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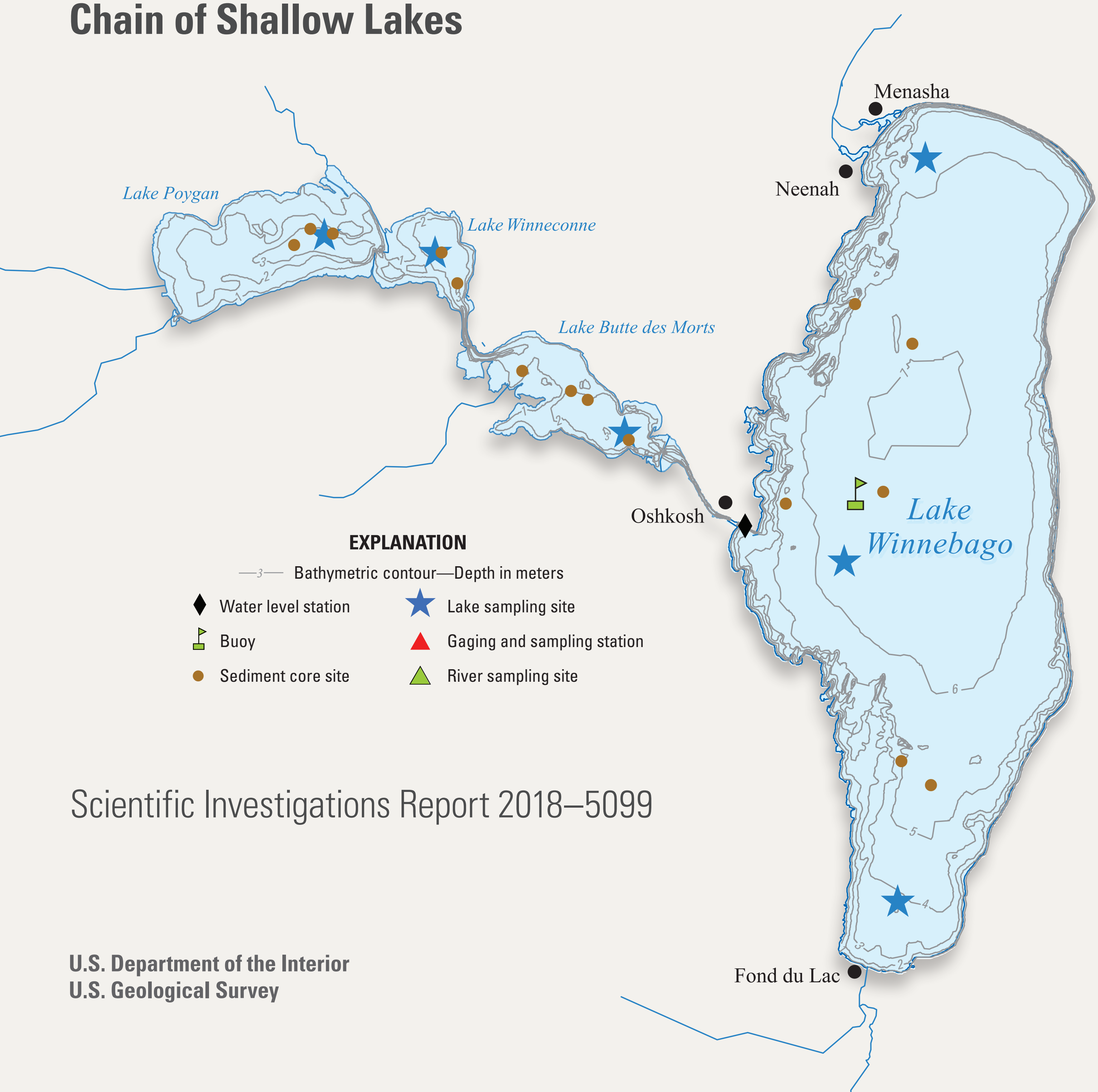
Water-Quality Response to Changes in Phosphorus Loading of the Winnebago Pool Lakes, Wisconsin, with Special Emphasis on the Effects of Internal Loading in a Chain of Shallow Lakes



Scientific Investigations Report 2018–5099

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

Cover figure. Winnebago Pool Lakes, Wisconsin, and their watersheds, with sampling locations identified.

Water-Quality Response to Changes in Phosphorus Loading of the Winnebago Pool Lakes, Wisconsin, with Special Emphasis on the Effects of Internal Loading in a Chain of Shallow Lakes

By Dale M. Robertson, Benjamin J. Siebers, Matthew W. Diebel, and Andrew J. Somor

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U.S. Geological Survey
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Conversion Factors

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
kilometer (km)	0.6214	mile (mi)
micrometer (µm)	0.00003937	inch (in.)
meter (m)	1.094	yard (yd)
	Area	
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
cubic meter (m^3)	264.2	gallon (gal)
cubic meter (m^3)	35.31	cubic foot (ft^3)
	Flow rate	
cubic meter per year (m^3/yr)	0.000811	acre-foot per year (acre-ft/yr)
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)
	Mass	
kilogram (kg)	2.205	pound avoirdupois (lb)

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 Datum of 1983 (NAD 83).

Supplemental Information

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

Loading or release rates are given in milligrams per square meter per day (mg/m^2/d) or milligrams per square meter per year (mg/m^2/yr).

Yields are given in meters (m) for flow and in kilograms per square kilometer (kg/km^2) for nutrient constituents.

Deposition rates are given in kilograms per hectare per year (kg/ha/yr).

Inputs are given in kilograms per year (kg/yr).

Sediment volume is given in cubic meters per square meter (m^3/m^2), bulk density is given in grams per cubic meter (g/m^3), and mass is given in grams per square meter (g/m^2) or milligrams per square meter (mg/m^2).

Note to USGS users: Use of hectare (ha) as an alternative name for square hectometer (hm^2) is restricted to the measurement of small land or water areas. Use of liter (L) as a special name for cubic decimeter (dm^3) is restricted to the measurement of liquids and gases.

Abbreviations

Chl-*a*	chlorophyll-*a*
DO	dissolved oxygen
DP	dissolved phosphorus
EGRET	Exploration and Graphics for RivEr Trends
NO_{2+3}	nitrate plus nitrite nitrogen as nitrogen
NWIS	National Water Information System
P	phosphorus
RMSE	root mean square error
SD	Secchi depth
SWAT	Soil and Water Assessment Tool
SWIMS	Surface Water Integrated Monitoring System database
TKN	total Kjeldahl nitrogen
TMDL	Total Maximum Daily Load
TN	total nitrogen
TP	total phosphorus
TSI	trophic state index
WDNR	Wisconsin Department of Natural Resources
WRTDS	Weighted Regressions on Time, Discharge, and Season program
WSLH	Wisconsin State Laboratory of Hygiene
WQ	water quality
WY	water year (the period from October 1 to September 30 of the specified year)
USDA–ARS	U.S. Department of Agriculture Agricultural Research Service
USGS	U.S. Geological Survey
$\bar{z}$	mean lake depth

Water-Quality Response to Changes in Phosphorus Loading of the Winnebago Pool Lakes, Wisconsin, with Special Emphasis on the Effects of Internal Loading in a Chain of Shallow Lakes

By Dale M. Robertson,[1] Benjamin J. Siebers,[1] Matthew W. Diebel,[2] and Andrew J. Somor[3]

Abstract

The Winnebago Pool is a chain of four shallow lakes (Lake Poygan, Lake Winneconne, Lake Butte des Morts, and Lake Winnebago) that are fed primarily by the Fox and Wolf Rivers, two large agriculturally dominated rivers in Wisconsin, United States. Because the lakes have received extensive phosphorus inputs from their watershed, they have become highly eutrophic with much phosphorus in the water column as well as trapped in their sediments. Each of the four Winnebago Pool Lakes has been included on the Wisconsin Department of Natural Resources impaired waters list because of their high total phosphorus concentrations, water-quality use restrictions, and excess algal growth. The study described in this report is part of a Total Maximum Daily Load investigation to determine what actions are needed to improve the water quality (trophic status) of these lakes and thus be able to be removed from the impaired waters list and restore their designated uses. As part of this study, data were collected to describe the existing water quality of the lakes, detailed phosphorus budgets were developed for each of the lakes to describe the different sources of the phosphorus, and two eutrophication models (BATHTUB and Jensen models) were used to determine how much of the phosphorus being input to the lakes needs to be reduced for the lakes to be removed from the impaired waters list and restore their designated uses.

In-lake water-quality data indicated that each of the lakes had extensive vertical mixing that resulted in their water quality deteriorating throughout summer. Each of the lakes had mean summer total phosphorus concentrations exceeding 0.088 milligram per liter (mg/L), well above the 0.040 mg/L criterion for the lakes. Detailed phosphorus budgets for the lakes indicated that the primary sources of phosphorus were from their tributaries (for the most upstream lake in the Winnebago Pool–Lake Poygan) or from a combination of input from the upstream lakes and phosphorus release from the bottom sediment when only the summer months were considered (for the other three lakes).

Model simulations with the BATHTUB and Jensen models indicated that (1) the lakes should have almost linear response in their total phosphorus concentrations to changes in their phosphorus inputs; (2) phosphorus inputs need to be reduced by about 60 percent to the Upper Pool Lakes and 69–73 percent to Lake Winnebago to reduce their mean summer total phosphorus concentrations to 0.040 mg/L; and (3) if all the anthropogenic phosphorus inputs to the lakes could be eliminated, their best possible mean summer total phosphorus concentrations should decrease to about 0.022–0.028 mg/L in the Upper Pool Lakes and to 0.032–0.033 mg/L in Lake Winnebago. The effects of any reduction in phosphorus loading will take many years (50 to more than 75 years) to be fully realized in lake water quality because of phosphorus release from the lake sediments. The effects of nutrient reductions in the watershed of a chain of lakes, such as the Winnebago Pool, gradually cascades down the chain, which has beneficial and detrimental effects. Any action made in the watershed of upstream lakes to reduce phosphorus inputs should improve the water quality of all downstream lakes; however, the upstream lakes delay the response in the downstream lakes, especially in lakes where internal phosphorus loading is important.

Introduction

The Winnebago Pool (also referred to as the "Pool Lakes") is a chain or series of four shallow lakes (Lake Poygan, Lake Winneconne, Lake Butte des Morts, and Lake Winnebago) primarily fed by the Fox and Wolf Rivers, two large agriculturally dominated rivers in Wisconsin, United States (fig. 1). The water quality (WQ) in these lakes, like many other lakes in agricultural areas, has declined (enhanced

[1]U.S. Geological Survey.

[2]Wisconsin Department of Natural Resources.

[3]The Cadmus Group.

eutrophication) because of excessive phosphorus (P) input from their watersheds (Wisconsin Department of Natural Resources, 2004; Schindler and others, 2016). Because of high total phosphorus concentrations (TP), all four Pool Lakes, which are classified as "shallow lowland drainage" lakes, were officially listed as "impaired" (mean June through September TP greater than 0.040 milligram per liter [mg/L]; Wisconsin Department of Natural Resources, 2016, 2017a), and a Total Maximum Daily Load investigation is being done to determine what actions are needed to improve their WQ and restore their designated uses. Designated uses for the Winnebago Pool, as defined by the Wisconsin Department of Natural Resources (WDNR), are for fish and aquatic life, recreational use, public health and welfare, and wildlife. The lake indicators of high TP are eutrophication, water-quality use restrictions, and excess algal growth. The Total Maximum Daily Load calculation is complicated by the short-term (annual) and long-term (multi-year) effects of P released from the bottom sediments in these shallow lakes and the effects of nutrient reductions cascading through the chain of lakes.

Many factors affect a lake's trophic state (including TP and chlorophyll-*a* [Chl-*a*] concentrations) and water clarity (Secchi depth [SD]). Usually, the main factor is external nutrient input from the watershed that varies with hydrology, geology, and long-term anthropogenic changes in the watershed. The effects of nutrient inputs (primarily P for most lakes; Schindler and others, 2016) are sufficiently understood such that many eutrophication models have been developed to estimate in-lake TP, Chl-*a*, and SD from their morphometry and external water and P loading (mass input during a specified period) (Cooke and others, 1993). Most of these eutrophication models describe the response in a single lake; only a few models describe the response in a chain of lakes. The BATHTUB model (Walker, 1996) has transport algorithms and can simulate changes in complex multibasin lakes or a chain of lakes. Other lake-specific empirical models (such as Carpenter and Lathrop, 2014) and complex mechanistic models (such as Epstein and others, 2013) have been developed for chains of lakes.

External P loading drives in-lake TP and productivity, and then the P is either exported out of the lake or deposited in the sediments. Not all P remains in the sediments; some is released back into the water column, referred to as "internal P loading." Some of the P in the bottom sediments can be released back into the water column when the water is aerobic (contains dissolved oxygen [DO]); however, when deep water of a productive lake (a lake with abundant plants or algae) becomes isolated from surface mixing because of thermal stratification, the sediment and the bottom water often become anaerobic or devoid of DO and the rate of internal P loading often increases dramatically (Mortimer, 1941). Internal P loading increases with increasing length of stratification (Kling and others, 2003), increasing water temperatures that promote microbial decomposition (James and Barko, 2004; Jensen and others, 2006), and increasing amounts of readily available P in the sediment (Rydin and Welch, 1999).

Productivity of shallow lakes, however, may respond differently than deep lakes because of internal P loading. Shallow lakes, lakes with a maximum depth less than 6 meters (m), typically experience deep mixing events throughout summer and are, therefore, referred to as polymictic lakes (Osgood, 1988). Osgood (1988) described the functional aspects of mixing from wind forcing based on a lake's mean depth ($\bar{z}$, in meters) and surface area (A, in square kilometers) in terms of the Osgood Index defined as $\bar{z} \div A^{0.5}$. Lakes with Osgood Index values less than 4 tend to be polymictic. Typically, P released from deep sediments in deep dimictic lakes (that is, lakes that remain stratified during the summer months) is retained in the hypolimnion (that is, the cool, lower layer of water in the lake) during summer and primarily released to near the surface of the water column during spring and fall turnover. Therefore, near-surface TP in dimictic lakes usually remains stable or decreases as summer progresses (Welch and Cooke, 1995). In shallow polymictic lakes, microzones of anaerobic activity can develop at or just below the sediment-water interface during the short periods of stratification. Then, frequent deep mixing events can cause an upward transport of the P, resulting in warmer bottom water and near-surface TP and productivity increasing throughout summer (Welch and Cooke, 1995). Robertson and others (2016) demonstrated differences in bottom temperatures and in surface TP and Chl-*a* between deep and shallow lakes by comparing the WQ in morphologically different basins in a multibasin lake in northern Wisconsin. In addition to mixing of the water column, strong winds can cause sediment resuspension that can increase sediment P release in shallow lakes (Søndergaard and others, 1992; Cyr and others, 2009; Huang and others, 2016). Søndergaard and others (1992) estimated that resuspension could enhance the rate of soluble P release from the sediment by 20–30 times compared to undisturbed sediment. In addition, aquatic animals can directly affect the cycling of P and increase internal P loading in lakes through feeding (which stirs up the sediments) and excretion (Andersson and others, 1988; Fischer and others, 2013). Based on enclosure studies, common carp (*Cyprinus carpio*) contribute as much as 4.4 percent (Lamarra, 1975) of their body weight as P per year into the water column. Fish feeding activity, which often uproots vegetation, also can create more flocculent sediments that are more prone to suspension from wind (Lin and Wu, 2013), thus increasing internal P release into a lake.

Lakes with high internal P loading may respond much slower to reductions in external P loading than other lakes (Jeppesen and others, 2005, Jensen and others, 2006, Meals and others, 2010). Even 20 years after large external P reductions, internal P loading continued to affect the trophic state of Shagawa Lake in Minnesota (Seo and Canale, 1999). Most eutrophication models do not simulate the transient effects of changes in external P loading; however, Jensen and others (2006) developed a simple empirical model that describes the continuous changes in in-lake TP and in-sediment P after external load reductions, which is a function of external P loading, initial available P in the sediment, hydraulic residence time, and water temperature.

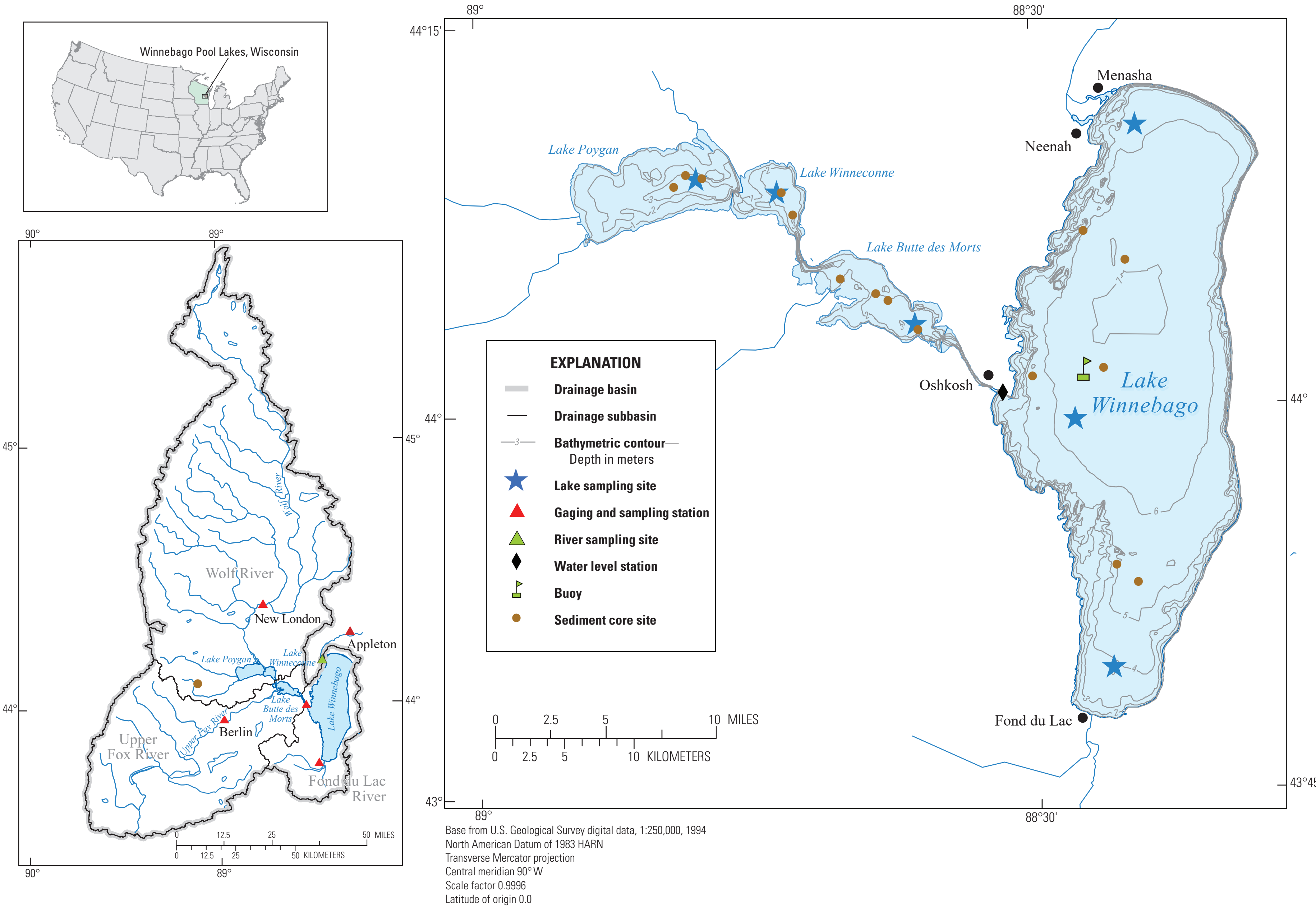


Figure 1. Winnebago Pool Lakes and their watersheds, with sampling locations identified, Wisconsin. Sampling sites are described in Table 2. Lake contours are in meters.

Typically, lakes are considered to be singular entities in the environment (Forbes, 1887) and are modeled that way. In a chain of lakes, such as the Winnebago Pool, however, the lakes interact; changes upstream in the watershed cascade down the chain. Lathrop and Carpenter (2014) and Carpenter and Lathrop (2014) modeled the effects of nutrient reduction strategies implemented in various parts of the Yahara watershed, Wisconsin, United States, cascading down through a chain of lakes. In a chain of lakes, actions done upstream in the watershed improve the WQ of all the downstream lakes; however, the response to external P load reductions also can be delayed in the downstream lakes, especially where internal loading is an important part of their P budget (Jeppesen and others, 2005; Jensen and others, 2006; Meals and others 2010).

Purpose and Scope

The purpose of this study was to (1) document the present (2009–11) WQ of each of the Winnebago Pool Lakes, (2) describe their water and P budgets including quantifying internal P loading using various approaches, (3) determine how the WQ of each lake should respond to changes in P loading (steady state and transient changes) using the BATHTUB and Jensen eutrophication models, (4) determine how much of the present P inputs would have to be reduced to attain nonimpairment and thus be able to delist each of the Pool Lakes for high TP, and (5) estimate the WQ in the lakes if all anthropogenic P inputs were removed (best attainable WQ). Because the Winnebago Pool is a chain of shallow lakes where internal P loading is expected to be important, specific approaches were used to determine how the trophic response to changes in P loading cascades down the chain and how internal P loading affects the steady state and transient responses of these lakes.

Methods

Study Area

The Winnebago Pool consists of four shallow lakes: Lake Poygan (58 square kilometers [km^2]; mean depth [$\bar{z}$]=1.8 m; the most upstream lake), Lake Winneconne (19 km^2; $\bar{z}$=1.6 m), Lake Butte des Morts (35 km^2; $\bar{z}$=1.6 m), and Lake Winnebago (532 km^2; $\bar{z}$=4.5 m; the most downstream lake) (fig. 1; table 1). Lakes Butte des Morts and Winnebago are connected by a 4.6-kilometer (km) widening of the Fox River. The Winnebago Pool has three main tributaries: Wolf River that drains into Lake Poygan, Fox River that drains into Lake Butte des Morts, and Fond du Lac River that drains into Lake Winnebago. Collectively, the Pool Lakes drain more than 14,000 km^2 of central Wisconsin before discharging into the lower Fox River, which is one of the largest tributaries to Lake Michigan. The watersheds of all the lakes are dominated by agriculture (table 1), but the Wolf River Basin upstream from Lake Poygan has more forested land than other areas, and the land immediately around Lake Winnebago has more urban development than other areas, including the cities of Oshkosh and Fond du Lac (fig. 1).

All four Pool Lakes were included on the WDNR 2016 impaired waters list for TP and suspended sediment (Wisconsin Department of Natural Resources, 2016). All four lakes are classified as “shallow lowland drainage” lakes with a TP criterion defined as mean TP concentration between June 1 and September 15 to not exceed 0.040 mg/L (Wisconsin Department of Natural Resources, 2017a).

Data Collection

Lake Water Quality

As part of this study, historical WQ (TP, Chl-*a*, and SD) data for each of the Pool Lakes were assembled, but only historical data for Lake Winnebago are presented in this report because of the lack of consistent data available for the other lakes. Because of their shallow depths, usually only surface WQ data are collected. Each of the Pool Lakes was originally sampled monthly as part of a University of Wisconsin-Oshkosh study from June 1976 to May 1977 (Sloey and Spangler, 1977). WQ in Winnebago was then consistently measured from 1989 to 2015 as part of the WDNR Long-Term Trend monitoring program; however, only limited sampling was done in the other Pool Lakes. The lakes also were sporadically sampled by local citizens as part of the WDNR Citizen Lake Monitoring Program (Wisconsin Department of Natural Resources, 2018); however, usually only SDs were measured. These data were obtained from the WDNR Surface Water Integrated Monitoring System (SWIMS) database at https://dnr.wi.gov/topic/surfacewater/swims/ (Wisconsin Department of Natural Resources, 2017b). The north end of Lake Winnebago also was sampled as part of limnological studies by Lawrence University (B. DeStasio, Professor, Lawrence University, unpub. data). The historical WQ data for Lake Winnebago are available at https://doi.org/10.5066/P9Y8BE4H (Robertson and Kennedy, 2018).

As part of this study, each of the Pool Lakes was sampled monthly by the WDNR during May through September 2009–11 near their deepest locations, except Lake Winnebago, which was sampled at three locations (fig. 1). During each visit, consistent protocols, similar to the WDNR Long-term Trend Monitoring Program (Wickman and Herman, 2005) protocols, were used that involved collecting profiles of water temperature, DO, specific conductance, and pH with a multiparameter meter, and collecting SD with a standard 20-centimeter (cm) diameter black and white Secchi disk. Near-surface grab samples were collected and analyzed for TP, dissolved phosphorus (DP), dissolved nitrite plus nitrate (NO_{2+3}), total Kjeldahl nitrogen (TKN) and Chl-*a* by the Wisconsin State Laboratory of Hygiene (WSLH; Madison, Wis.) in accordance with standard analytical procedures (Wisconsin State Laboratory of Hygiene, Environmental Science Section, 1993). All WQ data were obtained from the WDNR SWIMS database at https://dnr.wi.gov/topic/surfacewater/swims/ (Wisconsin Department of Natural Resources, 2017b).

Table 1. Morphometric characteristics of the Winnebago Pool Lakes, on the basis of 2014 Fishing Hotspots Lake Winnebago Map (https://www.fishinghotspots.com/e1/pc/ viewPrd.asp?idproduct=311) and 2007 Mapping Specialists Poygan, Winneconne, Butte des Morts Map (https://www.mappingspecialists.com/ store/lake-poygan-lake-winneconne-lake-butte-des-morts-fold-map/).

[m, meter; km, kilometer; km^2, square kilometer; km^3, cubic kilometer]

Lake or river	Primary water-quality station number	Maximum depth (m)	Mean depth (m)	Fetch (km)	Area (km^2)	Volume (km^3)	Residence time (days[a])	Osgood index	Lake drainage area (km^2)	Station drainage area (km^2)	Agriculture in the total drainage area (percent)[b]
Poygan	713121	3.35	1.78	12.4	58.0	0.103	13.2	0.23	9,455	5,794	35.5
Winneconne	713282	3.96	1.61	7.6	19.1	0.031	4.07	0.37	(Included with Lake Poygan.)		
Butte Des Morts	713252	3.66	1.56	11.3	35.4	0.055	3.7	0.26	13,510	9,200	40.3
Fox River—Connecting	713056	8.00	2.24	4.6	2.1	0.058	0.4	1.56	13,560	13,560	40.4
Winnebago[c]—		6.40	4.54	45.3	532.3	2.415	187.5	0.20	14,290	14,000	42.2
Center site	713245[d]										
North site	713243[e]										
South site	713244[f]										

[a]Based on total annual water inputs from water years 2009–11.

[b]Based on data from the National Land Cover Database (Fry and others, 2006).

[c]Summary statistics for Winnebago were computed for the entire lake, but data were collected at three stations.

[d]For long-term analysis the following station numbers were also included: 713302, 713338, 713339, 713342, 713355, and 714005.

[e]For long-term analysis the following station numbers were also included: 713301 and 713329.

[f]For long-term analysis the following station numbers were also included: 203101 and 203110.

All historical average monthly and seasonal WQ data for Lake Winnebago were obtained by averaging available data from the north, center, and south parts of the lake (station numbers for each location are given in table 1). To reduce potential biases from nonuniform sampling through time, individual WQ values were averaged monthly and then summer-average (June–September) WQ was computed from the monthly values; months with missing data were estimated by using the long-term (1973–2015) average for the missing month.

