensity with various meteorological, hydrologic, water quality, and in-lake physical factors, metalimnetic DO concentrations were lower during summers when metalimnetic water temperatures were higher, near-surface Chl-*a* and TP concentrations were higher, and Secchi disk depths were lower than in other years. Results of GLM–AED simulations of a wide range of P inputs to the lake indicated that external P loads would need to be decreased from 8,440 kg/yr to 4,060–5,370 kg/yr to reduce the occurrence of metalimnetic DO concentrations of less than 5 mg/L in over 75 percent of the years. This reduction equates to a 40–57-percent reduction in controllable external P sources: a reduction of 3,080–4,380 kg/yr from that measured in 2014–18. With more than a 57-percent reduction in external P loading, metalimnetic DO concentrations of less than 5 mg/L did not occur in any of the years simulated and, therefore, would be expected to become rare.

Large reductions in external P loading are expected to have an almost immediate effect on the water quality in Green Lake; however, because internal sediment recycling is an important source of P for the following spring, it may take several years for the full effects of the external P-load reduction to be observed in near-surface TP concentrations and metalimnetic DO concentrations. The time delay to achieve water-quality goals because of the effects of internal sediment recycling could be the subject of additional study.

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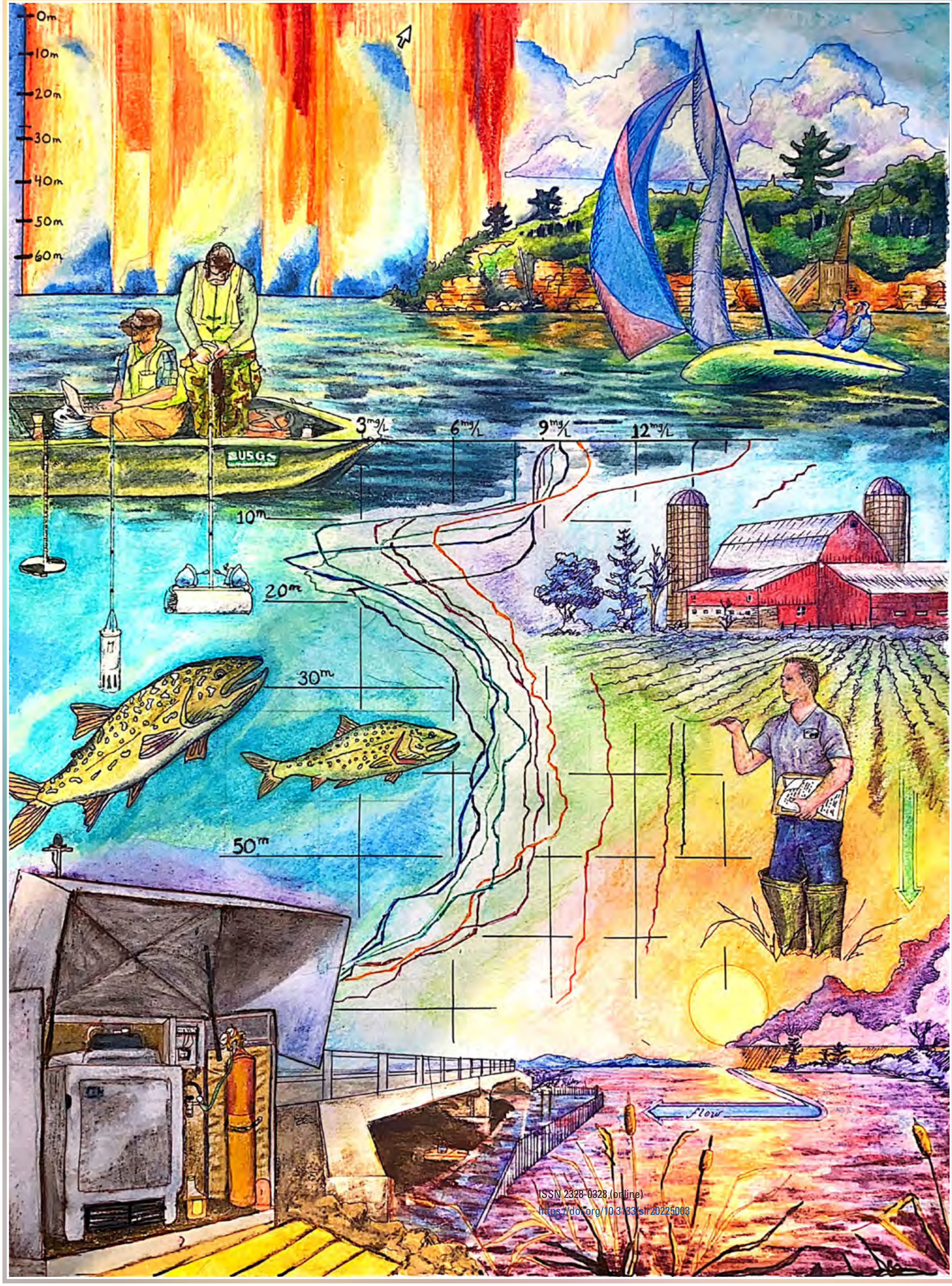

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