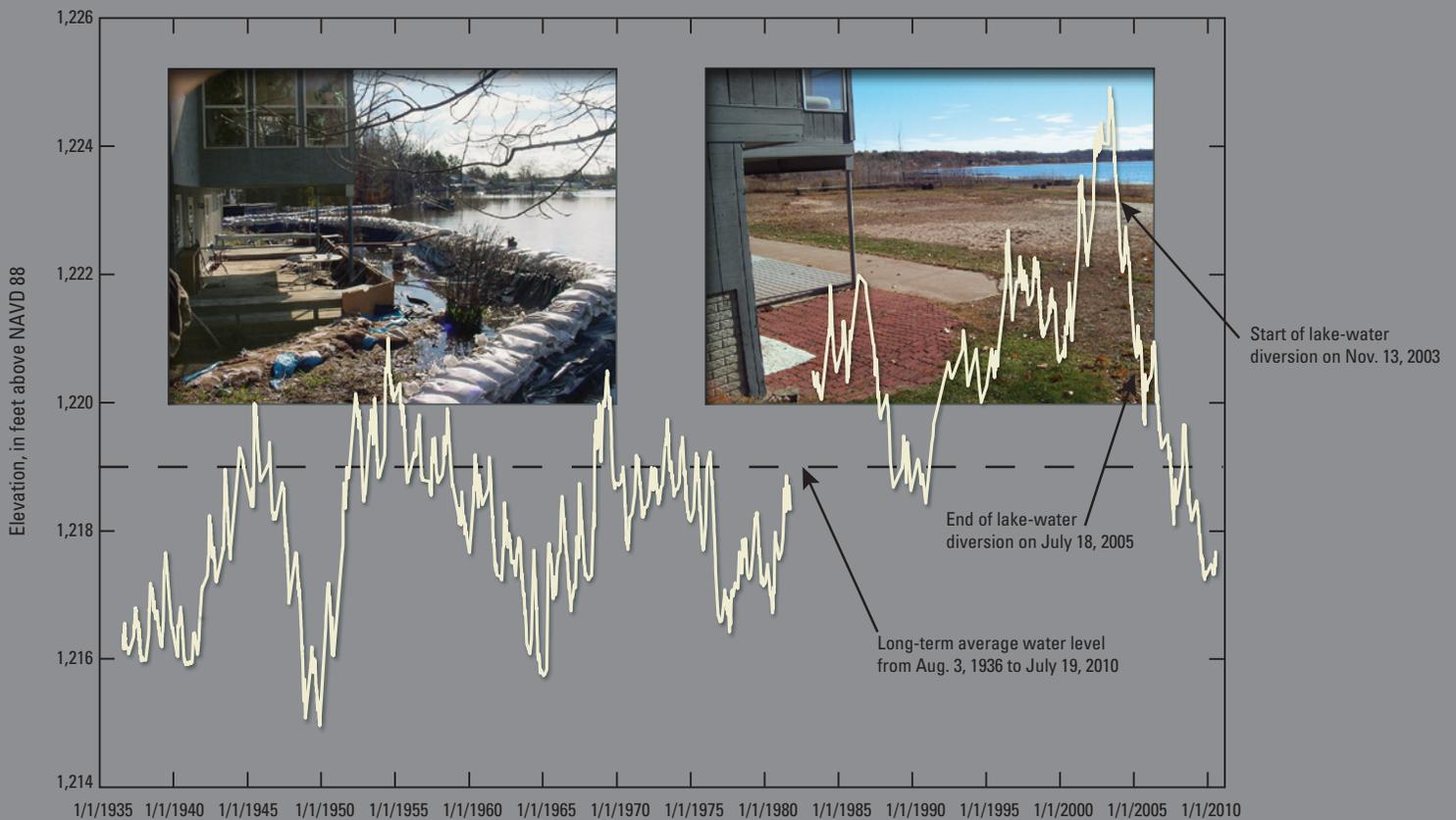


In cooperation with the City of Shell Lake, Wisconsin

Hydrology and Water Quality of Shell Lake, Washburn County, Wisconsin, With Special Emphasis on the Effects of Diversions and Changes in Water Level on the Water Quality of a Shallow Terminal Lake



Scientific Investigations Report 2013–5181

Cover photo left, high water level. Placement of sandbags around a home on the north shore of Shell Lake to minimize flooding, May 17, 2002. The water level on this day was 1224.11 feet above NAVD 88, which was about 0.8 foot below the peak water level of 1224.92 feet in May of 2003. Photo by John Haack, Regional Natural Resources Educator, UW-Extension.

Cover photo right, low water level. Low water level and large beach expanse as seen on November 12, 2013, around the same Shell Lake home as shown in the high water level photo, left. The water level on this day was 1216.65 feet above NAVD 88, which was about 8.3 feet lower than the peak water level. Photo by Dave Vold, Lake Coordinator, City of Shell Lake.

Hydrology and Water Quality of Shell Lake, Washburn County, Wisconsin, With Special Emphasis on the Effects of Diversions and Changes in Water Level on the Water Quality of a Shallow Terminal Lake

By Paul F. Juckem and Dale M. Robertson

In cooperation with the City of Shell Lake, Wisconsin

Scientific Investigations Report 2013–5181

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

U.S. Department of the Interior
SALLY JEWELL, Secretary

U.S. Geological Survey
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U.S. Geological Survey, Reston, Virginia: 2013

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Conversion Factors

Multiply	By	To obtain
Length		
inch (in)	2.54	centimeter (cm)
inch (in)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
Area		
acre	4,047	square meter (m ²)
acre	0.004047	square kilometer (km ²)
Volume		
million gallons (Mgal)	3,785	cubic meter (m ³)
cubic foot (ft ³)	0.02832	cubic meter (m ³)
acre-foot (acre-ft)	1,233	cubic meter (m ³)
Flow rate		
foot per second (ft/s)	0.3048	meter per second (m/s)
foot per day (ft/d)	0.3048	meter per day (m/d)
cubic foot per second (ft ³ /s)	0.02832	cubic meter per second (m ³ /s)
gallon per minute (gal/min)	0.06309	liter per second (L/s)
gallon per day (gal/d)	0.003785	cubic meter per day (m ³ /d)
million gallons per day (Mgal/d)	0.04381	cubic meter per second (m ³ /s)
inch per hour (in/h)	0.0254	meter per hour (m/h)
inch per year (in/yr)	25.4	millimeter per year (mm/yr)
Mass		
pound, avoirdupois (lb)	0.4536	kilogram (kg)
Hydraulic conductivity		
foot per day (ft/d)	0.3048	meter per day (m/d)
Transmissivity*		
foot squared per day (ft ² /d)	0.09290	meter squared per day (m ² /d)
Leakance		
foot per day per foot [(ft/d)/ft]	1	meter per day per meter

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

$$^{\circ}\text{F}=(1.8\times^{\circ}\text{C})+32$$

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

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

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

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

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

*Transmissivity: The standard unit for transmissivity is cubic foot per day per square foot times foot of aquifer thickness [(ft³/d)/ft²ft]. In this report, the mathematically reduced form, foot squared per day (ft²/d), is used for convenience.

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Local Project Coordinator

Dave Vold, Lake Coordinator, City of Shell Lake, Wisconsin (Wis.)

Data Collection and Analysis

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Michelle Greenwood, Reports Specialist, U.S. Geological Survey, Middleton, Wis.

Jim Kennedy, Geographer, U.S. Geological Survey, Middleton, Wis.

Leah Leonards, Office Automation Assistant, U.S. Geological Survey, Middleton, Wis.

Laura Nelson, Office Automation Assistant, Middleton, Wis.

Marta Anderson, Visual Information Specialist, Columbus Science Publishing Network, Madison, Wis.

Technical Reviewers

Jeff Raffensperger, Hydrologist, U.S. Geological Survey, Baltimore, Maryland

Norman Buccola, Hydrologist, U.S. Geological Survey, Portland, Oregon

Kathy Bartilson, Northern Region Basin Supervisor, Wis. Dept. of Natural Resources, Spooner, Wis.

Pamela Toshner, Water Resources Management Specialist, Wis. Dept. of Natural Resources, Spooner, Wis.

Editor

Mike Eberle, Technical Writer-Editor, U.S. Geological Survey, Columbus, Ohio

Groundwater Specialist Reviewer

Rodney Sheets, Groundwater Specialist, U.S. Geological Survey, Columbus, Ohio

Approving Official

Kevin Breen, Bureau Approving Official, U.S. Geological Survey Office of Science Quality and Integrity, New Cumberland, Pennsylvania

Hydrology and Water Quality of Shell Lake, Washburn County, Wisconsin, With Special Emphasis on the Effects of Diversion and Changes in Water Level on the Water Quality of a Shallow Terminal Lake

By Paul F. Juckem and Dale M. Robertson

Abstract

Shell Lake is a relatively shallow terminal lake (tributaries but no outlets) in northwestern Wisconsin that has experienced approximately 10 feet (ft) of water-level fluctuation over more than 70 years of record and extensive flooding of nearshore areas starting in the early 2000s. The City of Shell Lake (City) received a permit from the Wisconsin Department of Natural Resources in 2002 to divert water from the lake to a nearby river in order to lower water levels and reduce flooding. Previous studies suggested that water-level fluctuations were driven by long-term cycles in precipitation, evaporation, and runoff, although questions about the lake's connection with the groundwater system remained. The permit required that the City evaluate assumptions about lake/groundwater interactions made in previous studies and evaluate the effects of the water diversion on water levels in Shell Lake and other nearby lakes. Therefore, a cooperative study between the City and U.S. Geological Survey (USGS) was initiated to improve the understanding of the hydrogeology of the area and evaluate potential effects of the diversion on water levels in Shell Lake, the surrounding groundwater system, and nearby lakes. Concerns over deteriorating water quality in the lake, possibly associated with changes in water level, prompted an additional cooperative project between the City and the USGS to evaluate effects of changes in nutrient loading associated with changes in water levels on the water quality of Shell Lake.

Numerical models were used to evaluate how the hydrology and water quality responded to diversion of water from the lake and historical changes in the watershed. The groundwater-flow model MODFLOW was used to simulate groundwater movement in the area around Shell Lake, including groundwater/surface-water interactions. Simulated results from the MODFLOW model indicate that groundwater flows generally northward in the area around Shell Lake, with flow locally converging toward the lake. Total groundwater inflow to Shell Lake is small (approximately 5 percent of the water budget) compared with water entering the lake from precipitation (83 percent) and surface-water runoff (13 percent). The

MODFLOW model also was used to simulate average annual hydrologic conditions from 1949 to 2009, including effects of the removal of 3 billion gallons of water during 2003–5. The maximum decline in simulated average annual water levels for Shell Lake due to the diversion alone was 3.3 ft at the end of the diversion process in 2005. Model simulations also indicate that although water level continued to decline through 2009 in response to local weather patterns (local drought), the effects of the diversion decreased after the diversion ceased; that is, after 4 years of recovery (2006–9), drawdown attributable to the diversion alone decreased by about 0.6 ft because of increased groundwater inflow and decreased lake-water outflow to groundwater caused by the artificially lower lake level. A delayed response in drawdown of less than 0.5 ft was transmitted through the groundwater-flow system to upgradient lakes. This relatively small effect on upgradient lakes is attributed in part to extensive layers of shallow clay that limit lake/groundwater interaction in the area.

Data collected in the lake indicated that Shell Lake is polymictic (characterized by frequent deep mixing) and that its productivity is limited by the amount of phosphorus in the lake. The lake was typically classified as oligotrophic-mesotrophic in June, mesotrophic in July, and mesotrophic-eutrophic in August. In polymictic lakes like Shell Lake, phosphorus released from the sediments is not trapped near the bottom of the lake but is intermittently released to the shallow water, resulting in deteriorating water quality as summer progresses.

Because the productivity of Shell Lake is limited by phosphorus, the sources of phosphorus to the lake were quantified, and the response in water quality to changes in phosphorus inputs were evaluated by means of eutrophication models. During 2009, the total input of phosphorus to Shell Lake was 1,730 pounds (lb), of which 1,320 lb came from external sources (76 percent) and 414 lb came from internal loading from sediments in the lake (24 percent). The largest external source was from surface-water runoff, which delivered about 52 percent of the total phosphorus load compared with about 13 percent of the water input. The second largest source was from precipitation (wetfall and dryfall), which delivered

19 percent of the load compared to about 83 percent of the water input. Contributions from septic systems and groundwater accounted for about 3 and 2 percent, respectively. Increased runoff raises water levels in the lake but does not necessarily increase phosphorus loading because phosphorus concentrations in the tributaries decline during increased flow, possibly because of shorter retention times in upstream wetlands. Phosphorus loading to the lake in 2009 represented what occurred after a series of dry years; therefore, this information was combined with data from 2011, a wet year, to estimate phosphorus loading during a range of hydrologic conditions by estimating loading from each component of the phosphorus budget for each year from 1949 to 2011.

Comparisons of historical water-quality records with historical water levels and applications of a hydrodynamic model (Dynamic Lake Model, DLM) and empirical eutrophication models were used to understand how changes in water level and the coinciding changes in phosphorus loading affect the water quality of Shell Lake. DLM simulations indicate that large changes in water level (approximately 10 ft) affect the persistence of stratification in the lake. During periods with low water levels, the lake is a well-mixed, polymictic system, with water quality degrading slightly as summer progresses. During periods with high water levels, the lake is more stratified, and phosphorus from internal loading is trapped in the hypolimnion and released later in summer, which results in more extreme seasonality in water quality and better clarity in early summer.

Results of eutrophication model simulations using a range in external phosphorus inputs illustrate how water quality in Shell Lake (phosphorus and chlorophyll *a* concentrations and Secchi depths) responds to changes in external phosphorus loading. Results indicate that a 50-percent reduction in external loading from that measured in 2009 would be required to change phosphorus concentrations from 0.018 milligram per liter (mg/L) (measured in 2009) to 0.012 mg/L (estimated for the mid-1800s from analysis of diatoms in sediment cores). Such reductions in phosphorus loading cannot be accomplished by targeting septic systems or internal loading alone because septic systems contribute only about 3 percent of the phosphorus input to the lake, and internal loading from the sediments of Shell Lake contributes only about 25 percent of phosphorus input. Complete elimination of phosphorus from septic systems and internal loading would decrease the phosphorus concentrations in the lake by 0.003–0.004 mg/L. Therefore, reducing phosphorus concentration in the lake more than by 0.004 mg/L requires decreasing phosphorus loading from surface-water contributions, primarily runoff to the lake.

Reconstructed changes in water quality from 1860 to 2010, based on changes in the diatom communities archived in the sediments and eutrophication model simulations, suggest that anthropogenic changes in the watershed (sawmill construction in 1881; the establishment of the village of Shell

Lake; and land-use changes in the 1920s, including increased agriculture) had a much larger effect on water quality than the natural changes associated with fluctuations in water level. Although the effects of natural changes in water level on water quality appear to be small, changes in water level do have a modest effect on water quality, primarily manifested as small improvements during higher water levels. Fluctuations in water level, however, have a larger effect on the seasonality of water-quality patterns, with better water quality, especially increased Secchi depths, in early summer during years with high water levels.

Introduction

Shell Lake is a relatively shallow, soft-water, closed-basin, terminal lake in Washburn County, Wisconsin (Wis.) (fig. 1). A closed-basin lake has no surface-water outflows, and a terminal lake is a closed-basin lake that is fed by annual or perennial inflowing streams. Since water-level measurements began in 1936, the water level of the lake has fluctuated by about 10 feet (ft) (about 30 percent of its maximum depth). Starting around 1990, the lake level rose above average conditions; and in the early 2000s, numerous homes and municipal buildings were flooded (fig. 2). In an effort to reduce flooding, the City of Shell Lake (City) sought a permit from the Wisconsin Department of Natural Resources (WDNR) to divert water out of the lake to a nearby river. The City was granted a permit by the WDNR to divert lake water in August 2002. A pipeline to convey water from the lake to a discharge point on the Yellow River was constructed during 2003 (fig. 1), and water was removed from the lake during 2003–5. The permit required that the City comply with specific conditions. One such condition was the “hydrogeology condition” (part of permit condition number 62), which required the City to evaluate the hydrogeologic conclusions made in a previous study by Krohelski and others (1999) and to evaluate the effects of the water diversion on subsequent water levels in Shell Lake and other lakes in the immediate vicinity. The condition also required monitoring of near-lake groundwater quality to evaluate the possibility of contaminant migration from flooded local residences. Potential contaminants include nutrients from septic systems, volatile organic compounds from potentially submerged petroleum containers, and heavy metals. All of these contaminants, if present in groundwater, could travel toward the lake during potential groundwater gradient reversals associated with flooding followed by a drop in lake levels. A cooperative project between the City of Shell Lake and the U.S. Geological Survey (USGS) was initiated in 2006 to satisfy the permit-required conditions for evaluating the hydrology of the lake and its interaction with the groundwater-flow system and was partially funded by the Cooperative Water Program of the USGS.

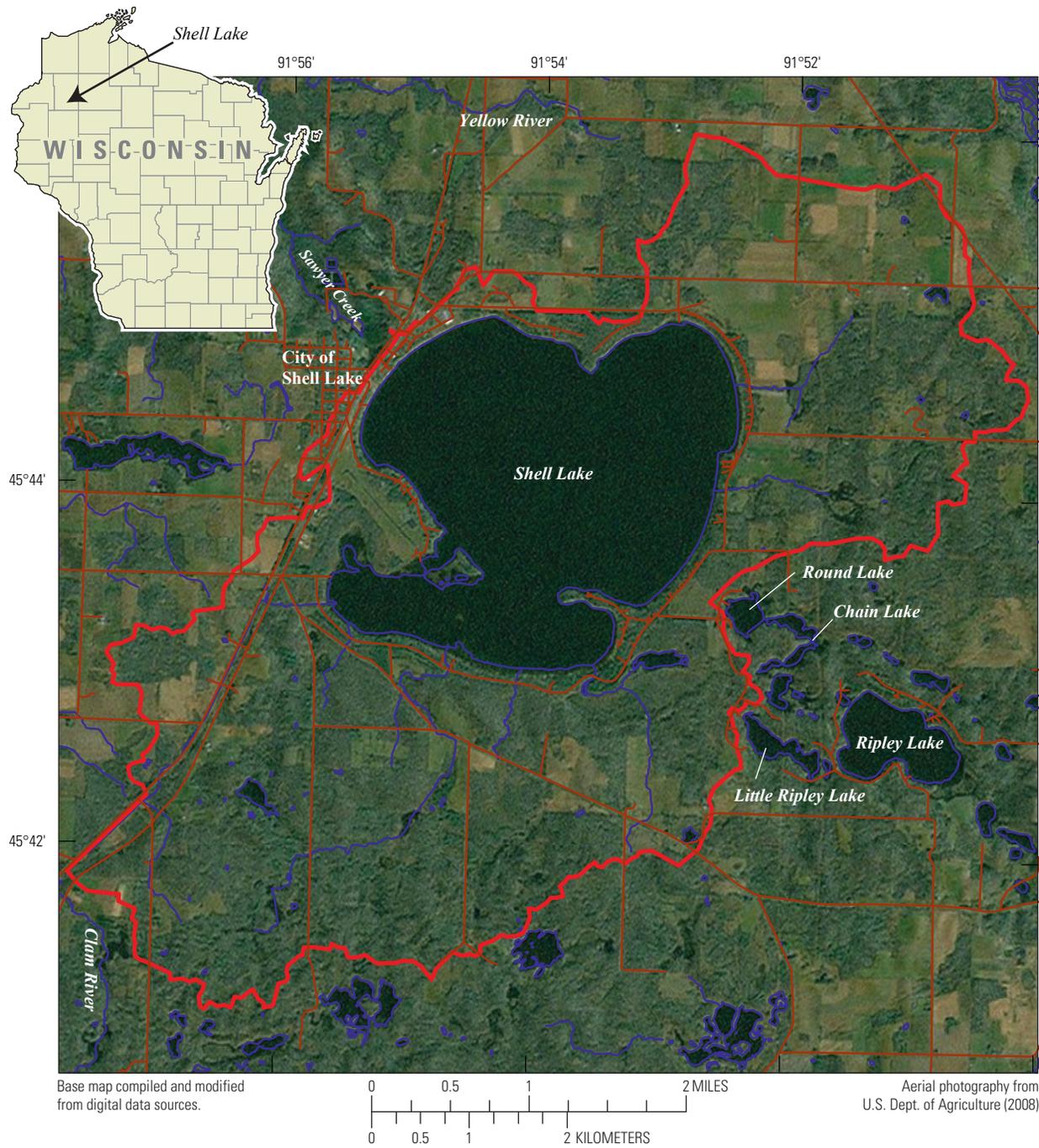


Figure 1. Shell Lake, Washburn County, Wisconsin, and its watershed.

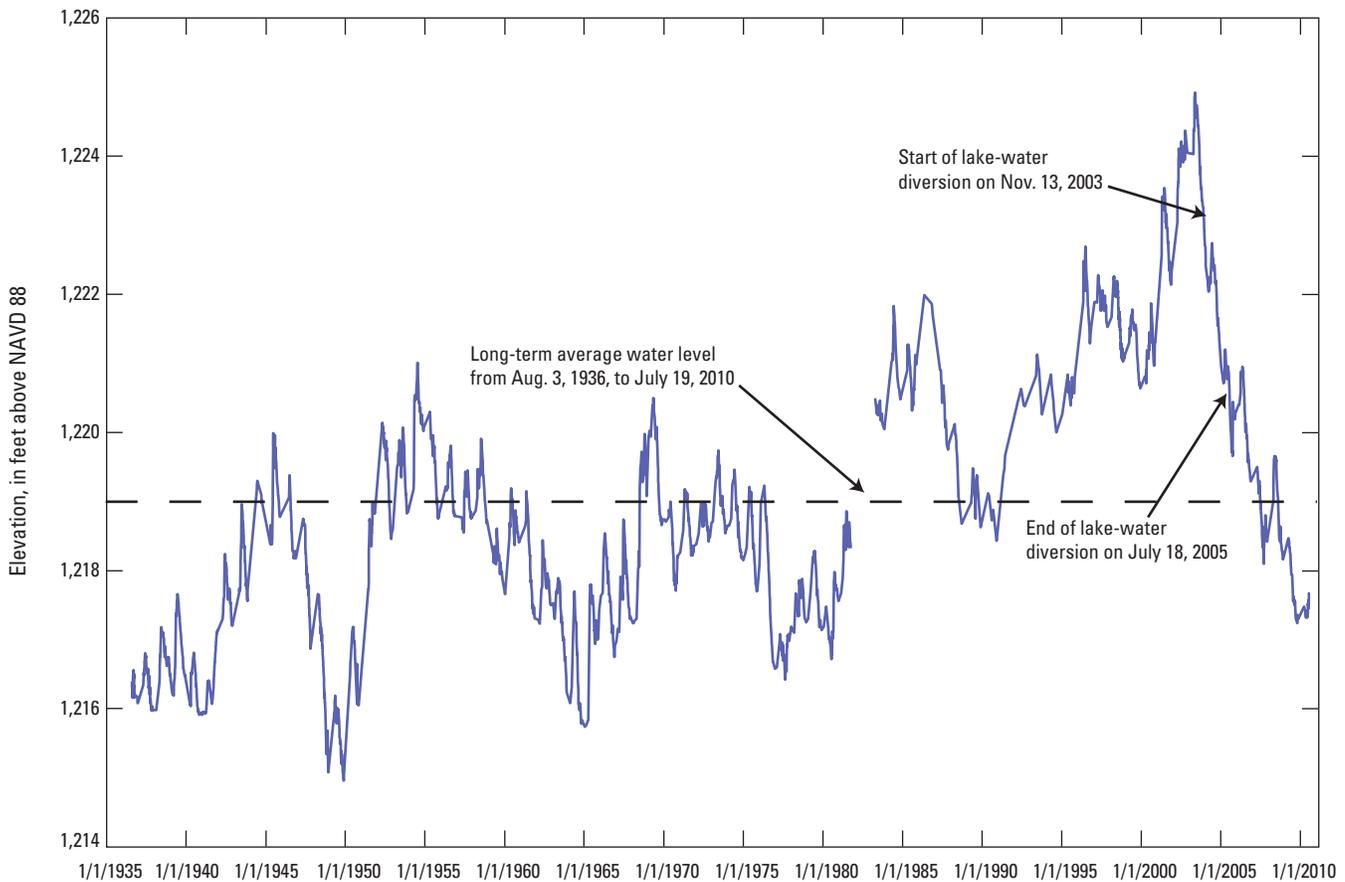


Figure 2. Historical water levels measured in Shell Lake, Wisconsin.

Concerns over deteriorating water quality in Shell Lake, especially associated with changes in water level and possibly associated with the water diversions, prompted an additional cooperative project between the City of Shell Lake and the USGS. As part of this project, further monitoring of the lake and groundwater system were conducted, primarily in 2009, to understand how changes in water level affect the water and nutrient inputs into the lake and, ultimately, the water quality of the lake. This project was partially funded through the Lake Protection Grant Program of the WDNR and the Cooperative Water Program of the USGS. This report describes both the permit-required lake-water/groundwater investigation (the hydrogeologic study) and the lake water-quality investigation.

Shell Lake and Its Watershed

Shell Lake (fig. 1) is a natural, closed-basin, terminal lake in Washburn County, in the Northern Lakes and Forests Ecoregion (Omernik and others, 2000). Precipitation that falls within the watershed of a closed-basin lake does not flow

out of the basin, and water leaves the lake only by evaporation and seepage into the underlying groundwater system. Six ephemeral stream channels contribute surface water to the lake, all of which drain mixed agricultural and forested areas, and have no contributions from point sources such as wastewater treatment plants (fig. 1). The sources of water to Shell Lake were described by the Wisconsin Department of Natural Resources (1982), which indicated that precipitation is the primary source of water, followed by tributary streamflows and groundwater inflow. Without a surface-water outlet, evaporation is expected to be the dominant process by which water leaves Shell Lake.

The area and volume of Shell Lake were given on the 1965 Wisconsin Conservation Department lake survey map as 2,580 acres and 60,378 acre-feet (acre-ft), respectively, at an elevation of 97.65 ft (equivalent to a lake stage of 1,217 ft). In this study, the morphometry of the lake was reevaluated on the basis of a 2005 aerial image obtained from the National Agricultural Imagery Program (U.S. Department of Agriculture, 2006). Through the use of geographic information system (GIS) techniques, the surface area of Shell Lake, at an

elevation of 1,217 ft, was determined to be 2,556 acres and the volume to be 59,600 acre-ft, with a mean depth of 23.3 ft. The water level of Shell Lake is prone to considerable annual and long-term variability. Over the period of record from 1936 to 2011, the average stage of the lake was 1,219 ft. At this stage, the lake is estimated to cover approximately 2,625 acres (4.1 square miles (mi²)) and to have a volume of approximately 65,600 acre-feet, with a maximum depth of about 38 ft and a mean depth of 25.0 ft.

Shell Lake is a relatively shallow lake. Shallow lakes, defined as having a maximum depth less than about 20 ft (Osgood and others, 2002), typically experience only short periods of stratification, when the cooler, more dense, bottom water (hypolimnion) does not mix with the warmer, less dense, surface water (epilimnion). Deep lakes tend to experience a single extended period of stratification from early summer through early fall. Lakes in which water mixes frequently throughout the open-water period are called “polymictic,” whereas lakes in which water mixes only during spring and fall and have extended periods of stratification in summer and under the ice are called “dimictic.” Lakes deeper than 20 ft with relatively large surface areas may also be prone to frequent deep mixing. Osgood (1988) described the functional aspects of mixing from the forces of wind based on the mean depth and surface area of lakes in terms of the Osgood Index. The Osgood Index is defined as the mean depth (z , in meters) of a water body divided by the square root of the surface area (A , in square kilometers), or $z/A^{0.5}$. Lakes with Osgood Index values greater than 9 are usually dimictic, and lakes with values less than 4 tend to be polymictic. Shell Lake has an Osgood Index of 2.3 at a stage of 1,219 ft, with the index ranging from about 2.2 (stage of about 1,215.6) to about 2.6 (stage of about 1,224). Therefore, Shell Lake is expected to be polymictic in most years; however, the extent of mixing may vary depending on its stage.

Phosphorus that enters a lake from its watershed drives lake productivity (plant and algal growth) and then is either exported out its outlet or deposited in the sediments in the deep areas of the lake. Not all of the phosphorus that is deposited in the bottom sediments, however, remains in the sediment; some of it is released back into the water column. This release from the bottom sediment is referred to as “internal phosphorus loading.” Typically, in deeper, dimictic lakes, phosphorus released from the deep sediments is retained in the hypolimnion of the lake and is released to the shallower epilimnion primarily at fall turnover. Therefore, phosphorus concentrations near the surface (termed “near-surface concentrations”) in dimictic lakes usually remain stable or decrease as summer progresses (Welch and Cooke, 1995). In polymictic lakes, however, phosphorus that is released from the sediments is frequently delivered to the surface of the lake throughout summer. The frequent mixing results in near-surface concentrations of phosphorus increasing throughout summer and the lake being more productive in late summer than earlier in the year (Welch and Cooke, 1995). Several factors affect the rate

at which phosphorus is released from the bottom sediments. When deep water of a productive lake (a lake with abundant algae or plants) becomes isolated from surface mixing because of thermal stratification, the bottom water often becomes anaerobic or devoid of dissolved oxygen because of the consumption of oxygen associated with the decomposition of organic materials. When oxygen is completely depleted (anoxia) near the bottom sediment, the rate of phosphorus release from the deep sediments often increases dramatically (Mortimer, 1941). Another factor affecting the rate of phosphorus release is the water temperature at the sediment interface. James and Barko (2004) found that phosphorus release rate increases as the water temperature above the sediment increases. Because near-bottom temperatures in shallow lakes are typically higher than in deep lakes, the rate of phosphorus release in shallow lakes can be higher than in deep lakes. The importance of internal phosphorus loading in Shell Lake may vary through time because of changes in depth and potential changes in the extent of stratification.

Polymictic lakes like Shell Lake can be extremely vulnerable to phosphorus contributions from areas outside of the lake (termed “external loading”) and can be difficult to rehabilitate. Once phosphorus concentrations in the bottom sediments of polymictic lakes become elevated, a condition that tends to increase internal loading, the lakes usually do not respond quickly to decreases in external phosphorus inputs (Welch and Cooke, 1995; Nurnberg, 1998; Newton and Jarrell, 1999).

Land Use and Population

Shell Lake’s watershed (excluding the 4.1-mi² area of the lake) is about 13.1 mi², and it contains primarily a mixture of forest (54.1 percent), agriculture (26.0 percent), wetlands (8.3 percent), and open space and grassland (6.5 percent); the remainder consists of smaller areas of urban and residential land (3.0 percent) and open water (2.1 percent) (fig. 1; Multi-Resolution Land Characteristics Consortium, 2001). Most of the wetlands in the basin are south and east of the lake and do not intercept the water draining from the agricultural areas on the north side of the lake.

The City of Shell Lake, on the northwest side of the lake, is a rural community that has not experienced the recent rapid growth that other areas in the northwoods have (Garn and others, 2010). The population as reported in decennial censuses from 1910 to 1970 hovered near 900. From 1970 to 2010, the population increased from 928 to 1,353 (U.S. Census Bureau, 2012). Although the population in the area has not increased very much since 1910, the number of lakefront homes around the lake has increased. In addition, in many cases, small summer cottages have been replaced by large urban-style homes accompanied by larger impervious surfaces, clearing of natural vegetation from shorelines, and creation of lawns down to the edge of water.

Previous Scientific Investigations of Shell Lake

Shell Lake has a long history of water-level and water-quality monitoring and studies. As mentioned previously, the water level of Shell Lake has been prone to considerable annual and long-term variability. Based on measurements beginning in 1936, the water level of Shell Lake commonly fluctuates about 0.5–2 ft annually but has fluctuated by nearly 10 ft over the period of record, from a low stage of 1,214.96 ft (above NGVD 29) in 1949 to a high of 1,224.92 ft in 2003 (fig. 2). The average water level for the period of record from 1936 to 2011 is 1,219 ft. A study of the lake by the WDNR in 1980 (Wisconsin Department of Natural Resources, 1982) described the water level of Shell Lake prior to when water-level measurements were begun by the USGS in 1936. That report mentioned an 1852 atlas that showed an outlet stream from Shell Lake to Sawyer Creek. Water levels were apparently high from the early 1880s to about 1911 but receded by more than 17 ft by the early 1930s; however, the exact water levels and dates used to make this determination are unknown.

Concerns arose in the 1930s in response to decreasing water levels in the lake; in response, a diversion of flow from the Clam River watershed into the Shell Lake watershed was constructed in 1942 (Wisconsin Department of Natural Resources, 1982) that increased Shell Lake's watershed by about 1 mi². Concerns over water levels again arose in the 1980s, and because of high water levels in Shell Lake, the Clam River diversion was plugged by 1986 (U.S. Army Corps of Engineers, 1988); however, water levels continued to rise after the Clam River diversion was discontinued. Consideration for an outlet diversion from the lake began during this period of high water and led to a report by the U.S. Army Corps of Engineers (1988) on the economics and feasibility of an outlet diversion. It also led to a study by Krohelski and others (1999), who used field measurements and a water-budget model to simulate the potential effect of lake-water withdrawal on the water levels of Shell Lake. Simulated results indicated that diverting 1,000 gallons per minute, or 526 million gallons per year, of lake water could lower lake levels by about 1 ft in 1.5 years. Results of these simulations were used to design capacity for a diversion pipe for Shell Lake. In 2002, the City of Shell Lake was granted a permit to divert water out of the lake to the Yellow River from the northwest side of the lake (fig. 1). A pipeline to convey water from the lake to a discharge point on the Yellow River was constructed during 2003. The diversion pipe was used to divert lake water at various rates and for various durations from November 2003 to July 2005.

Concerns arose over how the water diverted from Shell Lake affected the groundwater system and the water level of nearby lakes. Because the water-budget model of Krohelski and others (1999) incorporated limited groundwater/

lake-water interaction and focused solely on Shell Lake, it thus could not be used to assess effects of the diversion on the water level of nearby lakes; therefore, alternative approaches to examine the hydrology near Shell Lake were considered. Following completion of the water-budget model by Krohelski and others (1999), a lake-budget module for the USGS groundwater-flow model, MODFLOW (Harbaugh, 2005), was developed and published (Merritt and Konikow, 2000). This lake-budget module is similar to the Shell Lake water-budget model of Krohelski and others (1999), but because it is coupled to a groundwater-flow model, it can solve for and simulate gradients between the lake and the groundwater system at discrete intervals (for example, every 150 ft along the Shell Lake shoreline). This groundwater-flow model also enables the simulation of water levels for nearby lakes and links the lakes together through the simulation of the shallow groundwater-flow system. This new version of MODFLOW with the lake-budget module was used in the current study described in this report.