One method of classifying the WQ of a lake is with trophic state index (TSI) values based on near-surface concentrations of TP and Chl-*a*, and SD, as developed by Carlson (1977), and total nitrogen (TN), as developed by Kratzer and Brezonik (1981). The indices were developed to place these four characteristics on similar scales to allow trophic comparisons among constituents. TSI values based on TP (in milligrams per liter; TSI_{TP}), Chl-*a* (in micrograms per liter; $TSI_{Chl\text{-}a}$), SD (in meters; TSI_{SD}), and TN (in milligrams per liter; TSI_{TN}) were computed for each sampling date using equations 1–4. Individual TSI values were averaged monthly, and the monthly average values were then used to compute summer-average (May through September) TSI values:

$$TSI_{TP}\ 4.15 + 14.42 \times (\ln [TP \times 1{,}000]) \tag{1}$$

$$TSI_{Chl\text{-}a} = 30.6 + 9.81 \times (\ln Chl\text{-}a) \tag{2}$$

$$TSI_{SD} = 60 - 14.41 \times (\ln SD) \tag{3}$$

$$TSI_{TN} = 54.45 + 14.43 \times (\ln TN) \tag{4}$$

Oligotrophic lakes have TSI values less than 40; have a limited supply of nutrients; typically have low TP, low algal populations, and good water clarity; and contain DO throughout the year in their deepest zones (Carlson and Simpson, 1996). Mesotrophic lakes have TSI values from 40 to 50; have a moderate supply of nutrients; tend to produce moderate algal blooms and moderate clarity; and occasionally have DO depletions in the deepest zones of the lake. Eutrophic lakes have TSI values greater than 50; have a large supply of nutrients; have severe WQ problems, such as frequent seasonal algal blooms and poor water clarity; and commonly have DO depletions throughout the deeper zones of the lake. Eutrophic lakes with TSI values greater than 70 often are classified further as hypereutrophic, and they typically have even more severe WQ problems, including frequent extensive algal blooms.

Lake Morphometry

Morphometric characteristics of Lake Winnebago were based on the 2014 Fishing Hotspots Lake Winnebago Map (https://www.fishinghotspots.com/e1/pc/viewPrd.asp?idproduct=311). Morphometric characteristics for the three Upper Pool Lakes (Lake Poygan, Lake Winneconne, and Lake Butte des Morts) were based on the 2007 Mapping Specialists Poygan, Winneconne, Butte des Morts Map (https://www.mappingspecialists.com/store/lake-poygan-lake-winneconne-lake-butte-des-morts-fold-map/). The surface areas and associated contour lines for the lakes were reevaluated and adjusted, if needed, on the basis of an aerial image obtained from the 2005 National Agricultural Imagery Program (U.S. Department of Agriculture, 2006). The area and volume of the lake at specific depths were computed from the depth contours of the map using a geographic information system (GIS).

Lake Stage

Daily water levels of Lake Winnebago have been measured near to where the Fox River enters Lake Winnebago (USGS station 04082500; fig. 1) since 1882. The U.S. Geological Survey (USGS) has monitored water levels at this location with a stage gage since 1938. Data from October 2008 to September 2014 were used in this study. All water-level data were obtained from the National Water Information System (NWIS) database at https://doi.org/10.5066/F7P55KJN (U.S. Geological Survey, 2018).

River and Tributary Monitoring and Load Computation

Five river sites (fig. 1; table 2) were instrumented to monitor flow continuously (at 5- to 15-minute intervals) and to estimate P loading at selected locations in the watershed or downstream from the lakes. At three sites (Wolf River at New London, Fox River at Berlin, and Fond du Lac River at Fond du Lac), water levels were measured and used to determine flow using standard stage-discharge relations (Rantz and others, 1982). Flow at the Fox River at Oshkosh (upstream from Lake Winnebago) and Fox River at Appleton (downstream from Lake Winnebago and used to estimate flow out of the lake at Neenah Menasha) were estimated 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 has been in operation since at least 1986, except the Fond du Lac River site, which was operated from 2007 to 2011. From the 5- to 15-minute flow data, daily average flows were computed. All flow data were obtained from the National Water Information System (NWIS) database at https://doi.org/10.5066/F7P55KJN (U.S. Geological Survey, 2018).

Table 2. River and tributary monitoring sites for the Winnebago Pool Lakes and their drainage basins, Wisconsin.

[km^2, square kilometer; WDNR ID, Wisconsin Department of Natural Resources station identification number; Wis., Wisconsin; WRTDS, Weighted Regressions on Time, Discharge, and Season program; GCLAS, Graphical Constituent Loading Analysis System]

USGS station name	USGS station number	Drainage area (km^2)	Streamflow period of record	Water-quality station WDNR ID	Used water-quality data	Water-quality sampling frequency	Increased frequency during study	Load computation technique	Percentage of agriculture in basin
Wolf River at New London, Wis.	04079000	6,123	1913–2018	693035	1977–2014	Monthly	About 3 times per month in 2010–12	WRTDS	31.4
Fox River at Berlin, Wis.	04073500	3,427	1898–2018	243020	1977–2014	Monthly	About 3 times per month in 2008–11	WRTDS	47.2
Fox River at Oshkosh, Wis.	04082400	14,070	1991–2018	713056	2009–14	Monthly	No extra sampling	Linear interpolation	80.1
Fond du Lac River at Fond du Lac, Wis.	04083545	435	2007–11	04083545	2008–11	About 40 times per year	No extra sampling	GCLAS	40.4
Fox River at Neenah Menasha,[a] Wis.	04084445[b]	15,570	1986–2018	713002	2009–14	Monthly	About 4 times per month in 2011–12	Linear interpolation	42.2

[a]Neenah and Menasha are technically two cities on opposite sides of the river

[b]Streamflow from Fox River at Appleton was used for this site, with with adjustment factors described in the text.

Samples for WQ analyses (TP) for all river sites, except the Fond du Lac River, were collected on a routine monthly basis using grab or equal-width increment methods (Edwards and Glysson, 1999) by the USGS or WDNR. As part of this study, additional samples were collected about weekly during a few years at all sites, except the Fox River at Oshkosh site, to better describe changes in concentrations. Samples at the small Fond du Lac River site with quickly changing flows were collected throughout high-flow events using an automated sampler and augmented by routine base-flow samples. All chemical analyses were done by the WSLH in accordance with standard analytical procedures (Wisconsin State Laboratory of Hygiene, Environmental Sciences Section, 1993). All river TP data are available from the WDNR SWIMS database at https://dnr.wi.gov/topic/surfacewater/swims/ (Wisconsin Department of Natural Resources, 2017b) or NWIS database at https://doi.org/10.5066/F7P55KJN (U.S. Geological Survey, 2018).

Because TP concentrations at the various riverine sites had different responses to changing flow conditions, different approaches were used to compute TP loads at each site. Daily TP loads for the Fox River at Oshkosh and Fox River at Neenah Menasha for water years (WY) 2009–14 were computed using a simple linear interpolation technique to estimate daily TP concentrations between measured concentrations because changes in concentrations downstream from the lakes varied slowly and had little relation to streamflow. Daily loads were then computed as the product of daily concentration and daily discharge. Daily TP loads at the Fond du Lac River site, with quickly changing flows, were computed for WY2009–11 using the Graphical Constituent Loading Analysis System (GCLAS; Koltun and others, 2006). GCLAS was developed to estimate loads of WQ constituents from continuous measurements of streamflow and instantaneous constituent concentrations. Generally, concentrations are linearly extrapolated between measurements except at the beginning and end of events with large changes in flow. Before the extrapolations, additional concentrations often are 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 estimated from concentrations measured during previous low-flow periods. Concentrations at the end of the events were estimated from concentrations measured shortly after the end of an event. Daily loads for WY2012–14 (unmonitored period) for the Fond du Lac River were estimated from measured daily flows multiplied by the volumetrically weighted concentration of the WY2009–11 period. Daily TP loads during WY2009–14 at the Wolf River at New London and Fox River at Berlin were computed using the Weighted Regressions on Time, Discharge, and Season program (WRTDS; Hirsch and De Cicco, 2014) using all available TP data for these sites from 1977 to 2014. WRTDS was implemented using the R Package (R Development Core Team, 2013) Exploration and Graphics for RivEr Trends (EGRET). WRTDS is used to develop nonlinear, time-varying relations between the logarithms of the constituent concentrations and explanatory variables consisting of decimal time, the logarithm of daily discharge, and sine and cosine transformations of decimal time (Hirsch and others, 2010; Lee and others, 2016). WRTDS derives these flexible relations using a unique weighted regression for each day of the estimation period. All default weighting windows in EGRET were used in this study. Weights for each day are based on differences in the values of the explanatory variables between the estimation and sample day. WRTDS uses a bias correction factor specific to each year, day, and discharge to adjust for retransformation bias (Hirsch and De Cicco, 2014). All daily loads for WY2009–14 for these sites are available in the NWIS database at https://doi.org/10.5066/F7P55KJN (U.S. Geological Survey, 2018).

Sediment P Release

Sediment P release from diffusion and sediment resuspension were estimated separately. To estimate the release rates of P from the lake sediments from diffusion alone during aerobic and anaerobic conditions (used to obtain initial internal loading estimates), sediment cores were collected with a Wildco® sediment-core sampler at 6 locations in Lake Winnebago (12 cores for aerobic and 6 cores for anaerobic analyses), 4 locations in Lake Butte des Morts (5 aerobic and 5 anaerobic analyses), 2 locations in Lake Winneconne (3 aerobic and 1 anaerobic analyses), and 3 locations in Lake Poygan (4 aerobic analyses) on August 10, 2010 (fig. 1). Anaerobic analyses were only done for cores collected at the deeper locations. In the laboratory, cores were maintained in either aerobic or anaerobic environments by gently bubbling air (aerobic) or nitrogen (anaerobic) through air stones placed just above the sediment interface. Samples for DP analysis were collected daily for 9 days from water above the sediment in each core and filtered through a 0.45-micrometer filter. All core processing and chemical analyses were done by the WSLH in accordance with standard analytical procedures (Wisconsin State Laboratory of Hygiene, Environmental Sciences Section, 1993). P release rates were calculated as the linear change in the P mass in the overlying water divided by the time and cross-sectional area of the incubation core (James and Barko, 1991; Robertson and others, 2012) and were combined to obtain average aerobic and anaerobic release rates for each lake and for all lakes combined. The area of aerobic and anaerobic conditions at the sediment-water interface is needed to determine what areas to apply the aerobic and anaerobic rates to compute the total P release from diffusion. This information was obtained from data collected at a buoy installed in Lake Winnebago (fig. 1) using NexSens T-Node temperature (NexSens Technology, Inc.) and RDO PRO® DO (In-Situ, Inc.) sensors placed at the sediment interface and about every meter to the surface of the lake. Each of the sensors was monitored every 5 minutes from May 10 to October 20, 2012. All temperature and DO data are available in the NWIS database at https://doi.org/10.5066/F7P55KJN (U.S. Geological Survey, 2018).

Total net internal P release, defined as P released from diffusion plus P released from sediment resuspension caused by strong winds and biological activity minus sedimentation to the lake bottom, was estimated for Lake Winnebago as the residual in a detailed P budget constructed for the lake. Because all the other terms in Lake Winnebago's P budget could be estimated accurately, the net P release rate could be estimated by adjusting the net release rates until estimated TP concentrations in the lake matched those measured in the lake during WY2009–14. The net P sediment release rates in all four lakes also were estimated using BATHTUB by adjusting internal loading rates until estimated TP concentrations in the lake matched those measured during May through September in 2009 through 2011. Since the total inputs to the Upper Pool Lakes could not be estimated as accurately as Lake Winnebago, their internal P loading rates could not be estimated as accurately as Lake Winnebago. Net P release rates, as well as gross P release and sedimentation, in all four lakes also were estimated from calibrated Jensen models (described later in the "Jensen Model" section).

Meteorological Data

Daily precipitation and air temperature data from October 2008 to December 2014 for all four Pool Lakes were obtained from the National Weather Service station at Oshkosh, Wis. (Cooperative Observer Program identification number 476330) (Midwest Regional Climate Center, 2015).

Numerical Watershed and Lake Models

Watershed Model—SWAT

To estimate the total P loading from the watershed to each Pool Lake, the P loads estimated at the monitored sites were extrapolated to the complete watershed by using output from the Soil and Water Assessment Tool (SWAT) model developed for the Upper Fox and Wolf River Basins (The Cadmus Group, 2018). SWAT is a continuous time step, semidistributed, process-based watershed model developed by the U.S. Department of Agriculture Agricultural Research Service (USDA–ARS; Arnold and others, 2012). SWAT includes key components contributed from several other USDA–ARS models. The SWAT model was calibrated and verified with streamflow and P loads measured at the five monitored sites included in this study and data from several additional small sites monitored in the basin, and then the SWAT model was used to estimate streamflow and TP loading throughout the Winnebago Pool watershed during WY2009–13 (The Cadmus Group, 2018).

Flow and P loading from the SWAT model were applied in this study using two different approaches: one for lake-model calibration and one for full lake-model prediction. For lake-model calibration, measured flows and loads at the monitored sites in the basin were extrapolated to their complete basin areas by multiplying the measured loads at the selected sites by either a simple drainage ratio (Fond du Lac River to its mouth) or a SWAT ratio for Lake Poygan and Lake Butte des Morts. A SWAT ratio is the total SWAT modeled flow and P load to a lake divided by that estimated by SWAT at an upstream gaging station. Downstream extrapolation of measured loads using SWAT ratios was used to reduce potential biases in SWAT model results. For the unmonitored nearshore area between Lakes Butte des Morts and Winnebago, the flows and loads were directly from SWAT for both applications. For the unmonitored nearshore area around Lake Winnebago, flow and P loads were obtained using SWAT ratios and the measured data from the Fond du Lac River site for WY2009–11, and completely from SWAT estimates when the Fond du Lac River was not monitored (WY2012–13). Flows and P loads were estimated for May–September 2009–11 for BATHTUB model calibration, and for WY2009–13 for Jensen model calibrations. For lake-model simulations, the full SWAT-estimated watershed loading to each lake was used, which was based on average total May–September loading for WY2009–13 for the BATHTUB model and the full year of monthly loads for WY2009–13 for the Jensen models.

Lake Models

Mass-Balance Model

To help estimate net internal P loading (from diffusion plus sediment resuspension and other internal sources minus sedimentation) to the lakes, a daily TP mass-balance model (eq. 5) was constructed for Lake Winnebago for WY2009–14:

$$\Delta TP \times Vol = (SW_{PIn} + PPT_P + GW_{PIn} + Pt_P + Septic_P + Sed_{PIn}) - (SW_{POut} + GW_{POut} + Sed_{POut}) \quad (5)$$

where

ΔTP is the change in TP concentrations in the water column,
Vol is the volume of the lake, in cubic meters,
SW_{PIn} is the surface-water inputs of P,
PPT_P is the precipitation inputs,
GW_{PIn} is the groundwater inputs,
Pt_P is the point sources inputs,
$Septic_P$ is the inputs from nearby septic systems,
Sed_{PIn} is the inputs from the bottom sediments of the lake,
SW_{POut} is the surface-water outputs,
GW_{POut} is the groundwater outputs, and
Sed_{POut} is the sedimentation to the bottom sediments of the lake.

The product of ΔTP and Vol is the change in the mass of P in the lake and is equal to the sum of the P inputs to the water column minus the sum of the P leaving the water column. Because each component in equation 5 could be accurately measured for Lake Winnebago, it is believed that net internal P loading (Sed_{PIn}–Sed_{POut}) in Lake Winnebago could be accurately estimated.

BATHTUB Model

Empirical lake models that relate P loading to measured WQ characteristics can be used to determine how changes in P loading to the Pool Lakes should affect their WQ. These models were developed on the basis of data from many different lake systems with specific measures describing lake WQ, such as near-surface TP, Chl-*a*, and SD. Some of these empirical lake models are contained within the BATHTUB model (Walker, 1996). The BATHTUB model includes a hydrologic transport algorithm and is, therefore, capable of simulating changes in complex lakes (multibasin lakes or a chain of lakes like the Winnebago Pool). Within BATHTUB, near-surface TP was simulated with the Canfield and Bachmann (1981) natural-lake equation (eq. 6), which relates hydrology and P loading to near-surface TP:

$$TP = \frac{L_A \div 1{,}000}{\bar{z} \times \left(0.162 \times \left(L_A \div \bar{z}\right)^{0.458} + 1 \div \tau\right)} \tag{6}$$

where

- TP is total P concentration, in milligrams per liter;
- L_A is the P-loading rate per unit area of the lake, in milligrams per square meter per year;
- $\bar{z}$ is the mean depth of the lake, in meters; and
- τ is the water residence time, in years.

Chl-*a* was then estimated from the estimated TP (eq. 6), residence time, and a model-estimated light limitation (that is the default Chl-*a* model in BATHTUB; Walker, 1996). SD was estimated from estimated Chl-*a* and a model-estimated nonalgal turbidity (this is the default model in BATHTUB). Normally, the appropriate averaging time period for water and constituent mass balance calculations is 1 year for reservoirs with relatively long residence times (typically greater than about 1 year) or seasonal (May through September) for reservoirs with relatively short residence times (Walker, 1996). Because of the short residence time for most of the Pool Lakes (table 1), BATHTUB was applied using flow and P inputs from May through September (extrapolated to 12 months by multiplying the seasonal inputs by 2.385) rather using inflow data from all 12 months. The model was used to simulate average TP for June through September; this is the period for which the phosphorus criterion was derived for the Pool Lakes (Wisconsin Department of Natural Resources, 2017a). Because Lake Winnebago had a relatively long residence time compared to the other lakes (table 1), equation 6 (applied external to BATHTUB) also was used with flow and P inputs for the full year to determine if changing its averaging period affected its simulated response to changes in P loading.

Jensen Model

BATHTUB can be used to simulate changes in seasonal average WQ at a new equilibrium condition; however, it does not describe shorter-term variability or transient changes in in-lake WQ after the changes in P inputs. To simulate the expected transient and equilibrium conditions in the lakes, the Jensen model (Jensen and others, 2006) was used. The Jensen model is an empirical model (eq. 7) that relates daily changes in in-lake TP concentrations (ΔTP, in milligrams per liter) to external P loading per unit area of the lake (L_A, in grams per square meter), initial recyclable P accumulated in the sediment (P_{Sed}, in grams per square meter), mean lake depth ($\bar{z}$, in meters), and water temperature (T, in degrees Celsius) by combining its simulations of sediment P release (P_R, in grams per square meter; eq. 8) and P sedimentation (P_S, in grams per square meter; eq. 9).

$$\Delta TP = \frac{\frac{Q}{Vol} \times \left(1 - f_d\right) \times L_A + P_R\left(eq.\ 8\right) - P_S\left(eq. 9\right)}{\bar{z}} \tag{7}$$

$$P_R = b_R \times (1 + t_R)T^{-20} \times P_{Sed} \tag{8}$$

$$P_S = b_S \times (1 + t_S)T^{-20} \times TP \tag{9}$$

where

- Q is the inflow to the lake, in cubic meters per day;
- Vol is the volume of the lake, in cubic meters;
- f_d hydraulic retention time factor equal to $1/(1 + \sqrt{V \sqrt{Q} \sqrt{365}})$;
- b_R is the sediment release constant, that is multiplied by a temperature dependence factor (t_R) and the P_{Sed} of the lake sediments; and
- b_S is the sedimentation constant, that is multiplied by a temperature dependence factor (t_S), and the TP concentration in the lake (TP).

The Jensen model was converted into an Excel worksheet and calibrated for WY2009–13 (that is, optimal values were determined for b_R, t_R, b_S, t_S, and P_{Sed}) using measured TP and total daily external P loadings to the lake (all P inputs to the lake except those from the sediments) using the Solver code in Excel®. Two Jensen models were first developed: one model for the three Upper Pool Lakes (collectively), and one model just for Lake Winnebago. The Upper Pool Jensen model was calibrated for WY2009–13 by using the daily total external P input to all three Upper Pool Lakes (Lake Poygan, Lake Winneconne, and Lake Butte des Morts) and measured TP at the Fox River at Oshkosh as its in-lake TP response variable. The Winnebago Jensen model was calibrated by using daily total external P input to Lake Winnebago and TP concentrations measured in the center of the lake combined with those measured at the Fox River at Neenah Menasha as its in-lake TP response variable. The additional TP data at Neenah Menasha were required to describe the full seasonality in TP concentrations. Daily output from the Upper Pool Jensen model was then used as input to the calibrated Winnebago Jensen model to obtain a linked Jensen model driven by complete watershed loading. The Jensen model uses continuous daily flows and P inputs from all 12 months. The Jensen models were used to simulate daily in-lake TP for the entire year, and geometric summer-mean TP concentrations were then computed for the TP criterion assessment period (June 1–September 15).

The BATHTUB and Jensen models were first calibrated by using loading from the watershed extrapolated from the monitored sites to minimize potential model biases and then the models were used to simulate specific scenarios using full SWAT simulated watershed inputs (and specific modification of those inputs) for WY2009–13 as a base-loading condition.

Lake Water Quality

Water temperatures in all the Pool Lakes gradually increased as summer progressed. Only weak thermal stratification was observed in the routine monitoring data. Water temperatures typically reached 24–26 degrees Celsius (°C) in July and August in 2009–11. All four lakes were polymictic, resulting in their near-bottom temperatures being similar to their surface temperatures. This extensive deep mixing was expected given that the Osgood Index (Osgood, 1988) for each lake was much less than 4 (table 1). Because Lake Winnebago is deeper than the other lakes, it had slightly stronger stratification than the other Pool Lakes. To document short-term changes in stratification, continuous recording sensors were installed throughout the water column in Lake Winnebago in 2012 (fig. 1), which further demonstrated only weak thermal stratification in the lake (fig. 2). Near-surface DO was near or above saturation throughout most of the open-water period. DO in the deep areas of all the lakes occasionally declined but rarely was measured below 5 mg/L (DO concentrations less than 5 mg/L exceed the DO criterion for lakes; Wisconsin Department of Natural Resources, 2017a). In 2012, the continuous recording sensors demonstrated that near-bottom DO occasionally dropped well below 5 mg/L but rarely dropped below 2 mg/L (fig. 2). These hypoxic occurrences were short-lived because of the frequent deep mixing.

Near-surface TP in all four Pool Lakes were relatively similar and had distinct seasonality (fig. 3). TP remained relatively steady during May and then increased dramatically from June through late September, often reaching 0.15 mg/L in late summer. During 2009–11, the average near-surface summer (June–September) TP was 0.097 mg/L in Lake Winnebago, 0.104 mg/L in Lake Butte des Morts, 0.088 mg/L in Lake Winneconne, and 0.094 mg/L in Lake Poygan (table 3). TSI_{TP} values for the Pool Lakes ranged from about 68 to 71; therefore, based on TP, the lakes are classified on the border between eutrophic and hypereutrophic and are classified as hypereutrophic in late summer. DP concentrations were quite different among lakes. In Lake Winnebago, DP gradually increased as summer progressed, typically exceeding 0.06 mg/L by late summer; therefore, P does not seem to be the factor limiting late summer productivity in Lake Winnebago. In the other three lakes, DP typically was low (less than 0.01 mg/L), but a few concentrations greater than 0.07 mg/L were measured. During 2009–11, summer-average near-surface DP was 0.029 mg/L in Lake Winnebago, 0.004 mg/L in Lake Butte des Morts, 0.006 mg/L in Lake Winneconne, and 0.011 mg/L in Lake Poygan (table 3).

TN, computed from the sum of NO_{2+3} plus TKN, was relatively similar among lakes and had little seasonality. During 2009–11, summer-average near-surface TN was 1.23 mg/L in Lake Winnebago, 1.65 mg/L in Lake Butte des Morts, 1.39 mg/L in Lake Winneconne, and 1.49 mg/L in Lake Poygan (table 3), resulting in TSI_{TN} values ranging from 57 to 62. TN greater than 0.74 mg/L indicates eutrophic conditions (Kratzer and Brezonik, 1981); therefore, all four lakes are classified as highly eutrophic.

Near-surface Chl-*a* also had distinct seasonality (increasing throughout summer) and was lower in Lake Winnebago than in the other lakes (fig. 3). During 2009–11, summer-average near-surface Chl-*a* was 30.5 micrograms per liter (µg/L) in Lake Winnebago, 52.1 µg/L in Lake Butte des Morts, 39.8 µg/L in Lake Winneconne, and 36.2 µg/L in Lake Poygan (table 3). $TSI_{Chl\text{-}a}$ values for the Pool Lakes ranged from about 62 to 68; therefore, based on Chl-*a*, the lakes are classified as highly eutrophic, especially in late summer.

Water clarity consistently decreased as summer progressed, especially in Lake Winnebago; lowest SDs occurred in mid-to-late summer (fig. 3). Water clarity in Lake Winnebago was better than in the upstream lakes in early summer, but clarity was similar among lakes by late summer. During 2009–11, summer-average SD was 1.15 m in Lake Winnebago, 0.54 m in Lake Butte des Morts, 0.68 m in Lake Winneconne, and 0.66 m in Lake Poygan (table 3). TSI_{SD} values for the Pool Lakes ranged from about 60 to 70; therefore, on the basis of SD, the Upper Pool Lakes are classified as borderline hypereutrophic.

To determine how WQ during 2009–11 compared with other recent years, and whether there were any long-term changes, data from 2009 to 2011 were compared with data collected in 1976 and 1989–2015 (fig. 4; table 4). Only Lake Winnebago had sufficient data for this long-term comparison. WQ during 2009–11 was similar or possibly slightly better (lower TP and Chl-*a*, and deeper SD) than the long-term average. The earliest data collected by Sloey and Spangler (1977) indicates that the WQ in Lake Winnebago may have improved; since 1976, TP seems to have decreased and water clarity has improved. However, only 1 year of data were available in 1976, and little data were available between 1976 and 1989.

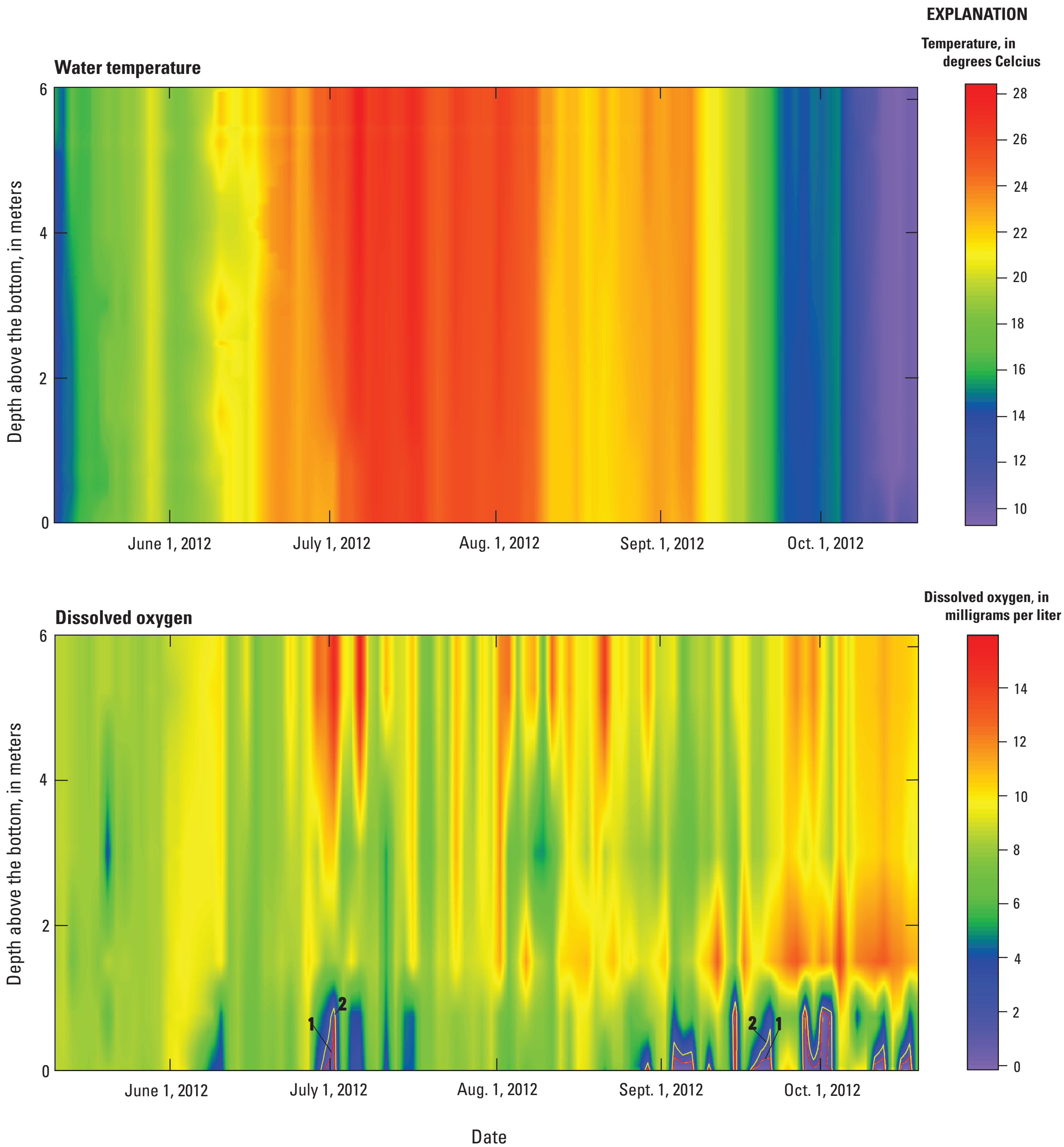


Figure 2. Distributions of temperature and dissolved oxygen in Lake Winnebago, Wisconsin, in 2012. The 1 mg/L and 2 mg/L isopleths for dissolved oxygen are identified.

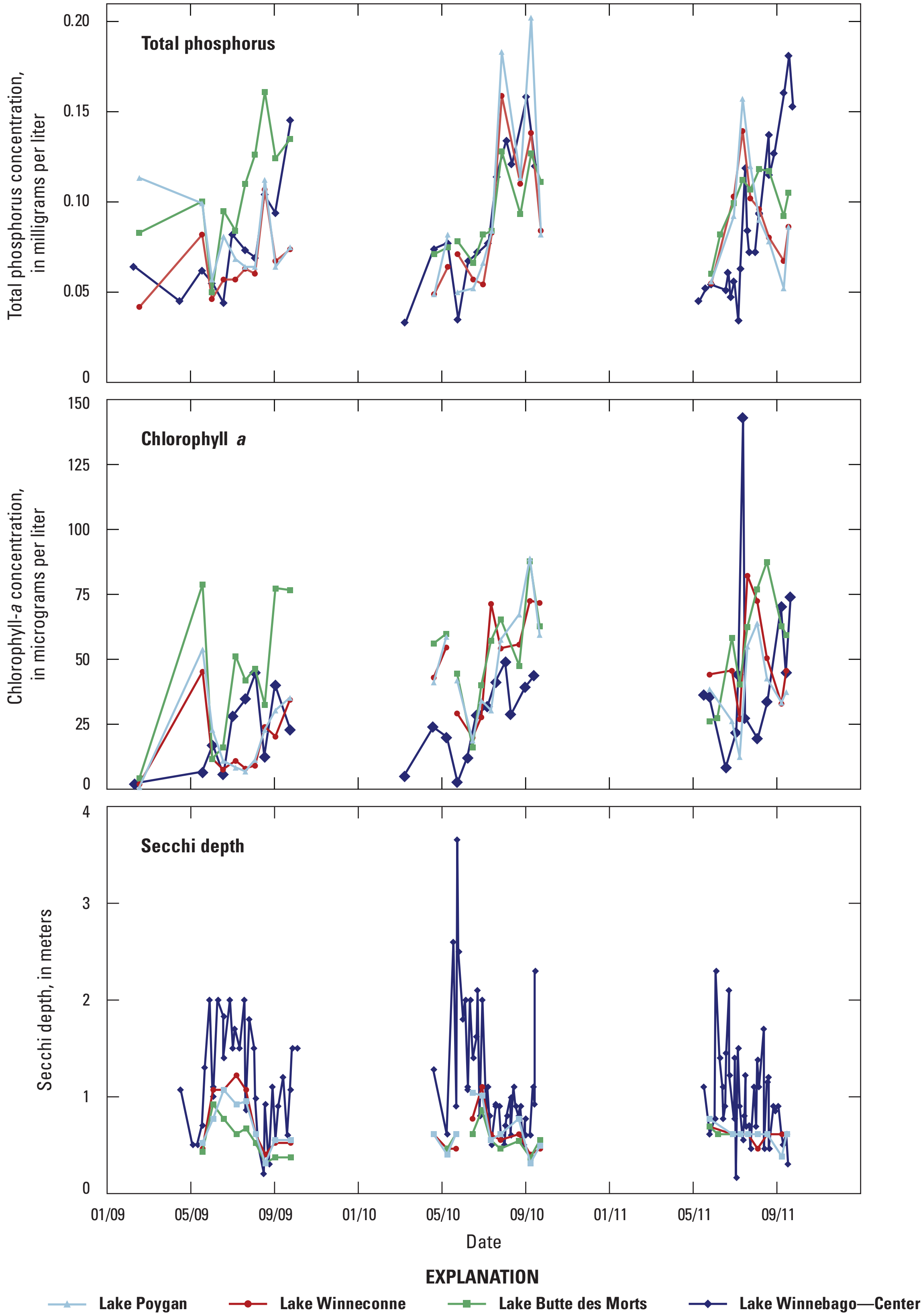


Figure 3. Near-surface total phosphorus, chlorophyll *a*, and Secchi depths in the Winnebago Pool Lakes, Wisconsin, 2009–11.

Table 3. Average June–September near-surface water quality in the Winnebago Pool Lakes, Wisconsin.

[TSI, trophic state index; --, not available]

Sampling site	Measurement				TSI	
	2009–11	2009–13	Sloey and Spangler[1] historical 1976	Long term 1976–2015	2009–11	Long term 1976–2015
Total phosphorus, in milligrams per liter						
Lake Poygan	0.094	—	0.136	--	68.6	--
Lake Winneconne	0.088	—	—	--	67.9	--
Lake Butte des Morts	0.104	—	0.155	--	70.7	--
Lake Winnebago—Fox River inlet	0.103	—	--	--	70.6	--
Lake Winnebago[2]	0.097	0.116	0.226	0.122	68.9	71.8
Lake Winnebago—Fox River outlet	0.105	—	--	--	69.7	--
Dissolved phosphorus, in milligrams per liter						
LakePoygan	0.011	--	--	--	--	--
LakeWinneconne	0.006	--	--	--	--	--
LakeButte des Morts	0.004	--	--	--	--	--
Lake Winnebago—Fox River inlet	0.005	--	--	--	--	--
Lake Winnebago[2]	0.029	--	--	0.029	--	--
Lake Winnebago—Fox River outlet	0.045	--	--	--	--	--
Total nitrogen, in milligrams per liter						
Lake Poygan	1.49	--	--	--	60.0	--
Lake Winneconne	1.39	--	--	--	58.9	--
Lake Butte des Morts	1.65	--	--	--	61.5	--
Lake Winnebago—Fox River inlet	1.66	--	--	--	61.6	--
Lake Winnebago[2]	1.19	--	--	1.28	57.0	57.7
Lake Winnebago—Fox River outlet	1.23	--	--	--	56.9	--
Chlorophyll *a*, in micrograms per liter						
Lake Poygan	36.2	--	90.2	--	63.7	--
Lake Winneconne	39.8	--	--	--	64.5	--
Lake Butte des Morts	52.1	--	47.1	--	68.3	--
Lake Winnebago—Fox River inlet	45.5	--	--	--	65.6	--
Lake Winnebago[2]	30.5	43.5	68.5	48.7	61.6	66.4
Lake Winnebago—Fox River outlet	25.5	--	--	--	56.4	--
Secchi depth, in meters						
Lake Poygan	0.66	--	0.58	--	66.7	--
Lake Winneconne	0.68	--	--	--	66.3	--
Lake Butte des Morts	0.54	--	0.48	--	69.7	--
Lake Winnebago[2]	1.15	1.04	0.66	0.98	59.9	61.5

[1]Sloey and Spangler (1977).

[2]Average of data collected at north, center, and south sites, whenever available.

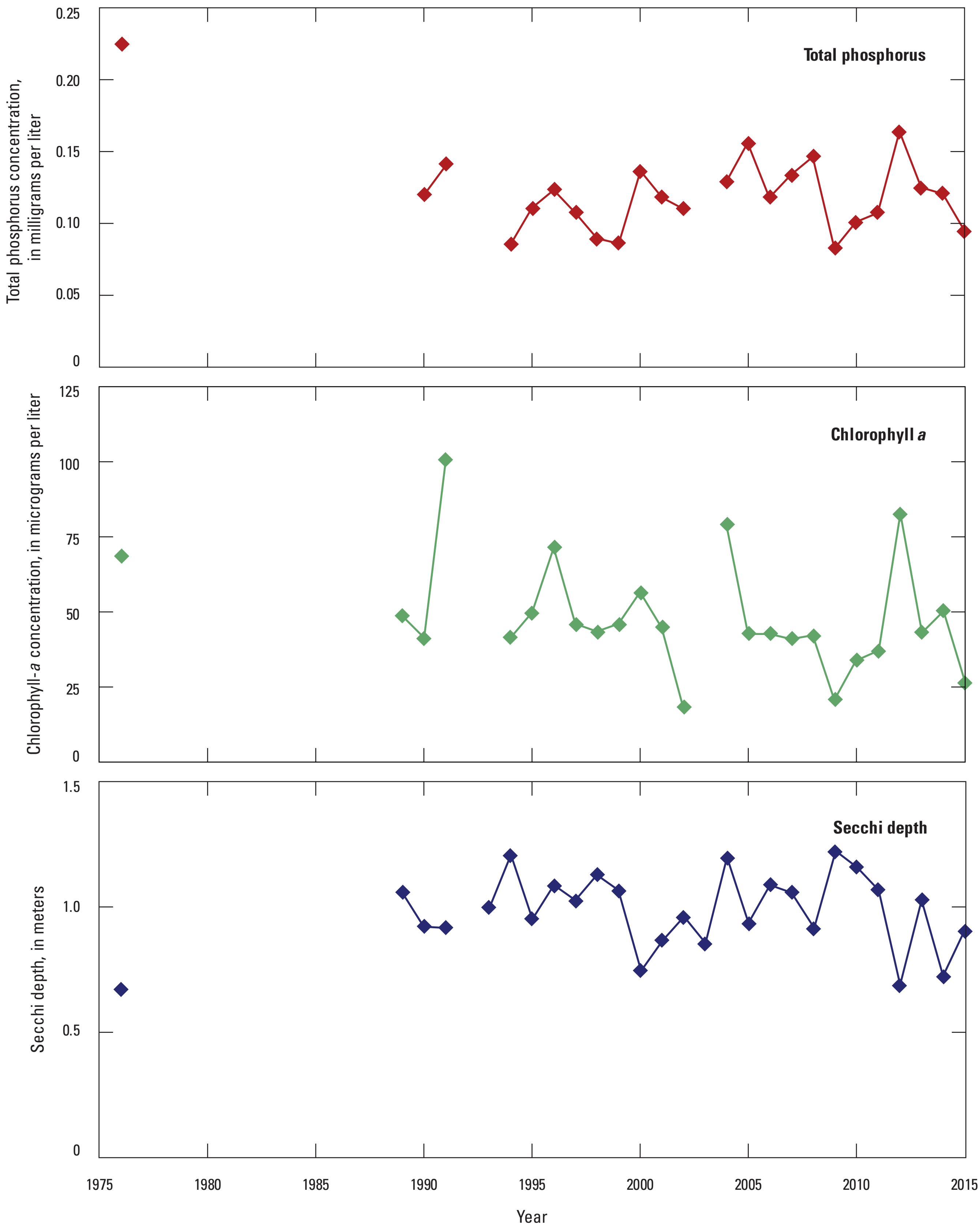


Figure 4. Summer-average near-surface total phosphorus, chlorophyll *a*, and Secchi depths in Lake Winnebago, Wisconsin, 1976–2015.

Table 4. Summer-average (June–September) near-surface total phosphorus, chlorophyll *a*, Secchi depths, and total nitrogen for Lake Winnebago, Wisconsin, 1976–2015.

[TP, total phosphorus; mg/L, milligram per liter; Chl-*a*, chlorophyll *a;* µg/L, microgram per liter; SD, Secchi depths; m, meter; TSI, Trophic State Index (values were computed using equations 1 through 4 in the text); ; TN, total nitrogen; NA, not available; water quality data were retreived from the sites described in table 1]

Year	TP (mg/L)	Chl-*a* (µg/L)	SD (m)	TN (mg/L)	TSI_{TP}	$TSI_{Chl\text{-}a}$	TSI_{SD}	TSI_{TN}
1976	0.225	68.6	0.67	NA	81.4	71.3	66.2	NA
1989	NA	48.7	1.06	1.182	NA	67.9	59.5	NA
1990	0.120	41.3	0.93	1.232	72.7	64.9	61.9	57.8
1991	0.142	100.7	0.92	1.794	73.5	72.1	63.0	61.0
1992	NA	NA	NA	NA	NA	NA	NA	NA
1993	NA	NA	1.00	NA	NA	NA	60.6	NA
1994	0.085	41.4	1.21	1.239	66.9	62.3	58.2	57.0
1995	0.111	49.7	0.96	1.083	71.3	68.6	61.8	55.5
1996	0.124	71.5	1.09	NA	71.7	69.1	59.2	NA
1997	0.108	45.9	1.03	1.221	70.4	67.1	60.8	57.8
1998	0.089	43.4	1.13	1.176	66.0	64.9	59.8	57.4
1999	0.086	45.9	1.07	1.192	67.3	65.2	59.4	56.3
2000	0.136	56.6	0.75	1.523	74.1	70.1	64.8	60.2
2001	0.118	44.9	0.87	1.138	70.8	67.2	64.0	57.0
2002	0.110	18.2	0.96	1.803	71.9	57.8	62.6	63.0
2003	NA	NA	0.86	NA	NA	NA	63.2	NA
2004	0.129	79.1	1.20	NA	72.9	71.8	61.7	NA
2005	0.156	42.8	0.94	0.982	76.5	67.4	61.4	53.9
2006	0.118	42.7	1.09	NA	71.7	64.3	59.4	NA
2007	0.134	41.0	1.06	1.172	73.9	66.2	59.8	57.3
2008	0.147	42.2	0.92	1.385	74.5	64.6	62.0	59.1
2009	0.082	20.6	1.22	1.165	66.9	59.5	57.5	56.6
2010	0.101	33.9	1.16	1.176	70.4	64.5	58.6	56.6
2011	0.108	37.1	1.07	1.239	70.3	64.1	59.9	57.3
2012	0.164	82.8	0.69	NA	76.5	72.3	67.0	NA
2013	0.125	43.1	1.03	NA	73.1	66.8	60.7	NA
2014	0.121	50.3	0.72	NA	70.5	68.2	64.8	NA
2015	0.095	26.5	0.91	NA	68.4	61.7	61.5	NA
				Average 2009–11				
2009–11	0.097	30.5	1.15	1.194	69.2	62.7	58.7	56.8
2009–13	0.116	43.5	1.04	1.194	71.4	65.4	60.7	56.8
				Long-term average 1976–2015				
1976–2015	0.122	48.7	0.98	1.277	71.8	66.4	61.5	57.7

Hydrology and Water Budget

Because productivity of the Pool Lakes is believed to be primarily controlled by P, except in late summer, changes in the P inputs should change in-lake TP and affect other trophic state constituents, like Chl-*a* and SD. Almost all P entering lakes is transported with water inputs; therefore, to quantify P inputs, it is helpful to first accurately quantify the water inputs. The water budget for a period of interest for a lake may be represented as follows:

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

where

- ΔS is change in the lake water volume, in cubic meters per day,
- SW_{In} is the surface-water inflow, in cubic meters per day,
- PPT is the precipitation, in cubic meters per day,
- GW_{In} is the groundwater inflow, in cubic meters per day,
- SW_{Out} is the surface-water outflow, in cubic meters per day,
- E is the evaporation, in cubic meters per day, and
- GW_{Out} is the groundwater outflow, in cubic meters per day.

The complete water budget could be quantified for Lake Winnebago; however, much of the information was not available for the other lakes. Therefore, the average-annual and growing-season (May–September) water budgets for WY2009–14 were estimated for Lake Winnebago, and this information was then used to help develop P budgets for all of the Pool Lakes.

Changes in Water Level and Lake Volume

Changes in the volume of Lake Winnebago (fig. 5) were determined from continuous water levels measured at the lake-stage gage near the Fox River inlet (fig. 1) and from lake morphometry based on the 2014 contour map for the lake. Water levels and volume varied seasonally and were typically highest in early summer and lowest in February. Over the entire WY2009–14 period, lake volume changed little overall.

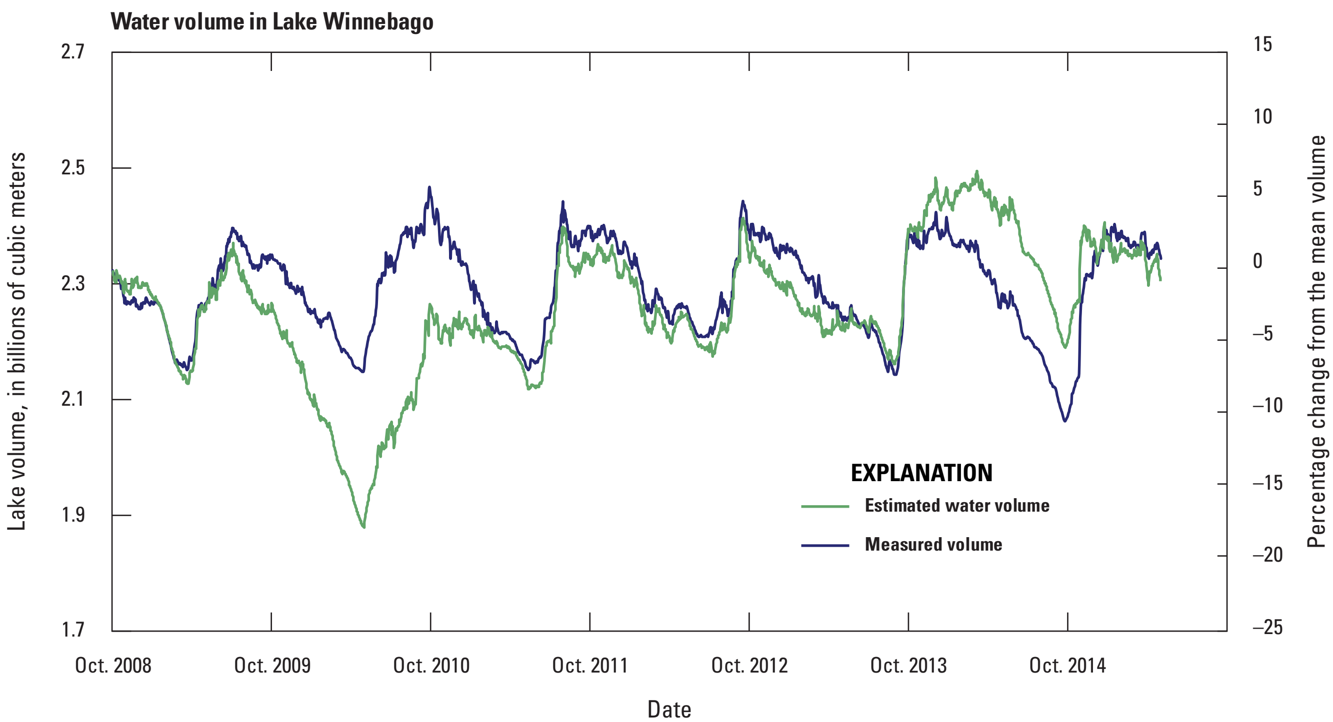


Figure 5. Measured and estimated volume of Lake Winnebago, Wisconsin, water years 2009–14.

Surface-Water Inflow and Outflow

Streamflow was measured at five sites in the watershed (fig. 1) and used to estimate average-annual and average-growing-season surface-water input and output from the Pool Lakes (tables 5 and 6). The magnitudes of the measured flows were directly related to the size of the drainage basins upstream of the gages. After compensating for the difference in drainage areas, flows per unit area (yields) were highest at the Fox River at Berlin. In general, flows were highest in WY2011, especially in the Wolf River, which resulted in the average annual flows for WY2009–11 being slightly higher than in other summary periods; however, the years with highest flows during May–September varied among sites.

Either drainage-area ratios (total area of interest divided by the area at a gaging station) or SWAT-flow ratios (total flow to a lake estimated with SWAT divided by the flow estimated at a gaging station with SWAT) based on WY2009–11 results were used to extrapolate the flows measured upstream in the watershed to the total flows into each lake. Drainage areas of the monitored sites and full drainage area for each lake used in SWAT are shown in figure 6. To estimate average flows for May–September, a SWAT-flow ratio of 1.700 was used to extrapolate the flow measured at the Wolf River at New London to the total flow into Lake Poygan, 1.152 was used to extrapolate the flow measured at the Fox River at Berlin to the total flow into Lake Butte des Morts, and 1.375 was used to extrapolate the flow measured at the Fond du Lac River at Fond du Lac to the total ungaged nearshore flow to Lake Winnebago. The measured flows estimated at the monitored stations, full drainage areas for each lake, and SWAT-flow ratios are given in table 7. To estimate annual flows, a SWAT ratio of 1.700 was used to extrapolate flow measured at the Wolf River at New London to the total flow into Lake Poygan, 1.141 was used to extrapolate the measured flow at the Fox River at Berlin to the total flow into Lake Butte des Morts, and 1.399 was used to extrapolate the flow measured for the Fond du Lac River to the ungaged nearshore flow to Lake Winnebago (table 7). To estimate the total flows from the Fond du Lac River, a 1.011 drainage-area ratio was applied to the data measured at the gage just upstream from its mouth.

Outflows from Lake Winnebago (Fox River at Neenah Menasha) were estimated from flows measured at the Fox River at Appleton station using a 1.01 ratio. The total inflow to Lake Winnebago from the Fox and Fond du Lac Rivers, the remaining nearshore areas, and the total outflow from the lake are given in table 6.

Groundwater

Groundwater inflow to Lake Winnebago was estimated from a regional groundwater-flow model developed for the Lake Michigan Basin (Feinstein and others, 2010). From this model, estimated direct groundwater inflow to Lake Winnebago was 12 million cubic meters per year (Mm^3/yr; 13.3 cubic feet per second [ft^3/s]), and estimated seepage out of the lake to groundwater was 0.9 Mm^3/yr (1 ft^3/s), for a net inflow of about 11 Mm^3/yr or 4.6 Mm^3 during May–September (table 6; D. Feinstein, U.S. Geological Survey, written commun., 2013). Groundwater inflow to the other lakes could not be estimated directly from this model; therefore, it was assumed that total groundwater inflows to Lake Poygan, Lake Winneconne, and Lake Butte des Morts were 2, 1, and 3 Mm^3/yr, respectively, which were based on their relative lengths of shoreline. There was assumed to be no losses of lake water to groundwater from the Upper Pool Lakes.

Precipitation

During 2009–14, daily precipitation on the surface of the lakes was estimated on the basis of measured data from Oshkosh, Wis. Based on data from this site, annual precipitation on the surface of each lake ranged from 0.739 m in WY2010 to 1.068 m in WY2008, and the total precipitation during May–September ranged from 0.336 m in WY2009 to 0.692 m in WY2010. Annual and seasonal rainfall on the surface of Lake Winnebago is given in table 6.

Evaporation

Total annual evaporation from Lake Winnebago (0.95 m) was assumed to be similar to that measured in evaporation pan measurements at Green Bay, Wis., which is about 45 km north of the lake (data were collected during 1956–70; Farnsworth and Thompson, 1982). The monthly evaporation rates were adjusted to better represent large lakes, like Lake Winnebago, with more evaporation in the fall and less in late winter and early spring. The total evaporation used for Lake Winnebago for May–September (0.71 m) is consistent with that estimated by Farnsworth and Thompson (1982) for those months.

Table 5. Flow, total phosphorus loads, and total phosphorus concentrations (volumetrically weighted), and flow and total phosphorus (P) yields in the Winnebago Pool tributaries, inlet to Lake Winnebago and outlet of Lake Winnebago in water years 2009–14.

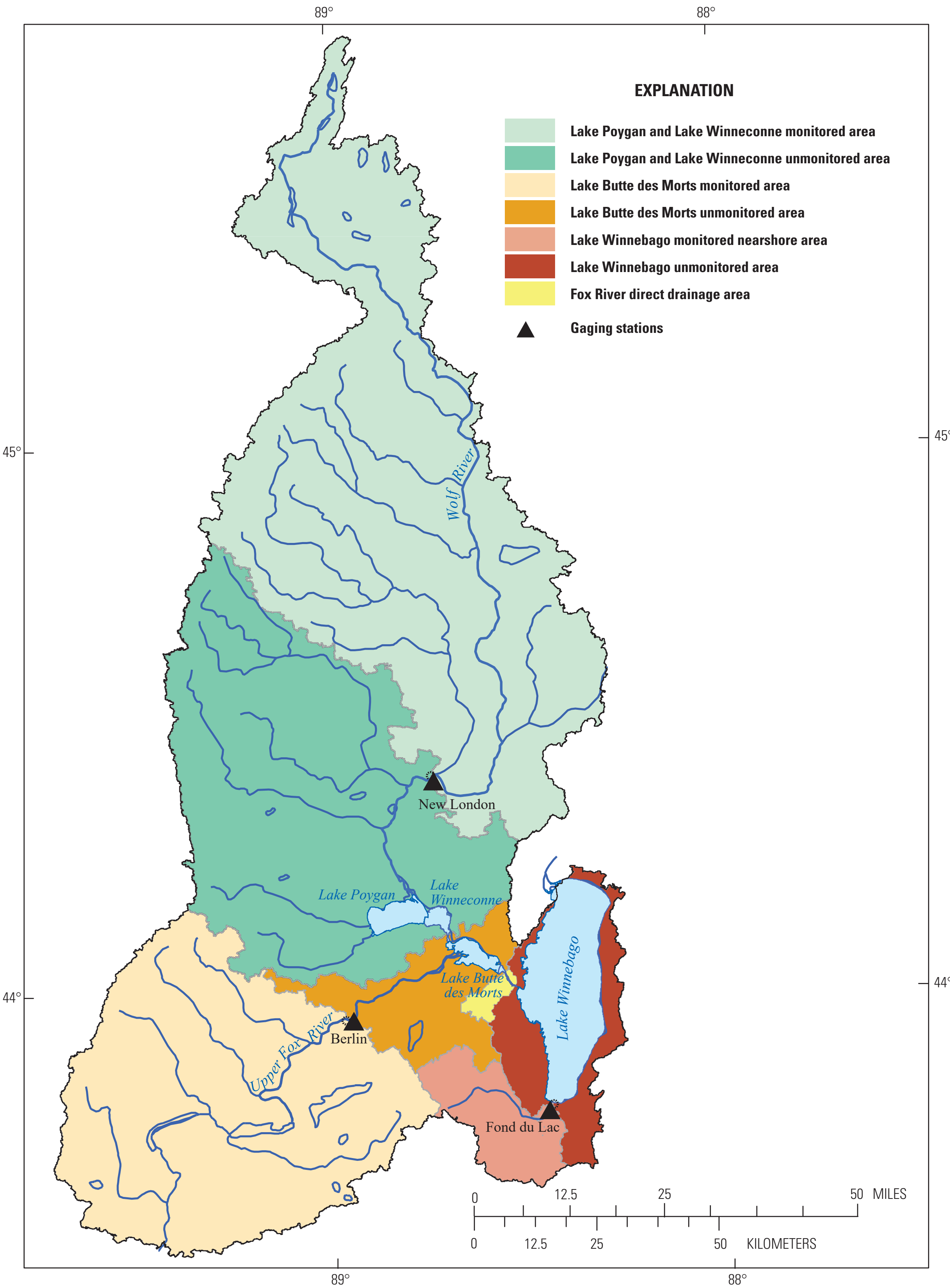
[m^3/s, cubic meter per second; P, phosphorus; kg, kilogram; VW, volumetrically weighted; TP, total phosphorus; mg/L, milligram per liter; m, meter; kg/km^2, kilogram per square kilometer]

Water year	Wolf River at New London			Fox River at Berlin			Fond du Lac River at Fond du Lac			Fox River @ Oshkosh			Fox River at Neenah Menasha		
	Flow (m^3/s)	P (kg)	VW TP (mg/L)	Flow (m^3/s)	P (kg)	VW TP (mg/L)	Flow (m^3/s)	P (kg)	VW TP (mg/L)	Flow (m^3/s)	P (kg)	VW TP (mg/L)	Flow (m^3/s)	P (kg)	VW TP (mg/L)
Annual summary															
2009	36	76,143	0.066	32	81,625	0.081	3.3	17,194	0.166	93	216,298	0.074	101	229,271	0.072
2010	47	124,879	0.084	37	101,995	0.088	3.6	27,676	0.241	120	271,839	0.072	129	320,447	0.079
2011	73	173,782	0.075	38	91,654	0.077	3.5	16,595	0.149	168	390,218	0.074	171	356,719	0.066
2012	46	91,001	0.062	32	72,376	0.072	2.3	11,004	0.150	115	247,118	0.068	121	328,288	0.085
2013	50	104,508	0.066	35	84,985	0.077	3.2	14,948	0.150	137	298,135	0.069	138	360,818	0.083
2014	59	128,474	0.069	34	83,445	0.077	3.4	16,256	0.150	144	460,969	0.101	160	480,655	0.095
2009–14	52	116,464	0.071	35	86,013	0.079	3.2	17,279	0.168	129	314,096	0.076	137	346,033	0.080
2009–11	52	124,935	0.075	35	91,758	0.082	3.5	20,488	0.186	127	292,785	0.073	134	302,146	0.072
2009–13	51	114,062	0.071	35	86,527	0.079	3.2	17,484	0.171	126	284,721	0.071	132	319,109	0.077
May–September															
2009	36	38,146	0.081	30	47,396	0.119	2.1	4,912	0.176	79	97,901	0.094	75	57,966	0.058
2010	59	87,573	0.112	44	69,889	0.119	4.2	17,728	0.322	164	185,575	0.085	160	236,629	0.112
2011	68	87,010	0.097	37	54,221	0.110	2.3	6,704	0.220	168	200,063	0.090	166	185,245	0.084
2012	36	36,488	0.076	30	38,837	0.099	1.6	3,256	0.150	90	124,346	0.104	92	138,895	0.114
2013	55	61,252	0.084	38	53,515	0.105	2.9	5,653	0.150	144	158,082	0.083	135	184,416	0.103
2014	69	81,291	0.089	42	54,723	0.099	4.9	9,690	0.150	172	244,530	0.108	186	308,886	0.126
2009–14	54	65,293	0.090	37	53,097	0.109	3.0	7,990	0.195	136	168,416	0.094	136	185,340	0.100
2009–11	54	70,910	0.096	37	57,169	0.116	2.9	9,781	0.240	137	161,180	0.090	134	159,947	0.085
2009–13	51	62,094	0.090	36	52,772	0.111	2.6	7,650	0.204	129	153,193	0.091	126	160,630	0.094

Water year	Wolf River at New London		Fox River at Berlin		Fond du Lac River at Fond du Lac		Fox River at Oshkosh		Fox River at Neenah Menasha	
	Flow yield (m)	P yield (kg/km^2)	Flow yield (m)	P yield (kg/km^2)	Flow yield (m)	P yield (kg/km^2)	Flow yield (m)	P yield (kg/km^2)	Flow yield (m)	P yield (kg/km^2)
Annual yields										
2009–14	0.268	19.0	0.318	25.1	0.234	39.7	0.290	22.3	0.277	22.2
2009–11	0.269	20.4	0.326	26.8	0.252	47.1	0.284	20.8	0.271	19.4
2009–13	0.261	18.6	0.319	25.2	0.231	40.2	0.283	20.2	0.268	20.5
May–September yields										
2009–14	0.117	0.0	0.142	0.0	0.091	0.0	0.128	0.0	0.115	0.0
2009–11	0.117	0.0	0.144	0.0	0.087	0.0	0.129	0.0	0.114	0.0
2009–13	0.110	0.0	0.139	0.0	0.079	0.0	0.121	0.0	0.107	0.0

Table 6. Summary of the water-budget components for Lake Winnebago, Wisconsin, water years 2009–14.