The water quality in Shell Lake has been examined as part of several studies. Shell Lake was one of many lakes studied by Edward A. Birge and Chancey Juday—the fathers of modern limnology at the University of Wisconsin—in the early 1900s. Some of the original data collected by Birge and Juday are described in the WDNR report on Shell Lake (Wisconsin Department of Natural Resources, 1982). Work by Birge and Juday focused on describing in-lake physical and chemical properties. Subsequent studies have continued those evaluations.

A study by Wisconsin Department of Natural Resources (1971) examined the water quality of Shell Lake and nutrient loading to the lake. That study classified Shell Lake as an oligotrophic lake (a relatively unproductive lake with limited plants and algae and low nutrient concentrations), despite reporting some questionably high phosphorus and nitrate concentrations. The study also evaluated sources of nutrients to the lake, including septic systems and tributaries. Evaluation of 72 septic systems led the authors to encourage the City to extend sanitary sewer lines around the lake to reduce potential nutrients leached from private septic systems, although the amount of nutrient loading (input over a specified time period) from septic systems was not quantified. From synoptic sampling of the tributary streams in spring 1971, the authors estimated that more than 200 pounds (lb) of phosphorus may have entered the lake from three ephemeral streams during a 7-day period.

The first detailed water and nutrient budget of Shell Lake was constructed as part of a study by WDNR in 1980 (Wisconsin Department of Natural Resources, 1982). That was the first study of Shell Lake aimed at identifying the major sources of water and nutrients to the lake. As part of that study, precipitation, groundwater, and tributary contributions were

estimated and used to compute phosphorus loads to the lake. The authors estimated that the total phosphorus load to the lake during 1980 was about 1,800 lb. As part of that study, various management scenarios were considered and used to estimate changes in potential phosphorus loading to the lake, and an empirical eutrophication model by Dillon and Rigler (1974) was used to simulate how these changes in loading would affect algal concentrations in the lake (as indicated by chlorophyll *a* concentrations). Although the study was robust for its time, factors such as three-dimensional groundwater flow and internal phosphorus loading from bottom sediments of the lake were not considered. Despite these limitations, that study provided numerous useful suggestions for watershed management that could improve the water quality of the lake, including land management that would minimize disturbance to the wetlands and minimize increases in nutrient loading to the lake.

Effects of Changes in Nutrient Loading, Water Level, and Climate Change on Water Quality

Degradation in the water quality of many lakes, including Shell Lake, has been caused by an increase in nutrient inputs from their watersheds. The effect of nutrient loading on a lake's water quality is sufficiently understood such that empirical eutrophication models have been developed that can predict in-lake nutrient (phosphorus) and algal (chlorophyll *a*) concentrations and water clarity (Secchi depth) from external phosphorus loading (Reckhow and Chapra, 1983; Cooke and others, 1993; Panuska and Kreider, 2003). Most of these models were derived from analyses of many lakes with very different nutrient loading rates and very different hydrologic conditions (for example, Dillon and Rigler, 1974; Vollenweider, 1975; Canfield and Bachmann, 1981).

Changes in nutrient loading to lakes are usually associated with changes in anthropogenic factors in the watershed, such as changes in land use or changes in contributions from point sources; however, changes in nutrient loading also occur naturally as a function of hydrologic changes. For example, more nutrients are usually delivered to a lake during wet years than during dry years. Changes in hydrology often result in fluctuations in a lake's water level. Fluctuations in water levels in lakes are natural phenomena; however, it is believed that global climate change can affect and may have already affected the extent of water-level fluctuations in lakes (Wantzen and others, 2008). On the basis of results from General Circulation Models (GCMs), many scientists believe that climate change will either cause systematic decreases or increases in water levels, or cause water levels to fluctuate more widely than they have previously in response to larger

fluctuations in precipitation and runoff (Mulholland and Sale, 1998; Bates and others, 2008). Predicting the effects of climate change on a particular lake from large global and regional models is difficult because a lake's hydrological cycle involves many interrelated components, including precipitation patterns, water temperatures, evaporation rates, groundwater inputs and outputs, and surface-water flow rates (Wittman, 2008). It is believed, however, that closed-basin lakes—lakes without outlets, such as Shell Lake—are most vulnerable to changes in climate because of their sensitivity to changes in the balance of inflows and evaporation (Bates and others, 2008). Results from some GCMs indicate that seasonal precipitation patterns will change and will result in more precipitation in winter and less in summer, which could cause groundwater levels to increase and cause a rise in the water levels of lakes that receive large groundwater inputs. Increased air temperatures, however, may cause an increase in evapotranspiration and cause water levels to drop (Wittman, 2008). Only a few studies have examined how changes in water level affect the water chemistry of lakes (Leira and Cantonati, 2008). Most of these studies primarily examined how fluctuations in water level are related to changes in primary ions and alkalinity (for example, Webster and others, 1996; White and others, 2008); only a few studies have examined the effects on the trophic state (nutrient and algal concentrations and water clarity) of lakes. Robertson and Rose (2011) found that fluctuations in the water levels of relatively deep lakes were directly related to changes in their water quality, with slightly higher nutrient and algal concentrations and poorer water clarity occurring when the lakes were deeper (high water level) because of increased hydrologic inputs. Hoyer and others (2005), who examined how fluctuations in the water level of relatively shallow lakes in Florida affect water quality, found no overall consistent relation between water-level fluctuations and changes in trophic-state variables; however, they did find statistically significant changes (both improving and degrading water quality) for individual lakes.

The recent high water levels in Shell Lake have led to several questions: Are there changes in water quality in Shell Lake that are associated with the changes in water level? If so, are they as large as the changes caused by anthropogenic factors in the watershed, and are the changes indicative of what may occur in the future with climate change?

There are several ways to predict how the water quality of a specific lake may respond to hydrologic changes and future climate change. One approach is to examine how the lake has responded to past changes in water level, if data are sufficient for such an analysis. Another approach is to use numerical models to simulate how the lake is expected to respond to specific hydrologic conditions.

Purpose and Scope

The primary purposes of the hydrogeologic and lake water-quality studies were to evaluate the effects of diverting water from Shell Lake to the Yellow River on the water level of Shell Lake and nearby upgradient lakes, increase the understanding of the hydrology and phosphorus budgets of Shell Lake, and estimate how the water quality of Shell Lake will respond to changes in phosphorus loading and changes in water level.

The scope of these studies was limited to the ground-water and surface-water systems in the Shell Lake area. The studies evaluated historical variations since 1860 but focused on the period from 2006 to 2009, when most data were collected. This report describes (1) the conceptualization of the groundwater-flow system and lake hydrology; (2) the hydrologic (surface-water and groundwater) conditions near Shell Lake; (3) the calibrated groundwater model and its parameters and sensitivities; (4) the water quality measured in Shell Lake; (5) the phosphorus loading to the lake, with a description of the relative importance of each of the sources (inputs from residential shoreline development were not directly measured in this study); (6) the response in the water quality of Shell Lake to changes in phosphorus loading as inferred from water-quality models; and (7) the interpretation of simulated results from both the groundwater and the water-quality models.

Study Methods and Sampling Sites

The hydrogeologic and lake water-quality studies each involved two components—an onsite data-collection component and a numerical computer-modeling component. Onsite data collection is described in detail in this section of the report. A brief overview of the numerical models used to improve understanding of lake/groundwater interaction (the hydrogeologic study) and lake water-quality response to change (lake water-quality study) is provided at the end of this methods section. Detailed descriptions of the lake/groundwater interaction model (MODFLOW) and the lake water-quality analysis are given later under the corresponding section headings.

As part of this study, USGS personnel collected ground-water, lake, and stream-quality data from December 2006 to November 2011. Groundwater levels were monitored from November 2006 to April 2010, and groundwater-quality samples were collected annually from 2007 to 2009. Lake and stream data were collected primarily during open-water periods from February to November 2009. Owing to below-average precipitation during 2008–9, which began in 2005, additional stream data were collected through November 2011 during which there was increased rainfall and increased runoff to the lake. All USGS sampling locations in and around Shell Lake are shown in figure 3.

In addition to USGS data, water-quality data for Shell Lake were collected by the WDNR and volunteers from the City of Shell Lake as part of the WDNR's Wisconsin Citizen Lake Monitoring Program (Wisconsin Department of Natural Resources, 2008). WDNR and Citizen Lake Monitoring data were obtained from the WDNR's Surface Water Integrated Monitoring System (SWIMS) database (<http://dnr.wi.gov/topic/surfacewater/swims/>, accessed December 12, 2011). All available data, but primarily near-surface concentrations of total phosphorus and chlorophyll *a*, plus Secchi depths, were used to characterize long-term changes in the water quality of the lake; however, data collected during 2009 were primarily used to describe the hydrology and phosphorus inputs to the lake.

Historical lake water-level data were obtained from the USGS National Water Information System (NWIS) database, available at <http://wdr.water.usgs.gov/>, and the City of Shell Lake, available at <http://shelllakeonline.com> (accessed December 12, 2011).

Groundwater Monitoring

Eight nested wells (four shallow and four deep) were installed around Shell Lake in October and November 2006. Figure 3 illustrates the well locations and the naming convention, where well numbers started at 10 at the northeast corner of the lake and increased in the clockwise direction around the lake, ending at well nest 40 at the northwest corner of the lake. The eight nested wells were installed to improve characterization of the aquifer and confining-unit properties, help define areas contributing groundwater to the lake, assess the potential for groundwater contamination from volatile organic compounds (VOCs) and heavy metals associated with flooding of residential properties prior to the diversion, and help quantify the phosphorus loading from groundwater. In addition, six shallow, small-diameter minipiezometers (1/2-inch (in.)-diameter observation wells) were installed in July 2009 around the southern and eastern periphery of Shell Lake, where groundwater inflow to the lake was expected (fig. 3). The six shallow minipiezometers were installed and sampled to increase the spatial detail of phosphorus concentrations in groundwater entering the lake and thus to refine estimates of phosphorus loading from groundwater. Details of the well construction for the eight nested wells and six minipiezometers are provided in table 1.

At each of the four well-nest sites, one well was drilled shallow (20–30 ft below land surface), and the second well was drilled deep (30–80 ft). The minipiezometers were installed approximately 2–3 ft below the water table, to depths of 3–6 ft. Water levels were recorded hourly in each of the nested wells from December 2006 to April 2010, and the average water level in each well is listed in table 1. Dissolved phosphorus concentrations were measured once annually from 2007 to 2009 in the nested wells and twice during 2009 for

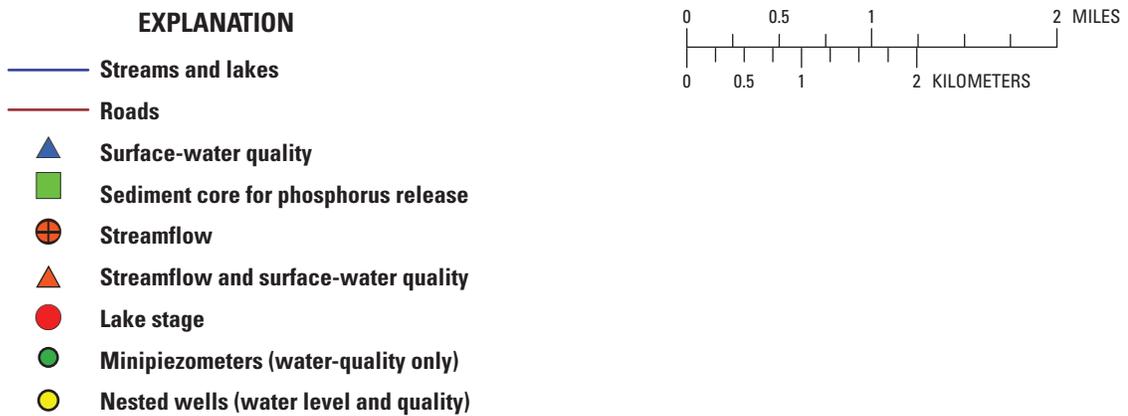
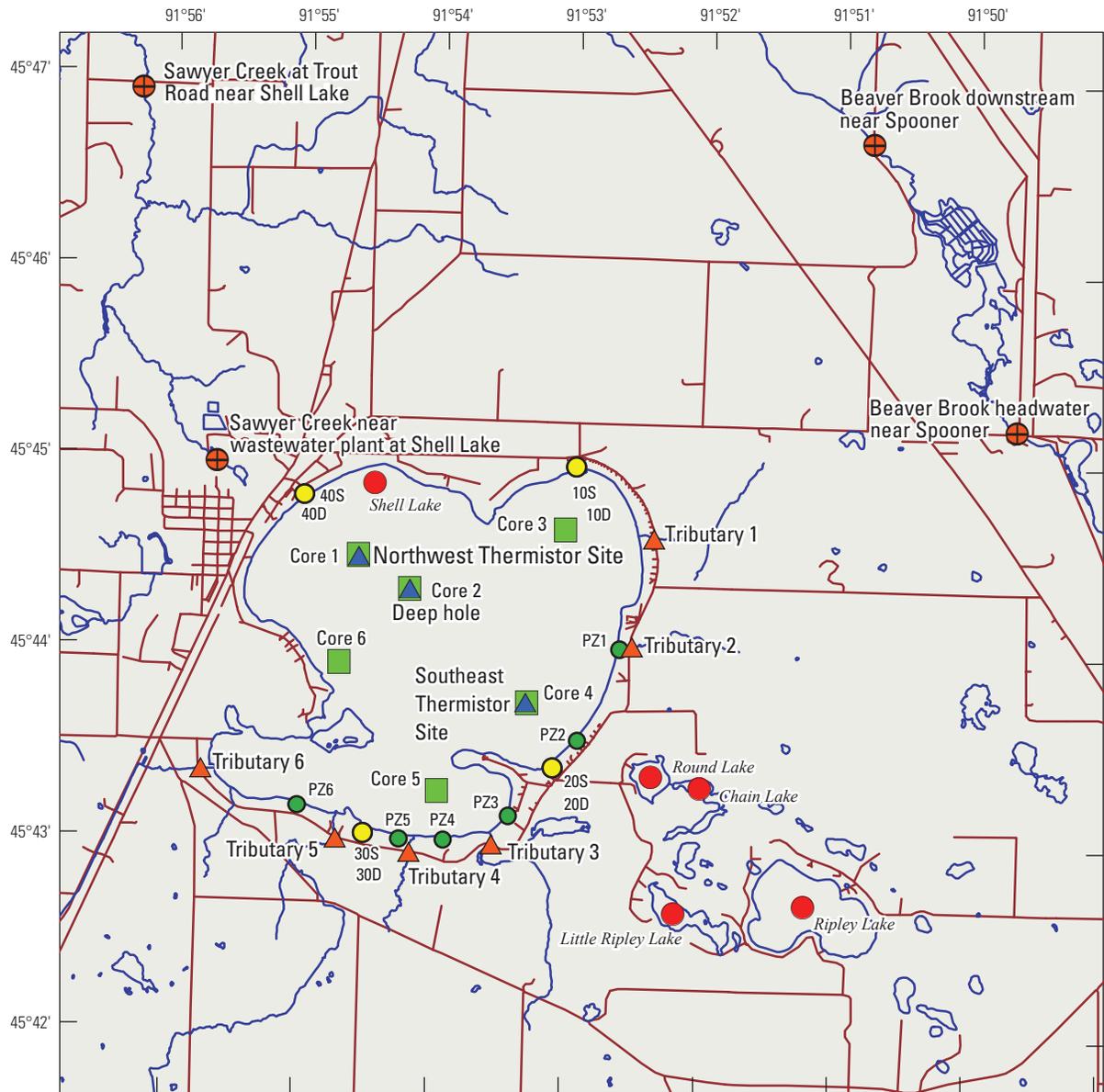


Figure 3. Locations and types of data-collection sites near Shell Lake, Wisconsin.

Table 1. Well construction details.

Well name	Well description	USGS station number	Date installed	Latitude	Longitude	Land elevation (feet above NAVD 88)	Height of casing (feet)	Depth of well (feet)	Length of screen (feet)	Diameter (inch)	Average water level 2007–8
10S	Shallow well at northeast nest	454458091525701	9/29/06	45°44'58.3"	91°52'57.3"	1220.85	2.34	12	5	1	1218.32
10D	Deep well at northeast nest	454458091525702	9/29/06	45°44'58.3"	91°52'57.3"	1221.12	2.33	73	3	1	1180.15
20S	Shallow well at southeast nest	454323091530501	9/30/06	45°43'23.5"	91°53'05.3"	1222.83	2.97	10	5	1	1221.14
20D	Deep well at southeast nest	454323091530502	9/30/06	45°43'23.5"	91°53'05.3"	1222.84	2.93	40	3	1	1211.13
30S	Shallow well at southwest nest	454301091542901	9/29/06	45°43'01.8"	91°54'29.7"	1221.34	2.52	11	5	1	1220.23
30D	Deep well at southwest nest	454301091542902	9/28/06	45°43'01.8"	91°54'29.7"	1220.86	2.85	45	3	1	1222.99
40S	Shallow well at northwest nest	454447091545901	11/15/06	45°44'47.7"	91°54'59.2"	1221.97	3.35	9	2.5	1	1215.57
40D	Deep well at northwest nest	454447091545902	11/15/06	45°44'47.7"	91°54'59.2"	1221.97	3.20	29	2.5	1	1205.78
PZ1	Minipiezometer on eastern shoreline near Tributary 2	454401091523700	7/14/2009	45°44'01.5"	91°52'36.8"	1220	--	5.9	0.5	0.5	--
PZ2	Minipiezometer on southeastern shoreline near well nest 20	454332091525500	7/14/2009	45°43'31.8"	91°52'55.1"	1220	--	4.8	0.5	0.5	--
PZ3	Minipiezometer on southeastern shoreline near Tributary 3	454308091532500	7/14/2009	45°43'07.8"	91°53'24.7"	1220	--	4.8	0.5	0.5	--
PZ4	Minipiezometer on southeastern shoreline near Tributary 4	454300091535400	7/14/2009	45°42'59.8"	91°53'53.6"	1220	--	3.3	0.5	0.5	--
PZ5	Minipiezometer on southern shoreline near Tributary 4	454300091541400	7/13/2009	45°42'59.5"	91°54'14.5"	1222	--	5	0.5	0.5	--
PZ6	Minipiezometer on southern shoreline near Tributary 5	454310091550000	7/14/2009	45°43'10.3"	91°54'59.9"	1223	--	4.3	0.5	0.5	--

each minipiezometer. Field measurements of water temperature, dissolved oxygen, specific conductance, and pH also were obtained for each well with a flowthrough chamber attached to a multiparameter meter prior to sample collection. All of the well and minipiezometer water-level and water-quality data collected by the USGS were published in the USGS online annual Water Data Report series (U.S. Geological Survey, 2006–10), available at <http://wdr.water.usgs.gov/index.html>.

Lake-Stage and Water-Quality Monitoring

The City of Shell Lake collects lake-stage data on an approximately weekly basis during the open-water period. Lake stages also were recorded approximately weekly at four upgradient lakes during the open-water period from May 2006 to October 2008. Reference datums for each lake were tied to sea level (NGVD 29) by the Washburn County Surveyor's Office in 2005. All data-collection locations around Shell Lake are shown in figure 3.

Water-quality data for Shell Lake were collected by the WDNR in 1968–71 as part of a statewide assessment and from 1980 to 1999 and 2005 to 2011 as part of their Long-Term Trend monitoring program. Water-quality data for the lake also were collected by volunteers from the local community from 1993 to 2011 and by the USGS in 2009 as part of this study. All three groups collected data near the deepest location in the lake, hereafter referred to as the “deep-hole site” (fig. 3). The USGS site number for this site is 454419091541100. During 2009, all three groups sampled the lake: USGS sampled the lake once in March, April, and September and twice per month in June–August; WDNR sampled the lake in May, July, August and September; and volunteers measured the water clarity three times in September. All Citizen Lake Monitoring was done in accordance with protocols described by Wickman and Herman (2005). USGS protocols were similar to those of the WDNR Long-Term Trend monitoring program. These protocols involved collecting vertical profiles of water temperature, dissolved oxygen, specific conductance, and pH with a multiparameter meter, and water clarity (Secchi depth) with a standard 8-in.-diameter black-and-white Secchi disk during each visit. Near-surface samples were collected with a Van Dorn sampler and were analyzed for total phosphorus and chlorophyll *a* concentrations. Samples were occasionally collected for various nitrogen species (nitrite plus nitrate and Kjeldahl nitrogen). Samples collected near the lake bottom (near-bottom samples) were collected about 3 ft above the sediment-water interface and were analyzed for total phosphorus concentration. Surface-water samples collected by the WDNR during spring turnover in 1969–70, 1980, and 1989–99 were analyzed for common ions and other characteristics such as color, turbidity, alkalinity, and total residue.

All of the lake-stage, water-quality, and quality-assurance data collected by the USGS were published in the USGS online annual Water Data Report series (U.S. Geological

Survey, 2009–10), available at <http://wdr.water.usgs.gov/index.html>. Data collected by the WDNR are available online in the Surface Water Integrated Monitoring System (SWIMS) at <http://prodoasjava.dnr.wi.gov/swims>.

In addition to the sampling described above, the USGS collected continuous water temperature data at two locations in the lake (deep-hole site and on the southeast side of the lake, fig. 3) from June 16, 2009, to September 22, 2009, to describe short-term changes in the extent of vertical stratification and help describe variability in dissolved oxygen concentrations in the lake. At both locations, water temperature was measured every 10 minutes at 6 depths: 1.5, 3, 5, 8, and 15 ft above the bottom. Water temperatures were measured with HOBO Pro v2 U22 Water Temperature Data Loggers. These loggers have an accuracy of 0.2 °C and a resolution of 0.02 °C over a range of 0 to 50 °C.

Lake Classification

One method of classifying the water quality of a lake is with trophic state index (TSI) values based on near-surface concentrations of total phosphorus, chlorophyll *a*, and Secchi depths, as developed by Carlson (1977). The indices were developed to place these three characteristics on similar scales to allow comparison of different lakes. TSI values based on total phosphorus concentrations (TSI_p), chlorophyll *a* concentrations (TSI_c), and Secchi depths (TSI_{SD}) were computed for each sampling by use of equations 1–3. The individual index values were averaged monthly, and the monthly average values were then used to compute growing-season (May through September) average TSI values:

$$TSI_p = 4.15 + 14.42 [\ln \text{total phosphorus (in milligrams per liter} \times 1,000)] \quad (1)$$

$$TSI_c = 30.6 + 9.81 [\ln \text{chlorophyll } a \text{ (in micrograms per liter)}] \quad (2)$$

$$TSI_{SD} = 77.12 - 14.41 [\ln \text{Secchi depth (in feet)}] \quad (3)$$

Oligotrophic lakes have TSI values less than 40, a limited supply of nutrients, low phosphorus concentrations (typically), low algal populations, high water clarity, and contain oxygen throughout the year in their deepest zones (Wisconsin Department of Natural Resources, 1992). Mesotrophic lakes have TSI values between 40 and 50, a moderate supply of nutrients, a tendency to produce moderate algal blooms and moderate clarity, and occasional oxygen depletions in the deepest zones of the lake. Eutrophic lakes have TSI values greater than 50, a large supply of nutrients, severe water-quality problems such as frequent seasonal algal blooms and poor clarity, and oxygen depletions commonly throughout the deeper zones of the lake. Eutrophic lakes with TSI values greater than 60 are often further classified as hypereutrophic lakes, and they typically have even more severe water-quality problems, including frequent extensive algal blooms.

Stream Monitoring

To estimate runoff and phosphorus loading from the main tributaries to Shell Lake, USGS staff made intermittent streamflow measurements and collected water samples at six ephemeral streams that drain areas to the south and east of Shell Lake (fig. 3) approximately monthly from February to November 2009. Streamflow measurements were made with a hand-held acoustic Doppler velocimeter, and water samples were collected by using grab methods when water was present. Water samples were analyzed for concentrations of total phosphorus. During most sampling periods, most streams were not flowing and, therefore, water samples were not collected. Daily streamflows at each stream were estimated during 2009 by linearly interpolating between measured flows. Daily phosphorus loads from each stream were then computed by multiplying the volumetrically weighted phosphorus concentration from all of the samples collected at a given site in 2009 by the estimated daily flows. Monthly and annual total phosphorus loads were then computed by summing the daily loads.

All water-quality data and corresponding instantaneous flow measurements were published in the USGS Water Data Report Series (U.S. Geological Survey, 2009–10); data are available online at <http://wdr.water.usgs.gov/> and are also available digitally from the NWIS mapper (<http://maps.waterdata.usgs.gov/mapper/index.html>).

Because of the dry conditions near Shell Lake in 2009, the measured runoff was expected to be much less than normal, and the phosphorus concentrations measured in the streams were thought to be nonrepresentative of years with more runoff. Therefore, to get more representative phosphorus concentrations for runoff occurring in other years, three additional runoff events were measured from December 2009 to November 2010, and five additional runoff events were measured from January 2011 to November 2011.

Phosphorus Release Rates from Shell Lake Sediments

Release rates of phosphorus from the sediments during aerobic and anaerobic conditions (used to estimate internal loading) were determined from analysis of sediment cores collected at six locations in the lake (fig. 3) by using the sediment-core-incubation techniques described by James and Barko (1991). Sediment cores were collected from the lake on August 18, 2009, with a Wildco sediment-core sampler equipped with acrylic core liners. At the three deepest sites (core 1, core 2, and core 4), four cores were collected: two cores for aerobic analysis and two cores for anaerobic analysis. At the three shallowest sites (core 3, core 5, and core 6), only two cores were collected, both for aerobic analysis. After retrieval, the core liners, containing both sediment and overlying water, were immediately sealed with rubber stoppers and stored in a protective container for shipping to the

Wisconsin State Laboratory of Hygiene (WSLH). Additional near-surface lake water was collected for incubation with the collected sediment.

At the WSLH, the sediment cores were carefully drained of overlying water. Then additional water, collected from the lake, was filtered and siphoned onto the sediment contained in each of the core liners. All of the liners containing the sediment were placed in the dark and incubated at a constant temperature (approximately 22 °C) for 1–2 weeks. The incubation temperature was a constant temperature close to that above the sediments in Shell Lake during summer.

The aerobic/anaerobic environment of each core was controlled in the laboratory by gently bubbling either air (aerobic environment) or nitrogen (anaerobic environment) through an air stone placed just above the sediment surface. The limited bubbling action insured complete mixing of the water column but did not disrupt the sediment. Twelve of the cores (two from each location) were subjected to aerobic conditions, and six of the cores (two each from the three deepest sites) were subjected to anaerobic conditions. Therefore, all incubations were run in duplicate.

Samples for soluble reactive phosphorus analysis were collected daily from water above the sediment for each core and filtered through a 0.45-micrometer filter. The volume of water that was removed was replaced with an equivalent volume of lake water with a known concentration. Rates of phosphorus release from the sediments (in pounds per square foot per day) were then calculated as a linear change in mass in the overlying water divided by the time and the cross-sectional area of the incubation core liner. All results were corrected for the water (and phosphorus) that was removed from above the core and replaced with lake water. All of the aerobic and anaerobic sediment release rates were then averaged to obtain an aerobic release rate and an anaerobic release rate.

The amount of phosphorus released from the sediment (internal loading) is typically obtained by summing the amount released from the aerobic and anaerobic areas; however, no anaerobic areas were found in 2009. It was assumed that no phosphorus was released from areas less than 6.5 ft deep because these areas typically have little accumulated organic sediment. The daily sediment release is then typically computed as the product of the release rates and the respective aerobic and anaerobic areas.

Paleoecological Analysis

Separate paleoecological studies of the lake-bottom sediments of Shell Lake were done by the WDNR, Bureau of Integrated Science Services, and Jeremy Williamson (Polk County Land and Water Resources). The purpose of the WDNR study was to quantify historical changes in sedimentation rates in the lake and quantify changes in phosphorus concentrations and changes in the macrophyte community by examining changes

in historical diatom communities in the core (Garrison, 2012). The purpose of the study by Williamson was to provide insight into the limnological history of the lake, especially changes in the dissolved oxygen concentration in deep areas of the lake that may have resulted from anthropogenic changes in the watershed, by examining changes in Chironomid (aquatic insect) communities in the core (Jeremy Williamson, Polk County Land and Water Resources, written commun., 2012). Sediment from near the deep-hole site (fig. 3) was extracted by use of a piston coring device on Oct. 5, 2010, for use by both studies. Results of these paleoecological studies are described further under the heading “Inferred Lake Water Quality From Analyses of Lake Sediment.” Results of the paleoecological studies are compared with results from the eutrophication models (refer to “Relative Importance of Natural and Anthropogenic Changes in the Watershed to Productivity in Shell Lake”).

Numerical Model Simulations

Numerical model simulations were designed and run to improve understanding of the hydrology and water quality of Shell Lake and to evaluate how the hydrology and water quality are expected to respond to future changes that may occur in the watershed. The hydrologic system was simulated with the groundwater-flow model MODFLOW-2005 (Harbaugh, 2005) to directly simulate groundwater movement in the area around Shell Lake, including groundwater/surface-water interactions. Surface-water hydrology was simulated by using available packages in MODFLOW-2005, including the lake package (LAK; Merritt and Konikow, 2000), streamflow-routing package (SFR; Prudic and others, 2004; Niswonger and Prudic, 2005), and unsaturated-zone flow package (UZF; Niswonger and others, 2006). These packages in MODFLOW-2005 are further described under the heading “Simulation of the Lake/Groundwater System Near Shell Lake.” The MODFLOW model was first calibrated for steady-state conditions, or “average” conditions corresponding to January 1, 2007, to December 31, 2008, when the majority of data were collected for the hydrogeologic study and during a time when the water level of Shell Lake was relatively stable and near the long-term average (fig. 2). The steady-state simulation, for which water levels do not change over time, was performed to estimate spatial hydrogeologic properties, such as aquifer and confining-unit hydraulic conductivity. Following the steady-state calibration, the model was applied in transient mode, whereby annual average hydrologic conditions were simulated for each year from 1949 to 2009. Results from the transient model were used for evaluating the effects of the lake-water diversion on water levels as described under the heading “Simulation of the Lake/Groundwater System Near Shell Lake” and for estimating components of the Shell Lake water budget for the water-quality analysis.

The MODFLOW groundwater-flow model was used to simulate groundwater and surface-water interaction for Shell Lake and nearby lakes, but it has limited capability to simulate

event-based processes such as surface-water runoff to the lake, as needed for phosphorus-load calculations. Therefore, an empirical runoff model was developed and used to estimate the total annual runoff and phosphorus loading to Shell Lake for the water-quality analysis. This empirical model required a starting volume of water in the lake, annual groundwater input and output, evaporation from the MODFLOW model, and annual precipitation (refer to “Estimating Runoff from the Unmonitored Areas and Total Runoff to Shell Lake”).

After simulating the hydrology and estimating phosphorus loading to Shell Lake, water quality in the lake was simulated with eutrophication models that relate hydrology and phosphorus loading to measured water-quality characteristics in the lake. Empirical models, such as those used to simulate the water quality in Shell Lake, can be used to determine how changes in phosphorus loading to the lake—including that released from sediments on the lake bottom—could either improve or degrade water quality of the lake. These models were developed on the basis of comparisons of hydrologic and phosphorus loading estimates determined for many different lake systems with specific measures describing lake water quality, such as near-surface phosphorus and chlorophyll *a* concentrations and Secchi depth. The Canfield and Bachman (1981) natural-lake model was applied to Shell Lake to determine the following: how well the model simulated phosphorus concentrations in the lake, how changes in phosphorus loading affect phosphorus concentrations in the lake, and how changes in water level and phosphorus loading caused by changes in hydrology have affected water quality in the past and may affect water quality in the future. The total phosphorus concentrations predicted with the Canfield and Bachman model were then used to predict chlorophyll *a* concentrations and Secchi depths through the use of Carlson’s (1977) TSI equations (eqs. 2–3). In other words, the chlorophyll *a* concentrations and Secchi depths that yielded a similar TSI value as that computed from the predicted total phosphorus concentration were computed. Total phosphorus concentrations were also used to predict chlorophyll *a* concentrations through the use of a relation found by Hickman (1980). The lake water-quality response simulations are described further under the heading “Simulated Water Quality and Response to Changes in Phosphorus Loading.”

Historical climate changes have modified the hydrology and water level in Shell Lake, and future climate changes are expected to continue to modify the Lake’s hydrology and water level. These water-level fluctuations may affect the extent of stratification in the lake and, in turn, affect the amount of internal phosphorus loading and the timing when the phosphorus from the bottom sediments is released into the surface water of the lake. To determine how changes in water level may affect these processes, the Dynamic Lake Model (DLM; McCord and others, 2000; Robertson and others, 2002) was used to simulate how the extent of stratification could change in response to changes in water level. DLM is a mechanistic lake and reservoir model that was modified from the DYNAMIC REServoir Simulation Model, DYRESM

(Imberger and Patterson, 1981), by McCord and others (2000). Both DLM and DYRESM are process-based models that simulate the physical (temperature) structure of a lake on a daily basis using daily inputs of water and daily average meteorological conditions. These models have been successfully used to simulate vertical changes in water temperatures in many lakes and reservoirs around the world. The models are based on parameterizations of each of the individual mixing processes, so site-specific model calibration is generally not necessary. For a more complete description of DLM, see Robertson and others (2002). DLM was first evaluated for 2009, when the majority of in-lake data were collected. Then DLM was used to simulate the extent of stratification (thermal structure) with water levels similar to the extremes that have been experienced in the lake (3.3 ft lower than 2009 and 6.6 ft deeper than 2009).

Lake/Groundwater System near Shell Lake

The remainder of this report is subdivided into two sections. The first section, which starts here, describes the lake/groundwater system near Shell Lake. This section includes a conceptual model of the lake/groundwater system, environmental information needed for the groundwater model (such as precipitation, evaporation, runoff, and the hydraulic properties of the groundwater-flow system), a description of the steady-state and transient MODFLOW models used to simulate groundwater flow near Shell Lake, an application of the model to evaluate effects of a lake-water diversion on the lake/groundwater system, and limitations of the MODFLOW model. The second section of this report, “Lake Water-Quality Analysis,” focuses on the water quality in Shell Lake and how it responds to changes in nutrient loading and water level.