Year	Fox River	Fond du Lac River	Nearshore	Groundwater inflow	Precipitation	Total input	Neenah Menasha outlet	Evaporation	Groundwater outflow	Total output
Annual water inputs and outputs										
Water input and output, in millions of cubic meters										
2009	2,930	104	146	12	375	3,570	3,200	485	0.9	3,689
2010	3,780	116	162	12	545	4,610	4,070	518	0.9	4,599
2011	5,290	113	158	12	391	5,960	5,380	504	0.9	5,889
2012	3,640	74	104	12	458	4,290	3,840	504	0.9	4,350
2013	4,320	101	141	12	495	5,070	4,360	504	0.9	4,860
2014	4,550	110	154	12	565	5,390	5,040	503	0.9	5,550
Average 2009–14	4,080	103	144	12	471	4,810	4,320	503	0.9	4,820
Average 2009–11	4,000	111	155	12	437	4,710	4,220	502	0.9	4,720
Average 2009–13	3,990	102	142	12	453	4,700	4,170	503	0.9	4,670
	Percentage of total input						Percentage of total output			
2009–14	84.8	2.1	3.0	0.2	9.8	100	89.5	10.4	0.0	100
2009–11	84.8	2.4	3.3	0.3	9.3	100	89.3	10.6	0.0	100
2009–13	84.9	2.2	3.0	0.3	9.6	100	89.2	10.8	0.0	100
May–September water inputs and outputs										
Water input and output, in millions of cubic meters										
2009	1,040	28.2	39.4	5.0	178	1,290	996	379	0.4	1,380
2010	2,170	55.6	77.8	5.0	368	2,680	2,120	380	0.4	2,500
2011	2,220	30.8	43.1	5.0	199	2,500	2,200	379	0.4	2,580
2012	1,200	22.0	30.8	5.0	240	1,490	1,220	379	0.4	1,600
2013	1,900	38.2	53.4	5.0	232	2,230	1,790	379	0.4	2,160
2014	2,270	65.5	91.6	5.0	308	2,740	2,460	379	0.4	2,840
Average 2009–14	1,800	40.0	56.0	5.0	254	2,160	1,800	379	0.4	2,180
Average 2009–11	1,810	38.2	53.4	5.0	248	2,160	1,770	379	0.4	2,150
Average 2009–13	1,710	35.0	48.9	5.0	243	2,040	1,660	379	0.4	2,040
	Percentage of total input						Percentage of total output			
2009–14	83.5	1.9	2.6	0.2	11.8	100	82.5	17.4	0.0	100
2009–11	84.0	1.8	2.5	0.2	11.5	100	82.3	17.6	0.0	100
2009–13	83.7	1.7	2.4	0.2	11.9	100	81.4	18.6	0.0	100



Base from U.S. Geological Survey digital data, 1:1,000,000, 2018
North American Datum of 1983 HARN
Transverse Mercator projection
Central meridian 90° W
Scale factor 0.9996
Latitude of origin 0.0

Figure 6. Monitored and unmonitored areas of the watershed of the Winnebago Pool Lakes, Wisconsin. Gaging stations are identified on the figure.

Table 7. Calculation of Soil and Water Assessment Tool ratios used to extraplolate measured flows and phosphorus loads at selected sites in the Winnebago Pool Basin to unmonitored areas in the basin. All flows and phosphorus loads are average annual totals for the specified period simulated with SWAT for 2009 to 2011 (The Cadmus Group, 2018).

[SWAT, Soil and Water Assessment Tool; --, not available]

Flow/load parameter	Wolf River at New London	Entire Poygan and Winneconne basin	Fox River at Berlin	Entire Butte des Morts basin	Nearshore between Butte and Winnebago	Fond du Lac River at Fond du Lac	Winnebago nearshore not including contributions from the Fond du Lac River
Streamflow, in millions of cubic meters							
May–September							
Flow	693	1,180	455	524	16	60	82
SWAT-flow ratios	--	1.700	--	1.152	--	--	1.375
Annual							
Flow	1,600	2,710	1,030	1,180	34	106	149
SWAT-flow ratios	--	1.700	--	1.141	--	--	1.399
Phosphorus loading, in kilograms							
May–September							
Total load	63,900	108,000	45,100	51,600	4,560	10,600	7,080
Point source load	--	2,260	--	839	--	--	3,430
Nonpoint load	--	106,000	--	50,700	--	--	3,650
SWAT-load ratios	--	1.658	--	1.125	--	--	--
Annual							
Total load	115,000	194,000	71,200	82,000	10,500	20,600	14,400
Point source load	--	5,390	--	2,000	--	--	8,240
Nonpoint load	--	188,000	--	80,000	--	--	6,140
SWAT-load ratios	--	1.642	--	1.124	--	--	--

Water Budget Summary

While developing the water budget for Lake Winnebago, it was noticed that the measured and estimated volumes of the lake deviated from one another starting in October 2013. Moving the primary equipment used to estimate flow at the Fox River at Appleton may have caused this deviation (R. Waschbusch, U.S. Geological Survey, 2017, oral commun.); therefore, estimated outflows from the lake after September 2013 were increased by 9 percent, which resulted in equation 10 reproducing the changes in the volume of Lake Winnebago throughout WY2009–14 without any long-term bias (fig. 5). The discrepancy between measured and estimated water volume in 2009–10 may have been caused by using rainfall from only one location in the basin and using similar evaporation rates in all years.

During WY2009–14, the Fox River was the largest contributor of water to Lake Winnebago, contributing about 84 percent of its total water input for May–September and 85 percent for the entire year (table 6). Direct precipitation represented about 9–12 percent of the total input, and the Fond du Lac River and nearshore areas each represented about 2–3 percent of the total input. Based on these input rates during May–September, the residence time (time required for water entering the lake to completely replace its volume) ranged from 0.37 years during wet periods (2014) to 0.78 years during dry periods (2009), with an average of 0.47 years during May–September 2009–11.

On an annual basis, the total output of water from the lake was similar to the total input in all years, which resulted in little change in storage from the beginning to the end of each water year. On average, about 81–90 percent of the total water leaving the lake left through its outlets to the Fox River at Neenah Menasha. Evaporation represented between about 11 percent of water loss during the entire year and 17–18 percent during May–September (table 6).

Phosphorus Inputs to the Winnebago Pool Lakes

To describe where the P entering each of the Winnebago Pool Lakes originates from and how much may be controllable, detailed annual and May–September P budgets were computed for WY2009–14 for Lake Winnebago, and then this information was used to develop detailed P budgets for all four Pool Lakes for WY2009–11. A shorter period was used for all four lakes because information for some of the source components, primarily transport from the upstream lakes, were not available for Lake Winneconne and Lake Butte des Morts for the entire period. The P budget for a period of interest is described in equation 5. From equation 5, changes in TP concentrations in a lake can be estimated by dividing its total net daily P input by its volume.

Nonpoint Surface-Water Inputs

Measured tributary TP concentrations and loading.—TP concentrations were measured at five riverine sites in the Winnebago Pool watershed (fig. 1). TP concentrations at all sites in the watershed demonstrated distinct seasonality; highest concentrations in the rivers occurred around July of each year, which is a little before maximum concentrations were measured in the lakes (late September) (fig. 7). This seasonality resulted in the May–September average volumetrically weighted (VW) concentrations (total P load divided by total flow) being higher than the annual averages (table 5), and above the phosphorus criterion of 0.075 mg/L for wadeable streams (upstream sites) and similar to the 0.100 mg/L criterion for nonwadeable rivers (for the downstream Fox River sites) in Wisconsin (Wisconsin Department of Natural Resources, 2017a). Highest TP concentrations were measured in the Fond du Lac River, followed by the Fox River at Berlin and the Wolf River at New London. The relative magnitude in concentrations was related directly to the amount of agriculture in their basins (table 2). In general, TP concentrations measured in the Fox River at Oshkosh entering Lake Winnebago, measured within the lake, and measured in the Fox River at Neenah Menasha leaving the lake were all similar (fig. 7).

Total annual and seasonal (May–September) P loads and yields for the five monitored river sites are given in table 5. The annual yields from each subbasin were directly related to the amount of agriculture in their basins (table 2); highest yields (about 40 kilograms per square kilometer [kg/km^2]) were from the Fond du Lac River that has the most agriculture in its basin, moderate yields were from the Fox River at Berlin (about 25 kg/km^2), and lowest yields (about 19 kg/km^2) were from the Wolf River and New London that has the least agriculture in its basin. Annual yields immediately upstream and downstream from Lake Winnebago (about 19–22 kg/km^2) were lower than from most of the sites upstream in the basin, which indicates part of the P was deposited in the lakes.

The Fox River at Oshkosh was the largest tributary input into Lake Winnebago, contributing, on average, about 300,000 kilograms per year (kg/yr) (tables 5 and 8). Slightly more P left Lake Winnebago through the Fox River at Neenah Menasha over the entire year than entered through the Fox River at Oshkosh; however, inputs from the Fox River at Oshkosh were similar to those leaving the Fox River at Neenah Menasha during May–September.

Total nonpoint watershed loading.—Either drainage-area ratios or SWAT-load ratios were used to extrapolate the measured P loads at the upstream sites to the total nonpoint watershed P loads (not including input from point sources) into each lake for WY2009–11 (table 9). SWAT-load ratios were based on the total nonpoint P load to a lake estimated with SWAT for WY2009–11 divided by the total P load estimated at an upstream gaging station with SWAT for the same period (The Cadmus Group, 2018). For May–September P loads, a SWAT-load ratio of 1.658 was used for extrapolating P load from the Wolf River at New London to the total nonpoint surface-water

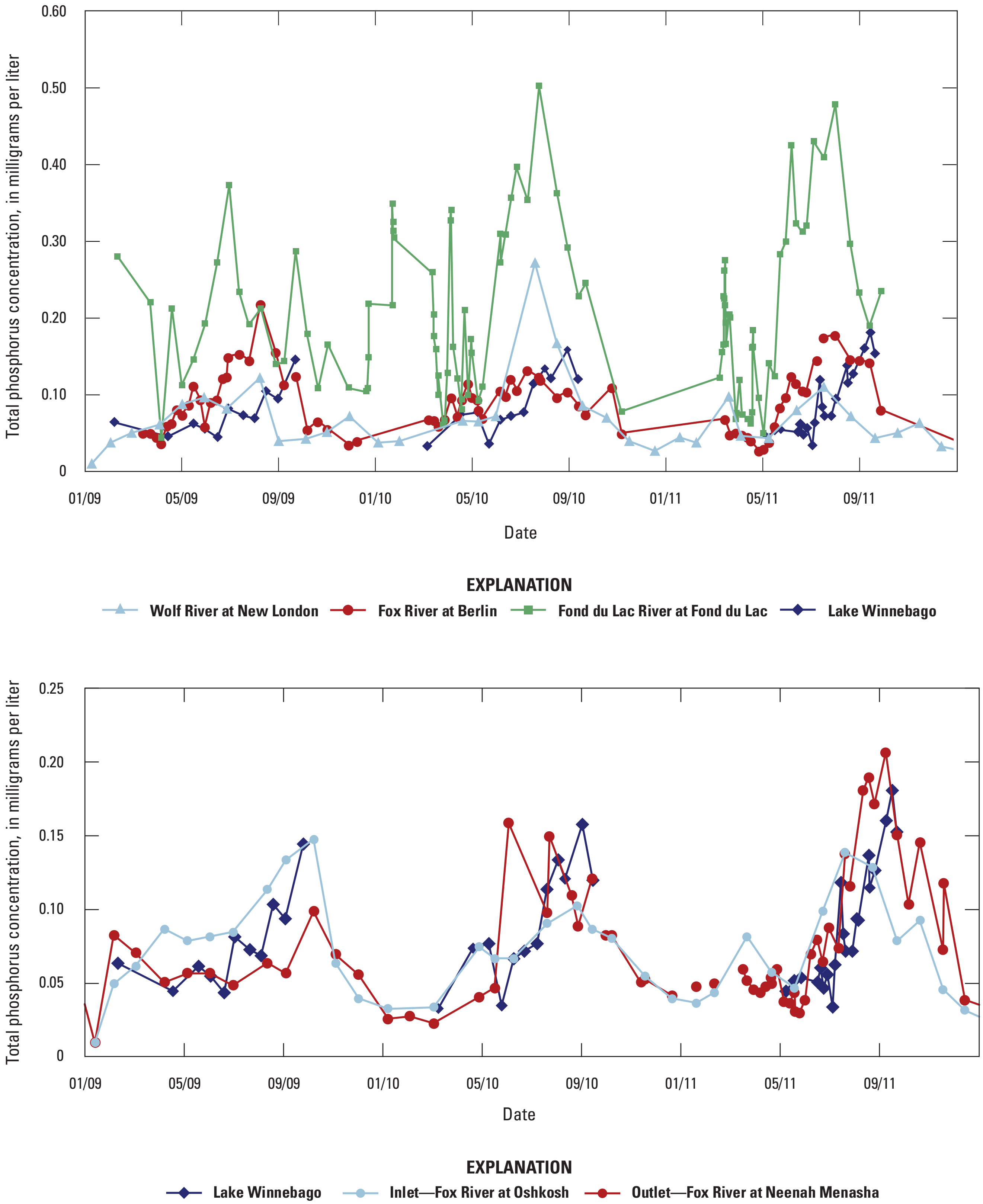


Figure 7. Total phosphorus concentrations measured in rivers throughout the Winnebago Pool watershed and in Lake Winnebago, Wisconsin, 2009–11.

Table 8. Summary of the phosphorus-budget components for Lake Winnebago, Wisconsin, water years 2009–14.

[NE, not estimated]

Year	Fox River	Fond du Lac River	Nearshore	Groundwater	Precipitation	Internal loading diffusion	Other internal sources	Septic	Point sources[1]	Total input	Total output Fox River at Neenah Menasha
Annual phosphorus loading											
Phosphorus loading, in kilograms											
2009	216,000	17,400	5,150	629	13,600	NE	NE	158	8,240	261,000	229,000
2010	272,000	28,000	8,300	629	19,800	NE	NE	158	8,240	337,000	320,000
2011	390,000	16,800	4,970	629	14,200	NE	NE	158	8,240	435,000	357,000
2012	247,000	11,100	3,300	631	16,700	NE	NE	158	8,260	287,000	328,000
2013	298,000	15,100	4,480	629	18,000	NE	NE	158	8,240	345,000	361,000
2014	461,000	16,400	4,872	629	20,600	NE	NE	158	8,240	512,000	481,000
Average 2009–14	314,000	17,500	5,179	629	17,200	NE	NE	158	8,240	363,000	346,000
Average 2009–11	293,000	20,700	6,141	629	15,900	NE	NE	158	8,240	345,000	302,000
Average 2009–13	285,000	17,700	5,240	629	16,500	NE	NE	158	8,240	333,000	319,000
Percentage of total input											
2009–14	86.5	4.8	1.4	0.2	4.7	NE	NE	0.0	2.3	100	[2]95.3
2009–11	85.0	6.0	1.8	0.2	4.6	NE	NE	0.0	2.4	100	[2]87.7
2009–13	85.5	5.3	1.6	0.2	4.9	NE	NE	0.0	2.5	100	[2]95.8
May–September phosphorus loading											
Phosphorus loading, in kilograms											
2009	97,900	4,970	1,470	264	6,490	33,900	204,000	66	3,450	252,000	58,000
2010	186,000	17,900	5,310	264	13,400	33,900	204,000	66	3,450	463,000	237,000
2011	200,000	6,780	2,010	264	7,240	33,900	204,000	66	3,450	457,000	185,000
2012	124,000	3,290	976	264	8,730	33,900	204,000	66	3,450	379,000	139,000
2013	158,000	5,720	1,690	264	8,450	33,900	204,000	66	3,450	415,000	184,000
2014	245,000	9,800	2,900	264	11,200	33,900	204,000	66	3,450	510,000	309,000
Average 2009–14	168,000	8,080	2,400	264	9,250	33,900	204,000	66	3,450	429,000	185,000
Average 2009–11	161,000	9,890	2,930	264	9,040	33,900	204,000	66	3,450	424,000	160,000
Average 2009–13	153,000	7,740	2,290	264	8,860	33,900	204,000	66	3,450	413,000	161,000
Percentage of total input											
2009–14	39.2	1.9	0.6	0.1	2.2	7.9	47.4	0.0	0.8	100	[2]43.2
2009–11	38.0	2.3	0.7	0.1	2.1	8.0	48.0	0.0	0.8	100	[2]37.7
2009–13	37.1	1.9	0.6	0.1	2.1	8.2	49.2	0.0	0.8	100	[2]38.9

[1]Point sources inputs were estimated based on a daily average input for 2011 for all periods.

[2]Percentage of the total input of phosphorus to Lake Winnebago.

Table 9. Summary of the total flow and phosphorus-budget components for the Winnebago Pool Lakes, Wisconsin, water years 2009–11.

[P, phosphorus; Mm^3, million cubic meter; p, phosphorus; kg, kilogram. Bold text refers to the total external loading and total input to each of the Winnebago Pool Lakes]

Source	Lake Poygan			Lake Winneconne			Lake Butte des Morts			Lake Winnebago		
	Flow (Mm^3)	Total P (kg)	Total P (percent)	Flow (Mm^3)	Total P (kg)	Total P (percent)	Flow (Mm^3)	Total P (kg)	Total P (percent)	Flow (Mm^3)	Total P (kg)	Total P (percent)
Total annual input												
Upstream Pool Lakes	0	0	0.0	2,790	220,000	97.3	2,790	209,000	65.2	3,960	276,000	80.0
Upstream point sources	0	0	0.0	0	5,270	2.3	0	5,390	1.7	17	17,200	5.0
Nonpoint watershed	2,780	205,000	96.6	0	0	0.0	1,270	103,000	32.1	260	26,900	7.8
Precipitation	48	1,740	0.8	16	573	0.3	29	1,060	0.3	437	15,900	4.6
Groundwater	2	106	0.0	1	53	0.0	3	159	0.0	12	629	0.2
Direct Point Sources	13	5,270	2.5	0	117	0.1	3	2,000	0.6	10	8,240	2.4
Septic	0.0	27	0.0	0	13	0.0	0	25	0.0	0	158	0.0
Total external loading	**2,850**	**212,000**	**100.0**	**2,800**	**226,000**	**100.0**	**4,100**	**321,000**	**100.0**	**4,700**	**345,000**	**100.0**
May–September input												
Upstream Pool Lakes	0	0	0.0	1,220	107,000	94.6	1,220.0	100,000	51.3	1,800	153,000	37.0
Upstream pointsources	0	0	0.0	0	2,210	1.9	0.0	2,260	1.2	7	7,220	1.7
Nonpoint watershed	1,220	118,000	83.2	0	0	0.0	568.0	64,300	32.9	91	13,500	3.3
Precipitation	27	987	0.7	9	324	0.3	16.6	603	0.3	248	9,040	2.2
Groundwater	1	44	0.0	0	22	0.0	1.3	67	0.0	5	264	0.1
Point sources	5	2,210	1.6	0	49	0.0	1.2	838	0.4	4	3,450	0.8
Septic	0	11	0.0	0	6	0.0	0.0	11	0.0	0	66	0.0
Total external loading	1,250	121,000	85.5	1,230	110,000	96.9	1,810.0	169,000	86.1	2,150	187,000	45.0
Internal loading[1]	0	20,400	14.5	0	3,500	3.1	0.0	27,100	13.9	0	228,000	55.0
Total input	**1,250**	**141,000**	**100.0**	**1,230**	**113,000**	**100.0**	**1,810.0**	**196,000**	**100.0**	**2,150**	**415,000**	**100.0**

[1]Internal loading estimates obtained from BATHTUB modeling.

P load into Lake Poygan, and 1.125 was used for extrapolating the P load at the Fox River at Berlin to the total nonpoint load into Lake Butte des Morts (table 7). A ratio of 0.369 was used for extrapolating the P load measured at the Fond du Lac River site to the ungaged nearshore P load to Lake Winnebago. This value was based on the SWAT estimated nonpoint P load for the nearshore area (3,650 kg; table 7) divided by the total load for the Fond du Lac River (9,890 kg; table 8). For annual loads, a SWAT ratio of 1.642 was used for extrapolating measured P load at the Wolf River at New London to the total nonpoint surface-water P load into Lake Poygan, 1.124 was used for extrapolating the P load at the Fox River at Berlin to the total P load into Lake Butte des Morts (table 7). A ratio of 0.296 was used for extrapolating the annual P load for the Fond du Lac River (20,700 kg; table 8) to the ungaged nearshore P load to Lake Winnebago from SWAT (6,140 kg; table 7). Total point sources of P above each of the monitored sites only represented 4.1 to 6.2 percent of the total P load if 100 percent of the point source contributions were delivered to the monitored location; therefore, these inputs were not included in this extrapolation process. To estimate the total P loads from the entire Fond du Lac River Basin, a 1.011 drainage-area ratio was applied to the load measured at the upstream gage.

The total nonpoint surface-water P loads to Lake Winnebago from the Fond du Lac River and the remaining nearshore areas are given in table 8, and the total nonpoint surface-water P loads to all four lakes are given in table 9.

Highest nonpoint watershed P loading into the Pool Lakes was from the Wolf River into Lake Poygan (table 9; fig. 8), even though the Wolf River had relatively low TP concentrations (table 5). The second highest tributary input came from the Fox River into Lake Butte des Morts. Nonpoint tributary loading to Lake Winnebago was relatively low even though the Fond du Lac River had highest TP concentrations (table 5) because most of surface-water loading to Lake Winnebago comes from the upstream lakes. The relatively low loads from the Fond du Lac River were because of its small drainage area and low flow volumes (table 5). The area directly around Lake Winnebago had low P loads (based on SWAT simulations) because of low estimated TP concentrations. No nonpoint watershed input was estimated into Lake Winneconne.

Precipitation

Atmospheric input of P to the Pool Lakes was determined from the volume of precipitation on surface of each lake and a constant P concentration of 0.036 mg/L. This concentration was obtained from the total precipitation volume and an annual deposition rate of 0.3 kilogram per hectare per year, which was applied to Wisconsin lakes by Panuska and Kreider (2003). This resulted in about 16,000 kg/yr to Lake Winnebago, but it varied from 13,600 kg/yr in WY2009 to 20,600 kg/yr in WY2014 (tables 8 and 9; fig. 8).

Groundwater

Input of P from natural groundwater inflow and septic systems were estimated separately. The total P inputs from natural groundwater inflows were computed from the estimated flows from a regional groundwater-flow model for the Lake Michigan Basin (Feinstein and others, 2010) and an assumed P concentration of 0.053 mg/L, which was based on concentrations measured monthly for 2 years in several shallow piezometers around the periphery of Nagawicka Lake (about 100 km south of Lake Winnebago) (Robertson and others, 2007). This resulted in a total input of 629 kg/yr of P from groundwater to Lake Winnebago, 159 kg/yr to Lake Butte des Morts, 106 kg/yr to Lake Poygan, and 53 kg/yr to Lake Winneconne (tables 8 and 9; fig. 8).

Septic Systems

P input from near-lake septic systems (M) was estimated separately from background groundwater inflow using an export coefficient (E_s), soil retention coefficient (S_R=0.7), and the number of people using septic systems around each lake each year (capita years) using equation 11 (Reckhow and others, 1980):

$$M = E_s \times (\text{number of capita years}) \times (1 - S_R), \qquad (11)$$

where

E_s is input of P per capita per year (0.68 kg; Garn and others, 1996).

The number of full-time and seasonal residences with near-lake septic systems (properties adjacent to the lakes) in 2011 were obtained from the East Central Regional Planning Commission (E. Fowle, written commun., 2013). It was assumed that seasonal residences used septic systems 3 months of the year, and full-time and seasonal residences each had three occupants. The number of capita years was then calculated by multiplying the number of residences by the number of persons per household by the fraction of the year occupied.

In 2011, there were 281 full-time residences and 336 seasonal residences using near-lake septic systems around the Pool Lakes, which equated to a total of about 1,096 capita years (table 10). Therefore, the total P load to the Pool Lakes contributed by septic inputs was estimated to be about 224 kg/yr and 158 kg/yr input into Lake Winnebago (table 10; fig. 8).

Point Sources

Inputs of P from point sources throughout the basin during WY2009–13 were obtained from the WDNR (J. Schmidt, written commun., 2014). Only point sources downstream from the monitored gages were included in the summaries in tables 8 and 9. To estimate the total point source contributions

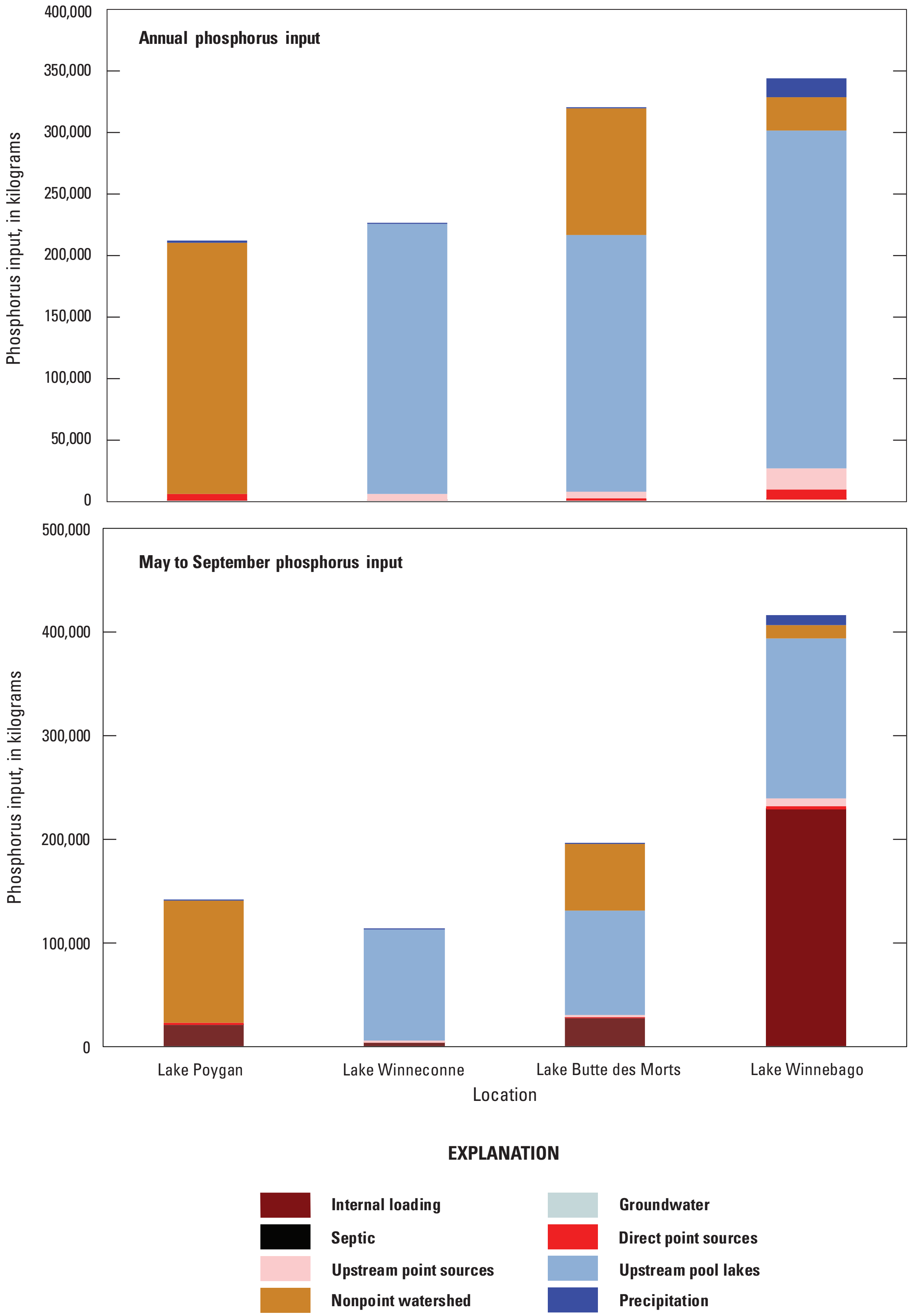


Figure 8. Average annual and average May–September phosphorus inputs into the Winnebago Pool Lakes, Wisconsin, by source, water years 2009–11. Inputs from groundwater and septic systems are not able to be displayed because of their relatively small contributions.

Table 10. Phosphorus input from septic systems into the Winnebago Pool Lakes, Wisconsin, 2011.

[E_S, kilogram input of phosphorus per capita per year; S_R, soil retention fraction; kg/yr, kilogram per year. Bold type indicates total values.]

Pool	E_S	S_R	Full-time residences			Seasonal residences			Total capita years	Septic system input (kg/yr)
			Number	Residents per unit	Fraction of year occupied	Number	Residents per unit	Fraction of year occupied		
Lake Poygan	0.68	0.7	30	3	1	54	3	0.25	131.0	26.7
Lake Winneconne	0.68	0.7	17	3	1	21	3	0.25	66.1	13.5
Lake Butte des Morts	0.68	0.7	35	3	1	26	3	0.25	124.0	25.3
Lake Winnebago	0.68	0.7	200	3	1	234	3	0.25	775.0	158.0
Total	**0.68**	**0.7**	**281**	**3**	**1**	**336**	**3**	**0.25**	**1,100.0**	**224.0**

to each lake (table 9), the point sources were divided into two groups: those in the direct drainage area of a specific lake (direct point sources), and point sources that contribute to upstream Pool Lakes in their basins and could affect their WQ. For example, the upstream point sources for Lake Winnebago includes all the point sources to the other Upper Pool Lakes and those from the city of Oshkosh that is upstream from the Fox River at Oshkosh gage. In estimating the contributions from upstream point sources, it is assumed that point sources above the monitored sites in the basin (upstream from the Wolf River at New London, Fox River at Berlin, and Fond du Lac River at Fond du Lac) were insignificant (they were, however, included in the gaged loads), and it is assumed that 100 percent of the P load from point sources below the monitored sites reached the lake and were then completely passed through the lakes. Summing the P load from point sources and upstream point sources provides an indication of the maximum potential point source contribution to a lake. The direct point sources to Lake Winnebago input was 8,240 kg/yr, and point sources upstream from the Fox River at Oshkosh may contribute about an additional 17,200 kg/yr (table 9), totaling about 25,400 kg/yr (fig. 8).

Upstream Pool Lakes

In a chain of lakes, such as the Winnebago Pool, much of P input into the downstream lakes may come from water released from the adjacent upstream lake. To estimate the amount of P transported downstream from each lake, the flow volume leaving the adjacent upstream lake and the TP concentration of that water are needed. This was only directly measured for Lake Winnebago at the Fox River at Oshkosh. For upstream P loading to Lake Winneconne, the flow from Lake Poygan was assumed to be equal to tributary input to Lake Poygan plus its input from groundwater (that is, it is assumed that precipitation equals evaporation), and the TP concentration was set equal to that measured in Lake Poygan (for estimating May–September loads) or estimated from historical measurements throughout the year in Lake Poygan (for estimating annual loads). To estimate the upstream P loading into Lake Butte des Morts, the flow from Lake Winneconne was assumed to equal the total upstream flow into Lake Poygan plus its groundwater input, and the TP concentration was set equal to that measured (for estimating May–September loads) or estimated throughout the year in Lake Winneconne (for estimating annual loads). The upstream P loading into Lake Winnebago was measured at the Fox River at Oshkosh.

The total upstream P input to Lake Winneconne was 220,000 kg/yr (table 9). The total upstream P input to Lake Butte des Morts was slightly less than this (209,000 kg/yr) because the TP concentration in Lake Winneconne was a little less than that in Lake Poygan. The total upstream P input into Lake Winnebago was 276,000 kg/yr (fig. 8).

Surface-Water Outflow

Phosphorus is discharged out of Lake Winnebago into the Fox River, estimated at Neenah Menasha. Daily P loads into the Fox River at Neenah Menasha were computed using daily TP concentrations linearly interpolated from about monthly measured concentrations. Total annual P output from Lake Winnebago ranged from 229,000 kg in WY2009 to 481,000 kg in WY2014 (table 8). The average annual P output from Lake Winnebago for WY2009–14 was 346,000 kg and 185,000 kg during May through September.

Release of Phosphorus from Lake Sediments

Typically, internal P loading is thought to primarily happen as diffusion of P from the bottom sediments, especially when anoxic conditions are present. However, in shallow lakes, P release from sediment from resuspension of sediments caused by strong winds, fish feeding activities, and fish excretion also may be important. To estimate internal P loading to the Pool Lakes, first the P released from sediments by diffusion was estimated from laboratory P release rates estimated from sediment cores collected from each lake, and then the total P released from diffusion and all other mechanisms, including wind resuspension and fish activity, was computed using a P budget approach for Lake Winnebago and various modeling approaches for all four Pool Lakes.

Three types of information are needed to estimate the amount of P released by diffusion: the area of sediment from which P is being released, characteristics of the sediment surface (aerobic or anaerobic), and the P release rate. It was assumed that P was released only from areas greater than 1 m deep in Lake Winnebago (about 517 km^2 or about 97 percent of the surface area of the lake) because these areas typically accumulate organic sediment. It is important to determine where and how long the sediment interface is anaerobic because P release typically is much higher during anaerobic conditions than during aerobic conditions (Mortimer, 1941).

Rates of P release from aerobic and anaerobic sediments in the Pool Lakes were estimated from 36 sediment cores collected from 15 locations in the Pool Lakes (fig. 1) and analyzed by the WSLH. In Lake Winnebago, aerobic release rates ranged from 0.00 to 1.49 milligrams per square meter per day ($mg/m^2/d$), with an average aerobic release rate of 0.43 $mg/m^2/d$ (table 11). In the three Upper Pool Lakes, average aerobic release rates ranged from 0.48 (Lake Winneconne) to 0.80 $mg/m^2/d$ (Lake Poygan). The average aerobic release rate for all four lakes was 0.55 $mg/m^2/d$. In Lake Winnebago, daily anaerobic release rates ranged from 1.29 to 2.38 $mg/m^2/d$; the average anaerobic release rate was 1.78 $mg/m^2/d$. In the Upper Pool Lakes, average anaerobic release rates ranged from 1.27 (Lake Winneconne) to 1.41 $mg/m^2/d$ (Lake Butte des Morts). The average anaerobic release rate based on data from Lake Winnebago, Lake Butte des Morts, and Lake Winneconne was 1.49 $mg/m^2/d$. Anaerobic release rates were about three to four times greater than the aerobic release rates, and the release rates were comparable among lakes.

Table 11. Phosphorus release rates in the Winnebago Pool Lakes, Wisconsin, August, 2010.

[ID, identification number; mg/m^2/d, milligram per square meter per day; --, does not apply]

Lake	Core ID	Condition	Phosphorus release rate (mg/m^2/d)	Aerobic average by lake (mg/m^2/d)	Anaerobic average by lake (mg/m^2/d)
Winnebago	1A	Aerobic	0.25	--	--
Winnebago	1B	Aerobic	1.21	--	--
Winnebago	2A	Aerobic	0.20	--	--
Winnebago	2B	Aerobic	0.19	--	--
Winnebago	3A	Aerobic	0.39	--	--
Winnebago	3B	Aerobic	0.31	--	--
Winnebago	4A	Aerobic	0.10	--	--
Winnebago	4B	Aerobic	0.05	--	--
Winnebago	5A	Aerobic	0.58	--	--
Winnebago	5B	Aerobic	1.49	--	--
Winnebago	6A	Aerobic	0.37	--	--
Winnebago	6B	Aerobic	0.00	0.43	--
Winnebago	2C	Anaerobic	1.88	--	--
Winnebago	2D	Anaerobic	2.02	--	--
Winnebago	4C	Anaerobic	1.29	--	--
Winnebago	4D	Anaerobic	1.79	--	--
Winnebago	6C	Anaerobic	2.38	--	--
Winnebago	6D	Anaerobic	1.34	--	1.78
Butte des Morts	B1–A	Aerobic	0.80	--	--
Butte des Morts	B2–A	Aerobic	0.47	--	--
Butte des Morts	B3–A	Aerobic	0.45	--	--
Butte des Morts	B3–B	Aerobic	0.48	--	--
Butte des Morts	B4–A	Aerobic	0.32	0.50	--
Butte des Morts	B1–B	Anaerobic	1.05	--	--
Butte des Morts	B2–B	Anaerobic	1.16	--	--
Butte des Morts	B2–C	Anaerobic	1.60	--	--
Butte des Morts	B3–C	Anaerobic	1.60	--	--
Butte des Morts	B4–B	Anaerobic	1.63	--	1.41
Winneconne	W1–A	Aerobic	0.43	--	--
Winneconne	W1–B	Aerobic	0.64	--	--
Winneconne	W2–A	Aerobic	0.37	0.48	--
Winneconne	W2–B	Anaerobic	1.27	--	1.27
Poygan	P1–A	Aerobic	1.35	--	--
Poygan	P2-A	Aerobic	0.69	--	--
Poygan	P2–B	Aerobic	0.53	--	--
Poygan	P3–A	Aerobic	0.61	0.80	--
Overall average of lakes	--	--	--	0.55	1.49

Based on data collected by the continuous recording sensors placed below the buoy on Lake Winnebago (fig. 1), DO rarely dropped below 2 mg/L near the sediment interface; therefore, only aerobic release rates were applied from May through September in Lake Winnebago. It is believed that aerobic conditions existed in the other three Upper Pool Lakes; therefore, only aerobic rates also were applied to each of them.

With an average aerobic release rate from the cores in Lake Winnebago applied from May 1 through September 30, the estimated amount of P released from diffusion from Lake Winnebago sediments was 33,900 kg (table 8). The amount of P released from diffusion before May 1 and after September 30 is expected to be low because release rates decrease when the water temperature is cool during spring, fall, and winter (James and Barko, 2004). In addition, during nonsummer months, it is expected that sedimentation of P would dominate over any diffusion of P from the sediments.

Accuracy of the Phosphorus Budget and Total Sediment P Release

To determine the accuracy of the initial P budget for Lake Winnebago, and whether additional sources of P might be missing from the budget (primarily additional P release from the sediments), the estimated changes in TP concentrations in Lake Winnebago with equation 5 (computed by dividing the daily estimated P mass in the lake by the corresponding daily lake volume) were compared with TP concentrations measured in or just downstream of Lake Winnebago during WY2009–14 (fig. 9). Equation 5 was applied starting on the closest date prior to May 1 of each year having a measured in-lake TP concentration. Initial TP concentrations were obtained as part of routine WDNR monitoring in 2009 through 2011, or from the outlet of the lake in 2012 through 2014. Because only limited sampling was done in the lake in 2012–14, additional WQ data collected at the outlet of the lake (assuming they represented TP concentrations in the lake) also were compared to the estimated P concentrations in figure 9. Since all the other terms in equation 5 could be accurately estimated, especially the major external P contributors, it is believed that any consistent discrepancy in measured and estimated TP concentrations in Lake Winnebago was because the P release on the basis of only diffusion was inadequate at describing the total net sediment P release from the bottom sediments.

Estimated in-lake TP concentrations in Lake Winnebago from about May 1 through September 30 for 2009–14, computed using equation 5 with sediment P release rates estimated on the basis of internal P loading only from diffusion from aerobic conditions, were consistently much less than those measured in the lake (fig. 9). The underestimation in TP concentrations indicate that although the TP concentrations in the Fox River at Oshkosh, which is the primary input to the lake, were close to those measured near the center of the lake (fig. 7), the contributions from all the defined P sources were not sufficient to cause the observed increase in in-lake TP concentrations. This indicates that sources of P in the P budget were underestimated or missing, most likely associated with additional internal P loading. To estimate this additional internal loading, internal P loading was increased until the estimated TP concentrations in the lake matched those measured during May through September in WY2009–14. To approximate the measured concentrations in the lake (or at its outlet) in figure 9, the total internal P loading had to be increased from 0.43 to 3.00 mg/m^2/d (an increase of 2.57 mg/m^2/d), which may have been caused by resuspension of bottom sediment from wind and fish feeding and excretion. Therefore, total net sediment release from diffusion and other internal sources during May through September was about 238,000 kg/yr (table 8) and seems to be greatly underestimated from laboratory diffusion rates alone (33,900 kg). Part of this P from internal loading may be a result of errors in the other components of the P budgets; however, given that most of the P load is from monitored tributary loading (Fox River and Fond du Lac River) and input from precipitation (that during the 6-year mass-balance period simulated the water level of Lake Winnebago accurately; fig. 5), this estimate of P from internal loading should be quite reliable. All of the data from the mass balance and estimation of internal P loading for Lake Winnebago are available at https://doi.org/10.5066/P9Y8BE4H (Robertson and Kennedy, 2018)

Summary of the Phosphorus Budgets for the Winnebago Pool Lakes

During WY2009–14, the average annual input of P from external sources into Lake Winnebago was 363,000 kg. Additional P is contributed by the sediments during summer months (a net contribution of about 238,000 kg during May–September); however, during nonsummer months a similar or slightly larger amount of P is deposited back into the sediments of the lake. The largest external source of P to Lake Winnebago was from the Fox River, which delivered about 87 percent of the total annual external P load (table 8), part of which represents upstream point sources. Contributions from direct point sources represent about 2.4 percent of the total annual external P load, but upstream point sources (including the Oshkosh wastewater treatment plant) could represent an additional 5.0 percent of the annual external P load (table 9). For the entire year, on average 346,000 kg of P were exported out of Lake Winnebago (table 8); therefore, there seems to be a small net retention of P into the sediments of Lake Winnebago.

During May–September, net internal P loading was very important to the P budget for Lake Winnebago because most of the P input from internal sources (bottom sediment and excretion) is during this time and represents more than 55 percent of the total P input into Lake Winnebago. Because of the large contribution from internal sources during May–September, all the external sources of P become relatively

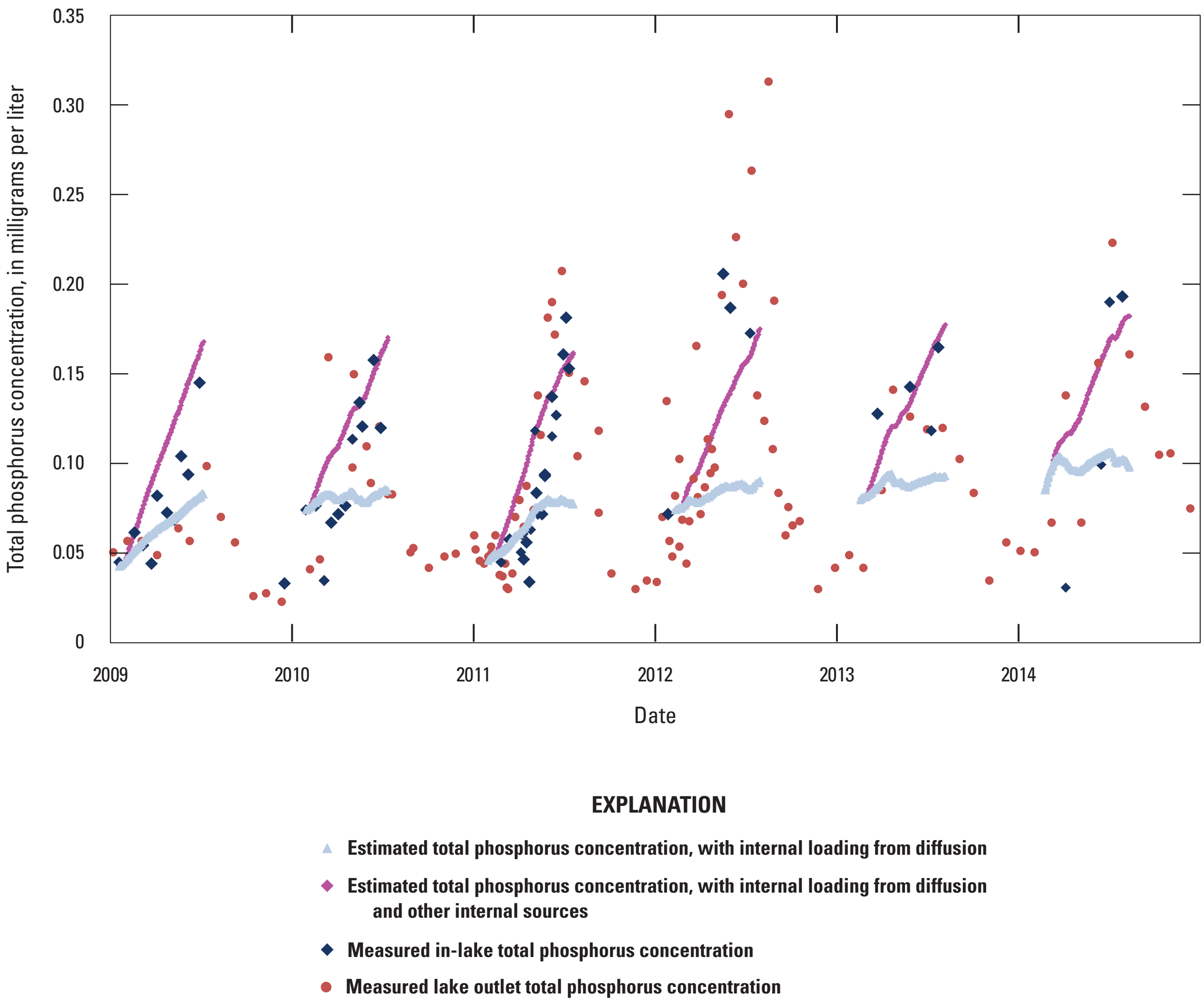


Figure 9. Measured and estimated near-surface total phosphorus concentrations in Lake Winnebago, Wisconsin, during 2009–14 using the mass-balance approach.

less important: Fox River contributed about 38 percent, direct point sources contributed 0.8 percent, and precipitation contributed about 2.2 percent of the total P input to the lake (table 8).

The relative importance of the various P sources varies greatly among lakes and season. For Lake Poygan, the primary P source is from the Wolf River (nonpoint watershed), and the secondary source during summer is from internal loading (fig. 8). For Lake Winneconne, the primary P source is from Lake Poygan (its upstream pool lake). For Lake Butte des Morts, the primary source is from Lake Winneconne (its upstream pool lake), and the secondary source is from the Fox River (nonpoint watershed). For Lake Winnebago, the primary source for the entire year is from its upstream pool lakes, but for the summer months it is from internal loading. For each of the lakes, the importance of internal loading increases when only the May–September period is considered.

Simulated Changes in Water Quality in Response to Changes in Phosphorus Loading

Water-Quality Modeling with BATHTUB

The BATHTUB model was used to estimate the expected changes in WQ in the Winnebago Pool Lakes in response to changes in P loading. In applying BATHTUB, first the average conditions for 2009–11 were used to calibrate the model, and then 21 scenarios were simulated with the calibrated model (table 12). Flow and P loading information from May–September 2009–11 were used to calibrate BATHTUB because this is when the most detailed WQ data were collected in the lakes and throughout the watershed. In model calibration, the measured daily loads within the basin during 2009–11 were extrapolated to the entire basin using SWAT ratios. After calibration, Scenario 1 simulated the average watershed conditions for 2009–13 based on full SWAT-estimated loadings from the watershed. Results from Scenario 1 are used as base conditions for which to compare results from all other scenarios. Nine scenarios (2–10) were used to simulate decreases in all P loading except inputs from precipitation and natural groundwater by 75, 73, 70, 65, 60, 50, 40, 25, and 10 percent; and five scenarios (11–15) simulated increases in all P loading except inputs from precipitation and natural groundwater by 10, 25, 50, 75, and 100 percent. Results from scenarios 1–15 are used to determine the amount of P load reductions that are required for the summer-average TP to reach 0.040 mg/L and thus be able to delist each of the lakes.

Scenarios 16–21 were then used to examine the effects of specific anthropogenic changes in P loading:

- Scenario 16 simulated the effects of removing all anthropogenic P sources, and therefore should represent the best WQ conditions possible in the lakes (that is, reference conditions). In this simulation, TP concentrations in all tributaries to the lakes were set to 0.020 mg/L, which is approximately the estimated median reference concentration determined by Robertson and others (2006) for streams in central Wisconsin. All inputs from point sources and septic systems were eliminated, and internal P loading was reduced by a percentage similar to the reduction in all other controllable P sources (78- to 80-percent reduction depending on the lake).

- Scenario 17 was used to describe how much of the anthropogenic inputs could be added back into the system in addition to reference loading (scenario 16) and yet maintain summer-average TP concentrations at or below 0.040 mg/L. To do this, all the anthropogenic sources were systematically added to the reference loadings until the summer-average TP concentration just reached 0.040 mg/L. In this process, internal P loading was changed by a percentage similar to the change in all other controllable P sources from base conditions.

- Scenario 18 simulated the effects of what would happen if all the tributaries were to reach TP concentrations similar to their P criterion of 0.075 mg/L for wadeable streams in Wisconsin (Wisconsin Department of Natural Resources, 2017a). Internal P loading was reduced by a percentage similar to the reduction in all other controllable P sources (18- to 26-percent reduction depending on the lake).

- Scenario 19 simulated the effects of removing inputs from all point sources throughout the Winnebago Pool basin. Internal P loading was reduced by a percentage similar to the reduction in all other controllable P sources (2- to 6-percent reduction depending on the lake).

- Scenario 20 simulated the effects of reducing all P sources, except precipitation and natural groundwater, by 73 percent (this percentage reduction was determined to be the reduction in loading required for Lake Winnebago to reach 0.040 mg/L), while maintaining internal P loading similar to that estimated during base conditions. The WQ from this simulation is what would be expected immediately after a 73-percent reduction in external P loading, and before internal P loading would respond.

- Scenario 21 simulated the effects of only removing all the P contributed by internal loading, while keeping all other inputs similar to base conditions. This simulation examined the effects of internal P loading independently of all other changes.

Table 12. BATHTUB simulation results. Results of changes in phosphorus loading to the Winnebago Pool Lakes, Wisconsin, simulated with BATHTUB for June–September for total phosphorus, chlorophyll *a*, and Secchi depth in Lake Winnebago. Percent changes in the total phosphorus load, including inputs from internal loading, are based on changes in only potentially controllable sources (i.e., inputs from precipitation and groundwater were unchanged). All changes in internal phosphorus loading were similar to percent changes in all controllable sources. Simulations were also performed with an additional 25-percent reduction in internal loading.

[P, phosphorus; kg, kilograms; TP, total phosphorus; conc., concentration; mg/L, milligram per liter; µg/L, microgram per liter; m, meter; —, does not apply; RD, results dependent upon changes in Upper Pool Lakes. Base conditions for all comparisons are in bold font.]

Scenarios	Change in internal phosphorus loading similar to change in all controllable sources							With an additional 25-percent reduction in internal loading			
	Scenario number	May–September P load (kg)	TP conc. (mg/L)	Chlorophyll-*a* conc. (µg/L)	Secchi depth (m)	Annual P Load (kg)	TP conc. (mg/L)	May–September P load (kg)	TP conc. (mg/L)	Chlorophyll-*a* conc. (µg/L)	Secchi depth (m)
Measured		422,000	0.097	30.5	1.2	573,000	0.097	—	—	—	—
Calibration		422,000	0.097	29.1	1.2	573,000	0.095	—	—	—	—
General response corresponding to percentage change in P load											
–75 percent	2	116,000	0.037	16.2	2.0	143,000	0.039	99,000	0.033	14.6	2.1
–73 percent	3	124,000	0.039	17.0	1.9	155,000	0.041	106,000	0.035	15.3	2.1
–70 percent	4	136,000	0.042	17.9	1.8	172,000	0.044	117,000	0.037	16.3	2.0
–65 percent	5	156,000	0.047	19.4	1.7	200,000	0.049	133,000	0.042	17.7	1.8
–60 percent	6	176,000	0.052	20.7	1.6	229,000	0.053	150,000	0.046	19.0	1.7
–50 percent	7	216,000	0.060	22.8	1.5	286,000	0.062	183,000	0.053	21.1	1.6
–40 percent	8	255,000	0.068	24.6	1.4	344,000	0.070	215,000	0.060	22.9	1.5
–25 percent	9	313,000	0.079	26.6	1.3	430,000	0.081	264,000	0.070	25.0	1.4
–10 percent	10	370,000	0.090	28.2	1.2	516,000	0.091	312,000	0.079	26.6	1.3
0 percent–Base	**1**	**408,000**	**0.096**	**29.0**	**1.2**	**573,000**	**0.097**	343,000	0.085	27.5	1.3
+10 percent	11	445,000	0.103	29.8	1.2	630,000	0.103	374,000	0.090	28.3	1.2
+ 25 percent	12	510,000	0.112	30.7	1.1	716,000	0.112	420,000	0.098	29.3	1.2
+ 50 percent	13	593,000	0.126	31.9	1.1	859,000	0.125	497,000	0.111	30.6	1.1
+ 75 percent	14	685,000	0.139	32.9	1.1	1,000,000	0.138	572,000	0.123	31.7	1.1
+ 100 percent	15	775,000	0.152	33.7	1.1	1,150,000	0.150	646,000	0.134	32.5	1.1
Specific natural and anthropogenic scenarios											
Reference conditions	16	95,000	0.032	14.2	2.2	RD	RD	81,363	0.028	12.7	2.4
Reference +9 percent (and +16 percent) anthropogenic sources	17	124,000	0.040	17.0	1.9	RD	RD	124,000	0.040	17.0	1.9
0.075 mg/L in all tributaries	18	308,000	0.078	26.5	1.3	RD	RD	259,000	0.069	24.8	1.4
Base with no point sources	19	398,000	0.093	28.6	1.2	RD	RD	333,000	0.084	27.3	1.3
73-percent reduction, full internal loading	20	318,000	0.080	26.8	1.2	RD	RD	252,000	0.068	24.5	1.4
Base with no internal loading	21	146,000	0.045	18.7	1.7	RD	RD	146,000	0.045	18.7	1.7

When BATHTUB is used to simulate the effects of changes in external loading, it is usually assumed that internal P loading to the lake will either be unaffected or it will eventually come to a new equilibrium with its new external P loading with a change directly proportional to the percentage change in all nonsediment sources (Robertson and others, 2018). Therefore, in all simulations, except those specifically looking at the effects of modifying internal P loading (scenarios 20 and 21), net internal loading was changed by a percentage similar to the change in all other external controllable P sources. The resulting changes in internal P loading are a little less than the total changes in external P loading to the lake because inputs form precipitation and groundwater were unchanged. This assumption results in net internal P loading reaching 0 mg/m^2/d before total external loading reaches 0 kg; with a 75-percent reduction in controllable sources, internal P loading into Lake Winnebago was estimated to be 0.70 mg/m^2/d compared to 0.82 mg/m^2/d if the change in internal P loading was based on the percentage of change in all external P loading to the lake. Thus, most of the outputs from the BATHTUB simulations should represent conditions in the future when the lake sediments come to equilibrium with their external P inputs. It is difficult to determine when these simulated changes would actually happen with BATHTUB, but this is examined later with the Jensen model.