Conceptual Model of the Lake/Groundwater System Near Shell Lake

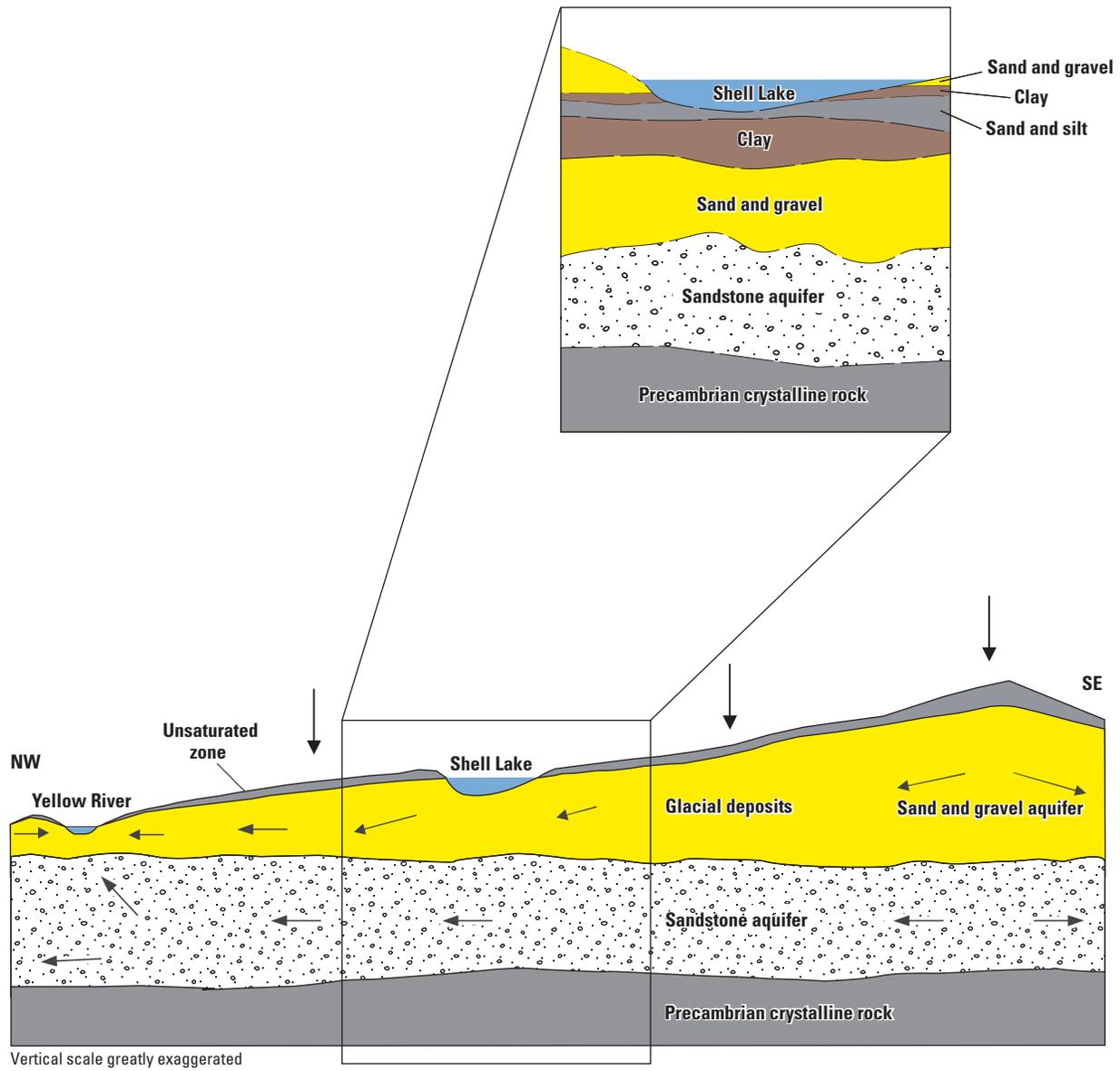
Prior to simulating the lake/groundwater system, a conceptual model is needed to aid in developing a computer model for the area near Shell Lake. The conceptual model is necessary to determine which components of the natural flow system are most important, because inclusion of all the complexities of the natural system into a computer model is not feasible. The conceptual model of the shallow groundwater-flow system near Shell Lake (fig. 4) was developed on the basis of soil-core analyses from the nested wells and previously published reports. The conceptual model describes the important hydrogeologic units of the groundwater-flow system and also describes the primary sources of recharge to the water table and locations of groundwater discharge. Boundaries of the groundwater system are described under the heading “Simulation of the Lake/Groundwater System near Shell Lake.”

Shell Lake was formed within glacial deposits that are underlain by sandstone rock. Precambrian crystalline rock underlies the sandstone and forms the lowermost boundary to the groundwater-flow system. The glacial deposits are a mixture of drift, outwash, and lake sediment near the Spooner Hills (Clayton, 1984; Johnson, 1986, 2000). Geologic cores around Shell Lake contained a complex mixture of sand, silt, and clay, which were generalized into six units of relatively homogeneous material for development of the conceptual model near Shell Lake. The six units, listed from shallow to deep, are as follows: (1) a relatively thin top layer of sand and gravel, (2) several feet of silt and clay, (3) a heterogeneous layer of predominantly sand and silt, (4) a relatively thick layer of clay, (5) a thick layer of sand and gravel, and (6) sandstone. The lateral extent and degree of variability within each of these units is unknown and was evaluated during the model calibration process.

Water generally enters the groundwater-flow system wherever groundwater does not discharge to a stream or lake. Most upland areas are generally considered to be groundwater-recharge areas, whereas groundwater discharge occurs in low-lying areas, such as many streams, wetlands, and lakes. A cursory evaluation of surface-water elevations in an area gives one a general sense for groundwater flow directions because groundwater tends to flow toward surface-water bodies with the lowest elevations. For example, Sawyer Creek is expected to receive substantial groundwater discharge (base flow) because of its relatively low elevation, also referred to as “low hydraulic head,” compared with other surface-water bodies in the area. The fact that many perennial streams in the Shell Lake area are approximately 40 ft below the elevation of Shell Lake and adjacent lakes is a strong indication that the lakes may have limited interaction with the regional groundwater-flow system due in large part to the thick layer of clay described above (unit 4). Locally recharged groundwater is expected to flow into Shell Lake and nearby lakes along their upgradient shores, and discharge from the lakes to the groundwater system is expected along the downgradient shores.

Hydrogeologic Characteristics of the Groundwater-Flow System

Initial estimates of hydraulic conductivity, recharge, and streambed and lakebed leakance for the three-dimensional MODFLOW groundwater-flow model were based on previous investigations in northern Wisconsin. Hydraulic conductivities of the geologic units in the area have been estimated by others (table 2) through the use of specific-capacity tests and hydrogeologic modeling. Horizontal hydraulic conductivities of sand and gravel aquifers are reported to range from 0.5 to 272 feet per day (ft/d). Measured horizontal hydraulic conductivities in the sandstone aquifer are lower than those in the sand and gravel aquifer. Vertical hydraulic conductivities in the sand and gravel aquifer and the sandstone aquifer are expected to be orders of magnitude lower than the



EXPLANATION

- ↓ Recharge
- ← Direction of groundwater flow

Figure 4. Conceptual model of groundwater flow near Shell Lake, Wisconsin.

corresponding estimates of horizontal hydraulic conductivity. Although these ranges are useful for characterizing the system, the MODFLOW model requires specific values for the hydraulic-conductivity variation in the system. Thus, hydraulic conductivities were treated as calibration parameters for the model. Final values used in the model were computed by use of pilot-point regularization with PEST (Doherty, 2008).

Rates of recharge are spatially variable because of differing soil percolation rates, slope, and relative topographic position, among other factors. This spatial variability is difficult to estimate and is naturally muted by the groundwater-flow system; thus, an average recharge rate was applied uniformly across the model domain. The average groundwater-recharge rate was expected to fall within the range of values reported for previously calibrated groundwater-flow models and recharge-estimation studies in the area (table 3). The steady-state recharge rate was treated as a calibration parameter, with the final values determined by use of the parameter-estimation code PEST to match observed water levels and streamflows. Transient recharge rates varied by year and were initially estimated as base flow in the St. Croix River at Danbury (the closest river with streamflow data from 1949 to 2009) divided by the watershed area. A single recharge multiplier was calibrated with PEST to adjust all annual recharge rates uniformly up or down for the transient calibration.

Estimates of streambed and lakebed leakance were needed to simulate the interaction between surface water and groundwater. Leakance is equal to the vertical hydraulic conductivity of a streambed or lakebed divided by its thickness. For this study, all streambeds were assumed to be 1 ft thick, and lakebeds were assumed to be 5 ft thick. A 5-ft-thick lakebed was used to represent the properties of the sandy shoreline around Shell Lake and adjacent lakes (the nearshore lakebed) as well as shallow clay that was observed during installation of wells and piezometers (fig. 5). Streambed hydraulic conductivity was expected to be within the range of values (1.6 to 37 ft/d) measured in streams in other parts of Wisconsin (Krohelski and others, 2000). Because of the relative insensitivity of the flow system to streambed leakance (see "Sensitivity Analysis" section), streambed hydraulic conductivity was not adjusted during calibration and was set at 2 ft/d for all rivers and streams. Simulated lake stages were sensitive to lakebed leakance, however, and were therefore estimated during the steady-state calibration. Vertical hydraulic conductivities for lakebed sediments were expected to be of similar magnitude as previously simulated values by Krohelski and others (1999) for Shell Lake (0.02 ft/d) and by Juckem and Hunt (2007) for lakes near Hayward, Wis. (0.02 to 0.1 ft/d). The calibrated vertical hydraulic conductivities for this study were estimated at 0.027 ft/d for the lakes (the equivalent lakebed leakance is 0.0053 day⁻¹).

Hydrologic Information for the Lake Water Budget

The water budget of a lake controls its water level. The water budget for Shell Lake can be described as follows:

$$(P + SW_{in} + GW_{in}) - (E + GW_{out}) = \Delta S, \quad (4)$$

where ΔS is the change in the volume of water stored in the lake and is equal to the sum of the volumes of water entering the lake minus the sum of the volumes leaving the lake. Water enters the lake as precipitation (P), surface-water inflow (SW_{in}), and groundwater inflow (GW_{in}). Water leaves the lake through evaporation (E) and groundwater outflow (GW_{out}). Shell Lake is a terminal lake, so there are no surface-water outflows from the lake. A diversion of Shell Lake water to the Yellow River through a system of pipes was incorporated into the water budget for the lake during 2003–5, when the diversion was operating, but is not routinely used and is not a regular component of the water budget.

Precipitation and Evaporation

Precipitation data from 1949 to 2011 were obtained from the National Weather Service Weather Station in Spooner, Wis., which is 6 mi north of Shell Lake. Precipitation on the lake surface was 28.9, 24.4, and 23.3 in. during 2007, 2008, and 2009, respectively; that is, precipitation was 96, 81, and 78 percent of the long-term average (30.0 in. for 1949–2009) in 2007, 2008, and 2009, respectively. The average annual precipitation for 2007–8 (26.6 in.) was applied to lakes in the steady-state groundwater simulation, whereas total annual precipitation was applied to the lakes during each year in the transient groundwater simulations.

The long-term average annual evaporation for shallow lakes in the area near Shell Lake was estimated to be 31 in. by Farnsworth and others (1982). This rate is close to the long-term average annual precipitation measured at Spooner (30.0 in. for 1949–2009). Thus, it was assumed for the steady-state groundwater model that evaporation from the lake surface was equal to precipitation. For transient simulations, annual evaporation was estimated from pan evaporation measured at Rainbow Reservoir in Oneida County in north-central Wisconsin (National Climatic Data Center, 2007–9) during May–October for 1949–96, and from Marshfield, Wis., in Wood County, central Wisconsin during May–September for 1997–2009 (National Climatic Data Center, 2007–9). The annual pan evaporation rates were adjusted by calibrated multipliers (1.28 for Rainbow Reservoir and 0.99 for Marshfield) for the transient simulation to account for differences

Table 2. Horizontal hydraulic conductivities from previous studies in northwestern Wisconsin.

Hydrogeologic unit	Horizontal hydraulic conductivity, in feet per day						This report
	Lenz and others (2003)*	Feinstein and others (2006)	Juckem (2007)**	Juckem and Hunt (2007)	Juckem (2009)***	Robertson and others (2009)	
Sand and gravel aquifer	0.5–230	1–20	21	8–58	13–272	45	1–120
	70				73.3		
Sandstone aquifer	0.1–10	5			--	--	21
	1.4						

* Values are the range estimated from specific capacity tests performed during private well constructions, and the calibrated value from a groundwater-flow model.

** Value represents only the Apple River Basin in Barron, Polk, and St. Croix Counties.

*** Values are the range and geometric mean of specific capacity tests from 19 municipal wells in the sand and gravel aquifer in Polk, Barron, Burnett, and Washburn Counties, Wisconsin.

Table 3. Recharge rates from groundwater-flow models and other methods in Northwestern Wisconsin. The calibrated recharge rate used for the steady-state simulation of Shell Lake was 7.9 inches per year.

Study area	Method description	Recharge value (in/yr)	Reference
Whittlesey Creek, Bayfield County, WI	Estimated recharge by base flow separation divided by watershed area, distributed among hydrogeologically diverse material	8–18	Lenz and others (2003)
	Calibrated value	17	
St. Croix River Basin, MN and WI	Three recharge zones trending roughly north to south in a 1-layer analytic element model	5.8 – 8.9	Feinstein and others (2006)
Apple River watershed, Wisconsin	Recharge zones simulated using a refined version of the model by Feinstein and others (2006)	6.6 – 9.5	Juckem (2007)
Lac Courte Oreilles reservation, Northwest Wisconsin	One recharge zone for a 1-layer analytic element model	10.1	Juckem and Hunt (2007)
Whitefish Lake area, Douglas County, WI	One recharge zone for a 1-layer analytic element model	11	Robertson and others (2009)
St. Croix and lower Chippewa River Basins in WI	Base flow divided by drainage area for gaging stations in the basin		Gebert and others (2007)
	St. Croix Basin:	Range: 6.7 – 10.8 Average: 8.8	
	Lower Chippewa Basin:	Range: 2.3 – 9.1 Average: 6.3	

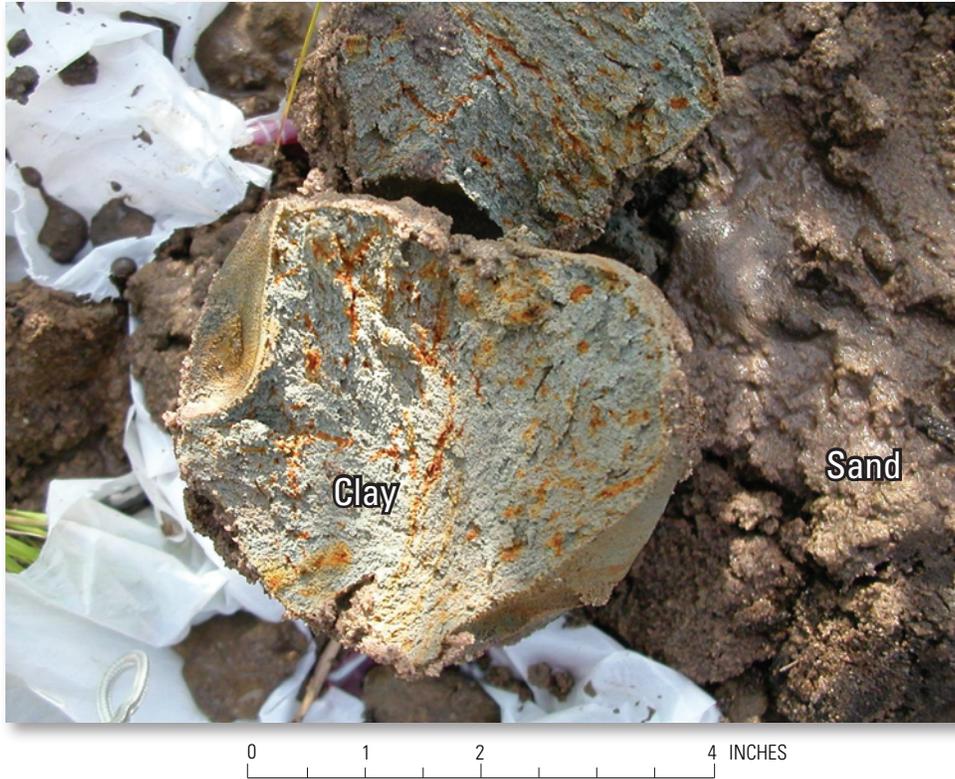


Figure 5. Shallow clay beneath sandy shoreline at PZ5 on the south side of Shell Lake, Wisconsin.

between pan and lake evaporation and climatic gradients from the Shell Lake area to the weather station locations. With these coefficients, the long-term average annual evaporation applied to lakes in the model was similar to the long-term average precipitation measured at Spooner. Annual data from Rainbow Reservoir were used when available because evaporation was expected to be more similar to that at Shell Lake because of their similarity in latitude and weather conditions, compared with Marshfield. Data from the Marshfield Experimental Farm were used because annual pan evaporation was not measured after 1996 at Rainbow Reservoir.

Surface-Water Inflow

Average annual streamflow was estimated with the unsaturated zone flow (UZF) package (Niswonger and others, 2006) within the MODFLOW model. In UZF, infiltrating water is rejected as runoff when the infiltration rate exceeds the saturated vertical hydraulic conductivity of the aquifer and (or) when the water table rises above the land surface. Rejected recharge is then routed to a nearby lake as surface-water runoff. In this way, wetlands were simulated as water bodies that captured precipitation and groundwater and released the water to adjacent lakes through ephemeral channels, with the amount of runoff determined by the long-term and annual precipitation cycles. Calcium (Ca) and magnesium (Mg) concentrations in water samples indicate a mixture of precipitation (low Ca and Mg concentrations) and groundwater (high Ca and Mg

concentrations) in the wetlands and support this conceptualization of wetland function as implemented in the model. The annual estimates of surface-water inflow were adequate for the steady-state and transient groundwater simulations; however, we the authors believed that the annual estimates did not adequately represent the interannual variability in runoff and, ultimately, phosphorus loading that is needed for simulating changes in the water quality of Shell Lake. By design, a groundwater model is not well suited for simulating short-term runoff events, which are important for understanding phosphorus loading into the lake. Therefore, these estimates of surface-water inflow were refined on the basis of data collected from 2009 to 2011 through the use of empirical runoff models developed specifically for Shell Lake and used to describe changes in the water quality of the lake, as described in the “Lake Water-Quality Analysis” section of this report.

Groundwater Inflow and Outflow

Groundwater inflow to and outflow from a lake is difficult to measure directly. Many methods have been developed for estimating groundwater inflow to lakes (Rosenberry and LaBaugh, 2008), including computation with Darcy’s Law using water-level gradients and inferred aquifer properties, which was the method used in previous studies of Shell Lake by Krohelski and others (1999) and the Wisconsin Department of Natural Resources (1982).

However, each method has limitations and biases that result in uncertainties in the computed values. Stauffer (1985) developed a technique to compute groundwater inflow for seepage lakes by using solute concentrations (primarily calcium, magnesium, and dissolved silica). The approach by Stauffer (1985) is based on a solute mass balance whereby measured concentrations in water inputs (precipitation and groundwater) can be used to compute the lake-water concentration. Because solute concentrations in the lake and in precipitation and groundwater are typically known with a higher degree of certainty than the amount of groundwater inflow, the solute mass-balance equation is used to estimate groundwater inflow. One important assumption of this method is that surface-water input to the lake is negligible, which is not an acceptable assumption for Shell Lake. Although the method cannot be quantitatively applied to Shell Lake because it is a terminal lake, qualitative application of the method demonstrates that groundwater inflow to Shell Lake is relatively small (likely less than 5 percent). The low concentration of Ca (about 3.5 milligrams per liter (mg/L)) and Mg (about 1.2 mg/L) in the lake water, combined with the relatively higher concentrations in groundwater (average of 30 mg/L for Ca and 12 mg/L for Mg in the three upgradient monitoring wells: 30S, 30D, 20S (fig. 3)) indicate that groundwater inflow is a relatively minor component of the total input of water to Shell Lake.

Groundwater inflow and outflow were simulated directly with the MODFLOW model. Calibration data (water levels in wells and lakes, and streamflows in nearby streams) were used to constrain model parameters that control groundwater inflow and outflow with the lake. The estimated groundwater inflow also was qualitatively compared with the estimates from solute chemistry described above.

Change in Storage

Changes in lake-water levels are caused by changes in storage, or changes in the volume of Shell Lake due to annual variability in components of the water budget (eq. 4). Stage-volume relationships for Shell Lake and the upgradient lakes were used in the MODFLOW model to convert changes in lake volume to changes in lake stage. During the intensive hydrologic monitoring period and steady-state calibration period of 2007 and 2008, the lake stage dropped by 1.3 ft (fig. 2), and the corresponding lake volume decreased by 1.45×10^8 cubic feet (ft³). Although a steady-state simulation implicitly assumes that no change in storage occurs, the observed change in storage is small (12 percent) in comparison with the change in storage over the longer period of record that was simulated with the transient model (1949 to 2009). For example, the change in storage from the historical minimum stage (1,214.96 ft on November 28, 1949) to the historical maximum stage (1,224.92 ft on May 19, 2003) was 1.2×10^9 ft³. During 2009, the lake dropped another 0.9 ft, which corresponds to a decrease in lake volume of 1.0×10^8 ft³.

Simulation of the Lake/Groundwater System Near Shell Lake

The lake/groundwater-flow system near Shell Lake was simulated with a three-dimensional groundwater-flow model developed by use of the USGS MODFLOW-2005 code (Harbaugh, 2005). Prior to development of the MODFLOW model, a two-dimensional analytic-element groundwater-flow model (GFLOW; Haitjema, 1995) was used as a simplified screening model to quickly test the conceptual model and derive hydraulic boundaries for the three-dimensional MODFLOW model (fig. 6). A complete description of analytic-element modeling is beyond the scope of this report, but Strack (1989) and Haitjema (1995) have written detailed descriptions of the analytic-element method. Hydraulic flux boundaries for the MODFLOW model were extracted from the GFLOW model by using methods described by Hunt, Anderson, and Kelson (1998) and Hunt, Kelson, and Anderson (1998).

To construct the GFLOW model, features that affect groundwater flow (such as surface-water bodies, aquifer characteristics, and recharge) were entered as mathematical elements or strings of elements. The amount of detail specified for each element depended on the distance from the area of interest—the area around Shell Lake. Each element was represented by an analytic solution. The effects of these individual solutions were added together to arrive at a solution for the groundwater-flow system. In GFLOW, the analytic elements are two dimensional and are used to simulate steady-state conditions.

The geometry of the single-layer model included a bottom elevation set at 800 ft and an average aquifer thickness of 400 ft, representing all glacial and sandstone aquifers. Recharge was applied uniformly across the simulated area. Surface-water features were simulated using several types of linesink elements. Line sinks are mathematical representations of streams and lakes. Most streams were expected to be in good hydraulic connection with the groundwater-flow system and were simulated as linesinks with relatively low resistance. Shell Lake was expected to have limited connection with the groundwater-flow system and was simulated with a lake element and high resistance. In analytic-element modeling, resistance is computed by dividing the streambed or lakebed sediment thickness by the vertical hydraulic conductivity of the sediment. Well-connected streams were simulated using linesinks with resistance of 0.05 day and widths ranging from 5 to 20 ft. Shell Lake was simulated with resistance of 586 days. This high resistance for Shell Lake reflected the low hydraulic conductivity of clay-rich sediments that separate Shell Lake from the deep regional aquifer. Resistance of 586 days corresponded to a 30-ft thick deposit with a vertical hydraulic conductivity of about 0.05 ft/d.

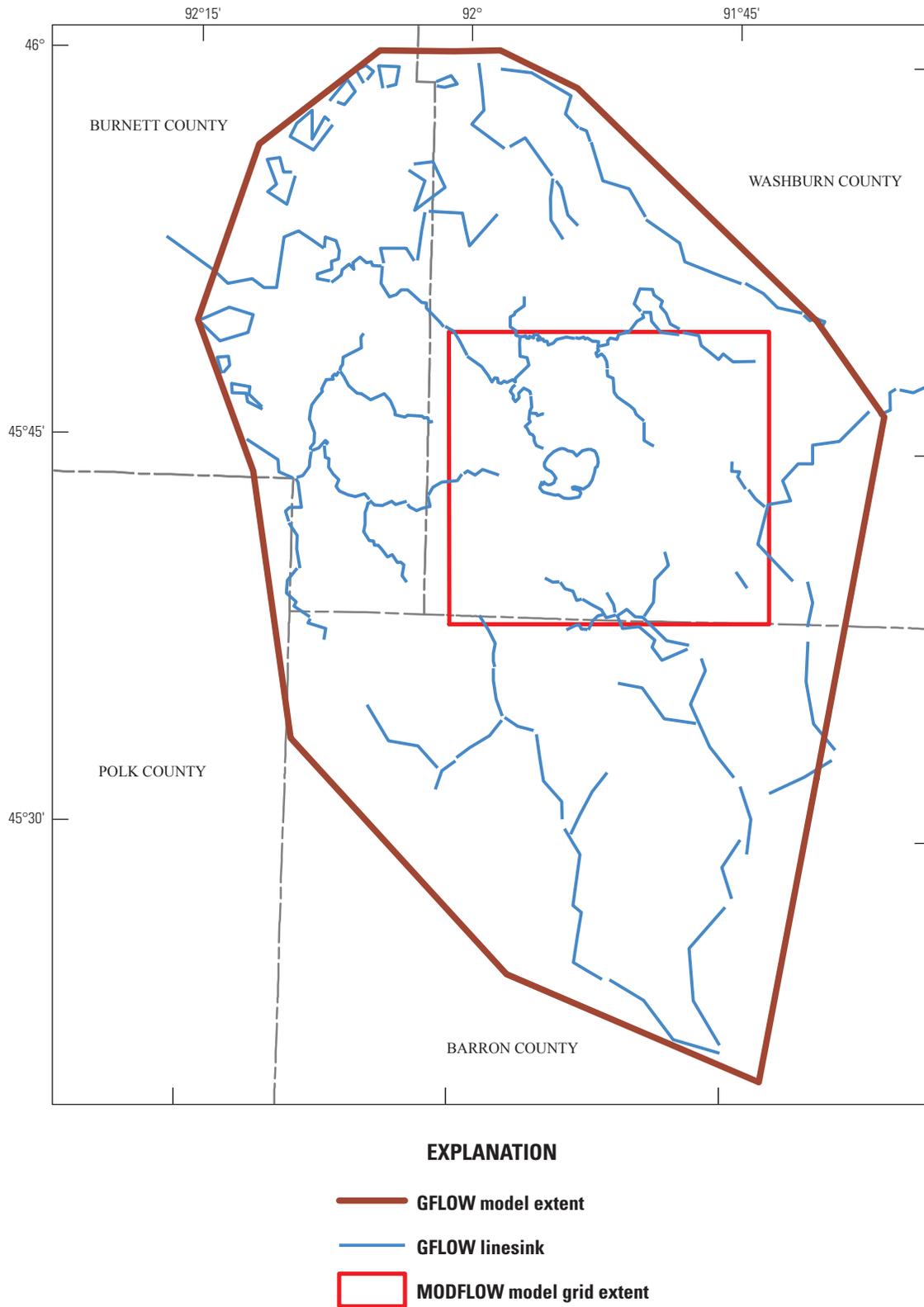


Figure 6. Extent of the analytic element model (GFLOW) and the inset MODFLOW model grid.

The calibrated GFLOW model had a recharge rate of 7.7 in/yr and hydraulic conductivity of 22 ft/d. These recharge and hydraulic conductivity values are similar to values used in previous GFLOW models developed in nearby areas (Juckem and Hunt, 2007; Juckem and Hunt, 2008; Juckem, 2009; Robertson, Rose, and Juckem, 2009; Hunt and others, 2010). While the GFLOW model was used only as a screening model to test the conceptual model and provide hydraulic boundaries, the model simulated heads and flows reasonably close to measured values. The mean difference and mean absolute difference (MAD) between simulated and measured groundwater levels were -12.2 and 23.8 ft, respectively. This represents less than 5 percent and 10 percent, respectively, of the full range in measured water levels over the simulated area. The mean and MAD for streamflows were 1.6 and 4.2 ft³/s, respectively. The simulated water level for Shell Lake was within 0.2 ft of the measured water level in 2007.

Two MODFLOW models were developed to simulate steady-state conditions (no change in water levels) and transient conditions (water-level fluctuation over time). The steady-state model was designed to focus on improving understanding of spatial patterns of groundwater flow based largely on hydrologic data collected during 2007 and 2008. The transient model was designed to improve understanding of changing hydrologic conditions over time by simulating average annual conditions for each year from 1949 to 2009. The steady-state model was constructed and calibrated first to inform the spatial properties of the hydrologic system around Shell Lake. The transient model was developed from the steady-state model to incorporate these spatial properties and then was calibrated to adjust hydrologic inputs to match average annual lake levels from 1949 to 2009 and to evaluate effects from a lake water diversion.

The steps involved in simulating three-dimensional groundwater flow near Shell Lake with MODFLOW were the following:

1. Construct the MODFLOW model grid, input perimeter boundary conditions, select appropriate aquifers and confining units as identified in the conceptual model, and represent surface-water features with advanced simulation packages (LAK, SFR, UZF).
2. Calibrate hydrogeologic properties of the flow system via the steady-state model through the use of the parameter estimation code PEST. This was done by adjusting selected parameters over realistic ranges until there was a reasonable match between measured and simulated groundwater levels, lake levels, and surface-water flows.
3. Calibrate annual recharge, evaporation, and aquifer storage via the transient model through the use of PEST. This was accomplished by comparing measured and simulated historical lake levels.
4. Evaluate the effect of the lake-water diversion from 2003 to 2005 by comparing simulated results of the calibrated transient model (which included the diversion) with results from a scenario that did not include the diversion.

Model Construction and Assumptions

MODFLOW-2005 uses a block-centered finite-difference method to solve the groundwater-flow equations (Harbaugh, 2005). Detailed discussions of finite-difference methods, MODFLOW input requirements, and theory are provided by McDonald and Harbaugh (1988), Anderson and Woessner (1992), Harbaugh and others (2000), and Harbaugh (2005). The MODFLOW model incorporates perimeter boundary conditions extracted from the analytic-element screening model, which were assigned as specified flux cells (by means of the MODFLOW well package). The flux values were determined from a single-layer MODFLOW extraction of the corresponding area from the GFLOW model and were then distributed among the MODFLOW layers on the basis of transmissivity. Details of the technique are described in more detail by Hunt, Kelson, and Anderson (1998). The boundaries of the MODFLOW model extended more than 4 miles from the shore of Shell Lake, or a distance equal to three times the estimated characteristic leakage length (Haitjema, 2006, eq. 5) based on initial estimates of aquifer transmissivity in the basal sand, gravel and sandstone layer and vertical resistance to flow across the deep clay layer. The characteristic leakage length (Haitjema, 2006, eq. 5) is equal to the square root of the product of aquifer transmissivity and the resistance of the overlying confining unit, with the confining-unit resistance computed as the thickness of the unit divided by the vertical hydraulic conductivity of the unit (the inverse of leakage). Haitjema (2006) demonstrates that 95 percent of vertical flow across a confining unit will occur within approximately three characteristic leakage lengths. Thus, any artifacts associated with distributing simulated fluxes from a regional 1-layer model across four layers along the perimeter the MODFLOW model are expected to be mitigated over the 4 miles separating the model boundaries and Shell Lake. Boundary conditions also were not expected to change substantially on an annual basis because groundwater flow directions were not expected to change substantially from one year to the next. That is, groundwater mounding, which can affect flow directions near the boundaries, is controlled by recharge, transmissivity, and the square of the distance between surface-water bodies (Haitjema, 2006). Of these three properties, only recharge (and a minor change in transmissivity caused by water table fluctuations associated with recharge) changes on an annual basis, and the effect of this change on flow directions is expected to be small considering that the distance between surface-water bodies (a squared quantity and, hence, highly influential) remains the same. Therefore, the boundary conditions extracted from the GFLOW model were used for both steady-state and transient MODFLOW model simulations.

MODFLOW requires input arrays (gridded data) that describe hydraulic parameters such as hydraulic conductivity and recharge, as well as top and bottom elevations for aquifers and confining layers. The area of interest around Shell Lake was simulated with uniform grid cells that measured 150 ft on each side, for which conditions within each cell (groundwater level and groundwater flow) are reduced to one average value for the entire cell. The six relatively uniform geologic units described in the conceptual model (fig. 4) were lumped into four layers in the MODFLOW model (table 4). The bottom of each layer was simulated as a horizontal plane with an elevation estimated from the geologic cores collected at the nested wells (fig. 3). The top of the MODFLOW grid was specified as the land-surface elevation on the basis of a digital elevation model (U.S. Geological Survey, 1998). MODFLOW layers 1, 2, and 3 are at least 30 ft thick, except in the north where the land surface drops below the bottom elevation of the layer (the aquifer or confining unit is conceptualized as being absent). There, the layers are simulated as 1 ft thick and assigned the hydraulic properties of layer 4; that is, where the land surface is below the original flat bottom elevation of a layer, that aquifer was considered to be absent and the model was adjusted to simulate the aquifer represented in layer 4 instead. Layer 1 in the model is represented by surficial sand and gravel deposits combined with a relatively thin layer of clay and has up to about 60 ft of saturated thickness. Layer 2 was simulated as a mixed sand and silt aquifer with a thickness of about 30 ft (except in the north, as described above). Layer 3 was simulated as a confining unit representing a relatively thick layer of clay (40 ft thick in the model, except in the north). Layer 4 of the model represented sand and gravel deposits below the clay as well as the sandstone aquifer that underlies the glacial sediments in this area and had a combined thickness of as much as about 320 ft.

Layer 1 was initially simulated as a confined aquifer with the saturated thickness estimated from observed water levels during early trial-and-error parameter adjustments. Prior to formal parameter estimation, layer 1 was specified as being unconfined, and cells simulated as being dewatered (dry) were converted to inactive cells. Layers 2, 3, and 4 were simulated as confined. Initial trial-and-error hydraulic-conductivity adjustments involved the use of uniform hydraulic conductivities for each layer. Final hydraulic conductivity values, which were estimated through pilot points and associated regularization by using the parameter estimation code PEST (see “Steady-State Calibration”), incorporated heterogeneity within layers 1, 2, and 3.