BATHTUB also was used to examine the effects of reducing internal P loading at a rate more than that for the other P sources to simulate what may happen if additional in-lake actions also are done. These may include actions that promote macrophyte growth and thus reduce wind induced resuspension or that reduce the population of rough fish, such as carp and thus reduce sediment resuspension during feeding and reduce P input from excretion. To examine the effects of reducing internal loading at a rate more than that projected to occur to external sources, all 21 scenarios were again simulated but with an additional reduction in the internal P loading to each lake by 25 percent. It is difficult to estimate the magnitude of these types of actions that would be required to achieve a specific additional P load reduction. The main reason for doing these scenarios is to illustrate that substantial in-lake management also may be required to meet WQ goals even if substantial watershed load reductions are achieved.

Data Requirements

Four types of data are required as input into BATHTUB: morphometric data (table 1), hydrologic data (table 9), P loading data (table 9), and WQ data needed to calibrate and verify the model (table 3). The BATHTUB model was set up with five interconnected lakes: Lake Poygan, Lake Winneconne, Lake Butte des Morts, widening of the Fox River, and Lake Winnebago. The widening of the Fox River between Lakes Butte des Morts and Winnebago was included as an additional lake to enable changes to happen between Lakes Butte des Morts and Winnebago. Lake Winnebago was treated as one well-mixed lake. The period for the hydrologic and P loading used in BATHTUB is dependent on the P-turnover ratio (number of times the P mass in the lakes is replaced in a specific period). Because of the short residence times of the Upper Pool Lakes, flow and P loadings for May through September were used as input to BATHTUB, which resulted in a P-turnover ratio of about 2, which is preferred for model calculations (Walker, 1996). The relations within BATHTUB simulate WQ during the summer growing season; therefore, BATHTUB was calibrated to WQ measured from June through September. Because detailed in-lake WQ data for all four lakes were only available for 2009–11, average flow and WQ data for these years were used to calibrate the relations in BATHTUB. The average precipitation and evaporation for May through September 2009–11 were 0.468 and 0.714 m, respectively. All input data used in the BATHTUB model and results from the model are available at https://doi.org/10.5066/P9Y8BE4H (Robertson and Kennedy, 2018).

Because Lake Winnebago had a relatively long residence time compared to the other lakes, the Canfield and Bachmann (1981) natural-lake equation, the algorithm selected within BATHTUB (eq. 6), also was used with flow and P inputs for the full year (table 9) to determine if changing its averaging period affected its simulated response to the changes in P loading in scenarios 1–15 (table 12). In this application, total P load was set to 573,000 kg (sum of all annual P loads in table 8, but with the internal P load estimated with the seasonal BATHTUB model), the mean depth ($\bar{z}$) was set to 4.54 m, and residence time (τ) was set to 0.51 year (table 1).

Phosphorus from internal P loading is not typically included as a source in BATHTUB, because most empirical models already indirectly incorporate this source. Additional internal loading, therefore, is typically only added when a lake or reservoir has abnormally high internal loading (Walker, 1996). It was believed that all the Pool Lakes have abnormally high internal loading, as was shown for Lake Winnebago (table 8; fig. 9); therefore, additional internal P loading was considered for each of the Pool Lakes. Internal P loading to each of the Pool Lakes was then estimated using BATHTUB by adjusting their internal P loading rate until the estimated average TP concentrations in each lake matched those measured during 2009–11. The resulting internal P loads for each lake were then used to complete their respective May–September P budgets in table 9.

Algorithms and Calibration

When applying BATHTUB, dispersion coefficients can be used to describe the upstream exchange of water with adjacent lakes or basins. A dispersion coefficient of 0.0 was applied between all adjacent basins, which indicates no upstream mixing with the adjacent upstream lakes. Almost all the algorithms selected in this application of BATHTUB were the default algorithms that have been shown to usually be appropriate for most lakes and reservoirs (Walker, 1996), except the algorithm to simulate changes in TP. The Canfield-Bachmann natural lake equation (algorithm 8 in BATHTUB) has worked well in simulating TP concentrations in lakes throughout Wisconsin (Robertson and others, 2012, 2018; Robertson and Rose, 2008; Juckem and Robertson, 2013); therefore, it was used in this application. Chl-*a* was simulated using estimated TP, residence time, and model-estimated light limitation (algorithm 2 in BATHTUB). SD was simulated with algorithm 1 in BATHTUB, which used estimated Chl-*a* and model-estimated nonalgal turbidity.

With average May–September loading data (flow and TP concentrations) from 2009 to 2011 and internal P loading rates based on laboratory aerobic release rates, BATHTUB was used to simulate the summer-average (June through September) WQ in each lake. The initial simulated (precalibration) TP and Chl-*a* concentrations were consistently less than those measured in the lakes, and simulated SDs were greater than to those measured (fig. 10). These differences between measured and simulated WQ indicate that the total P loads to the lakes were likely underestimated when aerobic internal P release rates were used to estimate the net internal P loading because of it not incorporating inputs from other internal sources; therefore, the internal P loading to all Pool Lakes were increased until the simulated TP concentrations (postcalibration) matched those measured in 2009–11 (fig. 10). The rates of net internal P loading were increased from 0.80 to 2.30 mg/m^2/d in Lake Poygan, increased from 0.48 to 1.20 mg/m^2/d in Lake Winneconne, increased from 0.50 to 5.00 mg/m^2/d in Lake Butte des Morts, and increased from 0.43 to 2.80 mg/m^2/d in Lake Winnebago. For Lake Winnebago, the final estimated internal P loading was a little less than that estimated with the mass balance approach (3.0 mg/m^2/d). The internal P loading rates for Lake Butte des Morts were not used to completely match measured in-lake concentrations because they also were used to try to match downstream P loading measured at the Fox River at Oshkosh, which also is computed in BATHTUB. During this calibration process, it was not possible to match the TP concentrations in Lake Butte des Morts without slightly overestimating the P load entering Lake Winnebago from the Fox River. Therefore, the Fox River segment was added to the model and a calibration coefficient (2.0) was applied to its TP concentration. This calibration factor reduced the TP in the Fox River segment and allowed the model to more closely match the P load measured at the Fox River at Oshkosh.

Based on the refined internal P loading estimates for all four Pool Lakes from BATHTUB, the total May–September P budget for each Pool Lake could be completed (table 9; fig. 8). During WY2009–11, the average May-September P input to Lake Winnebago was 415,000 kg, of which 187,000 kg came from external sources (45 percent) and 228,000 kg came from internal loading in the lake (55 percent). During May–September, net internal loading becomes much more important for each of the lakes. The importance of the net internal P loading varied from 3.1 percent for Lake Winneconne, to 13.9 percent for Lake Butte des Morts, to 14.5 percent for Lake Poygan, and to 55 percent for Lake Winnebago. Although the rate of internal P loading was relatively high in the Upper Pool Lakes, its relative importance to their total P budgets was less than for Lake Winnebago because of the magnitude of the other sources, primarily tributary loading and inputs from the adjacent upstream lake. Internal loading was omitted from the total annual inputs in table 9 because a net loss of P into Lake Winnebago was estimated from its annual P budget (table 8), which also was expected in the other lakes.

After adjusting the net internal P loading in the TP algorithms, the Chl-*a* algorithms for the Upper Pool Lakes in BATHTUB still did not simulate Chl-*a* well. Therefore, calibration coefficients were applied to the Chl-*a* algorithms for Lake Poygan (1.13), Lake Winneconne (1.9), and Lake Butte des Morts (2.2). No calibration was required for simulating Chl-*a* in Lake Winnebago. After adjusting net internal P loading in the TP models and calibrating the Chl-*a* algorithms for the Upper Pool Lakes, no calibration was needed for simulating SDs. The simulated (precalibration and postcalibration) and measured WQ values for 2009–11 are compared in figure 10. After model calibration, BATHTUB accurately simulated TP, Chl-*a*, and SD in all four lakes.

Response in Water Quality to Basinwide Changes in Phosphorus Loading

To estimate the expected response in WQ in the Winnebago Pool Lakes to changes in P loading from all potentially controllable sources (all inputs except those from precipitation and natural groundwater) and to determine the reduction in P loading required to each lake to reach a mean summer TP of 0.040 mg/L, 15 scenarios (1–15) were simulated with the BATHTUB model. Based on these simulations, TP in each Pool Lake should have a relatively linear response to changes in P loading (fig. 11*A*; table 12 for Lake Winnebago only). The changes in in-lake TP, on a percentage basis, are smaller than the changes in external P loading. A 75-percent decrease in all potentially controllable P sources into Lake Winnebago, which equates to a 72-percent decrease in total P loading from all sources, results in a 61-percent decrease in TP in Lake Winnebago; and a 75-percent increase in all potentially controllable P sources, which equates to a 68-percent increase in total P loading from all sources, results in a 45-percent increase in TP in Lake Winnebago.

Based on the TP response curve for Lake Winnebago, it would require a 73-percent reduction in P loading from all its potentially controllable sources to reach an average June–September TP concentration less than 0.040 mg/L (the P criterion for this lake). Based on the TP response curves for the Upper Pool Lakes, a slightly smaller reduction would be required for those lakes (about 60 percent) to reach 0.040 mg/L. However, if a 60-percent reduction in all potentially controllable sources were achieved for the Upper Pool Lakes, then Winnebago would not be able to reach 0.040 mg/L. Therefore, a 73-percent reduction in P input is needed from throughout the basin for all four Pool Lakes to reach 0.040 mg/L.

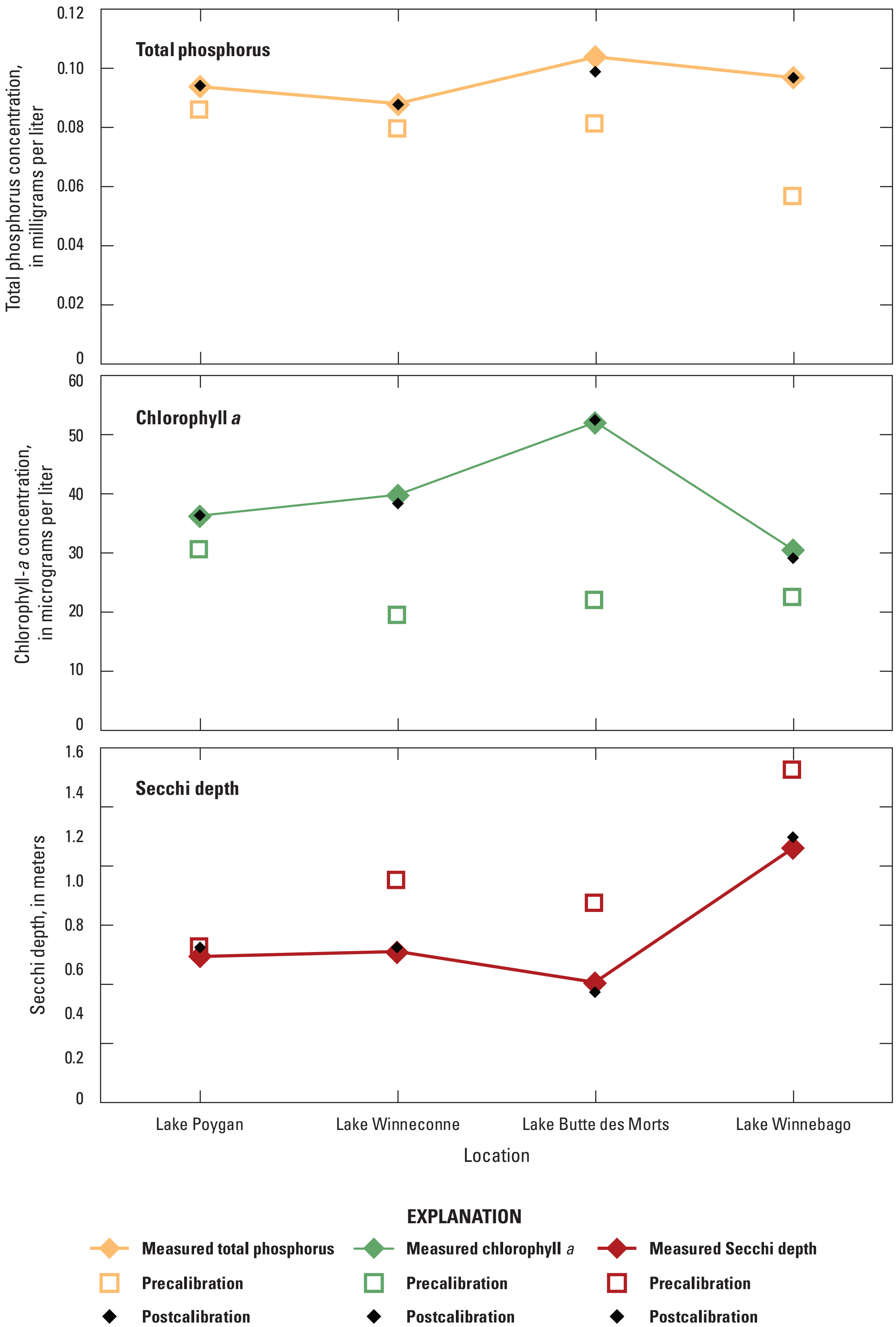


Figure 10. Measured and simulated (with the BATHTUB model) summer-average near-surface total phosphorus concentrations, chlorophyll-*a* concentrations, and Secchi depths in the Winnebago Pool Lakes, Wisconsin.

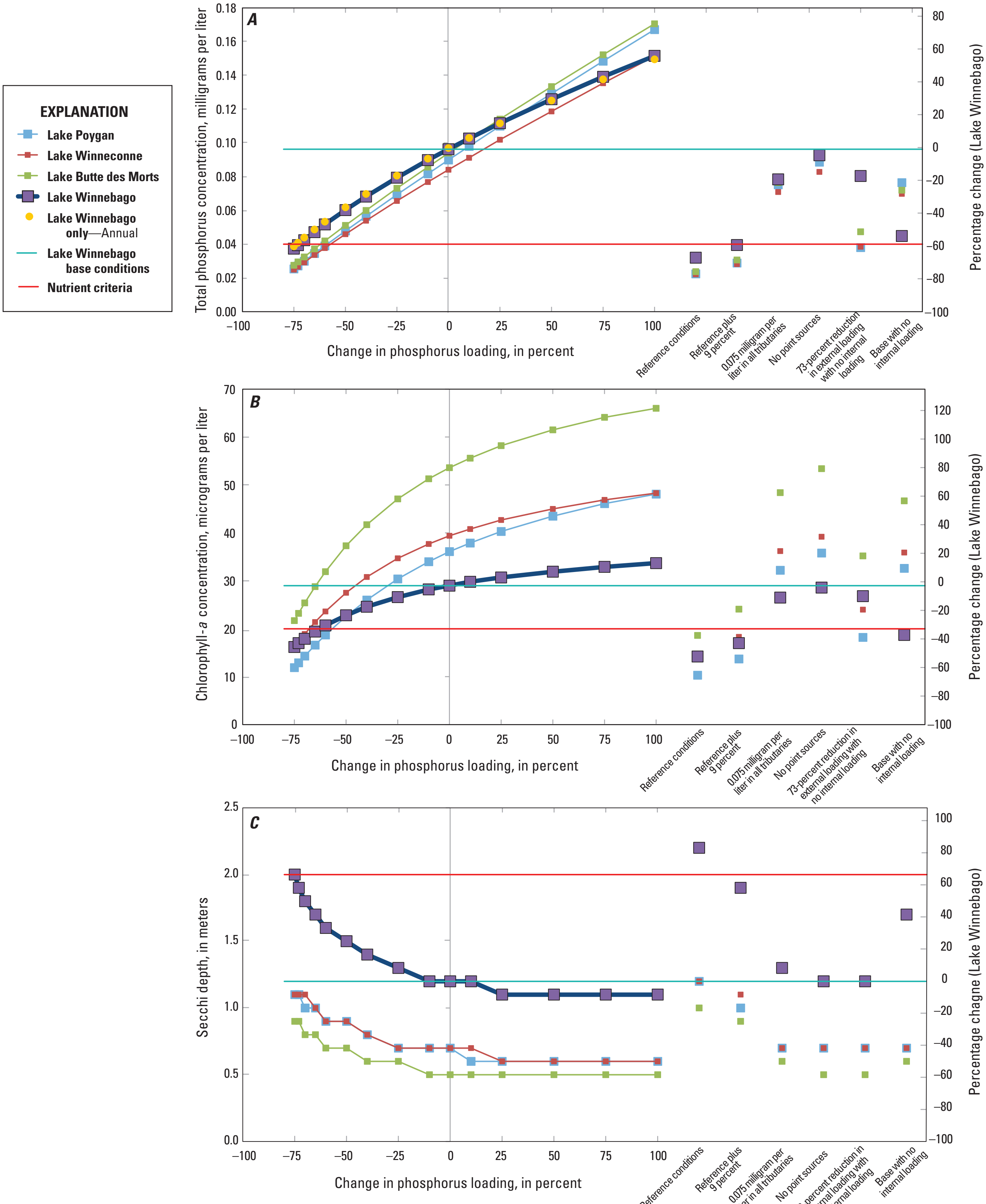


Figure 11. Simulated changes in summer-average water quality in the Winnebago Pool Lakes, Wisconsin, in response to various phosphorus-loading scenarios (table 12) using the BATHTUB model. *A*, Total phosphorus. *B*, Chlorophyll *a*. *C*, Secchi depth.

The Canfield and Bachmann (1981) natural-lake equation, algorithm within BATHTUB (eq. 6,) also was used with flow and P inputs into Lake Winnebago for the full year to determine if changing its averaging period affected its simulated response to the changes in P loading in scenarios 1–15. Equation 6 simulated growing season TP for 2011–13 to be 0.095 mg/L, close to that measured in the lake: 0.097 mg/L. Because of the small discrepancy in measured and simulated values, all simulated TP values were increased by 2 percent. A similar TP response curve to that obtained with BATHTUB using seasonal inputs was obtained for Lake Winnebago when equation 6 was used with flow and P inputs for the full year (fig. 11; table 12). With a 75-percent reduction in controllable P sources, simulated TP was only 0.002 mg/L more than that obtained using seasonal inputs in BATHTUB; and with a 100-percent increase in controllable P sources, the simulated TP was only 0.002 mg/L less than that estimated using seasonal inputs. When annual inputs were considered, a 74-percent reduction in P loading from all potentially controllable sources to Lake Winnebago was needed for it to reach an average June–September TP concentration less than 0.040 mg/L, which is almost identical to that determined using seasonal P inputs. Based on the similarity in these responses, similar results would be expected for all other scenarios if annual inputs were used; however, the actual response in Lake Winnebago depends on changes in the Upper Pool Lakes that cannot be simulated using equation 6 independent of the Upper Pool Lakes.

Simulated summer-average Chl-*a* had a nonlinear response to a linear change in P loading (fig. 11*B*; table 12 for Lake Winnebago). The response in Chl-*a* was larger (on a percentage basis) for decreases in P loading than for similar increases in loading. A 75-percent reduction in P loading to Lake Winnebago resulted in a 44-percent reduction in Chl-*a* (from 29.6 to 16.2 µg/L), whereas a 75-percent increase in P loading resulted in an increase of only 13.4 percent. A 73-percent decrease in P loading should reduce summer-average Chl-*a* in Lake Winnebago to 17.0 µg/L.

Simulated summer-average SDs also were more responsive to decreases in P loading than to increases in P loading (fig. 11*C*; table 12 for Lake Winnebago). When P loading to Lake Winnebago was decreased by 75 percent, SD increased to 2.0 m (67-percent increase); however, when loading was increased by 75 percent, SD decreased to 1.1 m (8-percent decrease). A 73-percent decrease in P loading should increase summer-average SD in Lake Winnebago to 1.9 m.

Summer-average TSI values indicate that all Winnebago Pool Lakes are highly eutrophic (TSI values between 60 and 70; table 3). Based on the simulations with BATHTUB, if P loading could be reduced by 60–75 percent from that measured in 2009–11, WQ in all Pool Lakes should improve, but the lakes would still be classified as eutrophic.

Potential Water Quality

To estimate the best potentially achievable WQ in the lakes (that is, reference conditions, scenario 16), TP concentrations in all the tributaries to the lakes were set to 0.020 mg/L (the approximate median reference concentration for streams in central Wisconsin when all agriculture and urban sources of P were simulated to be removed from their basins; Robertson and others, 2006), P inputs from point sources and septic systems were eliminated, and internal P loading was reduced by a percentage similar to the reduction in all the other controllable P sources (about an 80-percent reduction in internal P loading for each lake). Based on BATHTUB results (fig. 11; table 12 for Lake Winnebago), the best possible summer-average WQ should be TP concentrations ranging from 0.022–0.024 mg/L in the Upper Pool Lakes to 0.032 mg/L in Lake Winnebago; Chl-*a* concentrations ranging from 10 µg/L in Lake Poygan, to 14 µg/L in Lakes Winneconne and Winnebago, and to 19 µg/L in Lake Butte des Morts; and SDs ranging from 1.0 m in Lake Butte des Morts, to 1.2 m in Lake Poygan and Lake Winneconne, and to 2.2 m in Lake Winnebago. These conditions are better than the WQ criteria for shallow drainage lakes in Wisconsin (Wisconsin Department of Natural Resources, 2017a).

Effects of Specific Phosphorus Reductions from the Watershed

To determine how much of the anthropogenic P inputs could be added in addition to reference P loading and yet maintain summer-average TP concentrations at or below 0.040 mg/L, all the anthropogenic sources were systematically adjusted at similar rates, and internal P loading was adjusted at a rate similar to the change in all the other potentially controllable P sources. If all TP concentrations in the rivers, point sources, and septic systems increase less than 9 percent from their reference concentrations (and an associated 71- to 73-percent reduction in internal P loading from base conditions), the summer-average TP concentration in Lake Winnebago should be just less than 0.040 mg/L. The final results of this evaluation are described as scenario 17 (table 12; fig. 11).

To determine the potential effects of management strategies in the watershed that are sufficient to reduce the average TP concentrations in all the tributaries to the Pool Lakes to the current P criterion of 0.075 mg/L (scenario 18), all TP concentrations in the tributaries were set to 0.075 mg/L, and internal P loading was reduced by a percentage similar to the change in all the other P inputs (18 percent in internal P loading to Lake Poygan to 26 percent in Lake Winnebago). With all rivers entering at 0.075 mg/L, summer-average WQ should improve: TP concentrations should range from 0.071 to 0.077 mg/L in the Upper Pool Lakes to 0.078 mg/L in Lake Winnebago;

Chl-*a* concentrations should range from 27 µg/L in Lake Winnebago, to 32 µg/L in Lake Poygan, to 36 µg/L in Lake Winneconne, and to 48 µg/L in Lake Butte des Morts; and SDs should range from 0.7 m in the Upper Pools to 1.3 m in Lake Winnebago. These conditions in the Pool Lakes are still worse than the WQ criteria defined for shallow drainage lakes in Wisconsin (Wisconsin Department of Natural Resources, 2017a).

Scenario 19 simulated the effects of eliminating all the P inputs from point sources downstream from the monitored gaging stations. In this scenario, it was initially assumed that internal P loading would not change from baseline conditions; therefore, it would represent the immediate effects of point source removal. With P from all point sources removed (immediately after removing the point sources), summer-average WQ changed little: TP concentrations decreased by about 0.001 mg/L in all lakes, and little changes in Chl-*a* concentrations and SDs were measured. If the elimination of point sources were assumed to also reduce internal P loading by a percentage similar to the reduction in all the other potentially controllable P sources (future equilibrium), then a little more improvement in WQ was determined for Lake Winnebago, but no change was observed for the Upper Pool Lakes. If the elimination of P from point sources also fully affected internal loading in Lake Winnebago by a similar percentage, TP should decrease from 0.096 to 0.093 mg/L, and Chl-*a* should decrease from 29.0 to 28.6 µg/L, but no change in SD should be observed.

Effects of Internal Loading on In-Lake Water Quality

To estimate how important internal P loading is to WQ in the Pool Lakes, two simulations were done: in scenario 21, internal P loading was eliminated from the base simulation; and in scenario 20, the reduction in internal P loading was eliminated from the 73-percent reduction in P loading (scenario 3 but with no reductions in internal P loading). If the internal P loading estimated for 2009–11 could be eliminated but external P loading was left unchanged, WQ in all four Pool Lakes should greatly improve, especially the WQ in Lake Winnebago where the internal P loading was its largest source of P during summer (fig. 8). With internal P loading eliminated, summer-average TP concentrations should range from 0.045 mg/L in Lake Winnebago to 0.070–0.077 mg/L in the Upper Pool Lakes; Chl-*a* concentrations should range from 19 µg/L in Lake Winnebago, to 33 µg/L in Lake Poygan, to 36 µg/L in Lake Winneconne, and to 47 µg/L in Lake Butte des Morts; and SDs should range from 0.6–0.7 m in the Upper Pool Lakes to 1.7 m in Lake Winnebago.

Based on results from scenario 3 (73-percent reduction in all potentially controllable P sources), TP concentrations throughout the Winnebago Pool Lakes should eventually decrease to below 0.040 mg/L; however, these changes and the other changes in WQ will only be observed after the system (internal P loading) comes to equilibrium with its new external P loading. Scenario 20 simulates the immediate effects of a 73-percent reduction in external P sources. In this scenario, all external sources were reduced by 73 percent, but internal P loading was left unchanged (that is, did not have time to equilibrate with its P reduction from the other sources). In the years immediately after a 73-percent reduction in all external P sources, WQ should improve throughout the system (fig. 11), especially in the Upper Pool Lakes where internal P loading was a relatively small source (fig. 8). With a 73-percent reduction in all external loading and no change in internal P loading: TP concentrations should range from 0.038 mg/L in Lake Poygan and Lake Winneconne, to 0.047 mg/L in Butte des Morts, and to 0.080 mg/L in Lake Winnebago; Chl-*a* concentrations should range from 18 to 24 µg/L in Lake Poygan and Lake Winneconne, to 27 µg/L in Lake Winnebago, and to 35 µg/L in Lake Butte des Morts; and SDs should range from about 0.5 to 0.7 m in the Upper Pool Lakes to 1.2 m in Lake Winnebago. Therefore, large reductions in external P sources should result in an almost immediate improvement in WQ, especially in lakes where internal P loading is a relatively small source, but the internal P loading will delay the full effects of the reductions from being observed in the Winnebago Pool Lakes. It is not possible, however, to determine how long it will take the Winnebago Pool Lakes to come into this new equilibrium with BATHTUB.

BATHTUB was then used to examine the effects of reducing the inputs from internal P loading by 25 percent in all the scenarios to simulate what may happen if in-lake actions also are done in addition to any of the external P input changes described in table 12, such as activities to promote macrophyte growth that should reduce the wind resuspension or activities to reduce the population of rough fish that resuspend bottom sediment during feeding and have high P excretion rates.

With a 25-percent reduction in the inputs from internal P loading in all of the scenarios, all the simulated response curves shown in figure 11 shifted toward better WQ and the WQ for each of the individual scenarios improved (fig. 12; table 12). The improvements in WQ were larger for Lake Winnebago than for the other lakes because internal P loading was relatively more important for Lake Winnebago than the other lakes. If internal P loading was reduced by 25 percent, the overall external P load reduction needed for all four Pool Lakes to reach 0.040 mg/L decreased from a 73- to a 67-percent reduction from the loading for base conditions. Potentially achievable WQ in Lake Winnebago also improved: reference summer-average TP decreased from 0.032 to 0.028 mg/L, Chl-*a* decreased from 14.2 to 12.7 µg/L, and a SD increased from 2.2 to 2.4 m. The reduction in the inputs from internal P loading by 25 percent also increased the amount of additional external P loading that could happen above reference conditions and yet maintain a TP concentration in Lake Winnebago just below 0.040 mg/L from 9 to 16 percent.