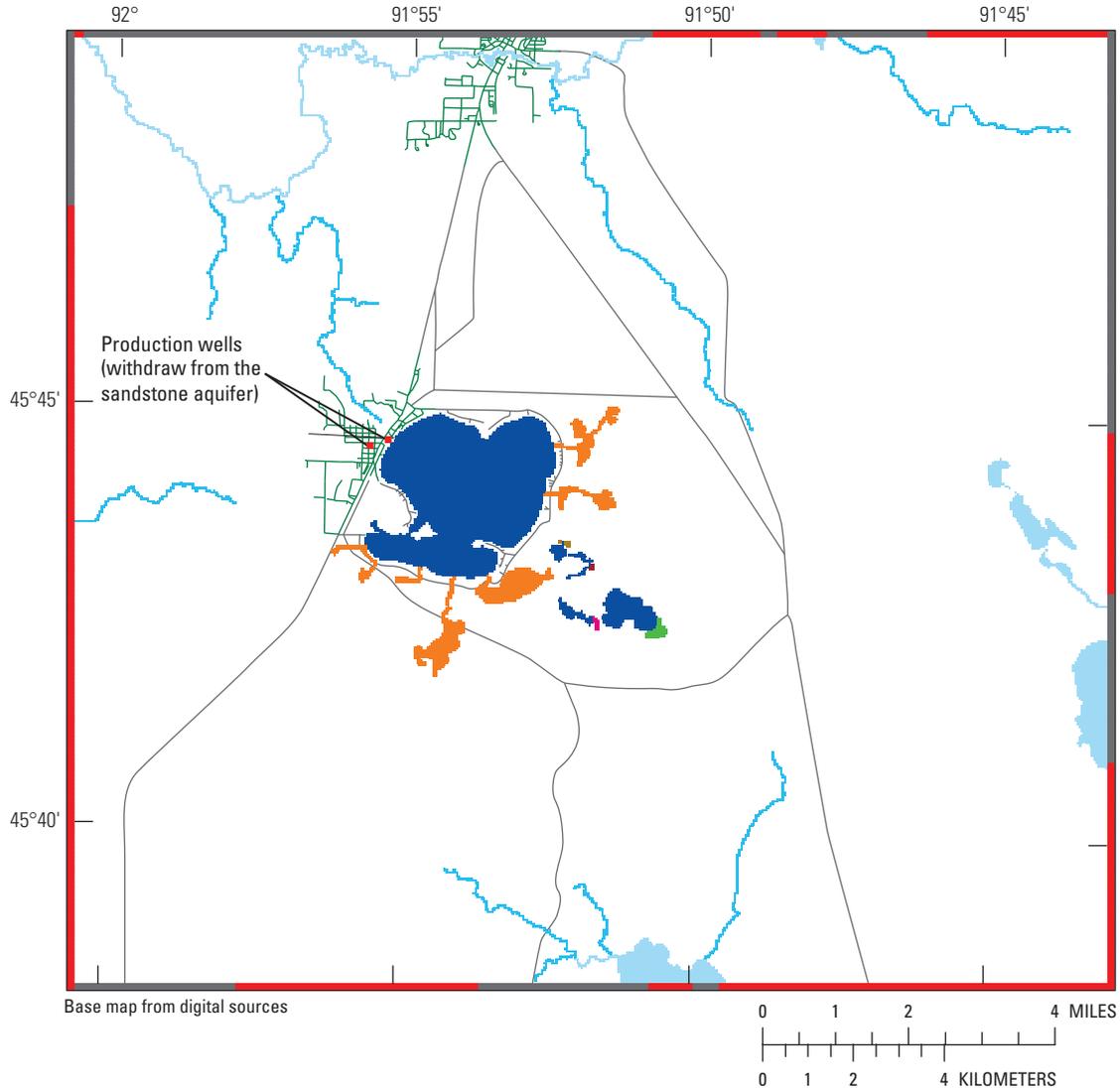
Groundwater recharge was simulated as a uniform rate across the model domain and applied by using the recharge (RCH) package (Harbaugh, 2005) in upland areas and the unsaturated-zone flow (UZF) package (Niswonger and others, 2006) in low-lying areas around lakes where wetlands are common. The RCH package applies recharge directly to the water table. The UZF package simulates infiltration through the unsaturated zone to the water table. The main advantage of using the more complex UZF package is that the amount of groundwater recharge can vary during the simulation in response to the rate of infiltration and the depth to the water table; that is, infiltrating water was rejected (as runoff) during the simulation when the infiltration rate exceeded the saturated vertical hydraulic conductivity of the aquifer and (or) when the water table rose above the land surface. Rejected recharge was routed to a nearby lake as surface-water runoff. Input values for recharge and infiltration in the RCH and UZF packages were set equal to each other and were estimated during the steady-state and transient calibrations (refer to “Steady-State Calibration” and “Transient Calibration”).

Internal hydrologic features in the MODFLOW model include lakes, streams, wetlands, and wells (fig. 7) that were simulated with the lake package (LAK; Merritt and Konikow, 2000), streamflow-routing package (SFR; Niswonger and Prudic, 2005), UZF package (Niswonger and others, 2006), and WEL package (Harbaugh, 2005), respectively. Wetlands that drain to adjacent lakes were simulated with the UZF package, as described above. Lakes and streams were simulated as head-dependent flow boundaries, for which groundwater flow to or from these surface-water bodies depends on the difference in surface-water and groundwater levels, the vertical conductivity and thickness of the streambed or lakebed (leakance), and the length and width of the stream or lake in the model cells that encompass the surface-water body. Streams near Shell Lake were simulated with the SFR package, which routes water from upstream reaches to downstream reaches to accumulate flow and directly solve for water level in the stream. Streams distant from Shell Lake were simulated with the more basic river (RIV) package (Harbaugh, 2005), which does not route flow from an upstream reach to a downstream reach. The specified streambed leakance [2 (ft/d)/ft] indicates a close hydraulic connection with the underlying aquifer; that is, the hydraulic conductivity of the streambeds is such that stream stages have a substantial effect on the local groundwater system.

Table 4. Attributes of the four layers simulated in the groundwater-flow model.

[ft, feet]

Layer	Maximum saturated thickness (ft)	Geologic material
1	60	Surficial sand and gravel underlain by clay
2	30	Mixed sand and silt
3	40	Predominately clay
4	320	Sand and gravel, and sandstone



EXPLANATION

- Lakes simulated with the LAK package
 - Streams simulated with the SFR package
 - Distant rivers and lakes simulated with the RIV package
- | | |
|--|--|
| <p>UZF runoff routed to:</p> <ul style="list-style-type: none"> Shell Lake Ripley Lake Little Ripley Lake Chain Lake Round Lake | <p>Regional groundwater flow simulated:</p> <ul style="list-style-type: none"> Out of the MODFLOW model Into the MODFLOW model Roads Village or city streets |
|--|--|

Figure 7. Hydrologic features simulated with the MODFLOW model.

Two production wells for the City of Shell Lake were each simulated as withdrawing 83,230 gallons per day (combined total of 60.9 million gallons per year; Public Service Commission of Wisconsin, 2006) from the sandstone aquifer (layer 4) using the WEL package (Harbaugh, 2005). This withdrawal represented total annual pumping for 2006 as simulated in the steady-state model. Data on long-term historical trends in groundwater pumping in the study area were not available, thus, the 2006 production well withdrawal was assumed to adequately represent historical pumping for each stress period of the transient model. Regardless, well withdrawals were expected to have little effect on the overall water budget for Shell Lake because clay layers separate the production wells from the lake. Private well pumping was not simulated because the amount of pumped water is typically small and expected to have minimal effect on the Shell Lake water budget.

Shell Lake and four nearby upgradient lakes were simulated with the LAK package (Merritt and Konikow, 2000), which simulates the water level in each lake on the basis of each lake's bathymetry and the amount of its inflows (precipitation, groundwater inputs, and runoff) and outflows (groundwater output and evaporation). Precipitation and evaporation were specified from measurements at weather stations, as described in the section entitled "Precipitation and Evaporation," for the steady-state and transient simulations. Groundwater inflow and outflow, along with surface-water runoff, were simulated in the model as a function of computed groundwater and surface-water levels and infiltration rates.

Steady-State Calibration

The steady-state model was designed to simulate average conditions that were approximated by hydrologic data collected during monitoring years 2007 and 2008. Average conditions over 2007–8 were chosen because it was the period with the greatest amount of hydrologic data (multiple well and lake levels) and because the average stage during 2007–8 (1,218.88 ft) was close to the long-term average from 1949 to 2009 (1,218.91 ft) and changed by less than 12 percent of the long-term average range in lake levels. Although average precipitation during 2007–8 (26.6 inches per year (in/yr)) was slightly lower than the long-term average (30.0 in/yr), evaporation from lakes was specified in the model such that the net precipitation in excess of evaporation remained near the long-term estimated average of 0 in/yr. Groundwater recharge, the primary source of water to the groundwater-flow system, was treated as a calibration parameter for the steady-state model, the final value being determined by matching simulated water levels and nearby streamflows to observed values.

Calibration of the steady-state model was aided by use of parameter-estimation techniques in PEST (Doherty, 2008) and was designed to inform the spatial properties of the hydrologic system around Shell Lake and the long-term water levels measured in Shell Lake. The primary benefit of parameter

estimation is the ability to automatically calculate parameter values (for example, hydraulic conductivity and recharge) that are a quantified best fit between simulated model output and observed data (for example, groundwater levels and streamflows). Other benefits include quantification of the quality of the calibration, parameter correlation (for example, hydraulic conductivity and recharge), and parameter sensitivity. Parameters estimated during the steady-state calibration were horizontal and vertical hydraulic conductivities for model layers, lakebed leakage, and recharge. Targets used to constrain the model parameters were measured water levels in wells, vertical differences in measured water levels, stream base flows, and lake levels.

Although a steady-state model simulates groundwater levels that do not change with time, measured water levels and base flows used for calibration spanned many years. Because of these inconsistencies, along with simplifications inherent in constructing a model, perfect agreement between the simulated and measured values was not expected. An approximate evaluation of data quality is included in the calibration via the weight assigned to each target in PEST (table 5). The weighted residuals between measured and simulated values were used by PEST to determine the model best fit. Water-level targets were arranged into four categories: (1) average water levels for lakes measured during 2007–8, (2) average water levels from the four deep USGS-installed monitoring wells monitored during 2007–8, (3) difference in vertical water level between Shell Lake and the deep monitoring wells at the four nested well locations, and (4) water levels from well-construction reports (WCRs; approx. 1988 to 2009) submitted to the Wisconsin Department of Natural Resources (2010). Water levels in the shallow USGS-installed monitoring wells were not used as calibration targets because of limitations with simulating small-scale three-dimensional flow near Shell Lake with 150-ft cells (Hunt and others, 2003); that is, based on the properties of the model, cells smaller than about 40 ft by 40 ft would have been needed to better simulate heads in the shallow wells adjacent to Shell Lake and thus minimize confounding factors for using the wells as targets for calibration. Higher weights were assigned to the deep USGS-installed wells and lake levels compared with well-construction report targets because of the more accurately measured reference-point elevations and longer monitoring periods. Finally, average annual base-flow estimates (table 6) also were used as calibration targets. Average annual base-flow values were estimated from four miscellaneous streamflow measurements by means of a statewide multiple-regression analysis (Gebert and others, 2007) that incorporated drainage area and a base-flow factor referenced to the 90-percent flow duration (the flow that is equaled or exceeded 90 percent of the time) for the St. Croix River at Danbury. Relatively lower weights were assigned to headwater flow-measurement sites where the measured flow was less than 0.5 cubic foot per second (ft³/s) and could be influenced by local hydrogeologic conditions not incorporated into the model. The base-flow targets were compared to simulated stream gains as part of the model calibration.

Table 5. Calibration results for water-level targets and associated weights used for calibration with the parameter estimation program PEST.

[ME, Mean error; MAD, Mean absolute difference; RMSD, Root mean square difference; n/a, not applicable]

Target type	Number of targets	ME (feet)	MAD (feet)	RMSD (feet)	Weight
Average water levels for lakes measured during 2007 and 2008	5	-0.2	0.7	1.0	1 – 4 Assigned a weight of 4 to Shell Lake based on the length of record and purpose of the model; assigned a weight of 1 to the four adjacent lakes.
Average values from deep USGS-installed groundwater-observation wells during 2007 and 2008	4	-0.3	1.5	2.0	1
Average differences in vertical water levels between Shell Lake and the deep monitoring wells at the four nested well locations	4	0.4	1.4	2.0	1
Well-construction report data	527	-7.8	20.3	27.3	0.005

Table 6. Calibration results for stream base-flow targets and associated weights used for calibration with the parameter estimation program PEST.

[ft³/s, cubic feet per second]

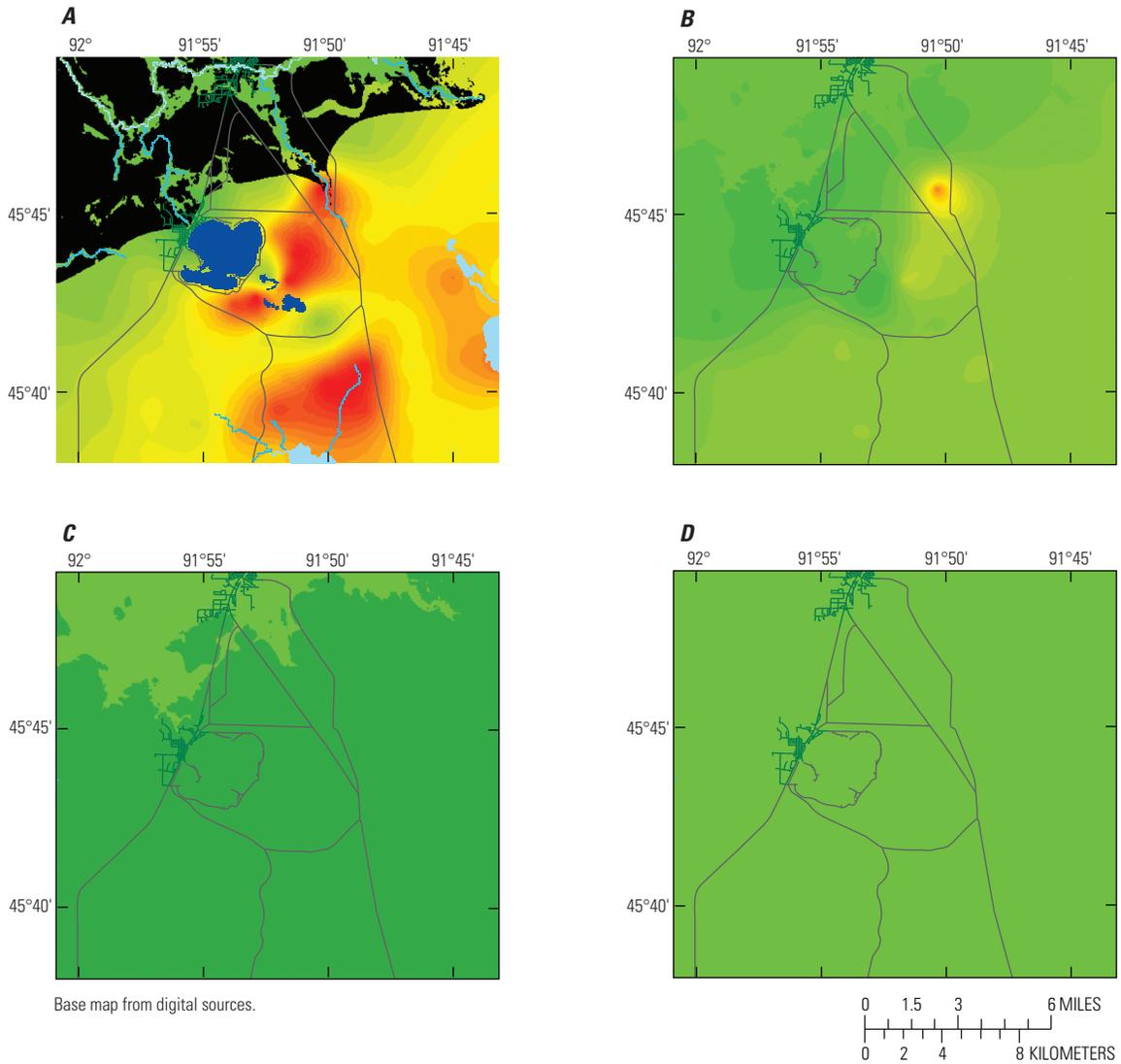
USGS station identifier	Station name	Measured base flow (ft ³ /s)	Target base flow (ft ³ /s)	Simulated base flow (ft ³ /s)	Difference (ft ³ /s and percent)	Weight
05334010	Sawyer Creek near wastewater plant at Shell Lake, Wis.	0.19	0.15	0.16	-0.01 (-5%)	3.0e-4
05334120	Sawyer Creek at Trout Rd near Shell Lake, Wis.	8.44	7.27	6.02	1.2 (17%)	1.6e-5
05333987	Beaver Brook headwater site near Spooner, Wis.	.41	1.13	1.63	-0.5 (-45%)	4.1e-5
05333990	Beaver Brook downstream site near Spooner, Wis.	1.59	5.36	4.23	1.13 (21%)	2.2e-5

Because of the difficulty with using discrete layers to simulate heterogeneous glacial aquifers, regularized inversion techniques were used to calibrate the hydraulic-conductivity fields of several layers; that is, the horizontal conductivity (K_h) field was allowed to vary on a cell-by-cell basis in the aquifers of layers 1 and 2; the vertical conductivity (K_v) field was allowed to vary on a cell-by-cell basis in layers 1, 2, and 3. PEST was used with pilot points to estimate a reasonable field of K_h and K_v values that balanced the desire to match measured water levels and flows with the limitation of minimizing the amount of heterogeneity, or variability, introduced to the hydraulic-conductivity fields. A single value was estimated for horizontal and vertical hydraulic conductivity in layer 4 and horizontal hydraulic conductivity of the clay-rich confining unit represented by layer 3. A single value was appropriate because data to inform heterogeneous conductivity fields for these properties was scarce and because the groundwater-flow system exhibited relatively low sensitivity to the bulk conductivity values of these layers (see “Sensitivity Analysis”). Some parameters were excluded from automated calibration because the groundwater system is relatively insensitive to the parameter values over a reasonable range. In these cases, the parameter values were either set equal to a value within the range reported in table 2 or tied to another parameter at a specified ratio. For example, a 10:1 ratio was specified between the horizontal and vertical hydraulic conductivities of the deep sandstone aquifer (layer 4), owing to the relative insensitivity of the flow system to vertical hydraulic conductivity in this deep aquifer. In addition, a single lakebed leakance value (0.0053 day^{-1} , or 5-ft-thick sediments with a vertical conductivity of 0.027 ft/d) was estimated for all simulated lakes. The steady-state recharge rate (7.9 in/yr) was estimated directly and compared with a range of reasonable rates from previous studies (table 3). Values for hydraulic conductivity used in the final calibrated model are listed in table 2 and shown in figures 8 and 9.

Application of pilot points with regularized inversion in PEST affords great flexibility for model parameters in order for the model to match observation targets. The approach allows for estimation of parameter values at specific locations in the model (at pilot points) and uses a kriging algorithm to interpolate parameter values for each cell between the pilot points; that is, each cell in a layer with pilot points is assigned an interpolated parameter value based on the proximity to nearby pilot points and the parameter values computed for those pilot points. (Readers interested in additional information on the use of pilot points for model calibration are referred to Doherty and others (2010).) However, to balance the computational power of PEST to obtain parameters that best match target values with the need for hydrogeologically reasonable parameter fields, a “lower limit” to the calibration (the match between simulated and observed target values, referred to as PHI) is specified in PEST (PHIMLIM in the PEST manual, Doherty, 2008). This “lower limit”, or PHIMLIM, represents the target measurement objective function

and controls the strength of the parameter regularization. Low values of PHIMLIM weaken the regularization constraint and result in overfitting of observations and parameter “bulls eyes” near observations; high values of PHIMLIM result in stronger smoothing, but increase misfit and can result in underfitting of observations (Fienen and others, 2009). Thus, an ideal value for PHIMLIM will balance the goal of matching observed target values against the potential for producing unrealistic parameter fields while in pursuit of an unrealistically close match to observed values. An extremely close match to observed values should never be expected of a model because all models incorporate necessary simplifications that preclude the model from obtaining perfect agreement with the physical system they represent (Hunt and others, 2007), and target values themselves incorporate measurement error and interpolation errors in space and time.

Fienen and others (2009) theorized that a reasonable value for PHIMLIM may be the number of targets used for calibration, if the target weights are based solely on measurement error. Weights assigned to targets in the steady-state model were not solely related to measurement error but followed a general pattern of expected accuracies. For example, WCR targets were assigned weights such that individual WCR targets had relatively less influence on the calibration than the Shell Lake stage, which was a primary focus of the model simulation. Doherty and others (2010) suggest an alternative method for estimating PHIMLIM, whereby at the beginning of a parameter-estimation process PHIMLIM is set very low and over-fitting is allowed to occur. The user can then review interim results from this preliminary effort and select a more informed value for PHIMLIM that “provides a good fit between model outputs and field measurements, but shows no signs of over-fitting ... judged to be geologically unreasonable.” Following the general methods of Fienen and others (2009) and Doherty and others (2010), the value of PHIMLIM used for pilot point regularization with PEST for the steady-state model was set as 3 times the sum of all lake level targets (5), deep USGS well targets (4), head-difference targets (4), and base-flow targets (4), and then increased by two (2) to account for the low-weight WCR targets. Therefore, PHIMLIM, the “lower limit” for the sum of squared weighted residuals was set to 57 ($3 * 19$ targets) in the PEST control file. An evaluation of the hydraulic-conductivity fields in figures 8 and 9 indicates that this value of PHIMLIM was appropriate because the conductivity fields appear reasonable and the heterogeneity added by PEST was limited to areas where the target data supported the addition of heterogeneity; that is, the hydraulic-conductivity fields display smooth transitions with “bulls eyes” limited to areas supported by field data, such as the relatively high vertical hydraulic conductivity along the southeastern shoreline of Shell Lake in the confining unit represented by layer 3 (fig. 9C), where measured hydraulic gradients indicated the presence of local downward vertical flow across the confining unit.



EXPLANATION

- Inactive and dry cells
- Lakes simulated with the LAK package
- Streams simulated with the SFR package
- Distant rivers and lakes simulated with the RIV package
- Roads
- Village or city streets

Horizontal hydraulic conductivity
(logarithmic scale in feet per day)

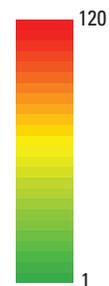
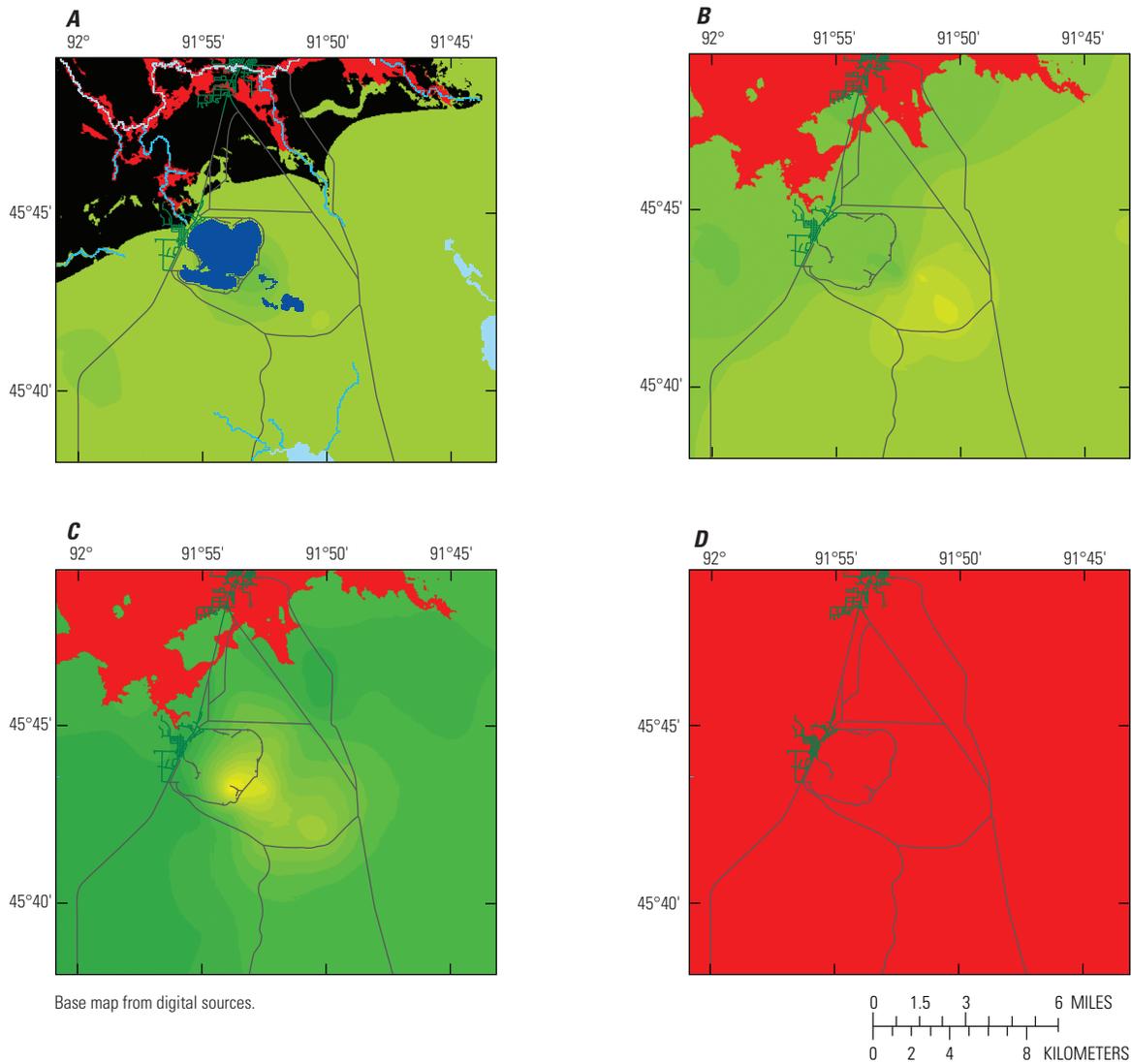


Figure 8. Horizontal hydraulic conductivity of simulated layers in the MODFLOW model. (Diagrams A–D represent layers 1–4 in the same order.)



EXPLANATION

- Inactive and dry cells
- Lakes simulated with the LAK package
- Streams simulated with the SFR package
- Distant rivers and lakes simulated with the RIV package
- Roads
- Village or city streets

Vertical hydraulic conductivity
(logarithmic scale in feet per day)

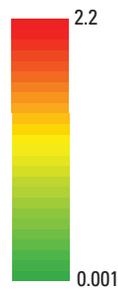


Figure 9. Vertical hydraulic conductivity of simulated layers in the MODFLOW model. (Diagrams A–D represent layers 1–4 in the same order.)

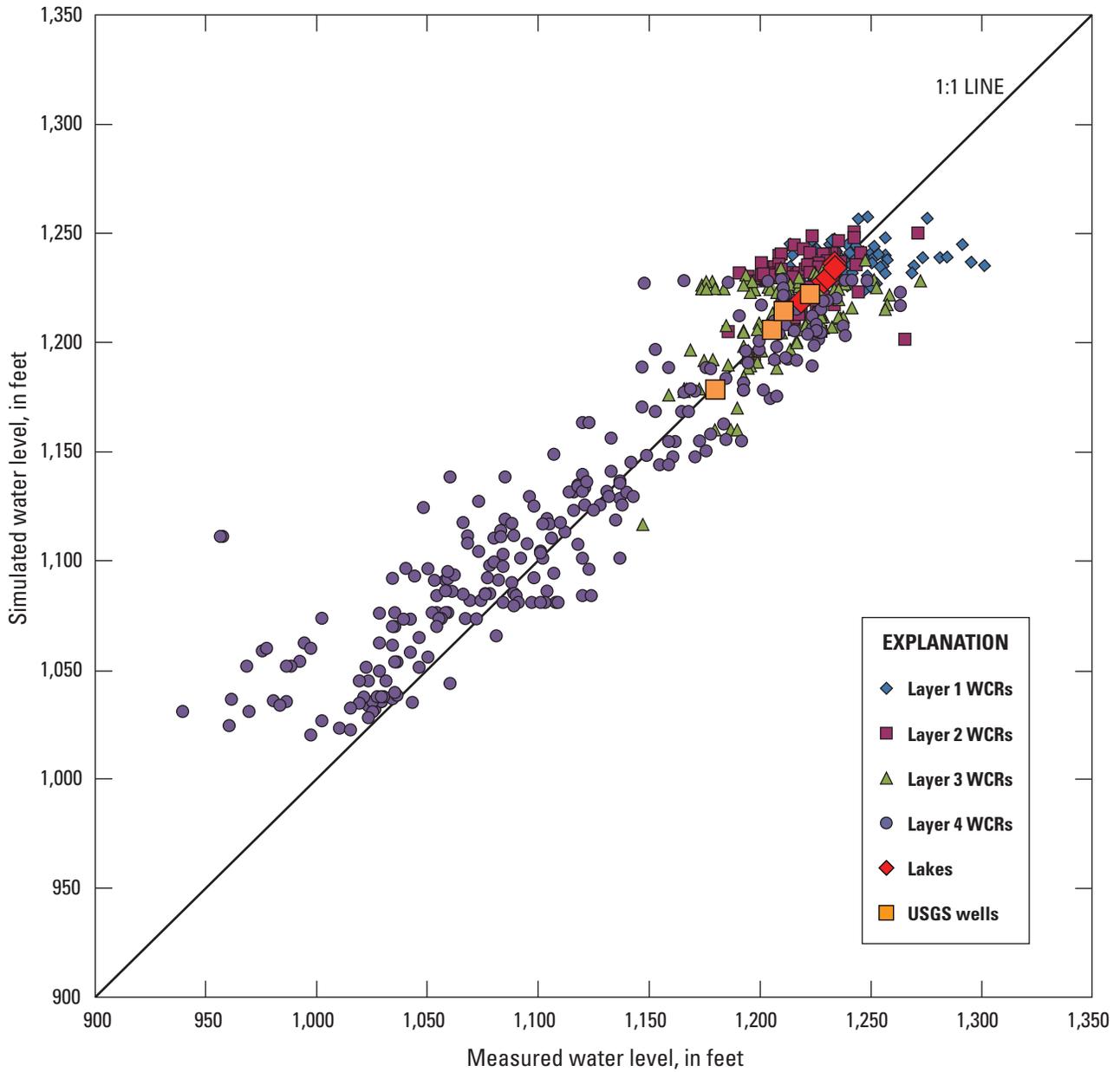
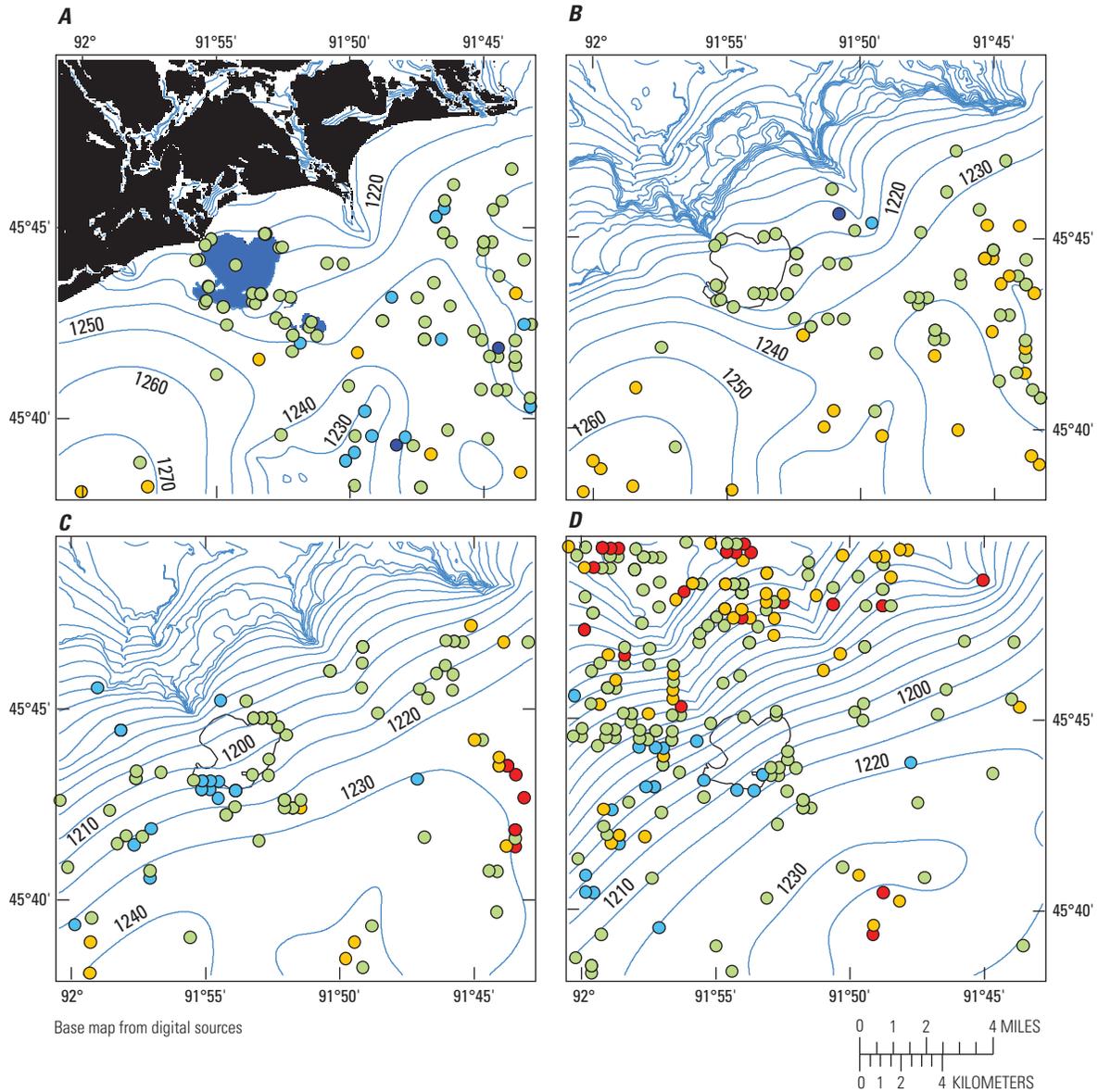


Figure 10. Measured and simulated water-level targets. (WCRs, Well Construction Reports.)

Measured groundwater and lake levels were compared to model-calculated groundwater and lake levels to evaluate the model fit (fig. 10). Summary statistics for the water-level calibration (table 5) are similar to what has been observed in other models in Wisconsin. The mean error (a measure of the model bias) of the non-WCR groundwater and lake level targets in the model is -0.05 ft (a negative value indicates that measured values were less than simulated values). The mean absolute difference (MAD) and root mean square difference (RMSD) between non-WCR measured and simulated water levels are 1.2 and 1.7 ft, respectively. Figure 10 and table 5 illustrate that the model tends to simulate water levels in layer 4

that are higher than measured water levels for WCRs in some areas. This bias was deemed insignificant because the purpose of the model is to simulate groundwater/surface-water interaction for Shell Lake and the adjacent lakes. Attempts to minimize the bias in simulated WCR water levels in layer 4 degraded simulated water levels for the higher priority lake and USGS monitoring-well targets and was therefore not pursued. In addition to comparing measured and modeled water levels with summary statistics, spatial comparisons between the measured and simulated water levels in the four model layers are shown in figure 11. The agreement between simulated water levels and measured water levels in observation wells is



EXPLANATION

- | | | | |
|---|--|---|---|
|  | Inactive and dry cells |  | Water level simulated
Greater than 50 feet too high |
|  | Lakes |  | 21–50 feet too high |
|  | Simulated groundwater elevation,
in 10-foot intervals above NAVD 88 |  | Within 20 feet of measured value |
|  | Shell Lake outline |  | 21–50 feet too low |
| | |  | Greater than 50 feet too low |

Figure 11. Simulated groundwater elevation contours and water-level residuals in the MODFLOW model. (Diagrams A–D represent layers 1–4 in the same order.)

generally close and shows little spatial bias, except for WCR targets in the northern part of layer 4. Shell Lake is upgradient of this area and buffered from biases in this area by the Yellow River, Sawyer Creek, and other streams in the area. Finally, simulated base flows compared well with target base flows at the four calibration locations in the model (table 6).

Transient Calibration

The transient model was designed to simulate annual average conditions for 61 years, from 1949 to 2009 (fig. 12). Initial conditions for the transient simulation were derived from a dynamic-average steady-state simulation designed to produce hydrologic conditions similar to those at the start of the transient simulation in 1949. This dynamic-average steady-state solution incorporated the long-term average recharge rate (7.9 in/yr) from the calibrated steady-state model, precipitation representative of 1949 (27.7 in/yr), and evaporation set slightly higher than 1949 precipitation (29.0 in/yr) to represent a system that was emerging from an extended dry period. The resulting initial conditions produced water levels similar to those observed in 1949.

Starting with the dynamic-average steady-state conditions, the transient model simulated average water levels from 1949 to 2009 with 61 one-year stress periods. The transient model calibration focused on matching simulated and measured water levels for Shell Lake (fig. 12) and the four upgradient lakes through 2009 using PEST (Doherty, 2008). A uniform weight of 1.0 was assigned to all annual lake water level targets in the transient model.

As described in the section entitled “Precipitation and Evaporation,” annual precipitation data from the National Weather Service Weather Station in Spooner, Wis., were assigned to the 61 stress periods from 1949 to 2011 for each lake simulated with the LAK package. Evaporation from each lake and groundwater recharge to the aquifers were adjusted annually for each of the 61 simulated years using adjustment factors estimated with the use of PEST (Doherty, 2008). Annual evaporation for each LAK-package lake was adjusted from pan evaporation data at Rainbow Reservoir (Oneida County; 1949–96) and Marshfield (Wood County; 1997–2009) using calibrated evaporation adjustment factors of 1.28 for Rainbow Reservoir and 0.99 for Marshfield to account for differences between pan and lake evaporation and climatic gradients from the Shell Lake area to the weather station locations. With these adjustment factors, the long-term average annual evaporation applied to lakes in the model was similar to the long-term average precipitation measured at Spooner. Groundwater recharge for the RCH and UZF packages was estimated as the annual base flow in the St. Croix River at Danbury divided by the watershed area at Danbury, then multiplied by a recharge adjustment factor to account for local hydrogeologic conditions. The calibrated recharge adjustment factor was 0.81 which, after it was applied to the input estimates, produced an average recharge (7.7 in/yr) over the period 1949–2009 that was similar to the steady-state calibrated recharge (7.9 in/yr).

In addition to the time-dependent hydrologic variables, values for storage parameters (specific yield for the water table aquifer and storage coefficient for the confined aquifers) were needed for the transient model. A specific yield of 0.3 (unitless) was estimated during the transient calibration for layer 1 and the “eroded” parts of layers 2, 3, and 4, where they are conceptualized as water-table aquifers. Specific yield is the volume of water per volume of aquifer that is drained by gravity from an unconfined aquifer in response to a decline in water level (Anderson and Woessner, 1992; Fetter, 1994). For the “uneroded” parts of layers 2, 3, and 4, storage coefficient values (unitless) of 0.0006, 0.0008, and 0.0064, respectively, were specified. Storage coefficient is the volume of water released per unit area of aquifer per unit decline in head for a confined aquifer (Anderson and Woessner, 1992). Storage coefficients were specified as the product of the layer thickness and a specific storage value of $2.0 \times 10^{-5} \text{ ft}^{-1}$ estimated for dense sand and gravel material (Domenico, 1972, as adapted by Anderson and Woessner, 1992). The transient calibration was relatively insensitive to storage parameters (see “Sensitivity Analysis”).

The calibrated transient model reproduced the general pattern of lake stage for Shell Lake (fig. 12), with a mean difference and mean absolute difference of -0.1 and 0.6 ft, respectively, between simulated and measured Shell Lake stages over the 61-year period. This represents less than 1.5 percent and 7 percent, respectively, of the full range in measured lake stage over the simulated period. The mean difference and mean absolute difference between measured and simulated lake stages for the four upgradient lakes during the three years with measured water levels (2006–8) were 0.8 and 1.4 ft, respectively.

Sensitivity Analysis

Some uncertainty in model accuracy is inevitable because the model parameter values are never exactly known. However, the importance of each input parameter and its effect on simulation results can be evaluated through sensitivity tests in which the value of a parameter, such as hydraulic conductivity, is adjusted above or below the calibrated value and the magnitude of change in simulated water levels and base flows are quantified. In this study, PEST was used to calculate the sensitivity of all water-level and base-flow residuals (difference between target and simulated values; tables 5 and 6) to changes in each parameter value calculated during the steady-state and transient model calibration processes. PEST uses a commonly accepted algorithm for computing sensitivities, as described in the PEST manual (Doherty, 2008; available at <http://www.pesthomepage.org/Downloads.php>). For the final steady-state calibrated parameter values, sensitivities computed by PEST indicate that water levels and base flows were most sensitive to hydraulic-conductivity parameters for layers in which pilot points were applied to describe heterogeneity within the layer (K_h of layers 1 and 2; K_v of layers 1, 2, and 3). The high sensitivity to hydraulic conductivity in these layers

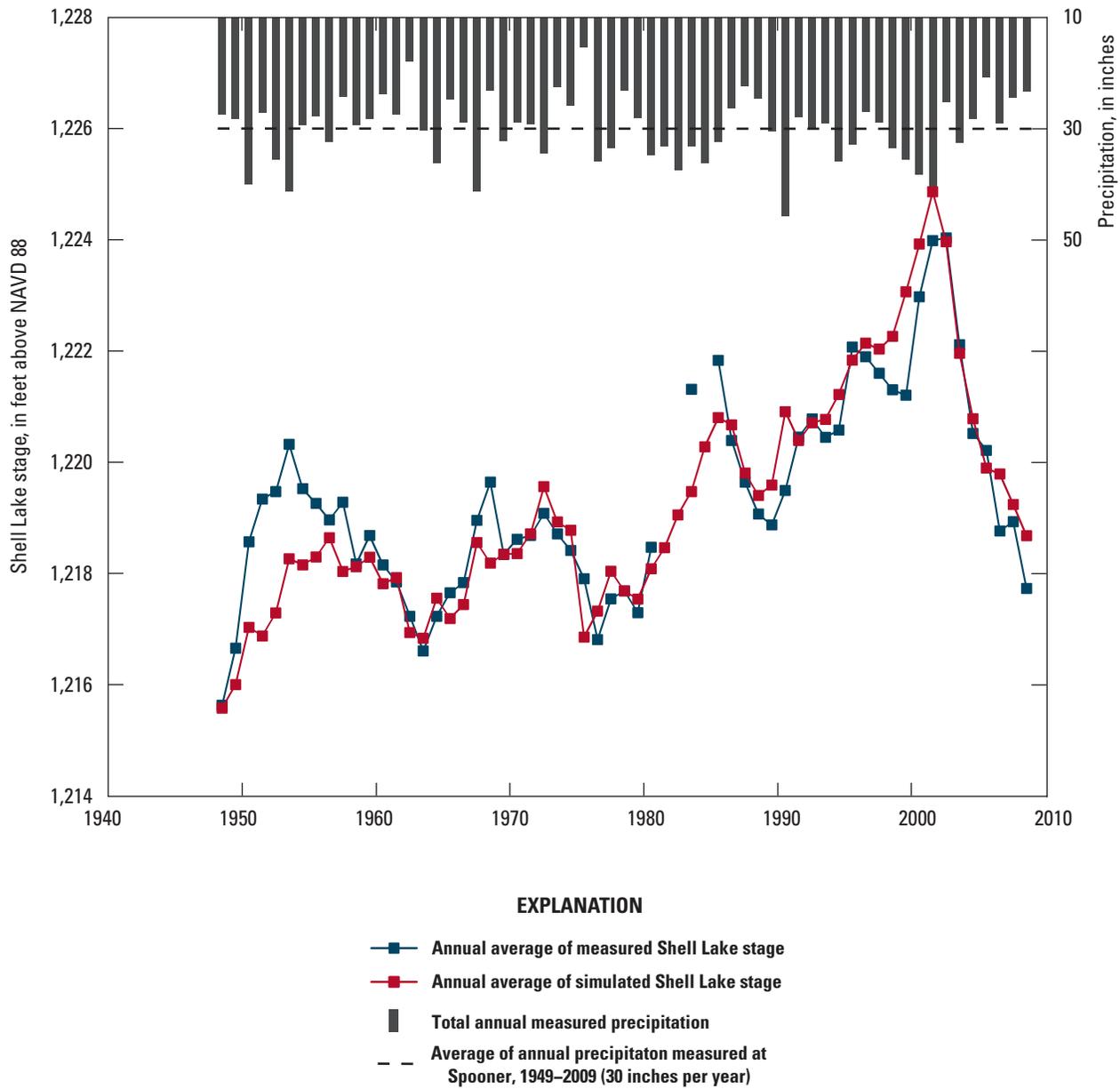


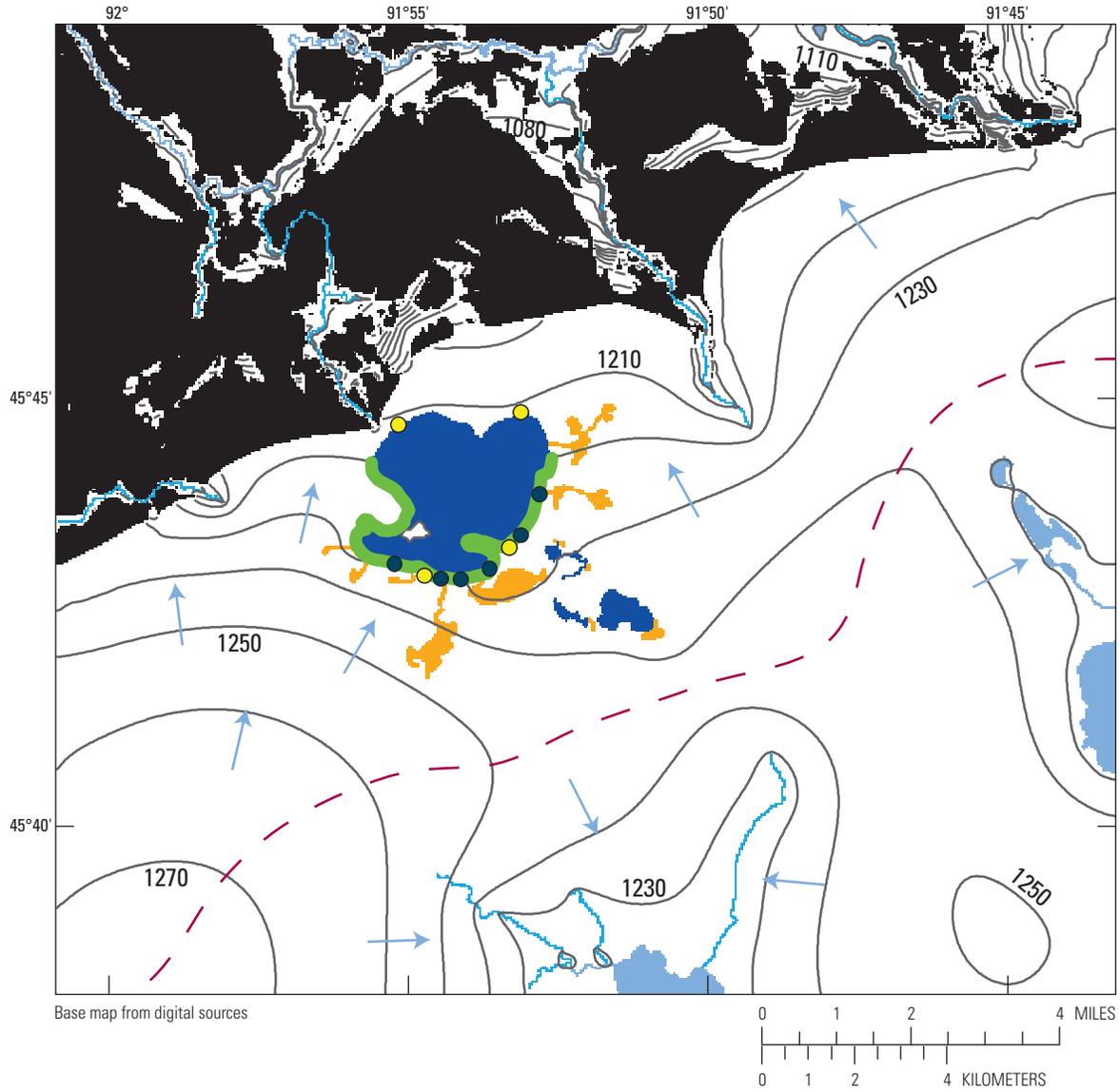
Figure 12. Measured and simulated average annual Shell Lake stage and measured precipitation at the Spooner weather station.

further supports the use of pilot points in that the target observations contained sufficient information to describe heterogeneity within these layers. Less sensitive parameters included hydraulic conductivity of the combined sand, gravel, and sandstone aquifer (layer 4), recharge, and lakebed leakance. The least sensitive parameters included streambed leakance and horizontal hydraulic conductivity of the clay-dominated confining unit (layer 3). Initial sensitivity analyses showed similar results and were used to guide selection of parameters for estimation with the steady-state model. For the final transient-model calibrated parameter values, sensitivities computed by PEST indicate that lake levels were most sensitive to the recharge multiplier and the evaporation multiplier for Rainbow Reservoir, followed by the evaporation multiplier for the Marshfield Experimental Farm. The evaporation multiplier for Rainbow Reservoir was likely more sensitive than the multiplier for the Marshfield Experimental Farm because pan evaporation data from the Rainbow Reservoir were used to estimate lake evaporation for 48 of 61 years, compared with the 13 years of pan evaporation data from Marshfield. Aquifer storage parameters were the least sensitive parameters for calibration of the transient model.

Results of Simulated Groundwater Movement Near Shell Lake

Simulated results from the MODFLOW model indicate that groundwater flows generally north in the area near Shell Lake, with flow converging toward the lake (fig. 13). A regional groundwater divide separates groundwater flowing north toward Shell Lake and the Yellow River watershed (part of the St. Croix River watershed) from groundwater flowing south toward Bear Lake and the Red Cedar River watershed (part of the Chippewa River watershed). Groundwater gradients immediately around Shell Lake indicate potential groundwater inflow to the lake along much of the Shell Lake shoreline (all but the northern shoreline), a result supported by measured hydraulic gradients from this study and a previous study by the Wisconsin Department of Natural Resources (1982). An advancement of using the MODFLOW model over previous studies is that, in addition to mapping groundwater flow directions and gradients, the model simulates total groundwater inflow and outflow through the lake as part of computing the historical lake volume and stage. As a result, although much of the shoreline appears to potentially gain groundwater, the amount is limited, especially compared with the amount of water lost from the lake to the groundwater system (see “Water-Budget Summary” under “Lake Water-Quality Analysis”). Minipiezometers were installed for sampling phosphorus in shallow groundwater immediately upgradient of the lake along the southern and eastern borders of the lake where the largest groundwater gradients were simulated (fig. 13).

In addition to understanding horizontal groundwater flow directions, the MODFLOW model was integral to understanding three-dimensional groundwater flow in the vicinity of Shell Lake, as well as the hydrogeologic properties that control groundwater flow around and into the lake. For example, the measured groundwater level in well 20D (fig. 3) was consistently measured about 8 ft lower than the lake level, indicating movement of groundwater away from the lake. This contradicted the measured water level in the shallow well (well 20S), which was consistently about 2 ft higher than the lake level and supported model results that showed groundwater flowing toward the lake along the southeastern part of the lake. Similar divergent hydraulic gradients were observed in a previous study (Wisconsin Department of Natural Resources, 1982). Allowing for flexibility in the conceptual model (fig. 4) through the use of regularized inversion with PEST during the steady-state calibration proved instrumental for understanding the hydrogeologic structure that could allow for such diverging hydraulic gradients. First, it is important to note that Sawyer Creek, a perennial stream with flow that originates less than 3,000 ft from Shell Lake, has a water level that is at least 50 ft lower in elevation than the water level of Shell Lake. In addition, the measured water level in well 10D, which is screened below the clay unit represented by layer 3 in the MODFLOW model, was consistently measured about 39 ft below the stage of Shell Lake, despite being located along the Shell Lake shoreline. These large vertical differences in water level indicate that the clay layer represented by layer 3 in the MODFLOW model must be widespread and have sufficiently low vertical hydraulic conductivity so as to be able to maintain an approximately 40-ft vertical gradient across a 2-mi-wide lake. Calibrated vertical hydraulic conductivities computed by PEST for layer 3 agreed with this conceptualization (fig. 9) and result in relatively low hydraulic heads in the deep aquifer upgradient of Shell Lake near well 20D (fig. 11). However, the second component to understanding the complex three-dimensional flow pattern near Shell Lake is that the clay unit must also have sufficient heterogeneity (the range in permeability) to allow for zones of higher permeability where the low hydraulic head in the aquifer below well 20D could be transmitted through the clay and into the mixed sand and silt aquifer represented by layer 2, which is the depth at which well 20D is screened; that is, a zone of relatively higher vertical hydraulic conductivity in the clay layer below well 20D was estimated through the regularized inversion process by using pilot points with PEST during the steady-state calibration process (fig. 9). As a result of implementing advanced calibration routines with a flexible conceptual model, the MODFLOW model was able to simulate diverging vertical flow paths, where very shallow groundwater upgradient of Shell Lake discharges into the lake but deeper groundwater (approximately 40 ft below land surface) flows beneath Shell Lake and discharges to Sawyer Creek and the Yellow River. Moreover, adequate representation of the clay units was important for simulating the complex flow paths as well as identifying the three-dimensional distribution of groundwater and the amount of groundwater-associated phosphorus expected to discharge into Shell Lake.



EXPLANATION

- Inactive cells (dry)
- Lakes
- Streams
- Distant rivers and lakes
- Wetlands
- Shoreline with simulated groundwater gradient toward Shell Lake
- Simulated water-table contour (above NAVD 88), interval 10 feet
- Simulated groundwater divide
- Simulated groundwater flow direction
- Minipiezometer
- Nested well

Figure 13. Simulated groundwater-flow directions and the shoreline with simulated groundwater gradients toward Shell Lake, Wisconsin.

Simulated Response to a Lake Water Diversion

In addition to improving the understanding of the hydrologic system near Shell Lake, the hydrogeologic study was also designed to evaluate effects of the lake-water diversion on water levels in Shell Lake and adjacent lakes. The rate of water diverted from Shell Lake from November 13, 2003, to July 18, 2005, was estimated by the City of Shell Lake (David Vold, City of Shell Lake, written commun., May 6, 2008). Total water diverted during each calendar year was applied directly to the calibrated transient model for 2003, 2004, and 2005. During 2003–5, the total diverted water (4×10^8 ft³, or 3 billion gallons) accounted for 27 percent of the water leaving the lake; evaporation and discharge to groundwater accounted for 52 and 21 percent of the outflow, respectively. Cumulative outflow from the lake (1.47×10^9 ft³, or 53.9 in/yr) during 2003–5 exceeded cumulative inflow (1.03×10^9 ft³, or 37.6 in/yr) by 4.46×10^8 ft³ (16.3 in/yr) during the 3-year simulated diversion period. Therefore, the simulated lake level decline over the 3-year diversion period was 4.1 ft. However, not all of the decline during the 3-year period can be attributed to the diversion, because estimated evaporation exceeded precipitation during two of the three years and likely contributed to some of the water level decline.

To directly evaluate the effect of the diversion on water levels, the diversion was removed from the transient simulation, and the resulting simulated water levels were compared with water levels from the calibrated model that included the diversion (figs. 14 and 15). The maximum difference in simulated annual average water levels for Shell Lake (approximately 3.3 ft) occurred at the end of the diversion, in 2005 (fig. 15). It is important to keep in mind that the model simplifies the natural physical system and encapsulated the seasonality of weather patterns into annual average conditions; therefore, results are expected to incorporate a higher level of uncertainty than the precision with which the model results are graphed. Nonetheless, the simulated difference in Shell Lake water level caused by the diversion is reasonable and is expected to be useful for future decisions about diverting water from Shell Lake. The results in figure 15 also illustrate that the effect of the diversion on the water level of Shell Lake decreased after the diversion ceased. This response is attributed to the artificially lower lake level compared with surrounding groundwater levels that were less affected by the diversion. Over the 4 years of simulated recovery (2006 to 2009), drawdown attributable to the diversion decreased by about 0.6 ft (fig. 15) because of increased groundwater inflow and decreased lake-water leakage to groundwater, which is associated with the relatively lower hydraulic gradients between the lake and the water table. Thus, of the 4.1-ft simulated decline in the average annual water level of Shell Lake from 2003 to 2005 (fig. 12), approximately 3.3 ft of the decline is attributable to the diversion. However, in the 4 simulated years after the diversion was halted, model results suggest that the decline due to the diversion alone diminished by about 0.6 ft. This 0.6-ft “recovery” may seem counterintuitive

because the measured lake level continued to decline during that period because of an extended drought in the area. However, the simulated 3.3-ft decline and 0.6-ft recovery represent the amount of water level change due *only* to the diversion and illustrate the power of computer models to separate effects of the diversion from effects of the drought (figs. 14 and 15).

A delayed response in drawdown associated with the water diversion from Shell Lake was transmitted through the groundwater-flow system to upgradient lakes. At the monitoring-well locations, timing of the simulated maximum groundwater drawdown ranged from coincident with the diversion to more than 4 years after the diversion, and water-level decline ranged from less than 0.5 ft to more than 2 ft (fig. 15). Differences in the response to the diversion are likely associated with heterogeneity in the aquifers and hydraulic gradients both within the aquifers and between Shell Lake and the aquifers. Simulated drawdown in upgradient lakes was further delayed, and the amount of drawdown was less than that observed in Shell Lake or at any of the monitoring wells. Four years after the diversion stopped, drawdown due to the diversion was less than 0.5 ft in all upgradient lakes and diminished with distance from Shell Lake (fig. 15). On the basis of simulation results, it appears that the effect of the diversion on drawdown in upgradient lakes had not peaked by 2009 but was increasing by a smaller amount each year (“leveling off”). The limited effect of the diversion on water levels in upgradient lakes is likely related to the relatively small amount of lake/groundwater interaction for Shell Lake and the upgradient lakes, due in part to extensive layers of shallow clay.

One challenge with predicting the effect of the diversion on lake levels is that prediction results can be sensitive to parameters that were found to have relatively low sensitivity during the calibration. For example, aquifer storage properties, which had low sensitivity for the transient calibration, were not expected to have increased sensitivity for the prediction because groundwater levels had a relatively minor influence on Shell Lake water levels during both the calibration and prediction because groundwater is a minor component of the total water budget. Nonetheless, sensitivity of lake level declines due to the lake-water diversion (the model prediction) was evaluated by testing a range of specific-yield values (0.1 to 0.3). This sensitivity test indicated that results of the prediction remained relatively insensitive to aquifer storage properties, with drawdowns increasing by less than 0.1 ft in all lakes and less than 0.5 ft at all groundwater monitoring well locations (over the tested range of reasonable specific yield values) compared with predicted results from the calibrated model.

Limitations of Groundwater-Flow Simulations

As is the case with all models, the groundwater-flow models described in this report are simplifications of the physical system and have corresponding limitations in model accuracy and how the models should be used. For example, as a result of the MODFLOW model discretization (150-ft-wide

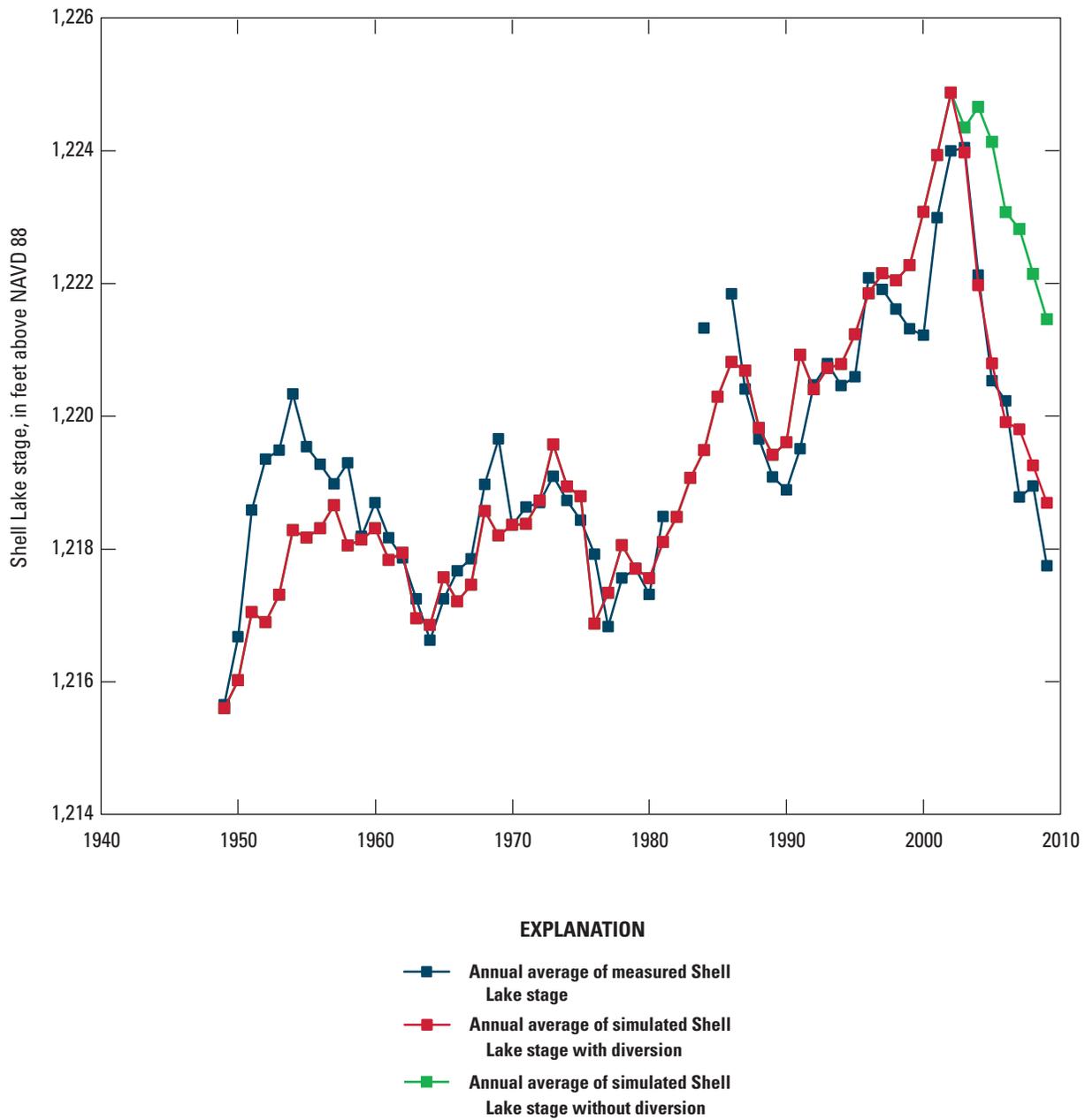


Figure 14. Measured and simulated (with and without the 2003–5 diversion) annual average Shell Lake stage.

cells), the conditions within each cell (groundwater level) are reduced to one average value for the entire cell. Although this cell size is adequate for the simulation objectives described in this report, new analyses requiring finer spatial detail would benefit from refinement of the model. In addition, a modeling artifact (dry cells) affected simulated groundwater levels and flow patterns north and west of Shell Lake. Although simulation of the historical Shell Lake hydrograph and annual water budgets are representative, modeling local flow patterns near the dry cells may require additional refinement or modification

to the model. Also, the model may not perform equally well in all locations because of local geologic complexities that were not incorporated into the model; that is, the calibrated hydraulic-conductivity fields were designed to maintain homogeneity where target data were insufficient to support additional complexity (heterogeneity). Thus, new simulations that focus on areas where little data were available to estimate the local hydraulic-conductivity field may require additional hydrologic data collection for recalibration so that heterogeneity is properly simulated in the new area of interest.

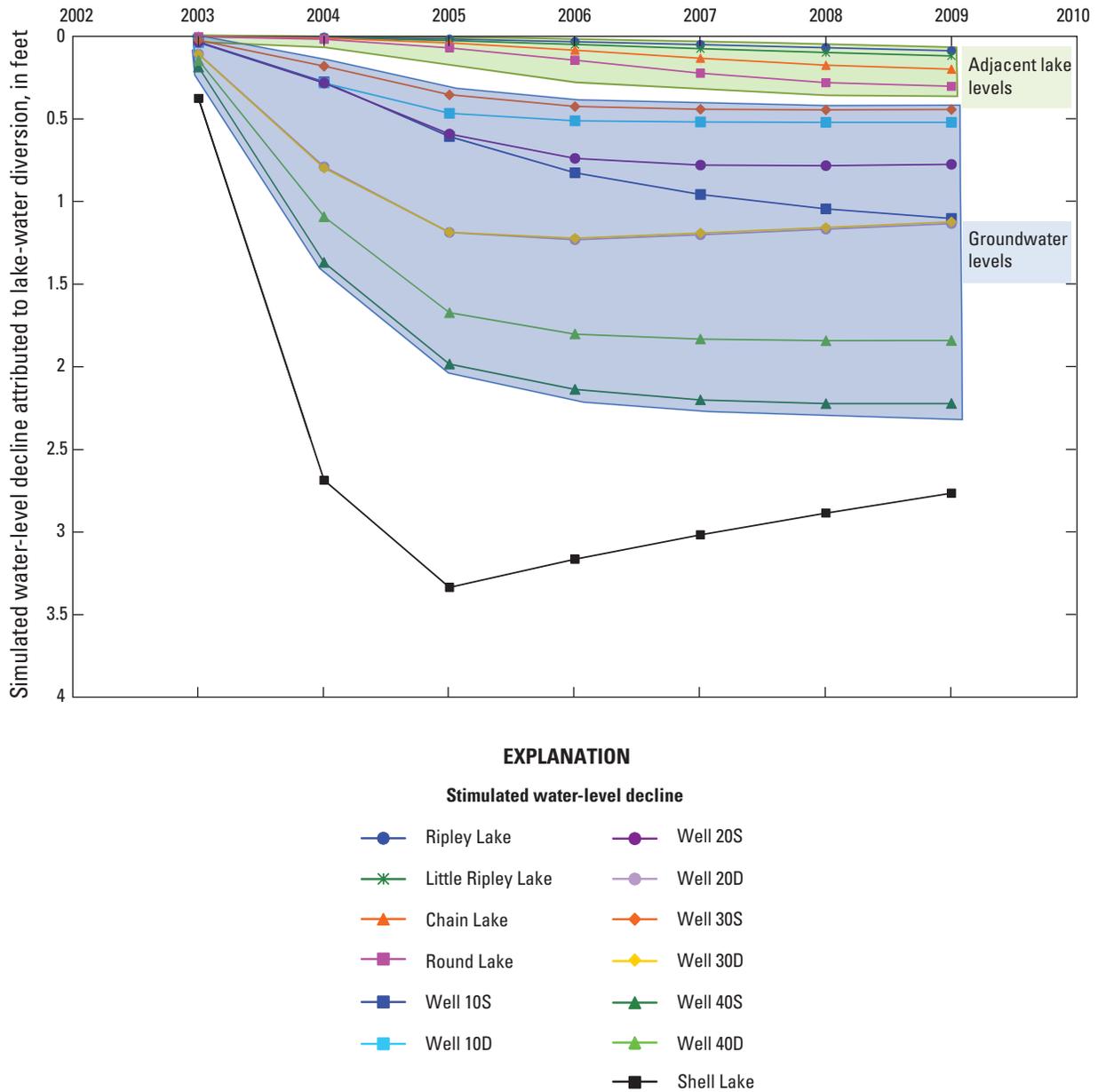


Figure 15. Simulated declines in water levels associated with the 2003–5 diversion.

The model-calibration process focused on annual (transient) and longer (steady state) water-level and base-flow targets to estimate hydrologic and hydrogeologic properties. As a result, the model cannot simulate hydrologic phenomena for periods shorter than 1 year (for example, seasonal water-level fluctuations). Moreover, simply refining the discretization of time in the model (from years to months or days) may not be sufficient to simulate subannual phenomena with the current model design; that is, hydrologic processes (notably recharge, infiltration, and runoff) are currently designed to be simulated by using annual average conditions wherein seasonal wet

and dry periods are expected to be sufficiently offset so as to be represented by a single value for the year and, therefore, adequately represented by the MODFLOW module used to simulate the respective process (RCH, UZF, LAK packages). Refinement of the time discretization would likely require alternative means of representing these hydrologic processes. In addition, target water levels currently represent annual averages and would need to be adjusted prior to a recalibration to inform subannual hydrologic processes. Also, current simulated results approximate average annual conditions and are not precisely representative for each year. Specifically, the

model cannot be expected to exactly match observed Shell Lake water levels every year; instead, the general pattern and cycle of annual water levels, and the change in these patterns caused by a stress (the diversion) were the focus of the simulations and represent the most appropriate timescale for interpretation. For example, the MODFLOW model has limited ability to simulate large interannual or seasonal fluctuations in runoff into Shell Lake. Because of the importance of phosphorus loading from runoff for understanding lake water-quality responses to climatic and land-management changes, the “Hydrology and Water Budget” expanded upon the MODFLOW model results by using a modified method for computing runoff into Shell Lake.