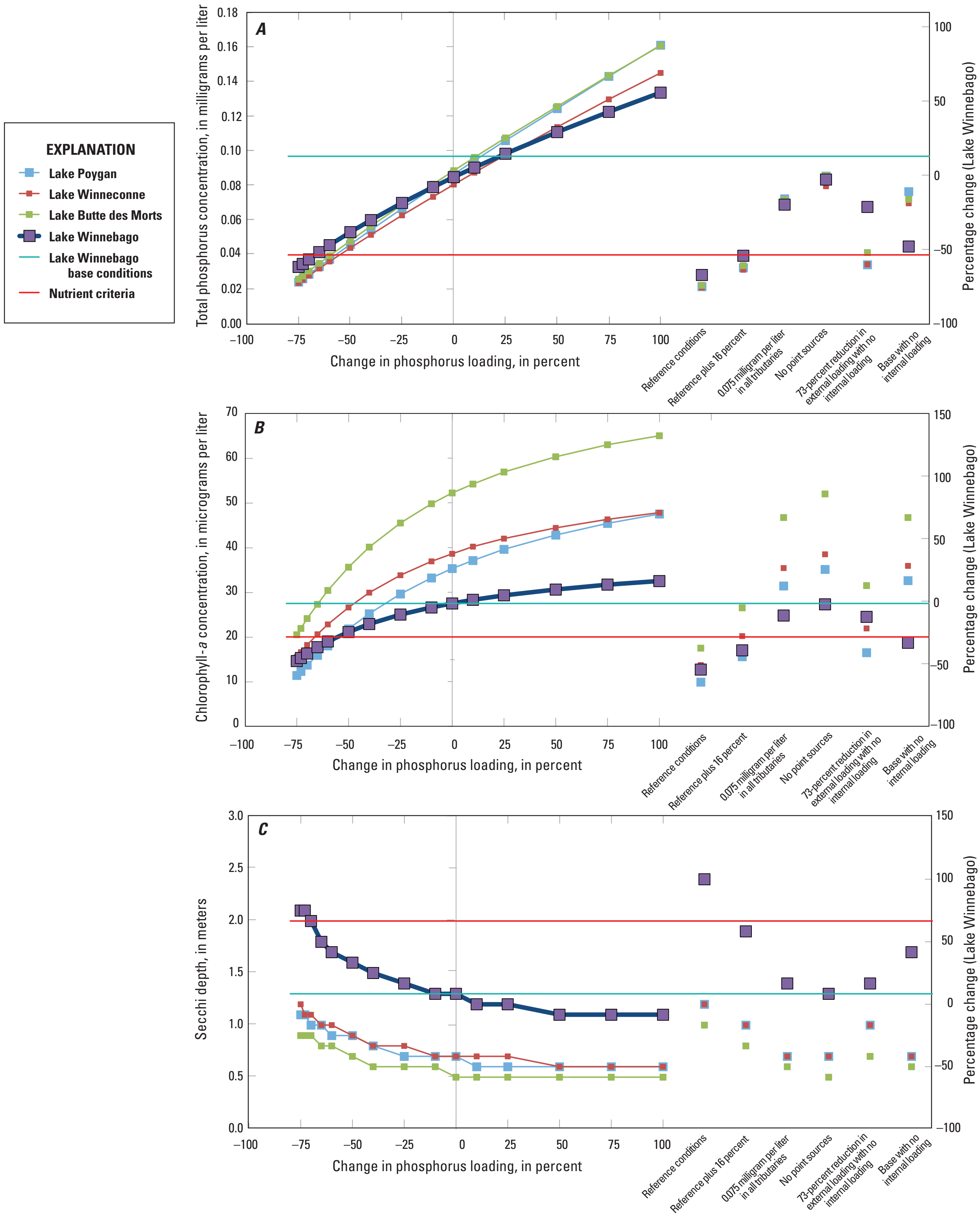


Figure 12. Simulated changes in summer-average water quality in the Winnebago Pool Lakes, Wisconsin, in response to various phosphorus-loading scenarios (table 12), with internal loading reduced by 25 percent using the BATHTUB model. *A*, Total phosphorus. *B*, Chlorophyll *a*. *C*, Secchi depth.

Water-Quality Modeling with the Jensen Model

Results from BATHTUB simulations were used to describe the changes in summer-average WQ that should happen in the future when the internal P loading in the lakes comes into a new equilibrium with the changes in external P inputs; however, BATHTUB does not describe the transient changes in WQ after the changes in external P inputs or the seasonal changes that happen throughout the year. Results from the Jensen model (Jensen and others, 2006) can be used to describe these shorter-term changes in WQ and the temporal changes in internal P loading. In this study, the daily P loadings and measured TP concentrations in the lakes and their outlets during WY2009–13 (extrapolated using SWAT ratios based on annual average flows and loads) were used to calibrate two independent Jensen models: one for the Upper Pool Lakes (Lake Poygan, Lake Winneconne, and Lake Butte des Morts collectively) terminating at the Fox River at Oshkosh, and one for Lake Winnebago terminating in the lake. Then, output from the Upper Pool Lakes Jensen model was used as input to the Lake Winnebago Jensen model to create one linked Jensen model for the entire Pool terminating in Lake Winnebago, but simulating changes at the downstream end of the three Upper Pool Lakes and in Lake Winnebago. Calibrations were based on WY2009–13 for the independent models, and the results are shown for the linked Jensen model. After calibration with extrapolated measured loads from the watershed, 18 P loading scenarios (1–J through 18–J) were simulated with the calibrated linked Jensen model (summarized in table 13). Scenario 1–J was used to simulate WQ in the lakes given the watershed conditions during WY2009–13 but based on loadings from full SWAT estimations, which provides base conditions for which to compare results from all other Jensen simulations. Then, 13 scenarios were used to examine the effects of changes in basinwide inputs: 8 scenarios (2–J through 9–J) simulated the effects of decreases in all P loading (except inputs from precipitation and natural groundwater that remained unchanged) by 80, 75, 70, 69, 65, 50, 25, and 10 percent, and 5 scenarios (10–J through 14–J) simulated the effects of increases in all P loading by 10, 25, 50, 75, and 100 percent. Results from these simulations were again used to determine the reduction in P loading required for summer-average TP concentrations in the lakes to decrease below 0.040 mg/L. Scenario 15–J (similar to scenario 16 with the BATHTUB model) simulated the effects of removing all anthropogenic P sources and, therefore, represents the best conditions possible in the lakes (that is, reference conditions). In scenario 15–J, TP concentrations in all tributaries to the lake were set to 0.020 mg/L, and all P inputs from point sources and septic systems were eliminated. Scenario 16–J was then used to demonstrate how much anthropogenic input could be added in addition to reference P loadings and yet maintain a summer-average TP at or below 0.040 mg/L (similar to scenario 17 with BATHTUB). Scenarios 17–J and 18–J were then used to demonstrate the effects of Lake Winnebago being the most downstream lake of a chain of lakes by examining changes in its response to a 75-percent reduction in external P loading (scenario 3–J) but with two changes. Scenario 17–J simulated the positive consequences to Lake Winnebago from P reductions applied only to the upstream lakes (upstream from the Fox River at Oshkosh) with no reductions in P loading from the area immediately around Lake Winnebago or from the point sources or septic systems effluent entering Lake Winnebago. Scenario 18–J simulated the negative consequences of Lake Winnebago being downstream from other lakes and simulated the delayed response to P reductions that were upstream in the watershed. This was done by showing the decrease in transient TP concentrations in Lake Winnebago when the TP concentrations leaving the Upper Pool Lakes were immediately set to the concentrations simulated 125 years after a 75-percent reduction in export from their watersheds (using output from scenario 3–J for the Upper Pool Lakes).

Algorithms, Data Requirements, and Calibration

The Jensen model (Jensen and others, 2006) (eq. 7) describes changes in in-lake TP concentrations caused by daily external P loading and water temperature, given the initial recyclable P accumulated in the sediments of the lake (P_{Sed}), by combining its estimations of P release from the sediments (eq. 8) and P sedimentation (eq. 9).

Six types of data are required as input into Jensen models: morphometric data (volume and mean depth from table 1), the initial recyclable (available) P accumulated in the sediments of the lake (P_{Sed}), daily water temperatures in the lake, daily flows into the lake, daily P loading to the lake, and TP concentration data from the lake to calibrate the models. All input data used in the Jensen models and results from the models are available at https://doi.org/10.5066/P9Y8BE4H (Robertson and Kennedy, 2018).

For comparison with results from model calibrations, the recyclable P accumulated in the sediment of Lake Winnebago was also estimated from information collected by Gustin (1995). Gustin (1995) estimated that the average depth of the zone of active TP recycling in Lake Winnebago was 0.3 m; therefore, the volume of active sediment was 0.3 cubic meter per square meter of bottom sediment. If the bulk density of wet lake sediment is assumed to be 100,000–200,000 grams of sediment per cubic meter, then the mass of active sediment is 30,000–60,000 grams of sediment per square meter. Gustin (1995) also estimated that the average recyclable P content at the sediment surface was 17.6 micromoles per gram of sediment (or 0.00054 gram of P per gram of sediment). If it is assumed that there was no recyclable P at 0.3 m and there was a linear decline in recyclable P from the surface to 0.3 m, then the average recyclable P was 0.00027 gram of P per gram of sediment. Therefore, the total recyclable P accumulated in the sediment should be about 8 to 16 grams per square meter (g/m^2).

Table 13. Linked Jensen model simulation results. Phosphorus load changes to the Winnebago Pool Lakes, Wisconsin, and the resulting geometric mean total phosphorus concentrations, for the period June 1 through September 14, are given for 125 years after the changes in phosphorus loading from the linked Jensen model. Calibrations were conducted from October 1, 2008 through September 30, 2013.

[kg, kilogram; conc., concentration; mg/L, milligram per liter. Base conditions for all comparisons are in bold font.]

Scenarios	Scenario	Upper Pool Lakes		Lake Winnebago	
		5-year average annual phosphous load (kg)	Total phosphorus concentration (mg/L)	5-year average annual phosphorus load (kg)	Total phosphorus concentration (mg/L)
		Calibration			
SWAT extrapolations	Calibration	305,600	0.117	333,100	0.100
SWAT extrapolations	Full linked model	305,600	0.117	347,700	0.101
		General response corresponding to percentage change in P load			
–80 percent	2–J	61,500	0.024	86,300	0.029
–75 percent	3–J	76,000	0.030	103,000	0.034
–70 percent	4–J	90,400	0.036	119,000	0.038
–69 percent	5–J	93,300	0.037	122,000	0.039
–65 percent	6–J	105,000	0.041	135,000	0.043
–50 percent	7–J	148,000	0.059	184,000	0.058
–25 percent	8–J	220,000	0.087	266,000	0.083
–10 percent	9–J	264,000	0.104	315,000	0.098
0 percent–Base	**1–J**	**293,000**	**0.116**	**348,000**	**0.108**
+10 percent	10–J	322,000	0.127	380,000	0.117
+25 percent	11–J	365,000	0.144	429,000	0.133
+50 percent	12–J	437,000	0.173	511,000	0.157
+75 percent	13–J	509,000	0.201	593,000	0.182
+ 100 percent	14–J	581,000	0.230	674,000	0.206
		Specific natural and anthropogenic scenarios			
Reference conditions	15–J	81,300	0.027	103,000	0.033
Reference plus 9 percent of the anthropogenic-affected sources	16–J	100,000	0.035	125,000	0.040
75-percent reduction to only phosphorus sources upstream of the Fox River at Oshkosh	17–J	76,000	0.030	133,000	0.043
Immediate upstream effects of a 75-percent reduction in phosphous sources to the Upper Pool Lakes	18–J	76,000	0.030	103,000	0.033

Continuous daily water temperatures in the lakes were estimated from air temperatures measured at the National Weather Service station at Oshkosh, Wis. Water temperatures were estimated using the average air temperature during the previous 5 days, with days having an estimated 5-day average air temperature less than 1 °C set to 1 °C. This resulted in a root mean square error (RMSE) between measured and estimated water temperatures of 2.56 °C based on water temperature data measured in Lake Winnebago from 1990 to 2013.

To calibrate the Jensen model, P_{Sed} and four additional coefficients need to be estimated: sediment release constant (b_R), temperature-dependence factor for P release (t_R), sedimentation constant (b_S), and temperature-dependence factor for sedimentation (t_S). Equations 7–9 were incorporated into Excel with daily flow, P loading, and water temperatures as inputs to the equations. “Optimal” values for the four coefficients and an initial P_{Sed} were obtained using the Solver routine in Excel, which was set up to minimize the sum of square errors between the daily measured and estimated TP concentrations in the lake. Solver adjusts the values in the decision variable cells (four coefficients and initial P_{Sed}) to minimize the value in an objective cell (sum of square errors between measured and estimated in-lake TP concentrations) using a Generalized Reduced Gradient Nonlinear method (Frontline-Solvers, 2017).

Because detailed in-lake data were only available for May through September for the Upper Pool Lakes and TP data are needed throughout the year, Jensen models could not be developed for each Upper Pool Lake. Instead, two independent Jensen models were initially constructed: one for all the Upper Pool Lakes using monthly TP data collected at their outlet (Fox River at Oshkosh) as its in-lake TP and one for Lake Winnebago using monthly or more frequent TP data collected in either the lake or its outlet (Fox River at Neenah Menasha) as its in-lake TP.

To calibrate the Jensen model for Lake Winnebago, Solver was used to find optimal values for the four coefficients and the initial P_{Sed} (table 14) for October 1, 2008, through December 31, 2013 (about 5 years of data) using flows and TP loads measured at the Fox River at Oshkosh and Fond du Lac River sites. The unmonitored flows and loads to Lake Winnebago were estimated from the Fond du Lac River site using SWAT-flow and SWAT-load ratios (1.399 [table 7] and 0.296, respectively). The 0.296 value was based on the SWAT estimated nonpoint P load for the nearshore area (6,140 kg; table 7) divided by the total load for the Fond du Lac River (20,700 kg; table 8). During calibration, P_{Sed} was estimated to be 15.0 g/m^2, which was within the range estimated from data collected by Gustin (1995). This value also kept P_{Sed} relatively stable throughout the 5-year calibration period. The RMSE of the calibrated independent Winnebago Jensen model was 0.043 mg/L based on 160 measurements.

After the Lake Winnebago Jensen model was calibrated, the model was used to simulate the daily P released from the sediments and daily P sedimentation from the water column (fig. 13*A*). The difference in these two rates represents the net internal P loading into the water column (fig. 13*B*) and should incorporate the effects of diffusion, wind resuspension, and other biological effects. During May through September of 2009–13, the average net P release from the sediments with the Jensen model was estimated to be 3.5 mg/m^2/d, which was a little higher than the 3.0 mg/m^2/d estimated using the mass-balance approach and 2.8 mg/m^2/d from BATHTUB. There was a net deposition of P into the sediments from September to April. The average net P release from the sediments for the entire year was 0.6 mg/m^2/d, which indicates that there should be a small net release of P into Lake Winnebago over the entire year.

Table 14. Parameters and coefficients used in the calibrated Jensen models for Lake Winnebago and the three Upper Pool Lakes.

Parameter Description	Parameter	Lake Winnebago	Upper Pool Lakes
Volume, in millions of cubic meters	*Vol*	2,415	189.1
Mean depth, in meters	$\bar{z}$	4.54	1.68
Initial sediment phosphorus content, grams per square meter	P_{Sed}	15.00	15.00
Sediment release constant	b_R	0.0004	0.0005
Temperature dependence of phosphorus release	t_R	0.3209	0.1175
Sedimentation constant	b_S	0.0523	0.0523
Temperature dependence of sedimentation	t_S	0.0606	0.0639
Summary statistic, sum of square errors	*SSE*	0.2895	0.0424
Summary statistic, root mean sum of square errors	*RMSE*	0.0425	0.0266

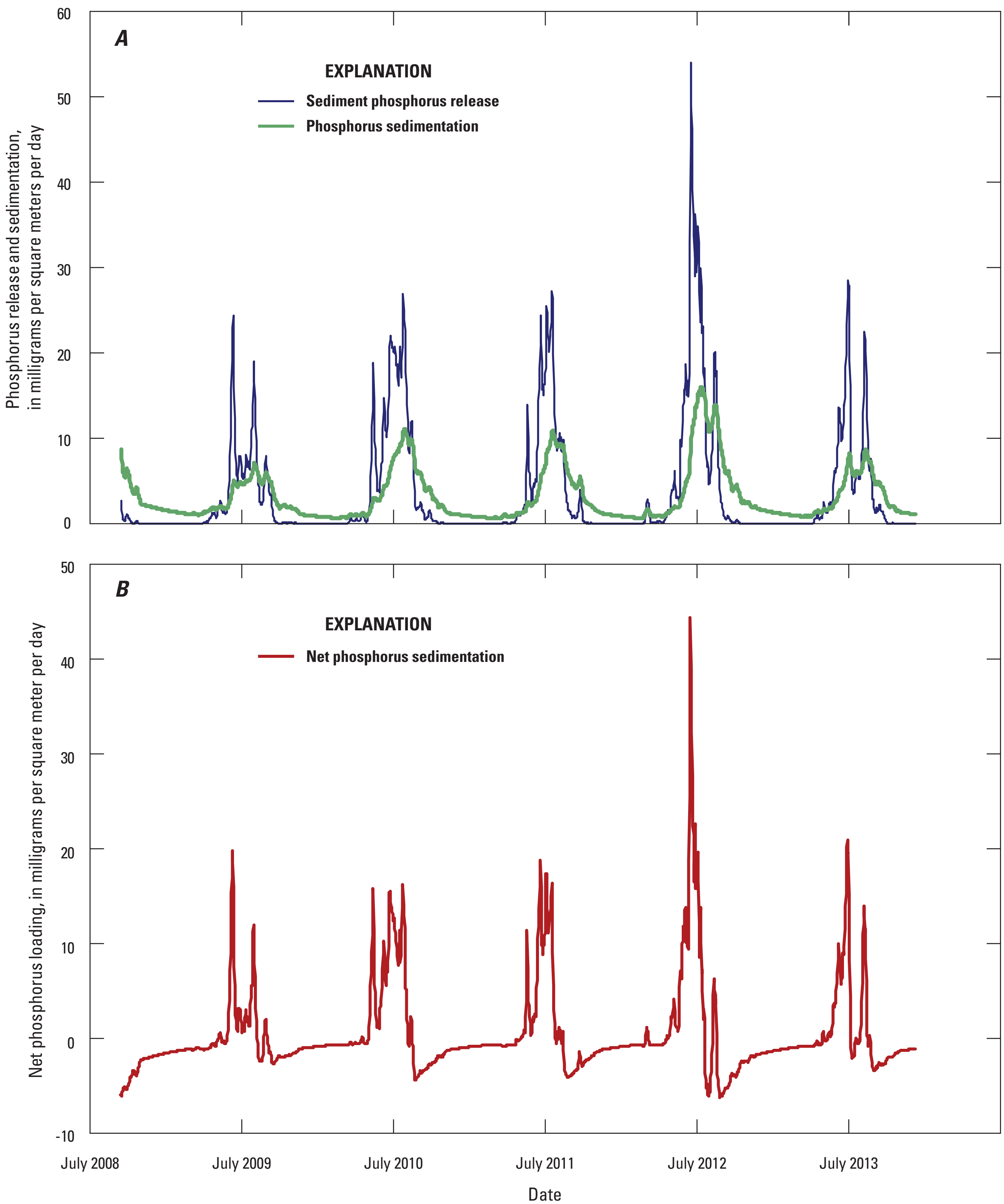


Figure 13. Phosphorus release, sedimentation, and net internal phosphorus loading for Lake Winnebago, Wisconsin, water years 2009–13, on the basis of Lake Winnebago Jensen model results. *A*, Phosphorus release and sedimentation. *B*, Net phosphorus sedimentation.

Solver was then used to calibrate the Jensen model for the combined three Upper Pool Lakes, for October 1, 2008, through December 31, 2013, using daily total flows and P loads estimated from measured flows and computed loads at the Wolf River at New London and Fox River at Berlin extrapolated to the entire basin using SWAT flow and load ratios (SWAT-flow ratios of 1.700 and 1.141 for the Wolf and Fox Rivers, respectively, and SWAT-load ratios of 1.642 and 1.124 for the Wolf and Fox Rivers, respectively, table 7). During initial calibration, the minimization process determined a sedimentation rate equal to 0, which is unrealistic; therefore, the sedimentation rate was set to a value similar to that set for Lake Winnebago, and then optimal values for all other coefficients were determined (table 14). During calibration, P_{Sed} was again estimated to be 15.0 g/m^2. This value kept P_{Sed} relatively stable throughout the 5-year calibration period. The measured and simulated TP concentrations for the Upper Pool Jensen model are compared in figure 14*A*. The RMSE of the final calibrated model for the Upper Pool Lakes was 0.027 mg/L based on 60 measurements.

Output from the calibrated Upper Pool Lakes Jensen model was then used as input to the calibrated Winnebago model to create one linked Jensen model, which used measured loads at the Wolf River at New London and Fox River at Berlin extrapolated to the entire Upper Pool Basin using SWAT-flow and SWAT-load ratios and direct flow and P loading to Lake Winnebago based on Fond du Lac River flows and P loads. Measured and simulated TP concentrations for Lake Winnebago from the linked Jensen model are compared in figure 14*B*. The RMSE for the measured compared to simulated TP concentrations in Lake Winnebago from the linked Jenson model was 0.043 mg/L, which is the same as the independently calibrated Lake Winnebago Jensen model. The measured and simulated annual geometric-mean TP concentrations for June 1 through September 15 (the period for which TP criterion for the lake were derived [Wisconsin Department of Natural Resources, 2017a]) only for days with coinciding data for the linked model are compared in figure 14*C*. The RMSE of the final linked model for annual estimated TP concentrations was 0.018 mg/L based on 5 years of data. These model results provided reasonable fit to the measured data, thus suggesting the models should provide accurate responses to changes in P loading.

Response in Water Quality to Basinwide Changes in Phosphorus Loading

In total, 14 scenarios (J–1 through J–14) were simulated with the linked Jenson model to estimate the expected response in the geometric-mean summer TP (June 1 through September 15) in the Upper Pool Lakes and Lake Winnebago to changes in controllable P sources and to determine the P load reduction required to reach the 0.040 mg/L TP criterion for the lakes For all simulations, the Jensen model was first used to simulate daily TP concentrations in the lakes during WY2009–13 using P loading from the watershed similar to that used in the calibration period, except the flows and TP loading from the watershed were replaced with flows and loadings fully estimated with SWAT (The Cadmus Group, 2018). Then the P inputs for those 5 years were modified based on the changes in loading being simulated (such as a 25-percent increase in loading simulated as a 25-percent increase in TP concentrations) for WY2014–18. These changes were applied to all P sources, except inputs from groundwater and the precipitation. Then the inputs for 2014 to 2018 were repeated over and over in 5-year replicates until 2145. Because the Jensen model simulates transient changes in lake TP after changes in P inputs, the expected future in-lake TP concentrations depends on the number of years after the change in P loading that is evaluated. The simulated changes in geometric-mean TP concentrations for the Upper Pool Lakes and Lake Winnebago after a 75-percent reduction in external P loading are shown in figure 15. After about 60 years, the recyclable P accumulated in the sediments and the in-lake TP in the Upper Pool Lakes approach a new equilibrium of about 4 g/m^2 and 0.030 mg/L, respectively; however, the recyclable P accumulated in the sediments and the in-lake TP in Lake Winnebago took much longer, over 100 years, to approach a new equilibrium of about 5 g/m^2 and 0.033 mg/L, respectively. Part of the reason for the difference in the responses in these two waterbodies was that the TP concentrations of all inflows to the Upper Pool Lakes in these simulations immediately changed in 2014; however, TP concentrations in the major inflow to Lake Winnebago was more gradual, taking several years to come to a new equilibrium because of the gradual transition of in-lake TP concentrations in the Upper Pool Lakes. Therefore, TP concentrations in Lake Winnebago should respond slower to external changes in P loadings from throughout basin than the Upper Pool Lakes. To describe this gradual decline in TP concentrations in the lakes, the geometric-mean TP concentrations for June 1–September 15 for the 5 years just before 10, 30, 50, 75, and 125 years after the load reduction are examined.

Based on Jensen model simulations, TP in the Upper Pool Lakes should have a linear response to changes in P loading, with a response that varies depending on the number of years after the P load changes (fig. 16*A*; table 13). The changes in in-lake TP concentrations in the Upper Pool Lakes gradually approach a similar percent change to that of the external P loading: 10 years after the P load reduction, a 75-percent change in external P loading (equating to a 74-percent change in total loading) should cause about a 50-percent change in TP concentrations, but after about 50 years, a 75-percent change in loading should cause about a 72-percent change in TP concentrations. The changes in in-lake TP concentrations in Lake Winnebago approach a similar change to that of the external P loading, but much more gradually than in the Upper Pool Lakes: after 75 years, a 75-percent change in external P loading (equating to a reduction in total loading of 70 percent) should cause about a 66-percent change in TP concentrations.

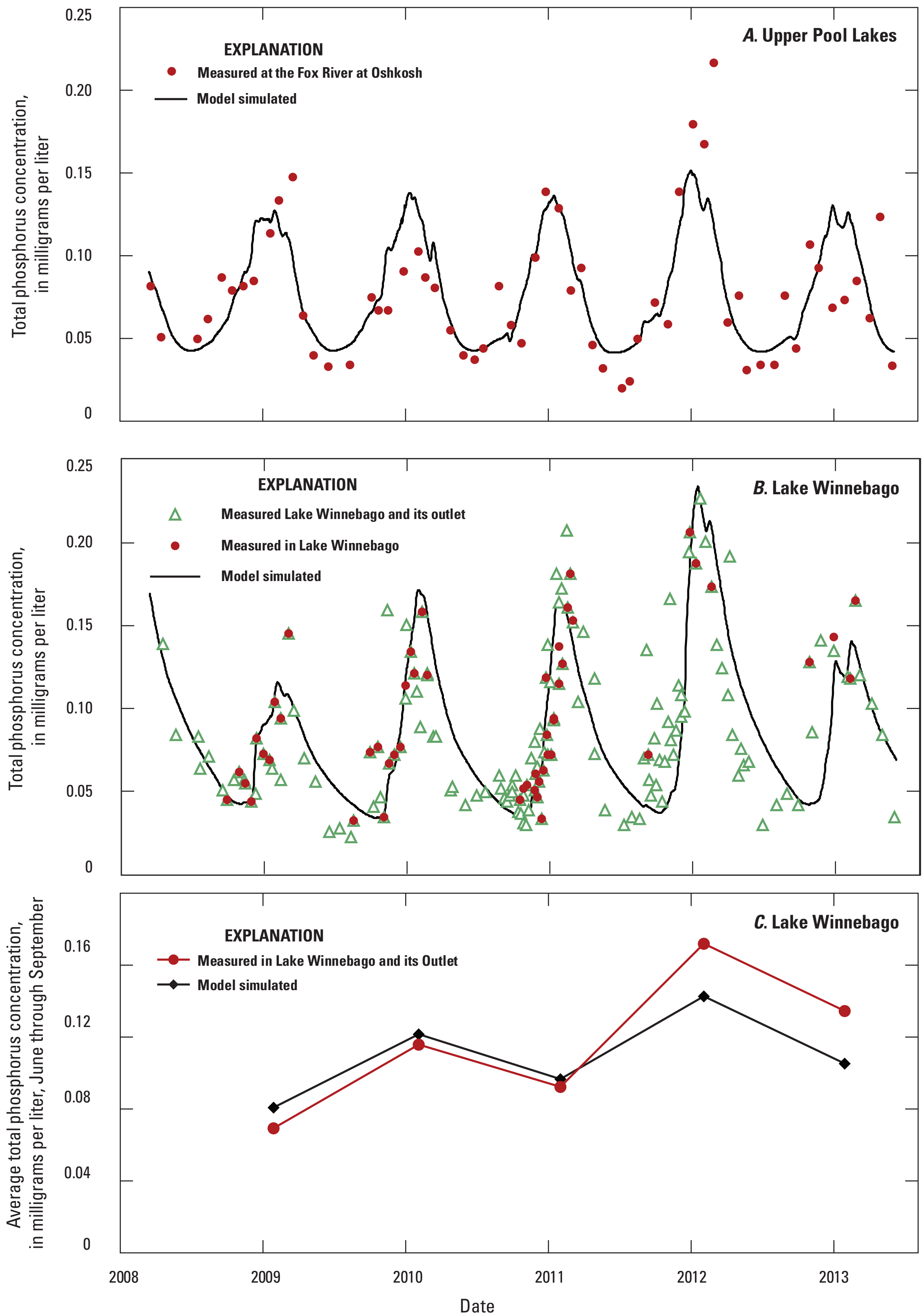


Figure 14. Measured and simulated (with the linked Jensen model) near-surface total phosphorus concentrations in the Upper Pool Lakes and Lake Winnebago, Wisconsin, water years 2009–13. *A*, Upper Pool Lakes. *B*, Lake Winnebago. *C*, Annual June 1 through September 15 geometric mean total phosphorus concentrations in Lake Winnebago.

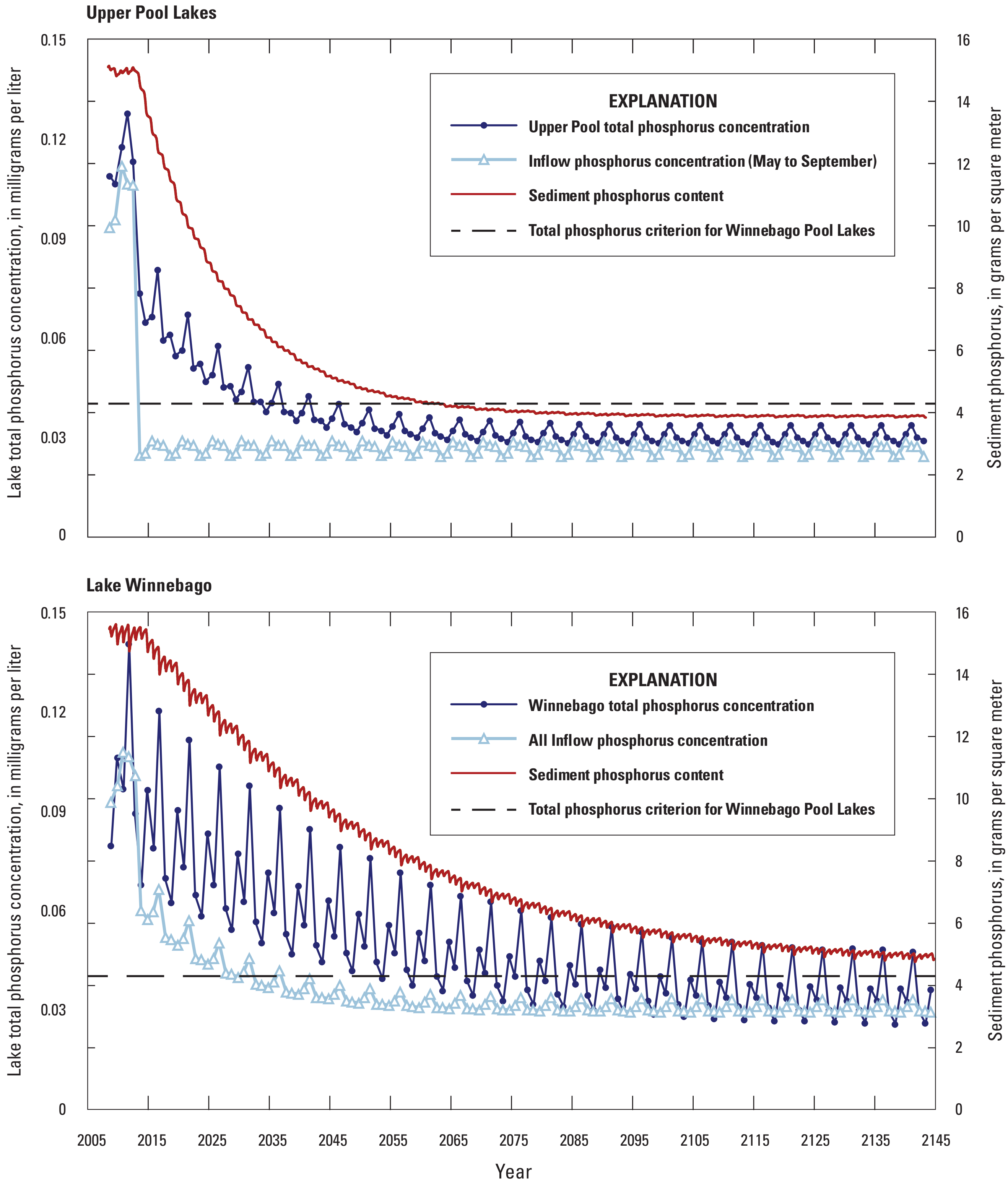


Figure 15. Simulated annual geometric mean total phosphorus concentrations for the Upper Pool Lakes and Lake Winnebago, Wisconsin, during water years 2009–2145 after a 75-percent reduction in external phosphorus loading using the linked Jensen model.

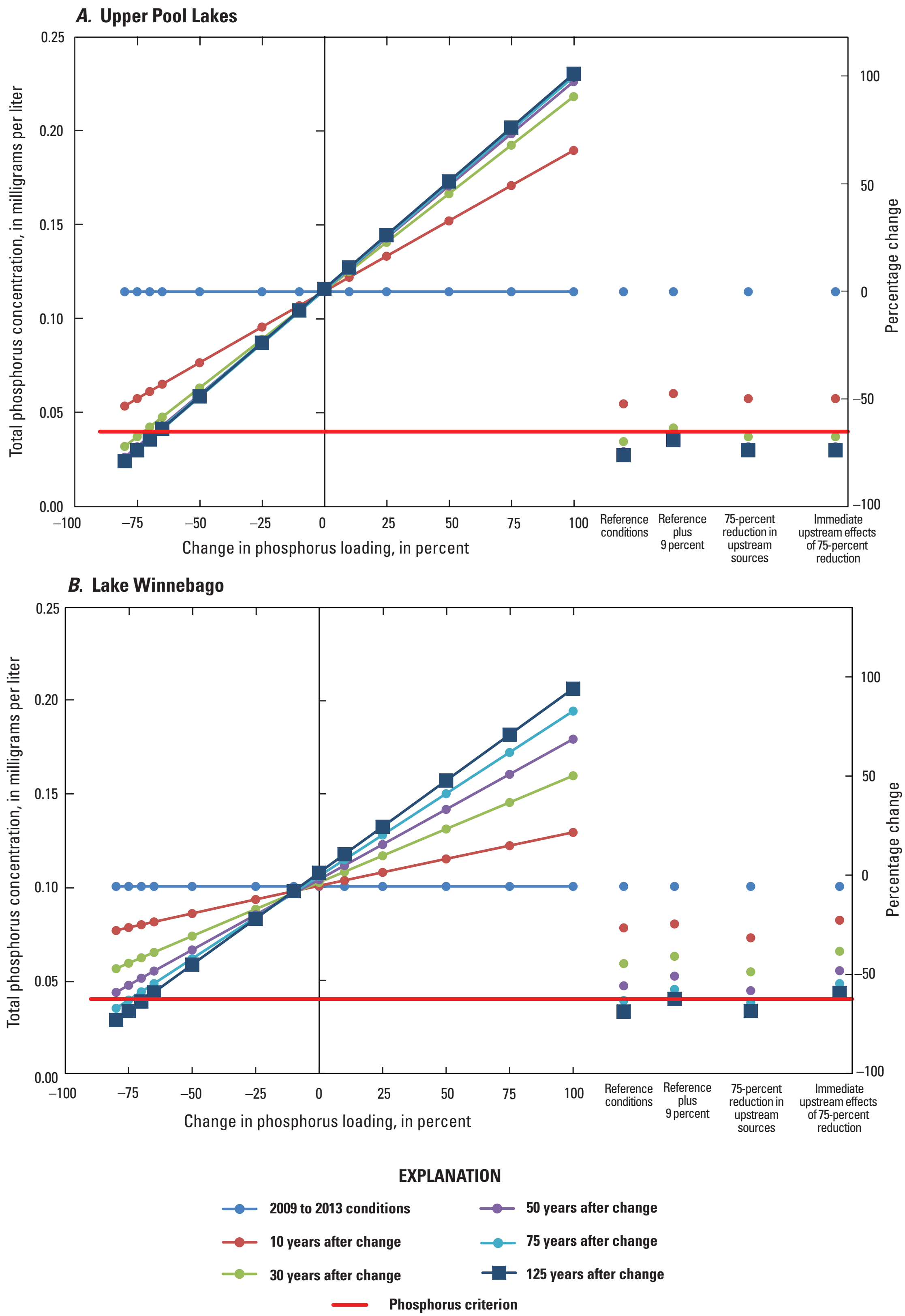


Figure 16. Simulated changes in the total phosphorus concentrations for the Upper Pool Lakes and Lake Winnebago, Wisconsin, in response to various phosphorus-loading scenarios (table 13) using the linked Jensen model. Response in the 5-year June 1 through September 15 geometric mean total phosphorus concentrations for A, the Upper Pool Lakes and *B*, Lake Winnebago.

Based on the response curves for the Upper Pool Lakes (fig. 16*A*), it would require about a 65-percent reduction in external P loading and about 125 years or a 75-percent reduction in external P loading and about 30 years for the Upper Pool Lakes to reach a geometric-mean June 1–September 15 TP concentration of 0.040 mg/L (the P criterion for these lakes). Based on the response curves for Lake Winnebago, a 75-percent reduction in P loading and 75 years are needed for the lake to reach a geometric-mean June 1–September 15 TP concentration of 0.040 mg/L or a 69-percent reduction in P loading and about 100 years. These reductions in P loading to reach the TP criterion are close to the 73-percent reduction estimated with BATHTUB.

Potential Water Quality

To determine the best potentially achievable WQ in the lakes (that is, reference conditions), TP concentrations in all tributaries to the lakes were again set to 0.020 mg/L, and P inputs from point sources and septic systems were eliminated (scenario 15–J, which is similar to scenario 16 with BATHTUB). Based on results from the linked Jensen model (fig. 16; table 13), the lowest possible geometric mean TP should be 0.027 mg/L for the Upper Pool Lakes and 0.033 mg/L for Lake Winnebago. These TP concentrations are better than the TP criterion for shallow drainage lakes in Wisconsin (Wisconsin Department of Natural Resources, 2017a).

Effects of Specific Phosphorus Reductions in the Watershed

To determine how much anthropogenic P inputs could be added to reference P loading and yet maintain a 5-year geometric-mean summer TP concentration at or below 0.040 mg/L, all the anthropogenic sources were systematically adjusted. If all TP concentrations in the streams, point sources, and septic systems increase less than 9 percent from their reference concentrations (scenario 16–J, which is similar to scenario 17 with BATHTUB), the geometric-mean TP concentration in the Upper Pool Lakes and Lake Winnebago in 125 years should both stay at or below 0.040 mg/L, which is similar to that determined using BATHTUB. However, Jensen model results demonstrate how long it will take to reach this concentration.

Quite similar reductions in external P loading required for TP in Lake Winnebago to reach 0.040 mg/L were determined using the Jensen and BATHTUB models even though each model used different inputs: the Jensen model used P loading for all 12 months, whereas the BATHTUB model used P loading only for May through September.

Importance of Internal Phosphorus Loading

Changes in external P loading drives changes in-lake TP (fig. 9) and productivity, and then the P is either exported out the outlet (in drainage lakes) or deposited in the bottom sediments. Not all P deposited in the sediments remains there because some is released back into the water column from internal P loading. Typically, P is released from the sediments by diffusion, which typically increases substantially when oxygen is completely depleted (anoxia) near or in the bottom sediment (Mortimer, 1941). The amount of P released from diffusion was quantified by incubating sediment cores from the Pool Lakes and measuring DO conditions in the lakes (in great detail in Lake Winnebago). Because of extensive mixing, anaerobic conditions were seldom, if ever, measured in routine monitoring or with continuous recording instruments deployed below a buoy in Lake Winnebago (fig. 2); therefore, the aerobic P release rates were used to estimate P released from diffusion. The amount of P released from the sediments on the basis of laboratory-estimated rates, however, was not sufficient to describe the changes in TP observed in the lakes using the phosphorus mass-balance approach (fig. 9). In addition to simple diffusion, sediment resuspension caused by strong winds can increase sediment P release in shallow lakes (Søndergaard and others, 1992; Cyr and others, 2009; Lin and Wu, 2013; Huang and others, 2016), and aquatic animals can increase internal P loading in lakes through their feeding behavior (stirring up the sediments) and excretion (Lamarra, 1975; Andersson and others, 1988; Fischer and others, 2013). These processes may increase the internal P released in the Winnebago Pool Lakes. Results from the mass-balance approach (fig. 9), the BATHTUB model, and the Jensen models all indicate that the net internal P release rate for May–September was about 2.8–3.5 $mg/m^2/d$, which is about 7 to 8 times more than laboratory rates for diffusion from aerobic sediments and about twice the rate estimated for diffusion from anaerobic sediments. Therefore, internal P release in these lakes seems to be much higher than that estimated directly from the sediment core studies. These release rates are similar to those estimated in other large shallow eutrophic lakes (for example, 5.56 $mg/m^2/d$ in Lake Taihu, China; Huang and others, 2016).

Relatively similar internal P loading to Lake Winnebago was estimated using three different approaches (mass-balance approach, the BATHTUB model, and the Jensen model); however, all three of the approaches were based on similar external P loading rates to the lake. Therefore, similar internal P loading may have been expected. Part of the P estimated to be from internal loading may be a result of errors in the other components of the P budgets; however, given that most of the P load to Lake Winnebago was from monitored tributary loading (Fox River and Fond du Lac River) and input from precipitation, this estimate of P from internal loading should be considered quite reliable.

Because of the P being released from processes in addition to diffusion, net internal P loading is an important source of P to each of the lakes, but it did vary in its relative importance among lakes because of the magnitude of the other sources and the varying residence time among lakes. On the basis of the net internal P loading rates estimated with BATHTUB, the importance of net internal P loading during May–September varied from about 3 percent for Lake Winneconne, to about 14 percent for Lake Poygan and Lake Butte des Morts, and to 55 percent for Lake Winnebago (table 9). Although most of the external P inputs to Lake Winnebago could be accurately estimated, more assumptions were needed to estimate the total external P inputs to the other Upper Pool Lakes; therefore, their estimated internal P loading should be considered less reliable than those for Lake Winnebago.

The internal P loading from the sediments during summer not only increases in-lake TP concentrations throughout summer, but it also results in an increase in Chl-*a* and a decrease in water clarity (SD) as summer progresses (fig. 3). These changes in WQ result in all the Winnebago Pool Lakes being classified as hypereutrophic by late summer.

Although the net internal P loading rate was relatively high in all Winnebago Pool Lakes during summer, results from the Jensen model indicate that there is a net P deposition back to the sediments during the rest of the year (fig. 13), which results in the sediments having little net effect on the annual P budget of Lake Winnebago. Based on the mass-balance approach, there was a slight accumulation of P in the lake (table 8), and based on the Jensen model there was a slight net release of P from the sediments (about 0.6 $mg/m^2/d$).

Lakes with high internal P loading may respond much slower to reductions in external P loading than lakes with relatively low internal P loading (Jeppesen and others, 2005, Jensen and others, 2006, Meals and others, 2010). Even 20 years after external P load reductions to Shagawa Lake, Minn., internal P loading continued to affect its trophic state (Seo and Canale, 1999). Results from the Jensen models demonstrate how P from internal loading can delay the full effects of external P load reductions from being achieved (figs. 15 and 16). Although changes in external P loading were simulated to happen immediately in all the Jensen simulations, it took several decades for the sediment P content (fig. 15) and its associated P release to reach equilibrium. About 10–15 years after the external P load reduction of 75 percent, the geometric mean for the May–September period for the Upper Pool Lakes should decrease from about 0.115 to about 0.050 mg/L, and the geometric mean for Lake Winnebago should decrease from about 0.101 to about 0.073 mg/L. TP concentrations reached equilibrium in the Upper Pool Lakes (0.031 mg/L) in about 50 to 60 years, but TP concentrations in Lake Winnebago reached its new equilibrium (0.033 mg/L) in about 100 years.

Effects of Changes in Phosphorus Loading Cascading Down a Chain of Lakes

Lakes typically are considered to be singular entities in the environment (Forbes, 1887) and, therefore, typically are modeled as such. However, in a chain of lakes, such as the Winnebago Pool, effects of changes upstream in the watershed affect the lakes downstream (Lathrop and Carpenter, 2014; Carpenter and Lathrop, 2014). The location of Lake Winnebago being at the downstream end of a chain of lakes can have beneficial and detrimental effects. Being downstream from other lakes, it should benefit from any actions done to improve the WQ of the upstream lakes; however, there should be a delayed response to changes made in the watershed, especially in lakes where internal P loading is important to its P budget (Jeppesen and others, 2005, Meals and others, 2010; Jensen and others, 2006). To examine the beneficial and detrimental effects caused by Lake Winnebago being the downstream member in a chain of lakes, two additional scenarios were simulated with the linked Jensen model (scenarios 17–J and 18–J). In figure 17, the 5-year moving-average geometric-mean summer TP concentrations are plotted for a 75-percent reduction in external loading throughout the entire watershed (scenario 3–J). Scenario 17–J simulates the effects of a 75-percent reduction only applied to the watersheds of the three upstream lakes, with no changes in external P loading directly around Lake Winnebago. Results of this simulation show that there is still a large decrease in TP in Lake Winnebago, from 0.116 to 0.048 mg/L (computed for the May–September period). Thus, there is a beneficial effect of being downstream in a chain of lakes. Actions improving the WQ of any upstream lake will also have beneficial effects to the downstream lakes.

The Upper Pool Lakes, however, also delay the effects of actions done throughout the watershed of a downstream lake, which can be seen in scenario 18–J. This scenario simulates what would happen if the full effects of a 75-percent reduction in P inputs are immediately input into Lake Winnebago. TP concentrations decline more quickly but ultimately reach the same TP concentration after about 100 years. After a 75-percent reduction, it should take about 25 years for 50 percent of the effects to be observed in Lake Winnebago; however, if the reduction in external loading was done immediately, it should only take about 20 years for 50 percent of the effects to be observed. The largest detrimental lag effect is seen about 15 years after the 75-percent reduction in P inputs, when TP concentrations in Lake Winnebago would have been about 0.007 mg/L lower if the Upper Pool Lakes did not delay the effects of the external P reductions. The delay in the changes in TP concentrations in Lake Winnebago primarily is because much of P loading to Lake Winnebago comes from the Upper Pool Lakes whose TP concentrations take several years to reach equilibrium. Therefore, the delayed reduction in P loadings to Lake Winnebago from the upstream lakes delays the changes in its sediment P content, which further delays the changes in its TP concentration.

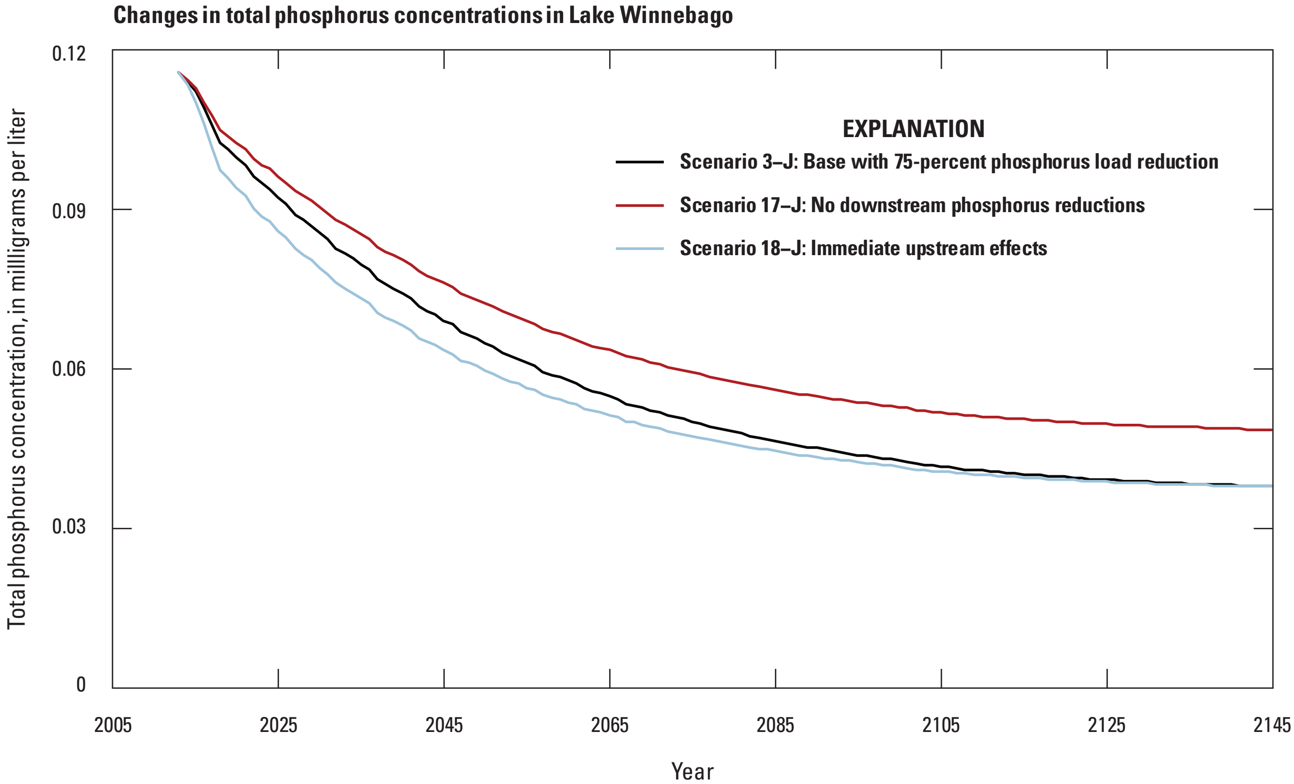


Figure 17. Simulated changes in the total phosphorus concentrations in Lake Winnebago, Wisconsin, in response to various phosphorus-loading scenarios using the linked Jensen model, demonstrating beneficial and detrimental effects of it being in a chain of lakes.

Summary and Conclusions

The Winnebago Pool is a chain of four shallow lakes that has received extensive phosphorus (P) inputs from its agricultural watershed during the past 150 years. These inputs have resulted in increased in-lake total phosphorus (TP) concentrations in all four lakes and much P trapped in their sediments. Because the in-lake TP concentrations exceed their TP criterion, all four Pool Lakes are now included on Wisconsin's impaired waters list. This study was done to determine how changes in P inputs to the lakes should affect their water quality (WQ) and how being a member of a chain of lakes affects this response. Further, the study determined the reduction in P loading needed for each lake to attain their TP criterion.

As part of the study, detailed WQ data were collected in the four lakes during 2009–11 to document their current WQ. During May–September of these years, all four lakes had extensive periodic vertical mixing resulting in relatively uniform WQ throughout the water column in each lake. Average TP concentrations for June–September ranged from 0.088 milligram per liter (mg/L) in Lake Winneconne to 0.104 mg/L in Lake Butte des Morts, average Chlorophyll-*a* (Chl-*a*) concentrations ranged from 30.5 micrograms per liter (µg/L) in Lake Winnebago to 52.1 µg/L in Lake Butte des Morts, and average Secchi depths (SDs) ranged from 0.54 meter (m) in Lake Butte des Morts to 1.15 m in Lake Winnebago. TP and Chl-*a* concentrations increased and SDs decreased as summer progressed, resulting in each of the lakes becoming hypereutrophic by late summer.

Phosphorus budgets were constructed for each lake. External P input from the watershed was the most important source of P to the most upstream lake (Lake Poygan) but became less important to downstream lakes as input from the adjacent upstream lake became more important: for Lake Poygan direct external P loading represented more than 96 percent of its total annual P input, compared to about 8 percent for Lake Winnebago, which had about 80 percent of its P input from the adjacent upstream Pool Lake (Lake Butte des Morts). The rates of net internal P loading during May–September (based on results from the BATHTUB model) varied among lakes from 1.2 milligrams per square meter per day ($mg/m^2/d$) in Lake Winneconne, to 2.3 $mg/m^2/d$ in Lake Poygan, to 2.8 $mg/m^2/d$ in Lake Winnebago, and to 5.0 $mg/m^2/d$ in Lake Butte des Morts. Similar net internal P load rates for Lake Winnebago were determined with a mass-balance approach and the Jensen model. These internal P loading rates were much higher than what could be explained from

P diffusion rates estimated from sediment cores incubated in the laboratory. The high internal P loading rates in these large shallow lakes are believed to be caused by sediment resuspension from wind and fish and by biological excretion. Based on the internal P loading rates from BATHTUB for May–September, the importance of the net internal P loading to their summer P budget varied from about 3 percent for Lake Winneconne, to about 14 percent for Lake Poygan and Lake Butte des Morts, and to 55 percent for Lake Winnebago. Part of the P from internal loading may be a result of errors in the other components of their P budgets; however, given that most of the P load to Lake Winnebago is from monitored tributary loading and precipitation, its estimate of P from internal loading should be quite reliable. However, more assumptions were needed to estimate the total external P inputs to the other Upper Pool Lakes; therefore, their estimated internal P loading may be less reliable than that for Lake Winnebago.

To determine how the WQ of the Pool Lakes should respond to changes in P loading, and ultimately the magnitude of actions that are needed to improve their WQ sufficiently such that they can be removed from Wisconsin impaired waters list, two eutrophication models (BATHTUB and Jensen models) were used to simulate the long-term and transient effects of various P-reduction strategies. Results from both models indicate that TP, Chl-*a*, and SD directly respond to changes in external P input; and that the P loading into the three Upper Pool Lakes (Lakes Poygan, Lake Winneconne, and Lake Butte des Morts) needs to be reduced by about 60 percent and into Lake Winnebago needs to be reduced by about 69–73 percent for their mean summer TP concentrations to decrease to less than their 0.040 mg/L criterion. Results of both models with TP concentrations in all tributaries set to 0.020 mg/L (reference TP concentration for streams in central Wisconsin), and P inputs from point sources and septic systems eliminated, indicate that the best possible mean summer WQ is as follows: TP concentrations ranging from 0.022 to 0.027 mg/L in the Upper Pool Lakes to 0.032–0.033 mg/L in Lake Winnebago; Chl-*a* concentrations ranging from 10 µg/L in Lake Poygan, to 14 µg/L in Lakes Winneconne and Winnebago, and to 19 µg/L in Lake Butte des Morts; and SDs ranging from 1 m in Lake Butte des Morts, to 1.2 m in Lake Poygan and Lake Winneconne, and to 2.2 m in Lake Winnebago. All these values are below the Wisconsin Department of Natural Resources WQ criteria for shallow drainage lakes.

BATHTUB was then used to examine the effects of reducing internal P loading by an additional 25 percent in each of the scenarios to simulate what may happen if in-lake actions also are done, such as activities to promote macrophyte growth that should reduce wind resuspension or activities to reduce the population of rough fish, such as carp, that stir the bottom during feeding and have high excretion rates. If internal P loading was reduced an additional 25 percent, the overall external P load reduction needed for the all four Pool Lakes to reach 0.040 mg/L decreased from a 73-percent reduction in all controllable P sources to a 67-percent reduction.

Input of P from internal loading not only increases the TP concentrations in the lakes as summer progresses, but it also increases Chl-*a* concentrations, reduces water clarity (SD), and should delay the timeframe for the lakes to experience the full effects of external P load reductions. The final equilibrium TP concentrations after a 69-percent reduction in external P loading to the Upper Pool Lakes (0.037 mg/L) was in about 50–60 years, and about 100 years in Lake Winnebago (0.039 mg/L).

Changes in WQ resulting from changes in the watershed of a chain of lakes, such as the Winnebago Pool Lakes, gradually cascades downstream in the chain. Lake Winnebago being the most downstream lake in this chain has beneficial and detrimental implications. Because Lake Winnebago is downstream from other lakes, results of any action done in the watershed of upstream lakes to reduce their P inputs should also improve the WQ of all the downstream lakes. However, the upstream lakes also delay the response in WQ of the downstream lakes to actions done in the watershed, especially in lakes where internal P loading is important to their P budgets.

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