Finally, potential future applications of the model for which groundwater-flow velocities are important (for example, mapping a time-referenced contributing area to a well, such as a “10-year area contributing recharge to a well”) would benefit from field measurements and analyses of porosity, groundwater age, and traveltimes.

Potential Groundwater Contamination from Flooding of Residential Properties

A concern in the wake of the high lake levels was the potential for possible infiltration of contaminants (volatile organic compounds (VOCs) and heavy metals) from flooded supplies of vehicle fluids and household and lawn chemicals stored in garages or basements into the groundwater system near Shell Lake after drawdown of the lake because of the diversion. To address this concern, the four shallow nested wells were sampled in July 2007 and analyzed for a small set of water-quality constituents (calcium, magnesium, chloride, alkalinity, and nutrients) and trace concentrations of 7 heavy metals and 68 VOCs. Insufficient water in wells 20-shallow (20S) and 30-shallow (30S) prevented sampling of those wells for heavy metals in 2007. The shallow wells were sampled for only the small set of water-quality constituents in 2008 and 2009. The four deep wells were sampled and analyzed for only the small set of water-quality constituents in 2007, 2008, and 2009. All water samples were sent to the Wisconsin State Laboratory of Hygiene for analysis.

Results of these analyses are provided in the appendixes. Almost all samples for VOCs in the four shallow wells and all samples analyzed for heavy metals in the two sampled shallow wells were below detection limits. Only carbon disulfide, which can be found in some soil disinfectants, herbicides, and grain fumigants (Montgomery and Welkom, 1989), was detected at low concentrations in one well. The general absence of contaminants in groundwater near the nested wells indicates that few, if any, contaminants entered the groundwater system after flooding of homes and subsequent drawdown of the lake. Nonetheless, it is possible that some contaminants may have entered the groundwater-flow system, but any such contamination was not sufficiently widespread to be detected at the well locations.

Lake Water-Quality Analysis

This second section of the report focuses on the water quality in Shell Lake and how the water quality in the lake has responded in the past and should respond in the future to changes in phosphorus loading and water level. Much of the information used to develop and evaluate the MODFLOW groundwater model also was used with the water-quality data collected in Shell Lake and its tributaries to evaluate how the water quality of Shell Lake responds to changes in phosphorus loading and water level.

Stratification and Dissolved Oxygen

During 2009, Shell Lake behaved as a polymictic lake, meaning that it frequently mixed throughout the open-water period. Water temperatures gradually increased throughout the lake as summer progressed, with near-surface water temperatures reaching about 22 °C in August (fig. 16). Temperatures in the deepest areas of the lake also gradually increased, indicating that deep mixing was frequently occurring, but increased to only about 20 °C in late summer. A weak thermocline (the depth interval where the temperature decreases rather abruptly) frequently developed but broke down relatively quickly because of wind and (or) surface cooling. Figure 16 was constructed from biweekly sampling data and therefore may not demonstrate the effects of rapid mixing events. To better define the extent of vertical stratification and the extent of deep mixing, water temperature loggers (thermistors) were suspended throughout the water column at two locations in the lake (fig. 3) from June through September 2009. Results from the loggers (fig. 17) demonstrated that relatively strong stratification developed in the lake (such from June 16 to the beginning of July and from August 10 to August 17, as illustrated by large vertical differences in water temperature), but this stratification was broken down by a series of mixing events (such as from the beginning of July to July 18, illustrated by temperatures throughout the water column beginning to converge). Extensive mixing also occurred over prolonged periods (such as from July 20 to August 9 and from August 21 to September 4, illustrated by little difference in temperature throughout the water column) as a result of wind mixing and surface cooling.

Dissolved oxygen concentrations were near saturation throughout the lake from just after the ice melted until the lake stratified in June. Once a thermocline developed, concentrations in the deep areas gradually decreased, while concentrations in shallow waters remained near saturation (fig. 16). During the prolonged stratification periods, dissolved oxygen concentrations near the bottom decreased, but only once were concentrations (measured bimonthly) below 1 mg/L and unable to support fish. Results from the temperature loggers (fig. 17) indicate that stratification persisted longer than inferred from the biweekly sampling; however, the frequent mixing prohibited dissolved oxygen from decreasing much lower than that indicated in figure 16.

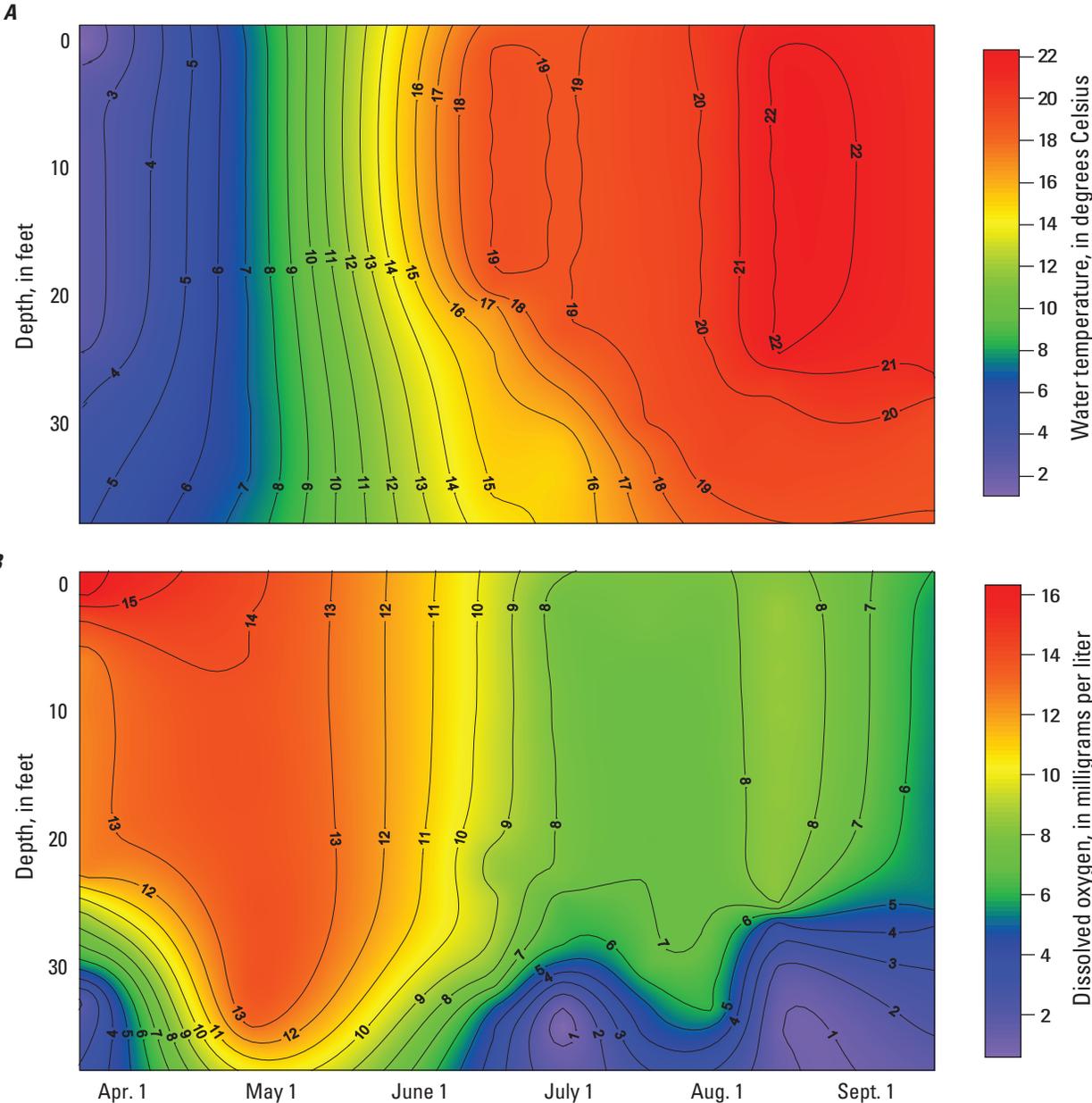


Figure 16. Temperature and dissolved oxygen distributions during the open-water period of 2009, based on data collected at the deep-hole site in Shell Lake, Wisconsin.

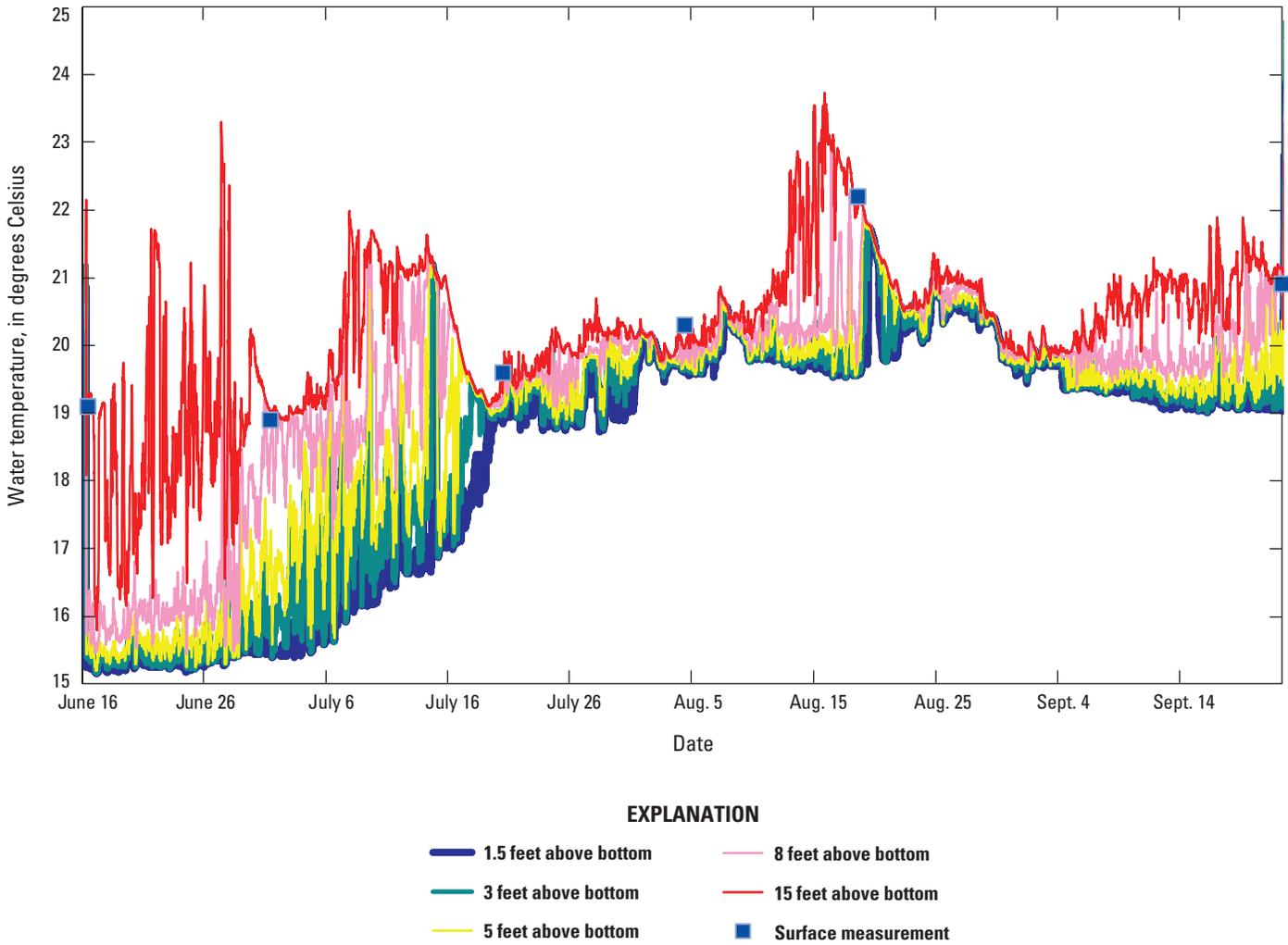


Figure 17. Continuous water temperature data collected at the deep-hole site in of Shell Lake, Wisconsin, from June 16, 2009, to September 22, 2009.

Water Chemistry and Trophic Conditions

Hardness of water is caused primarily by the presence of calcium and magnesium. The hardness of water in Shell Lake was about 13–14 mg/L as calcium carbonate (CaCO_3) (measured by the WDNR between 1980 and 1999), which classifies the lake as having soft water (Hem, 1985). According to Lillie and Mason (1983), relatively low concentrations of calcium, magnesium, and alkalinity characterize lakes in northwestern Wisconsin. They found that for 282 lakes sampled in this region, average concentrations of calcium, magnesium, and alkalinity were 7, 3, and 27 mg/L, respectively. During 1960–99, concentrations of calcium, magnesium, and alkalinity in Shell Lake as measured by the WDNR were 2.2–5.0, 0.5–1.3, and 9–15 mg/L, respectively; therefore, Shell Lake is a little softer than most lakes in northwestern Wisconsin, reflecting the limited amount of groundwater entering the lake.

Specific conductance throughout the lake during spring and fall overturns and near the surface during other periods was about 45 microsiemens per centimeter ($\mu\text{S}/\text{cm}$) but increased to about 55 $\mu\text{S}/\text{cm}$ near the bottom when dissolved oxygen concentrations were low. The pH throughout the lake fluctuated from about 6.7 to 8.4, with values generally decreasing with depth.

Chloride concentrations during spring overturn, measured by the WDNR during 1960–97, were about 1–3 mg/L. The median concentration of chloride in lakes in this area reported by Lillie and Mason (1983) was 2 mg/L. Concentrations of chloride are indicators of urban development in the watershed and road salting in the winter. Concentrations of chloride in Shell Lake are still very low when compared to lakes in southeastern Wisconsin. According to the U.S. Environmental Protection Agency (1988) water-quality criteria, freshwater aquatic organisms should not be affected if the 4-day average concentration of dissolved chloride remains less than 230 mg/L.

Phosphorus and nitrogen are essential nutrients for plant growth and are the nutrients that usually limit algal growth in most lakes. Elevated nutrient concentrations can cause high algal populations or “blooms”; therefore, increases in nutrient inputs can cause accelerated eutrophication (that is, accelerated aging and greater lake productivity). Concentrations of near-surface total phosphorus less than 0.012 mg/L indicate oligotrophic conditions, between 0.012 and 0.024 mg/L indicate mesotrophic conditions, and greater than 0.024 mg/L indicate eutrophic conditions (Carlson, 1977). Near-surface phosphorus concentrations in Shell Lake demonstrated distinct seasonality, with concentrations increasing as summer progressed (fig. 18A). During 2009, concentrations increased from 0.007 mg/L in March to 0.020–0.028 mg/L in September. The average near-surface growing-season (May–September) phosphorus concentration during 2009 was about 0.018 mg/L. Therefore, on the basis of total phosphorus concentrations, Shell Lake would be classified as mesotrophic during the growing season but near oligotrophic in early summer and near eutrophic in late summer (fig. 19). During 2009,

near-bottom total phosphorus concentrations were similar to those near the surface, except in late summer when near-bottom concentrations were about 0.006–0.008 mg/L higher than near the surface.

Water-quality monitoring in Shell Lake began in 1944 (table 7) but was conducted in an inconsistent manner with relatively intense sampling in 1980, during 1986–97, and during 2005–9, but very limited sampling in other years. Near-surface total phosphorus concentrations have ranged from 0.003 mg/L (July 1944) to 0.070 mg/L (August 1970). Concentrations measured prior to 1980 were either very low or quite high, and the data were of questionable quality; therefore, most water-quality data prior to 1980 are not discussed in this report. Since 1980, the lowest near-surface concentration measured in the lake has been 0.004 mg/L (March 1992) and highest near-surface concentration has been 0.044 mg/L (August 2007). No consistent long-term trends in phosphorus concentrations are apparent; however, there was consistent seasonality, with monthly average concentrations being lowest in early summer (June and July) and highest in late summer (August) (fig. 20A).

Although near-bottom total phosphorus concentrations were similar to those near the surface in 2009, bottom concentrations increased well above near-surface concentrations as summer progressed between 1995 and 2007 (fig. 21). During those years, near-bottom concentrations were occasionally greater than 0.100 mg/L, with the highest concentration of 0.25 mg/L measured in August 2005. High near-bottom concentrations were measured when the elevation of the lake was high; however, no bottom sampling was conducted during 2001–3, when lake elevations were highest.

Since 1960, total nitrogen concentrations, typically computed as the sum of Kjeldahl nitrogen¹ and dissolved nitrite plus nitrate, have ranged from 0.17 to 1.15 mg/L. Kjeldahl nitrogen usually represented more than 80 percent of the total nitrogen. The ratio of the near-surface concentrations of total nitrogen to total phosphorus (N:P ratio) is often used to determine the potential limiting nutrient for productivity in a lake. The specific value of this ratio that determines which nutrient potentially is limiting differs under different ambient conditions, such as water temperature, light intensity, and nutrient deficiencies (Correll, 1998); however, a ratio greater than about 16:1 by weight usually indicates that phosphorus should be the potentially limiting nutrient. The N:P ratios for Shell Lake ranged from 9:1 to 62:1, but the average was 27:1. This indicates that phosphorus should typically be the potentially limiting nutrient for Shell Lake, but nitrogen may occasionally limit productivity.

Any decreases in phosphorus concentration should not only increase the N:P ratio but also favor the growth of green algae over cyanobacteria, commonly referred to as “blue-green algae.” Many types of blue-green algae are not limited by nitrogen because they can fix nitrogen from the atmosphere.

¹ Kjeldahl nitrogen is the sum of free-ammonia and organic nitrogen compounds (Wisconsin State Laboratory of Hygiene, 1993).

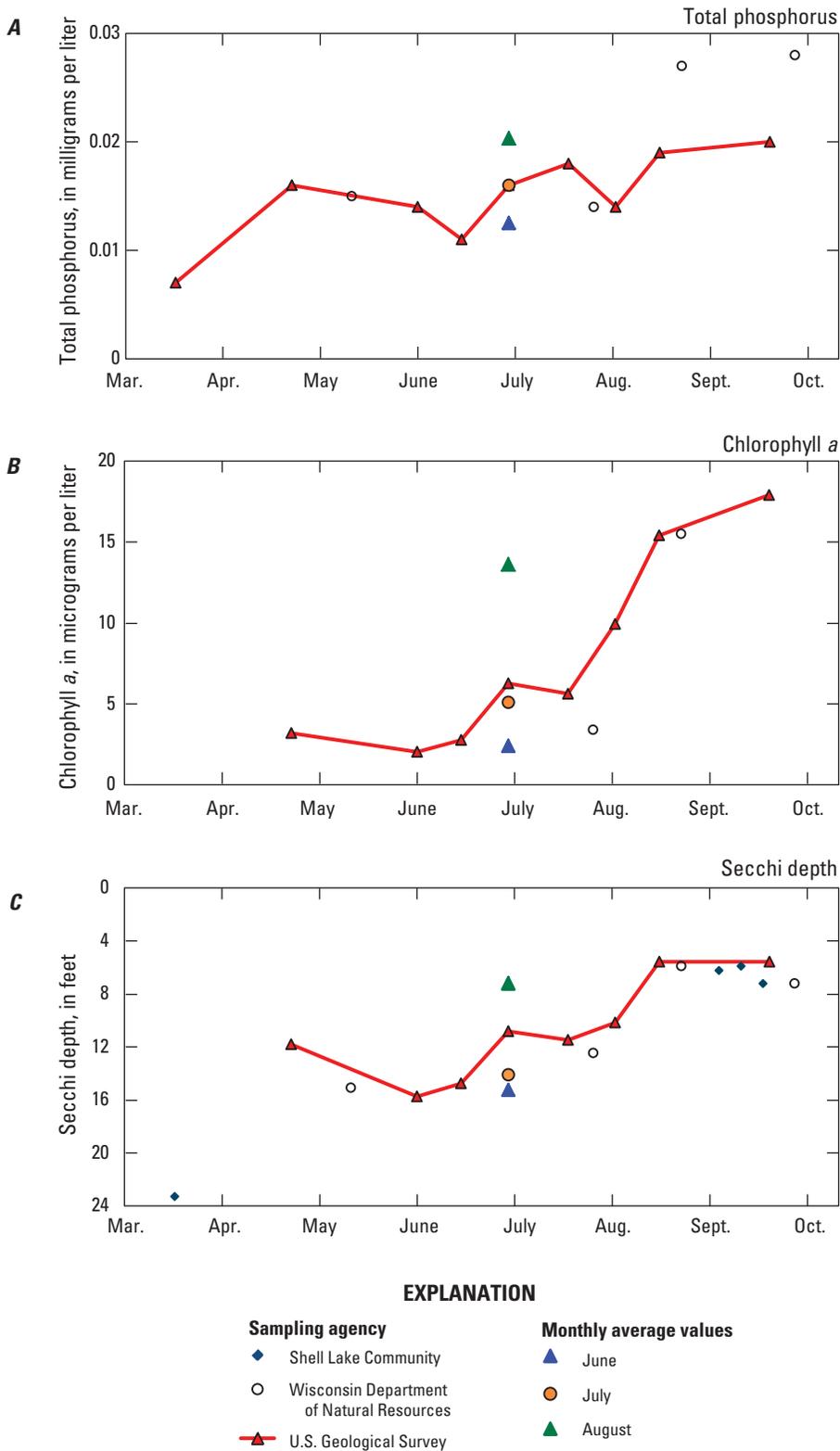


Figure 18. Near-surface water-quality characteristics in Shell Lake, Wisconsin, during 2009. *A*, Total phosphorus concentrations. *B*, Chlorophyll *a* concentrations. *C*, Secchi depths. The monthly average values for June, July, and August for 2009 are plotted at July.

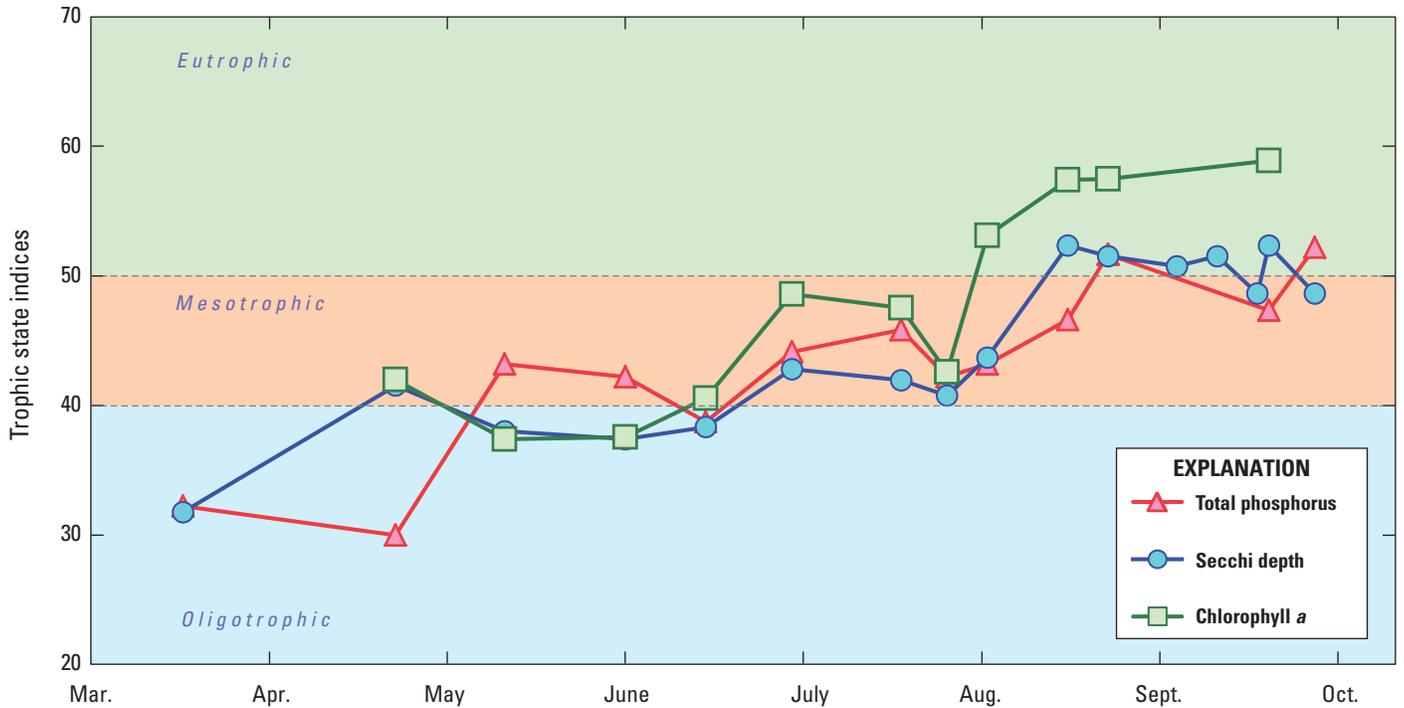


Figure 19. Trophic State Index values based on water-quality samples from Shell Lake, Wisconsin, during 2009.

Blue-green algae are the least desirable type of algae because they commonly form extensive blooms or scums on the water surface, are potentially toxic, and are usually the least preferred food by grazing zooplankton. Therefore, phosphorus should be the nutrient of concern when considering management efforts to improve or prevent degradation of the water quality of Shell Lake.

Chlorophyll *a* is a photosynthetic pigment found in algae and other green plants. Its concentration is commonly used as a measure of the density of the algal population in a lake. Average concentrations less than 2 micrograms per liter ($\mu\text{g/L}$) indicate oligotrophic conditions, between about 2 and 7 $\mu\text{g/L}$ indicate mesotrophic conditions, and greater than 7 $\mu\text{g/L}$ indicate eutrophic conditions (Carlson, 1977). During 2009, concentrations increased as summer progressed, with concentrations of 2–3 $\mu\text{g/L}$ in spring and early summer and 18 $\mu\text{g/L}$ in fall (fig. 18B). The growing-season-average near-surface chlorophyll *a* concentrations during 2009 was about 8 $\mu\text{g/L}$. Therefore, Shell Lake would typically be classified as eutrophic on the basis of growing-season-average condition but near oligotrophic in early summer and eutrophic in late summer (fig. 19).

Since 1980, chlorophyll *a* concentrations have ranged from less than 1 $\mu\text{g/L}$ to about 30 $\mu\text{g/L}$. No consistent long-term trends in chlorophyll *a* concentrations are apparent; however, there was consistent seasonality, with concentrations being lowest in early summer (June and July) and generally highest in late summer (August) (fig. 20B).

During 2009, water clarity, as measured with a Secchi disk, consistently decreased as summer progressed. The average Secchi depth during summer was about 11 ft, but individual measurements decreased from about 16 ft in June to about 6 ft in September (fig. 18C). Secchi depths greater than 13.1 ft indicate oligotrophic conditions, between 6.6 and 13.1 ft indicate mesotrophic conditions, and less than 6.6 ft indicate eutrophic conditions (Carlson, 1977). Therefore, on the basis of Secchi depths, Shell Lake was classified as mesotrophic relative to growing-season-average conditions but oligotrophic in early summer and eutrophic in late summer (fig. 19).

Since 1968, Secchi depths have ranged from 3 ft in July of 1993 to 33 ft in March of 1970. Since 1980 (fig. 20C), Secchi depths have had consistent seasonality, with clarities typically best in June and worst in August. No trends in clarity were apparent in any month; however, the extent of the seasonal variability has fluctuated during this time period. Although water clarity was almost always better in June than in July and August, water clarities in June during 2000–2008 were much better than in July and August. This extreme seasonality during 2000–2008 was measured when the elevation of the lake was high. During June 2011, water clarity was much better than later in the summer; however, this difference was based on only one Secchi-depth measurement in June (32 ft).

Table 7. Near-surface monthly (June, July, and August) average total phosphorus, chlorophyll *a*, and Secchi depths for Shell Lake, Wisconsin, 1968 to 2012.

Year	Total phosphorus (milligrams per liter)			Chlorophyll <i>a</i> (micrograms per liter)			Secchi depth (feet)		
	June	July	August	June	July	August	June	July	August
1968	--	--	0.039	--	--	--	--	--	15.0
1969	--	--	0.065	--	--	--	--	--	17.0
1970	--	--	0.070	--	--	--	--	--	6.0
1971	--	--	--	--	--	--	--	--	8.5
1980	0.010	0.010	0.030	1.7	3.0	8.7	15.1	13.6	11.4
1986	--	0.020	0.020	--	10.0	15.0	0.0	10.2	5.9
1987	0.008	0.010	0.013	3.0	2.5	5.0	19.7	14.8	11.2
1988	0.007	0.010	0.020	2.0	3.0	22.0	11.5	11.5	6.2
1989	0.009	0.008	0.015	3.0	2.0	7.0	13.9	15.0	10.0
1990	0.008	0.010	0.014	2.0	3.0	16.0	15.3	12.6	8.4
1991	0.007	0.006	0.020	2.0	2.0	9.0	14.0	17.0	15.0
1992	0.008	0.012	0.018	4.0	4.0	9.3	12.0	10.4	9.5
1993	0.008	0.012	0.013	3.8	5.0	20.6	15.0	9.4	8.7
1994	0.009	0.011	0.014	3.9	3.1	15.9	16.7	14.4	7.5
1995	--	0.008	0.011	0.6	--	6.6	19.7	17.9	12.8
1996	0.009	0.020	0.008	3.2	3.6	3.2	16.2	14.4	14.4
1997	0.005	0.008	0.009	2.4	1.6	2.6	15.1	14.4	24.9
1999	--	--	0.021	--	--	23.0	--	--	5.9
2000	--	--	--	--	--	--	20.0	13.1	11.2
2001	--	--	--	--	--	--	--	--	9.2
2002	--	--	--	--	--	--	--	20.5	10.2
2003	--	--	--	--	--	--	21.7	15.9	9.7
2004	--	--	--	--	--	--	21.8	--	12.1
2005	0.007	0.013	0.015	1.5	2.6	10.0	15.8	14.0	12.2
2006	0.020	0.019	0.024	1.8	4.0	13.4	21.5	13.5	8.5
2007	0.010	0.014	0.044	3.9	8.2	25.0	13.6	8.0	5.2
2008	0.018	0.016	0.019	2.9	--	8.3	20.0	14.1	11.5
2009	0.013	0.016	0.020	2.4	5.1	13.6	15.3	11.6	7.2
2010	--	0.020	0.009	--	28.9	4.6	--	7.3	5.7
2011	--	0.013	0.027	--	4.9	27.5	32.2	14.0	5.9
2012	0.020	0.017	0.026	3.9	5.2	19.0	12.1	11.6	8.5
Average: 1968–2012	0.010	0.013	0.023	2.7	5.4	13.0	16.4	13.3	10.2
Average: 2005–12	0.015	0.016	0.023	2.7	8.4	15.2	18.6	11.7	8.1

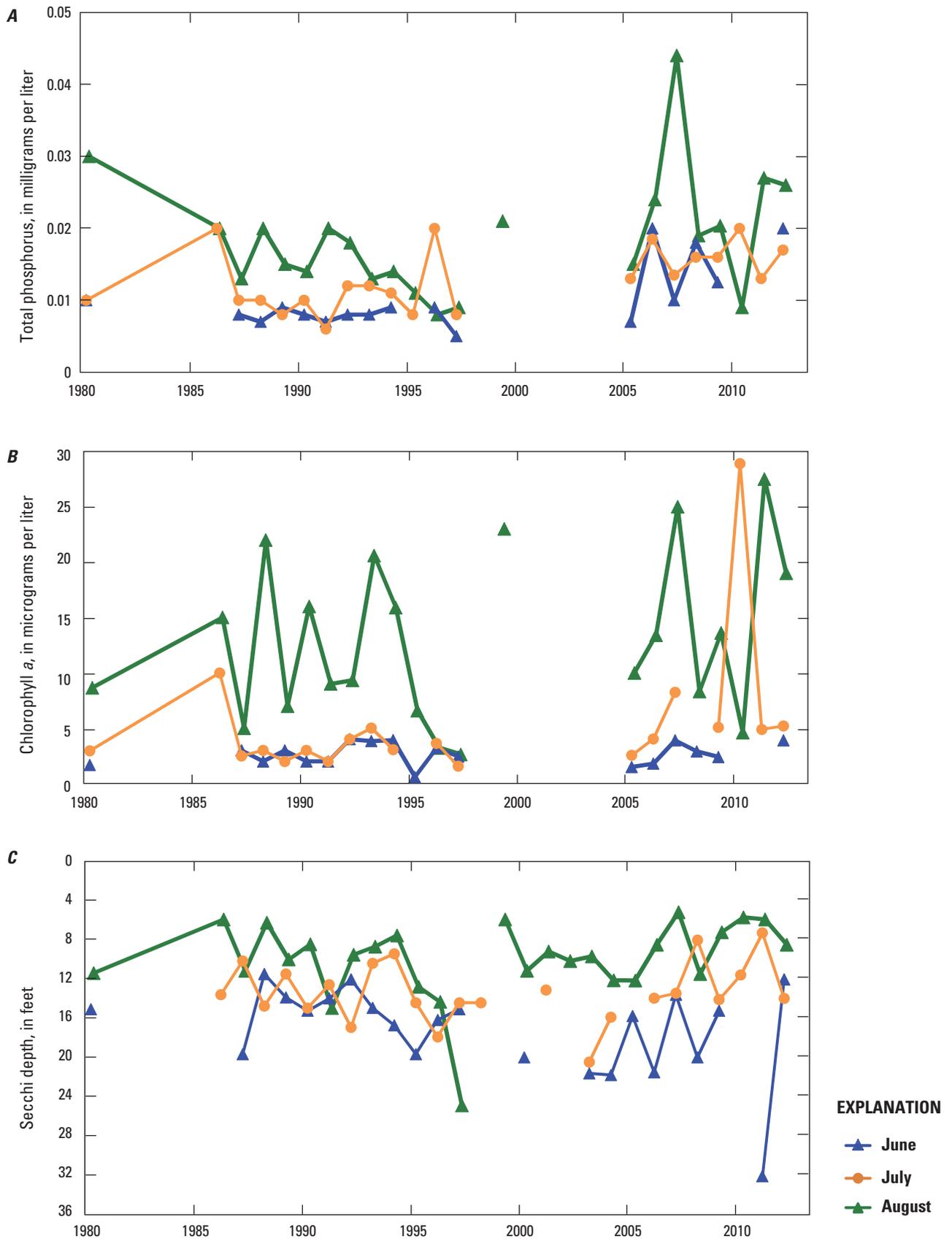


Figure 20. Monthly average near-surface water-quality characteristics in Shell Lake, Wisconsin, 1980–2012. *A*, Total phosphorus concentrations. *B*, Chlorophyll *a* concentrations. *C*, Secchi depths.

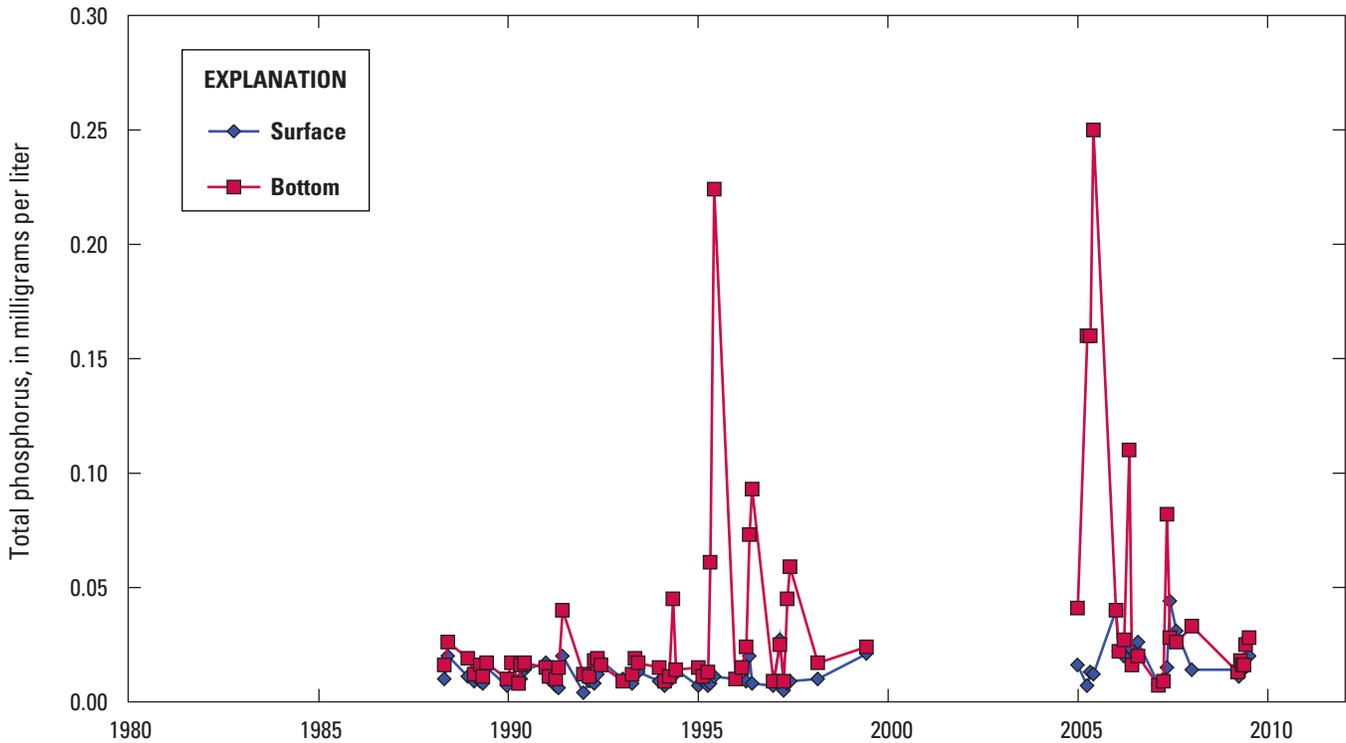


Figure 21. Near-surface and near-bottom total phosphorus concentrations Shell Lake, Wisconsin, 1988–2009.

TSI values based on near-surface concentrations of total phosphorus, chlorophyll *a*, and Secchi depths indicate that since 1980, Shell Lake has been typically oligotrophic-mesotrophic in June, mesotrophic in July, and mesotrophic-eutrophic in August (fig. 22). All three indices generally demonstrate similar interannual variability in the trophic state in the lake; however, during August, chlorophyll *a* concentrations indicate that the lake was more eutrophic than indicated by phosphorus concentrations and Secchi depths. In other words, there was typically more chlorophyll *a* than expected given the measured phosphorus concentrations and clarities. This difference may be related to the scarcity of macrophytes (rooted aquatic plants) in the lake.

Inferred Lake Water Quality From Analyses of Lake Sediment

Results of analyses of a sediment core extracted from the deep area of Shell Lake were used to quantify historical sedimentation rates and reconstruct changes in the water quality and the amount of macrophytes in the lake (Garrison, 2012). The overall average annual sedimentation rate for Shell Lake was 0.013 gram per square centimeter (g/cm^2), which is below

average for the lakes that have been studied in Wisconsin. In the early 1800s, the annual sedimentation rate in Shell Lake was $0.005 \text{ g}/\text{cm}^2$, but it steadily increased to about $0.020 \text{ g}/\text{cm}^2$ after the establishment of a sawmill on the lake in 1881. After the sawmill was closed in 1899, the sedimentation dropped to about $0.015 \text{ g}/\text{cm}^2$. Since 1910, sedimentation has fluctuated but has remained above the historical sedimentation rate possibly because of agricultural activities and urban development in the watershed. Around 2010, the sedimentation rate was about $0.018 \text{ g}/\text{cm}^2$.

Based on the analysis by Garrison (2012) of historic diatom assemblages, the macrophyte community has not changed much since the 1850s, and the density of macrophytes has remained quite low. Changes in diatom assemblages were also used to estimate changes in the phosphorus concentrations in the lake (fig. 23; Garrison, 2012). The diatom-inferred growing-season-average phosphorus concentration before the arrival of European settlers was about 0.012 mg/L. With the building of the sawmill in 1881 and the establishment of the village of Shell Lake, the phosphorus levels increased to 0.014–0.015 mg/L. The sawmill and associated activities increased the sedimentation rate but had only a small effect on the phosphorus concentrations in the lake. Land-use changes in the 1920s, likely in part from agriculture, had a relatively large effect

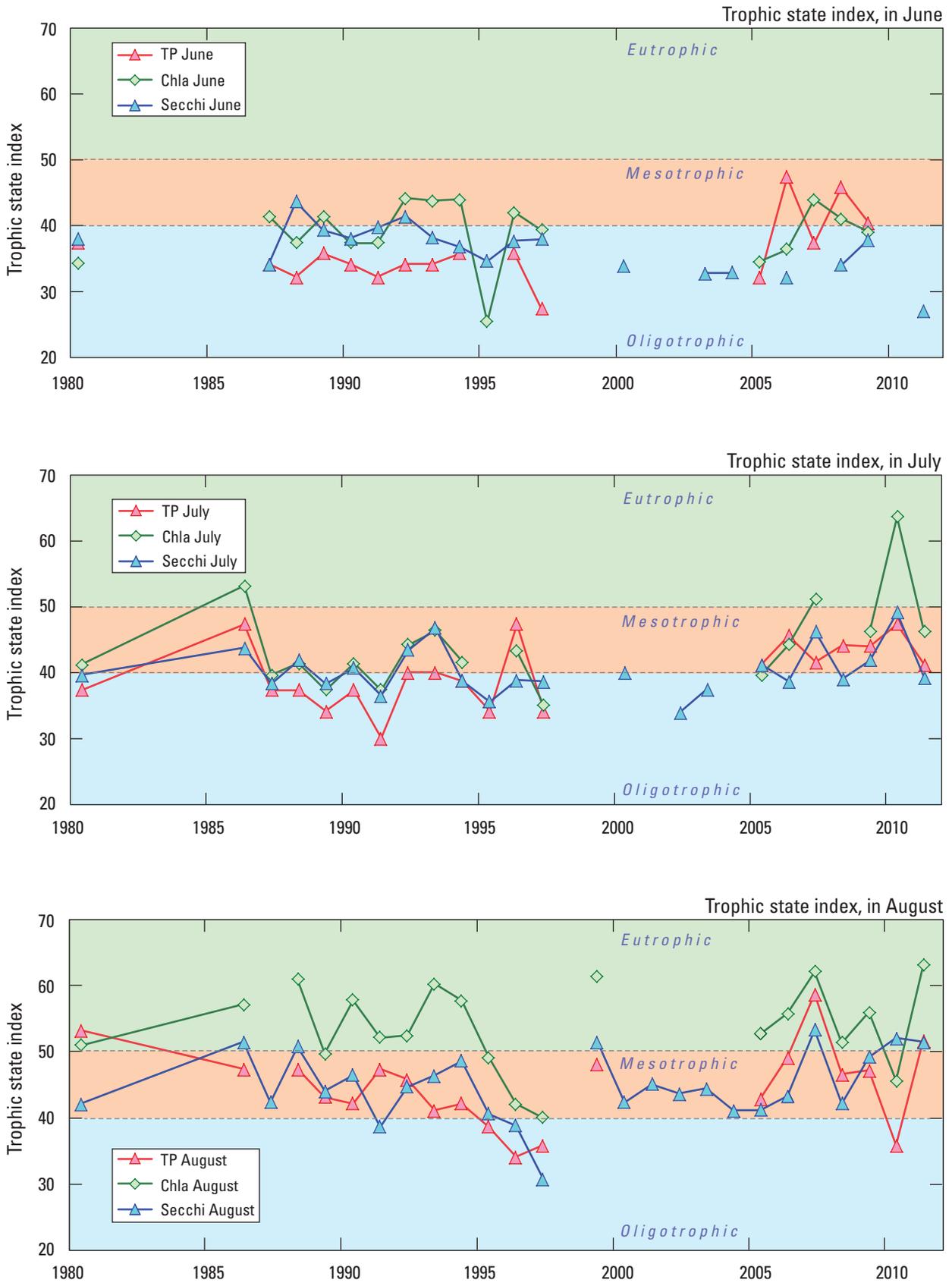


Figure 22. Monthly average Trophic State Index values based on water quality measured in Shell Lake, Wisconsin, from 1980 to 2011. Chlorophyll *a* is abbreviated as Chla in this figure.

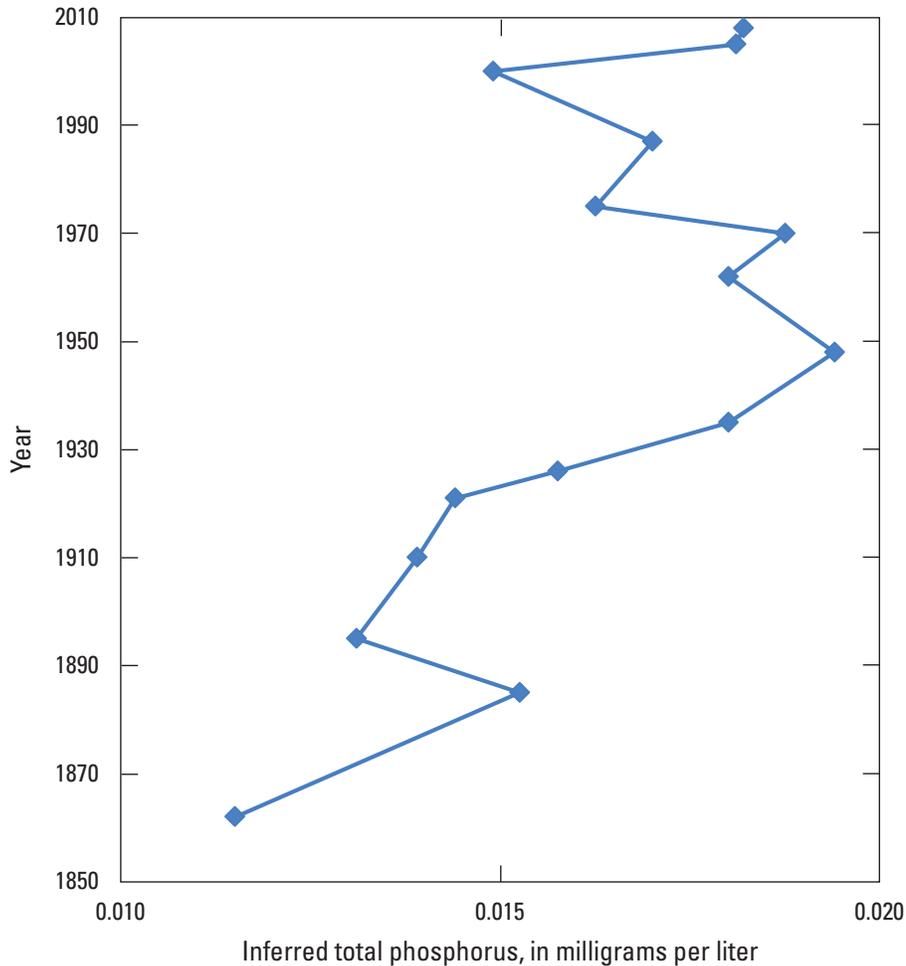


Figure 23. Estimated phosphorus concentrations from 1860 to 2010 inferred from changes in diatom assemblages in a sediment core collected in Shell Lake, Wisconsin (data from Garrison, 2012).

on the phosphorus concentrations in the lake. Phosphorus concentrations peaked around 1950, at 0.020 mg/L. Phosphorus concentrations appear to have declined slightly during 1950–2000, to about 0.015 mg/L. During the last decade, growing-season-average phosphorus concentrations based on the diatom assemblages from the sediment core appear to have again increased to near 0.018 mg/L.

One of the goals of the study by Garrison (2012) was to determine how natural fluctuations in water level affected water quality and whether those fluctuations were as important as anthropogenic changes in the watershed. Garrison found that anthropogenic changes in the watershed, especially between 1910 and 1950, had a much larger effect on growing-season-average water quality than the changes in water level, which appeared to have minimal effect on water quality on the basis of diatom assemblages.

A study by Jeremy Williamson (Polk County Wisconsin Land and Water Resources, written commun., 2012) examined changes in Chironomid (aquatic insects) communities in the same sediment core as Garrison (2012). Changes in

the Chironomid community structure have been shown to be related to changes in near-bottom dissolved oxygen concentrations, which may decrease as a result of increased productivity in the lake resulting from increased phosphorus loading from the watershed (Quinland and Smol, 2001). Changes in the Chironomid community structure in Shell Lake sediments indicated a similar trend as that found with changes in the diatom community: water quality in Shell Lake may have degraded as a result of the establishment of the sawmill around 1880, improved slightly when the sawmill was removed, and then degraded again beginning around 1920, likely in part from increased agriculture in the basin (Jeremy Williamson, Polk County Wisconsin Land and Water Resources, written commun., 2012). Changes in the Chironomid communities since the 1930s with respect to changes in water quality have been difficult to interpret because of the polymictic nature of Shell Lake and the complicating effects of changes in stratification. Changes in water level affect the intensity of stratification in the lake and near-bottom dissolved oxygen concentration; therefore, it is difficult to separate the effects of increases in

lake productivity caused by increases in phosphorus loading from the watershed and changes in near-bottom dissolved oxygen concentrations in the lake resulting from changes in the intensity of stratification in the lake.

Hydrology and Water Budget

Because the productivity in Shell Lake is typically limited by phosphorus (on the basis of N:P ratios) and because decreases in phosphorus concentrations should favor production of green algae over blue-green algae, decreases in phosphorus input to Shell Lake should reduce its phosphorus concentrations and thus improve its water quality. Therefore, it is important to understand the origins of the phosphorus reaching the lake. Almost all of the phosphorus entering the lake is transported by water inputs; therefore, to quantify phosphorus inputs, it is necessary to first quantify the water inputs. Preliminary annual water budgets (eq. 4) were constructed as part of the groundwater-modeling component of this study, in which precipitation, surface-water inflow, and evaporation were estimated (refer to “Hydrologic Information for the Lake Water Budget”), and groundwater input and output from the lake were computed by the transient MODFLOW model from 1949 to 2009. The water budgets constructed as part of the groundwater-modeling component of this study were better at estimating long-term changes in water level and lake volume than the original hydrologic model by Krohelski and others (1999; fig. 24). However, additional refinements of event-based runoff were made as part of the water-quality analysis; that is, the annual surface-water inflow estimates from the groundwater-modeling component of this study were adequate for simulating groundwater movement and long-term lake level fluctuations, but the estimated surface-water inflows may not have properly represented the expected interannual variability in runoff and ultimately phosphorus loading that is needed for simulating changes in the water quality of Shell Lake that occurs during a series of wet and dry years. Specifically, simulation of the lake/groundwater system with MODFLOW was not designed to simulate water contributions from unmonitored near-lake areas or event-based runoff, both of which were expected to be potentially important for understanding phosphorus loading to the lake.

Estimating Runoff From the Unmonitored Areas and Total Runoff to Shell Lake

Total surface-water runoff to Shell Lake is equal to that contributed by the watersheds of the six monitored streams described previously and that contributed from the remaining area draining to the lake (referred to as the “unmonitored area”). Runoff from the unmonitored area includes that conveyed in numerous small channels and from areas drained by unchannelized direct runoff to the lake, including the runoff from residential shoreline development. Because of the complex nature of the sources of this runoff, the use of a constant

or seasonally varying runoff coefficient applied to annual precipitation was thought to be inadequate. Instead, the total surface-water runoff to the lake for 2009 (and for each year from 1949 to 2011) was estimated from an empirical annual runoff model developed specifically for Shell Lake (described below). Runoff from the unmonitored area from 1949 to 2011 was then computed by subtracting that estimated from the monitored streams from the total estimated runoff. During dry years such as in 2009, the runoff from the unmonitored area is expected to be small.

The empirical annual runoff model (eqs. 5 and 6) was developed to describe the hydrology specific to Shell Lake; that is, more runoff to the lake occurs following a series of wet years when the soil moisture is high and potential for overland flow to the wetlands and lake is increased. During relatively dry years (eq. 5), total runoff (base runoff) to the lake is simulated as a simple fraction (α) of the total annual precipitation falling on Shell Lake ($PPTW$, in cubic feet), which was assumed to be equal to the precipitation measured at Spooner, Wis. (6 mi north of Shell Lake). After several wet years, however, soil moisture and runoff potential is elevated and water levels in the wetlands in the basin rise, increasing runoff to the lake. This latter process is simulated to occur when the 4-year moving average annual precipitation measured at Spooner, Wis., exceeds 27 in. (eq. 6). During these wet years, this “additional runoff” can originate from variable source areas (Dunne and Leopold, 1978) around Shell Lake, and is simulated as a nonlinear function (exponential function) of the annual precipitation (AP) rate (inches per year) for that year.

During dry years:

$$\text{Total Runoff} = \alpha (PPTW) \quad (5)$$

During wet years:

$$\text{Total Runoff} = \alpha (PPTW) + \beta(AP)^{\gamma} \quad (6)$$

The coefficients in equations 5 and 6 were estimated by finding the coefficients that best simulated the observed changes in the volume of the lake from 1949 to 2011, given a starting volume of water in the lake; the annual groundwater input, groundwater output, and evaporation estimated from the groundwater model; and estimated annual precipitation. The empirical annual runoff model was then used to estimate the total annual runoff to Shell Lake during 1949–2011. The annual phosphorus loads in the runoff were estimated by using the phosphorus concentrations measured during 2009–11.

Surface-Water Inflow

Surface-water inflows were monitored in 2009 and used to refine the annual surface-water inflow to the lake in 2009. Intermittent discharge measurements from six ephemeral streams (fig. 3) that drain areas to the south and east of Shell Lake were made on an approximately monthly basis during

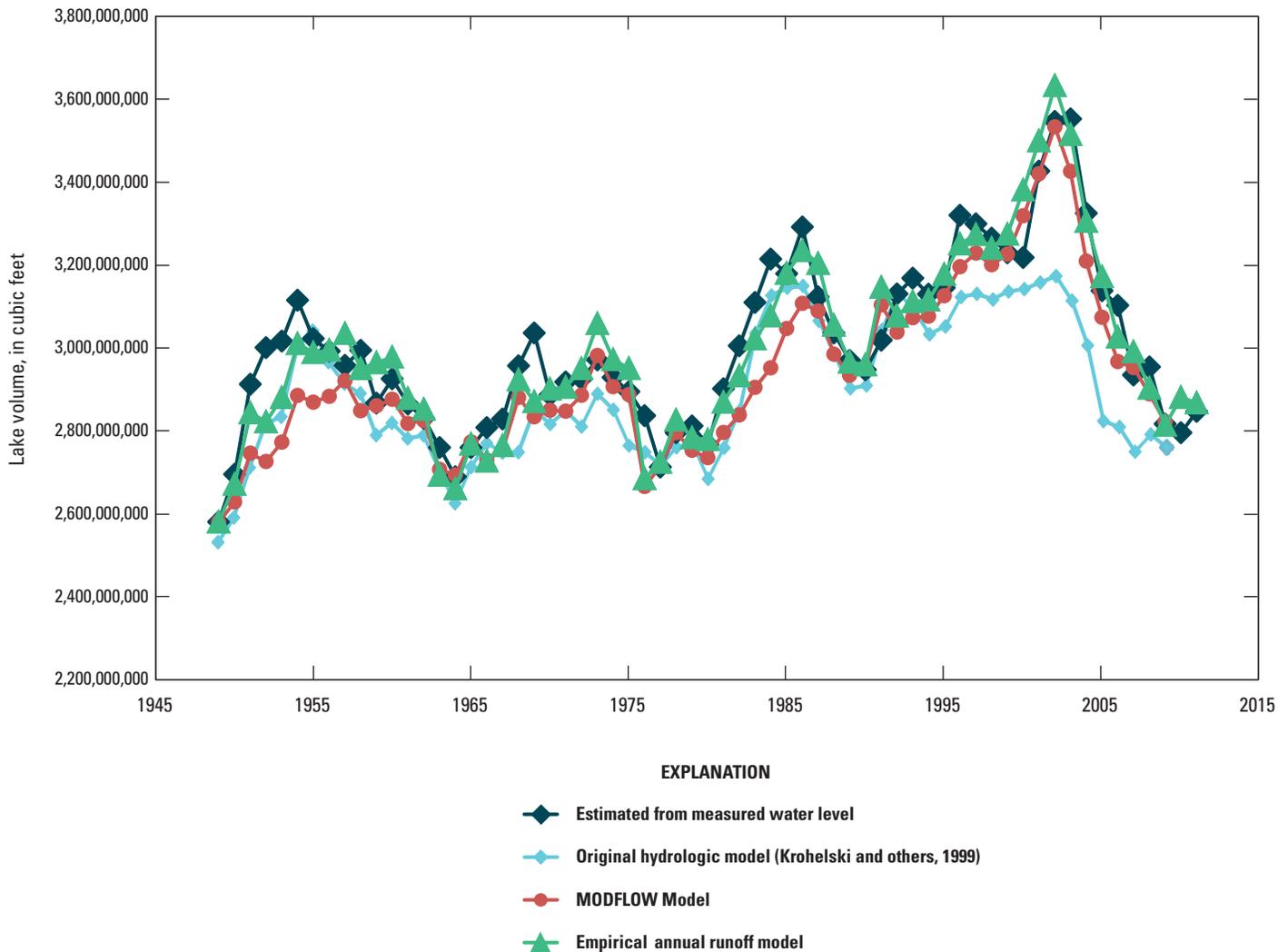


Figure 24. Measured and estimated changes in the volume of Shell Lake, Wisconsin, from three hydrologic models.

the ice-off period from February to November 2009. Flows ranged from 0 to 4.5 ft³/s (fig. 25). On many occasions, no flow was detected in most streams. The highest flows were measured in tributary 6 (diversion ditch) and tributary 5 (Johnson Creek). The intermittent measurements were used to estimate daily streamflows in each stream during 2009 by linearly interpolating between the measured flows. The total annual runoff for 2009 from the monitored tributaries was 31,500,000 ft³ (724 acre-ft, table 8), with the highest flows coming from the tributaries 5 (Johnson Creek, 36 percent) and 6 (diversion ditch, 30 percent).

Inflow (and phosphorus loading) to the lake in 2009 was expected to be much lower than normal because of the very dry conditions in 2009; therefore, the streams were intermittently monitored during 2010–11, when hydrologic conditions were closer to normal. During 2010–11, flows were much higher in the tributaries (fig. 25). The highest flows were measured in tributaries 3, 6 (diversion ditch), and 5 (Johnson Creek).

The three coefficients in equation 6 that best simulated the observed changes in the volume of the lake from 1949 to 2011 (fig. 24) were $\alpha=0.158$, $\beta=9,000,000$, and $\chi=0.4$. The annual water budgets constructed by using the surface-water runoff computed with the empirical runoff model were better (root mean square error, RMSE, reduced by 87 percent) at estimating long-term changes in water level and lake volume than the original hydrologic model by Krohelski and others (1999) and modestly better (RMSE reduced by 35 percent) than the groundwater modeling component of this study (fig. 24). More importantly, the empirical runoff model is expected to better simulate the interannual variability in runoff and ultimately phosphorus loading during a series of wet and dry years. During 2009, the second term ($\beta(AP)^{\chi}$) in equation 6 was equal to 0 because the 4-year moving average annual precipitation was less than 27 in.; therefore, there was no additional runoff to the lake caused by wet conditions in the watershed. Runoff from the unmonitored area did occur in

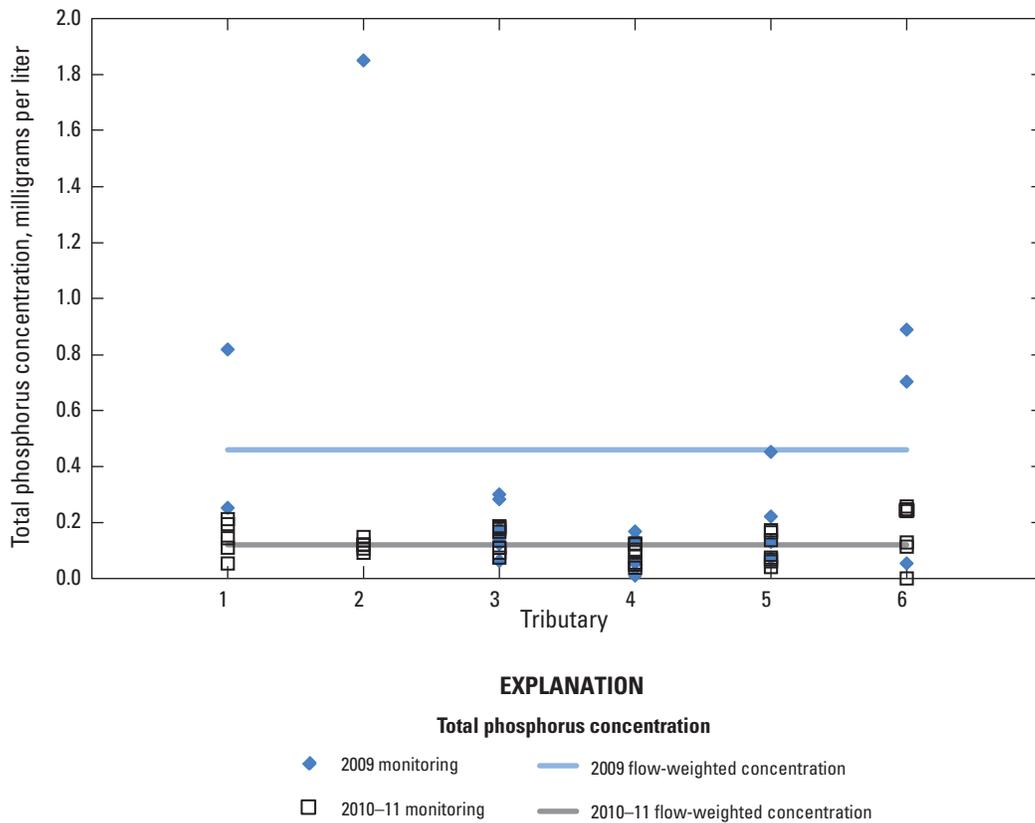
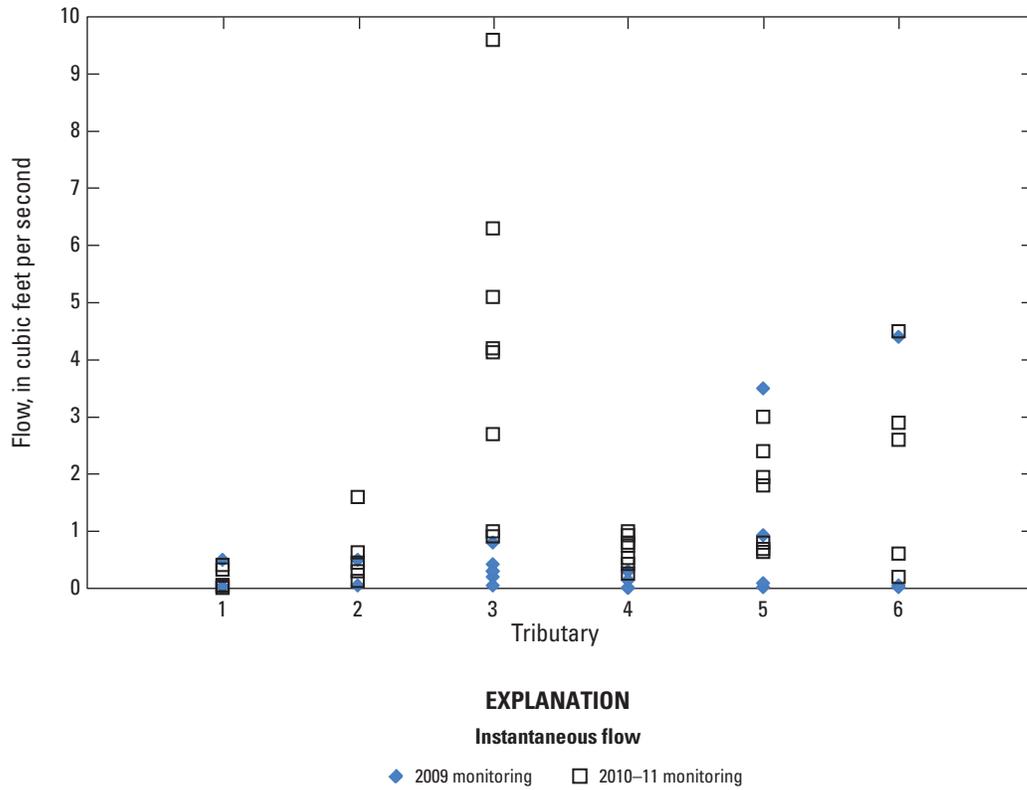


Figure 25. Intermittent flows and phosphorus concentrations measured in six tributaries to Shell Lake, Wisconsin, during 2009–11.

Table 8. Summary of the water-budget components for Shell Lake, Wisconsin, during 2009, 1980, and years when the lake was shallow, moderately deep, and deep.

[ft, feet; acre-ft, acre feet]

Hydrologic component	2009 (acre-ft)	Percent of total input	Average 1949–2011 (acre-ft)	Percent of average input	1980 (WDNR) (acre-ft)	Percent of 1980 (WDNR) input	Shallow lake (<24.8 ft) (acre-ft)	Shallow lake percent of total input	Middle depths (24.8-25.9 ft) (acre-ft)	Middle depths percent of total input	Deep lake (>25.9 ft) (acre-ft)	Deep lake percent of total input
Precipitation	5,076	82.6	6,634	76.1	5,647	58.1	6,006	75.6	6,758	76.1	7,156	76.5
Monitored tributaries	724	11.8	1,726	19.8	2,766	28.5	1,547	19.5	1,727	19.5	1,911	20.4
Unmonitored area	50.1	0.8	0.0 ¹	0.0 ¹	0.0 ¹	0.0	0.0 ¹	0.0 ¹	0.0 ¹	0.0 ¹	0.0 ¹	0.0 ¹
Groundwater	295.3	4.8	361	4.1	1,303	13.4	396.0	5.0	394.4	4.4	288.0	3.1
Total input	6,145	100.0	8,721	100.0	9,716	100.0	7,949	100.0	8,879	100.0	9,355	100.0
Evaporation	5,870	73.2	6,290	74.3	not est.	not est.	6,167	75.5	6,426	75.5	6,269	71.8
Groundwater	2,153	26.8	2,178	25.7	not est.	not est.	2,007	24.5	2,085	24.5	2,461	28.2
Total output	8,024	100.0	8,468	100.0	not est.	not est.	8,174	100.0	8,511	100.0	8,730	100.0

¹Water input from the unmonitored area is included in the values from the monitored tributaries.

2009 and was estimated by the difference in equation 5 and what was estimated from the monitored streams. After calibration, the empirical runoff model (eq. 5) was used to estimate total annual runoff to the lake in 2009 (33,700,000 ft³; 774 acre-ft). Given that the total annual measured runoff from the monitored tributaries was 31,500,000 ft³ (724 acre-ft) in 2009, the difference (2,180,000 ft³ or 50 acre-ft) represented runoff from the unmonitored area; that is, the unmonitored surface water inflow represented about 6.5 percent of the total runoff to the lake in 2009.

Water-Budget Summary

A final water budget for Shell Lake was constructed for 2009 by using precipitation from Spooner, Wis., surface-water inflow from the empirical annual runoff model (eq. 6), and evaporation, groundwater input, and groundwater output from the groundwater component of this study (table 8). During 2009, precipitation was the largest source of water to Shell Lake, supplying 82.6 percent of the total water input (fig. 26). Other sources were surface-water inflow (11.8 percent from the monitored tributaries and 0.8 percent from the unmonitored area), and groundwater (4.8 percent). The total input of water to the lake in 2009 was 268,000,000 ft³ (6,150 acre-ft, or 28.3 in.). The residence time—that is, the length of time required for water entering the lake to completely replace the volume of water in the lake—was 10.5 years, computed on the basis of the total inflow for 2009.

During 2009, the total annual output of water from the lake was more than the total input, resulting in a decrease in the volume of lake water storage from the beginning to the end of the 2009 study year and a drop in the lake water level of about 8.7 in. About 73 percent of the total water leaving the lake left through evaporation, and about 27 percent left through groundwater outflow (fig. 26; table 8). The total output of water from the lake in 2009 was 349,000,000 ft³ (8,020 acre-ft, or 36.9 in.). The residence time was 8.1 years, computed on the basis of the total outflow.

The quality or accuracy of the water budget was evaluated by comparing the observed decrease in water level (8.7 in.) with the change in water level estimated from the change in storage computed with the water budget (decrease in volume of 1,870 acre-ft). If it is assumed that the average area of the lake in 2009 was 2,611 acres, then a decrease in volume of 1,870 acre-ft equates to a decrease of about 8.5 in. On an annual basis, the percentage error was about 0.2 percent.

The hydrologic conditions documented in 2009 represent conditions following a series of dry years. To determine how the hydrologic loading to the lake changes under a wider range of hydrologic conditions, each of the components in the budget was estimated from 1949 to 2011 by using precipitation from Spooner, Wis., surface-water inflow from the empirical annual runoff model (eqs. 5 and 6), and evaporation, groundwater input, and groundwater output from the groundwater component of this study. During 2010 and 2011, groundwater

inputs and outputs were assumed to contribute 1.41 in. (307 acre-ft) and 10.31 in. (2,240 acre-ft), respectively, to the lake, similar to 2009. Since 1949, the average total annual water input to the lake has been 8,720 acre-ft but has varied from 4,220 acre-ft in 1976 to 13,000 acre-ft in 1991 (fig. 27). These changes in inputs resulted in an average residence time of 8.2 years but a range of 6.6 to 10.9 years. (All residence times were derived from the average of residence time based on total inflows and the residence time based on total outflows). Precipitation was always the major contributor of water to the lake. From 1949 through 2011, the input from precipitation averaged 6,630 acre-ft but varied from 3,290 acre-ft in 1976 to 10,100 acre-ft in 1991 (fig. 27; table 8). During the same period, the input from surface-water runoff averaged 1,730 acre-ft but varied from 520 to 2,870 acre-ft. During most years, the runoff model indicated that additional runoff was contributed because of high soil moisture and water levels in the wetlands that increase the flow in the tributaries; however, in dry years similar to 2009, there was little if any additional runoff. The input from groundwater averaged 361 acre-ft but varied from 159 to 489 acre-ft.

The computed water budget from this study indicates less total water moving through the lake than estimated in previous studies. In this study, the total annual inflow to Shell Lake in 2009 was 268,000,000 ft³ (6,150 acre-ft, or 28.3 in.). A Wisconsin Department of Natural Resources (1982) study of Shell Lake estimated that the average annual inflow to Shell Lake was about 4.23×10^8 ft³ (9,720 acre-ft, or 45.0 in.), with precipitation, surface-water runoff, and groundwater inflow accounting for 58, 29, and 13 percent of the total inflow to Shell Lake, respectively (table 8). Differences from the estimated values between the 1982 study and this report largely reflect the methods used to estimate groundwater and surface-water inflows. The 1982 study computed groundwater inflow by using permeability estimates and measured hydraulic gradients in wells surrounding Shell Lake, with no water budget or chemistry checks; surface-water inflows were estimated from previous regional watershed-scale analyses of total stream-flow per square mile (Young and Hindall, 1973). The study by Krohelski and others (1999) estimated that the average annual inflow to Shell Lake was about 4.17×10^8 ft³ (9,570 acre-ft, or 44.3 in.), with precipitation, surface-water runoff, and groundwater inflow accounting for 64, 36, and 0 percent of the total inflow to Shell Lake. Krohelski and others also estimated lake outflows, of which evaporation and discharge to groundwater accounted for 60 and 40 percent, respectively. Methods by Krohelski and others (1999) did not simulate the physics of groundwater flow and used the assumption that groundwater inflow was negligible, based on nearshore hydraulic gradients. Overland runoff was estimated by using seasonal runoff coefficients and seasonal precipitation. Although the previous study results are useful for comparison purposes in that they also identify precipitation and evaporation as the primary drivers of the Shell Lake water budget, results from the current study are expected to be more accurate because the models incorporate physics of the large-scale system and because the results produce reasonable estimates of in-lake water chemistry

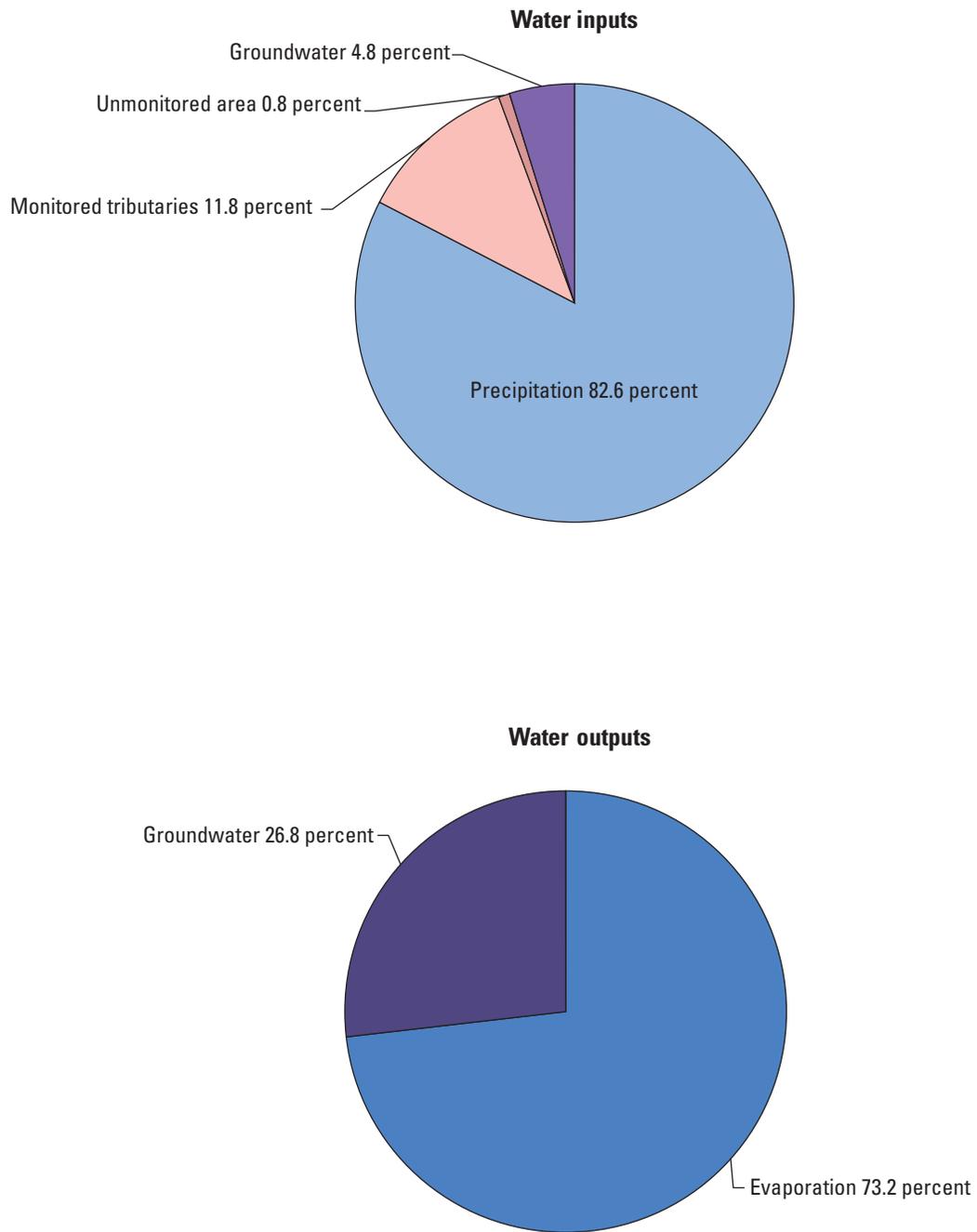


Figure 26. The water budget for Shell Lake, Wisconsin, for 2009.

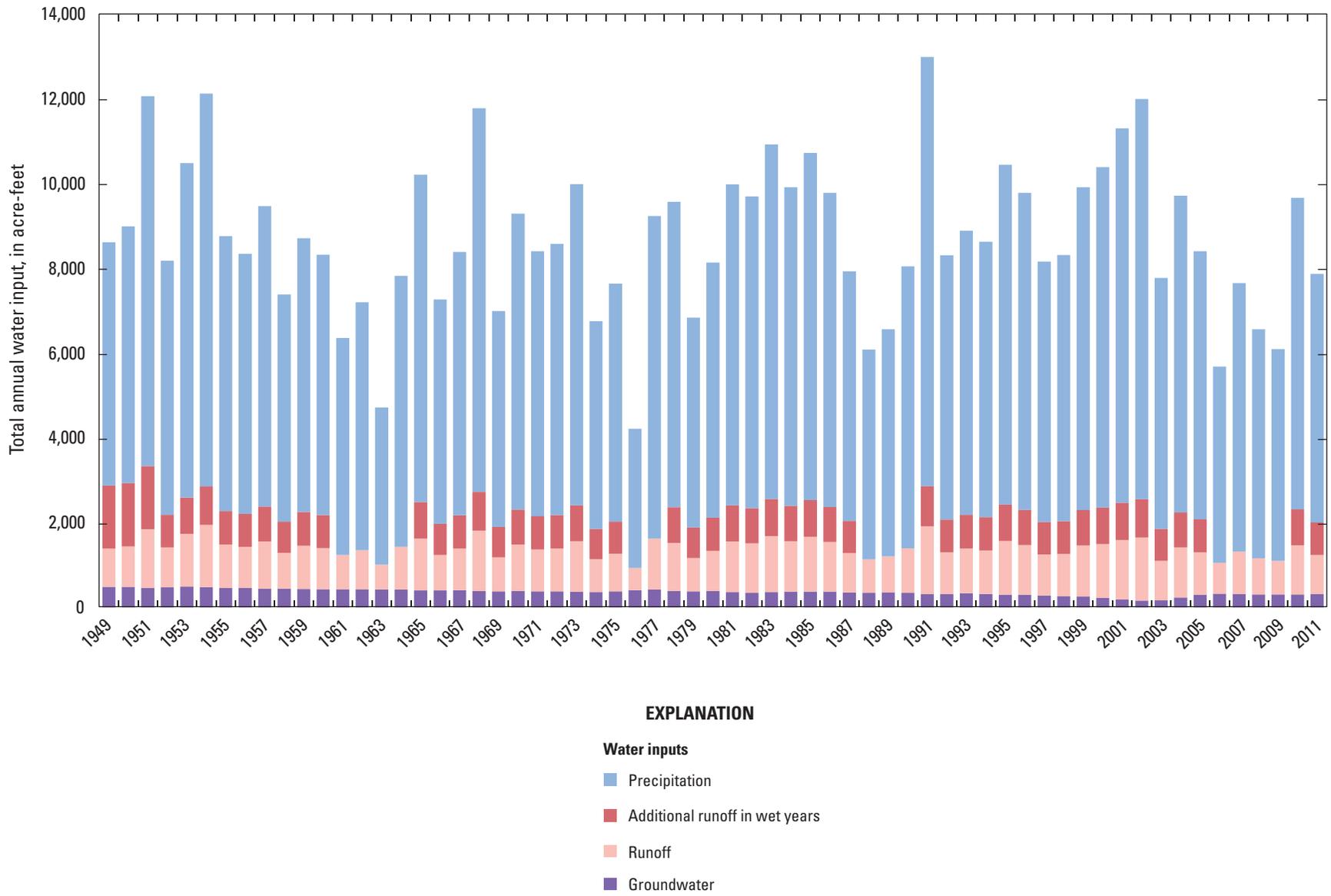


Figure 27. The water budget for Shell Lake, Wisconsin, from 1949 to 2011.

results produce reasonable estimates of in-lake water chemistry (Ca/Mg and phosphorus). Although 2007–9 was drier than average (25.34 in. of precipitation compared with the 1949–2011 average of 29.71 in.), inputs from the groundwater model and the empirical runoff model incorporate the interannual hydrologic variability and estimate the average inflow to the lake during 1949–2011 to be less than that computed by the Wisconsin Department of Natural Resources (1982) and Krohelski and others (1999).

Sources of Phosphorus

To help define where the phosphorus in Shell Lake originates and how much may be controllable, a detailed phosphorus budget was constructed for 2009 (fig. 28). This information was then used to estimate annual phosphorus budgets from 1949 to 2011 (fig. 29). External sources of phosphorus to the lake include precipitation, surface-water inflow, groundwater inflow, and contributions from septic systems. In addition to these external sources, phosphorus can be released from the bottom sediments of the lake, which is considered an internal source of phosphorus. Phosphorus is removed from the lake primarily through deposition to the lake sediments, although a small amount may be removed with water leaving the lake through the groundwater system.

Precipitation

Most water input into Shell Lake comes from precipitation on the lake surface (fig. 26); however, phosphorus deposition from the atmosphere is delivered from both precipitation (wetfall) and dryfall. Input from wetfall was determined by using phosphorus concentrations measured in rain and snow. Input from dryfall (dry particles from the atmosphere) was estimated from measured rates of dry deposition at a weather station previously operated at Whitefish Lake in Douglas County, Wis. (Robertson, Rose, and Juckem, 2009). Total annual wetfall was computed by multiplying the average estimated phosphorus concentration of 0.016 mg/L by the total annual precipitation (from Spooner, Wis.) on the lake. Phosphorus in dryfall was estimated from the deposition rate (26.1 pounds per square mile per year (lb/mi²/yr)) measured at Whitefish Lake, which was adjusted for lakes with few conifers in the surrounding area. Total phosphorus input to the lake from wetfall and dryfall, which hereafter we refer to as simply “precipitation,” was 327 lb (with about 68 percent coming from wetfall) in 2009 (table 9 and fig. 28).

Surface-Water Inflow

Phosphorus loading from surface-water inflow was subdivided into that contributed by the six intermittently monitored tributaries and by the unmonitored areas, the latter of which includes runoff from residential shoreline development.

Monitored Tributaries

Streamflow measurements and phosphorus concentration samples were collected in the six tributaries to Shell Lake (fig. 3) approximately monthly and shortly after occasional rainfall events during the ice-off period in 2009. On many occasions, no flow was detected and therefore no samples were collected for total phosphorus analysis. During 2009, phosphorus concentrations in the streams varied from 0.01 mg/L in tributary 4 in April to 1.85 mg/L in tributary 2 (Butler Creek) in March (fig. 25). The flow-weighted concentration of all tributaries for 2009, on the basis of only the measured concentrations and instantaneous measured flows, was 0.46 mg/L. Average flow-weighted concentrations were at or below 0.7 mg/L in all tributaries except tributary 2 (2.31 mg/L), which contains more agricultural area in its watershed than other tributaries. Daily phosphorus loads from each tributary in 2009 were estimated by multiplying their flow-weighted concentration by their estimated total annual flow (fig. 28). Loadings from each of the monitored tributaries, in descending order, were the following: tributary 6 (diversion ditch), 417 lb (0.31 pound per acre (lb/acre)); tributary 5, 186 lb (0.14 lb/acre); tributary 2, 182 lb (0.24 lb/acre); tributary 3, 53 lb (0.07 lb/acre); tributary 1, 21 lb (0.03 lb/acre); and tributary 4, 6 lb (0.01 lb/acre). Collectively, the six tributaries contributed 867 lb (0.15 lb/acre) of phosphorus in 2009.

Unmonitored Areas

It was assumed the phosphorus concentration of the unmonitored area, including the runoff from residential shoreline development, was equal to the average of the flow-weighted concentrations of all of the monitored tributaries (0.30 mg/L) except tributary 2, which was much different than the other tributaries. The volume of inflow from the unmonitored area was computed by using the empirical runoff model (see “Hydrology and Water Budget”). The total phosphorus load from the unmonitored areas was then computed by multiplying the total flow from this area by 0.30 mg/L. During 2009, the total phosphorus load from the unmonitored near-lake drainage area was 40.2 lb (table 9 and fig. 28). In this approach, it was assumed that phosphorus concentration in the runoff from the residential shoreline area was similar to that of the areas upstream of the five monitored tributaries from which the 0.30-mg/L average concentration was obtained. If phosphorus concentrations for the unmonitored area could be refined, a better estimate of phosphorus loading could be obtained. Such monitoring was beyond the scope of this study, but any difference from a refined estimate would be expected to represent a modest percentage of the total phosphorus load (fig. 28).

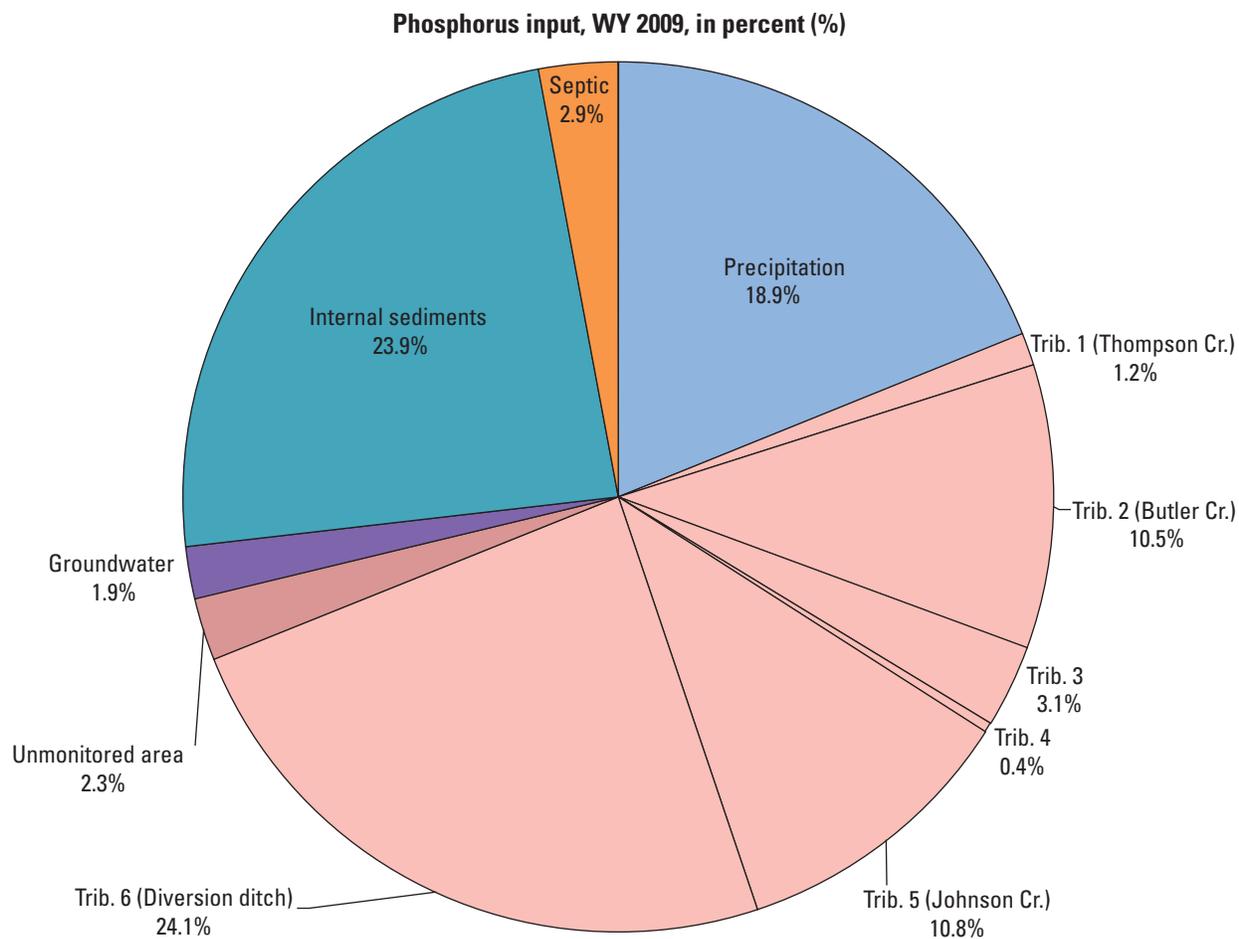


Figure 28. The phosphorus budget for Shell Lake, Wisconsin, for 2009.

Table 9. Summary of the phosphorus-budget components for Shell Lake, Wisconsin, for 2009, 1980, and years when the lake was shallow, moderately deep, and deep.

Budget component	2009 (pounds)	Percent of total input	1980 (this study) (pounds)	1980 (WDNR) (pounds)	Shallow lake (<25.2 feet) (pounds)	Middle depths (25.2-26.1 feet) (pounds)	Deep lake (>26.1 feet) (pounds)
Monitored tributaries	327	18.9	368	653	367	403	424
Unmonitored area	40.2	2.3	0.0 ¹	0.0 ¹	0.0 ¹	0.0 ¹	0.0 ¹
Groundwater	33.7	1.9	43.7	185.2	45.2	45.0	32.9
Septic systems	51	2.9	51.0	81.6	51.0	51.0	51.0
Total external input	1,319	76.1	1,253	1,783	1,181	1,235	1,167
Internal (sediment) loading	414	23.9	414 ²	414 ²	414 ²	414 ²	414 ²
Total external and internal input	1,733	100	1,666	2,197	1,594	1,649	1,581

¹Phosphorus loading from the unmonitored areas are included in the values from the monitored tributaries.

²Phosphorus input from internal loading was assumed to be similar to that estimated in this study.

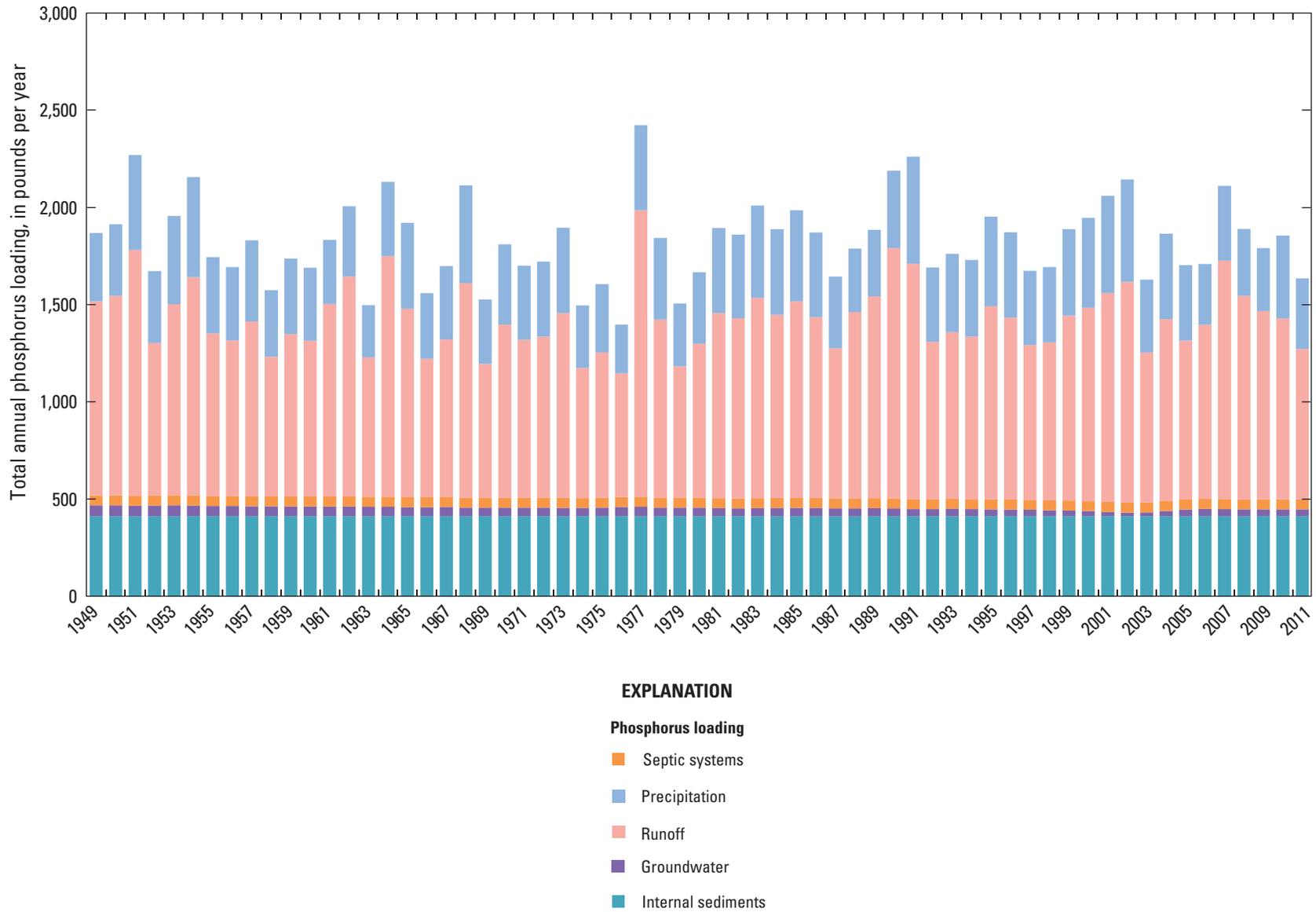


Figure 29. The phosphorus budget for Shell Lake, Wisconsin, 1949–2011.

Groundwater

Loadings of phosphorus from natural groundwater inflow and from contributions from septic inputs were estimated separately. To determine the natural groundwater contributions, dissolved phosphorus concentrations were measured in samples from the piezometers/wells installed around the lakeshore (fig. 3). As evidenced by the groundwater model results and water-elevation gradients between the piezometers/wells and water level of the lake, groundwater enters the lake primarily along the southern and eastern shorelines (fig. 13), represented by piezometers 1–6 and wells 20S and 30S (fig. 3). The average dissolved phosphorus concentrations in piezometers/wells in the areas draining to the lake ranged from 0.018 mg/L in piezometers 5 and 6 to 0.587 mg/L in piezometer 3. The average concentration of all samples in the area draining to the lake was 0.445 mg/L. Phosphorus concentrations at some sites, such as piezometers 2, 3, and 4, may be elevated above background concentrations and potentially affected by septic effluent. Therefore, to estimate the phosphorus input to the lake from natural groundwater inflow, it was assumed that the concentration of groundwater entering the lake was equal to the average concentration measured in piezometers 1, 5, and 6, and wells 20S and 30S, which was 0.042 mg/L. During 2009, the annual phosphorus load contributed by natural groundwater inflow was estimated to be 33.7 lb (table 9 and fig. 28).

Septic Systems

Phosphorus loading from near-lake septic systems was estimated separately from background groundwater inflow by applying export coefficients to the near-lake septic systems around Shell Lake. The site-specific export coefficients were estimated on the basis of soil and septic-system characteristics around Shell Lake. The number of septic systems was based on the latest population and occupancy information.

The input of phosphorus from septic systems (M) was estimated by use of equation 7 (Reckhow and others, 1980):

$$M = E_s * (\text{number of capita years}) * (1 - S_R), \quad (7)$$

where M is a function of a phosphorus export coefficient; E_s , a soil retention coefficient, S_R ; and the number of people using the septic systems annually (capita years). This method of estimating phosphorus loading from septic systems is described in more detail by Garn and others (1996). In applying equation 7, it was assumed that the most likely value for E_s was 1.5 lb of phosphorus per capita per year. Typical export coefficients range from about 1 lb per capita per year (Reckhow and others, 1980; Panuska and Kreider, 2003) to more than 2 lb per capita per year (Garn and others, 1996). Soils around Shell Lake consist of glacial deposits that are a mixture of drift, outwash, and lake sediment near the Spooner Hills (Clayton, 1984; Johnson, 1986, 2000). Therefore, an S_R value of 0.5 was used for these soils, plus an S_R value of 0.2 was added for the fraction of the phosphorus that is removed by storage of solids in the septic

tank. Therefore, a most likely S_R coefficient of 0.70 (70-percent retention) was used in septic-system input calculations.

In 2009, not all households and facilities around the lake generated septic discharge to the lake because part of the shoreline was served by the sanitary sewer and many residences were a long distance from the lake. It was estimated that 283 homes around the lake were not on the sewer system (David Vold, City of Shell Lake, oral commun., 2011), of which 40 were permanent residences and 243 were seasonal residences (occupied 3 months of the year) and 137 of which (48.4 percent) had holding tanks and no leach field and therefore did not release phosphorus. Of these 283 homes, 199 homes were in the area potentially contributing groundwater to the lake (fig. 13). Therefore, for the shoreline that is not serviced by the sanitary sewer, if it is assumed that the type of residences and type of septic systems (holding tank or leach field) were randomly distributed around the lake, then the area contributing groundwater to the lake should have 15 permanent and 88 seasonal residences with leach fields that release phosphorus to soils around Shell Lake. The average permanent household size around Shell Lake was estimated at 2.5 persons and average seasonal household size at 3.5 persons (David Vold, City of Shell Lake, oral commun., 2011). The number of capita years (113 capita years) was then calculated by multiplying the number of residences (15 permanent and 88 seasonal) by the number of persons per household (either 2.5 or 3.5 persons) by the fraction of the year occupied (either 1.0 or 0.25). Therefore, based on application of equation 7, the annual phosphorus load contributed by septic inputs in 2009 was about 51 lb (table 9 and fig. 28).

Release of Phosphorus from Lake Sediments

Three types of information are needed to estimate how much phosphorus may be released from the bottom sediments of the lake: the area from which phosphorus is being released, the aerobic or anaerobic characteristics of the sediment surface, and the rate of phosphorus release. It was assumed that the phosphorus release rate was only a function of whether the sediments were aerobic or anaerobic and that the rate was constant throughout the growing season. The amount of phosphorus released before May 1 and after September 30 is expected to be very low, because sediment release rates typically decrease when the water temperature is cool during fall, winter, and spring (James and Barko, 2004). During 2009, however, the biweekly measurements and the detailed thermistor information indicated that most, if not all, of the sediment surface area did not go anaerobic. It was also assumed that phosphorus was released only from areas greater than 6.5 ft deep because these areas typically accumulate organic sediment.

Rates of phosphorus release from the aerobic and anaerobic sediments in Shell Lake were estimated from 18 sediment cores collected from six different sites in the lake (fig. 3) and analyzed by the Wisconsin State Laboratory of Hygiene. Daily aerobic-release rates ranged from 0 lb/ft² (no net release) to 5.5×10^{-8} lb/ft², with an average aerobic release rate of

2.6×10^{-8} lb/ft² (fig. 30). Daily anaerobic-release rates ranged from 5.9×10^{-8} to 1.9×10^{-7} lb/ft², with an average anaerobic release rate of 1.3×10^{-7} lb/ft². The anaerobic release rate was about 5 times greater than the aerobic release rate.

All of the phosphorus was released from aerobic sediments from May 1 to September 30 because anaerobic conditions were not measured in 2009. Therefore, the annual input of phosphorus from internal loading from the sediments of the lake is estimated to be 414 lb (table 9 and fig. 28). This phosphorus is intermittently released to the surface of the lake during deep mixing events (fig. 17). Much more phosphorus would be released from the sediments if the period of stratification lasted long enough such that the bottom sediments became anaerobic.

Phosphorus-Budget Summary

2009 Phosphorus Loading

During 2009, the total annual input of phosphorus to Shell Lake was 1,730 lb (table 9 and fig. 28), of which 1,320 lb came from external sources (76.1 percent) and 414 lb came from internal loading from the sediments (23.9 percent). The largest external source of phosphorus was from surface-water runoff, which delivered 907 lb (867 lb from monitored tributaries and 40 lb from unmonitored areas) or about 52 percent of the total phosphorus load (fig. 28) compared with about 13 percent of the water input (fig. 26). The annual phosphorus yield from the Shell Lake watershed was 69.2 lb/mi² in 2009. This yield is higher than that measured from the relatively pristine Bear River watershed near Manitowish Waters, Wis., also in the Northern Lakes and Forests Ecoregion (24 lb/mi²; Corsi and others, 1997). The second largest source was precipitation, which delivered about 19 percent of the total phosphorus load compared to about 83 percent of the water input. Contributions from septic systems and groundwater accounted for about 3 and 2 percent of the total phosphorus load, respectively.

1949-2011 Phosphorus Loading

Phosphorus loading in 2009 represents what occurred after a series of dry years. To determine how phosphorus loading to the lake varies during a range in hydrologic conditions, loading from each component of the phosphorus budget was estimated from 1949 to 2011 on the basis of annual varying precipitation measured in Spooner, Wis., annual varying groundwater input from the groundwater model, and annual varying surface-water inflow from the empirical runoff model. Annual phosphorus inputs from precipitation were estimated by using similar concentrations in wetfall and dryfall rates as used in 2009. Annual phosphorus inputs from groundwater were estimated by using similar phosphorus concentrations as used in 2009. No adjustments were made to inputs from septic systems and internal loading. Internal loading might be expected to increase during periods of high water levels with potentially longer stratification; therefore, inputs from this source may be underestimated.

Phosphorus concentrations in surface-water runoff were expected to be different during wetter conditions from that measured during the 2009 dry conditions; therefore, the concentrations in the runoff in other years were estimated by using the phosphorus concentrations measured during the entire period from 2009 to 2011 (fig. 25). Typically, phosphorus concentrations are expected to increase during years with more precipitation because of increased runoff and associated higher erosion; however, the average phosphorus concentrations during 2010–11 were lower than those measured in 2009 in every tributary (fig. 25). Average flow-weighted concentrations were fairly similar among streams, ranging from 0.07 to 0.15 mg/L. The flow-weighted concentration of all six tributaries during 2010–11 was 0.12 mg/L. The lower concentrations in the tributaries during wet years may have been caused by an increased flow rate through upstream wetlands. It is hypothesized that the slower the water moves through the wetlands, the higher the rate of phosphorus release from the sediments of the wetlands will be because of higher decomposition rates caused by higher water temperatures. This, in combination with the longer residence time in the wetlands that results in more time for phosphorus to be transferred from the bottom sediments, results in higher phosphorus concentrations measured in dry years.

Annual phosphorus loads from surface-water runoff were computed by dividing the flow from the empirical runoff model into three components: base runoff from the six monitored tributaries (93.5 percent of the base runoff estimated for 2009; eq. 5), base runoff from the unmonitored nearshore areas (6.5 percent of the base runoff; eq. 5), and additional runoff occurring during wet years (the second term in eq. 6). Annual phosphorus loads in base runoff from the six monitored tributaries were computed by multiplying their estimated daily flow (eq. 5) by 0.46 mg/L (the flow-weighted concentration of all six tributaries measured during 2009). Annual phosphorus loads in base runoff from the unmonitored nearshore area was computed by multiplying its flow (eq. 5) by 0.30 mg/L (the flow-weighted concentration of the monitored streams, excluding tributary 2, measured during 2009). Annual phosphorus load from the additional runoff occurring during wet years (second term in eq. 6) was computed by multiplying its estimated flow by 0.12 mg/L (the flow-weighted concentration of all six tributaries during 2010–11).

Since 1949, the average annual total phosphorus loading to the lake has been 1,830 lb, but annual loads have varied from 1,400 lb in 1976 to 2,420 lb in 1977 (fig. 29). Input from surface-water runoff was always the major contributor of phosphorus to the lake. Since 1949, the average input from surface-water runoff has been 926 lb, but annual input has varied from 636 to 1,470 lb. The average annual yield from the Shell Lake watershed was 70.7 lb/mi². In all years, the additional loading occurring during wet years was only a small part of the loading from runoff because although the flow was relatively high, the phosphorus concentrations were estimated to be relatively low. Phosphorus loading from the lake sediments was usually the second largest contributor. Average annual phosphorus loading from precipitation was 398 lb but varied annually from 250 lb to 550 lb.

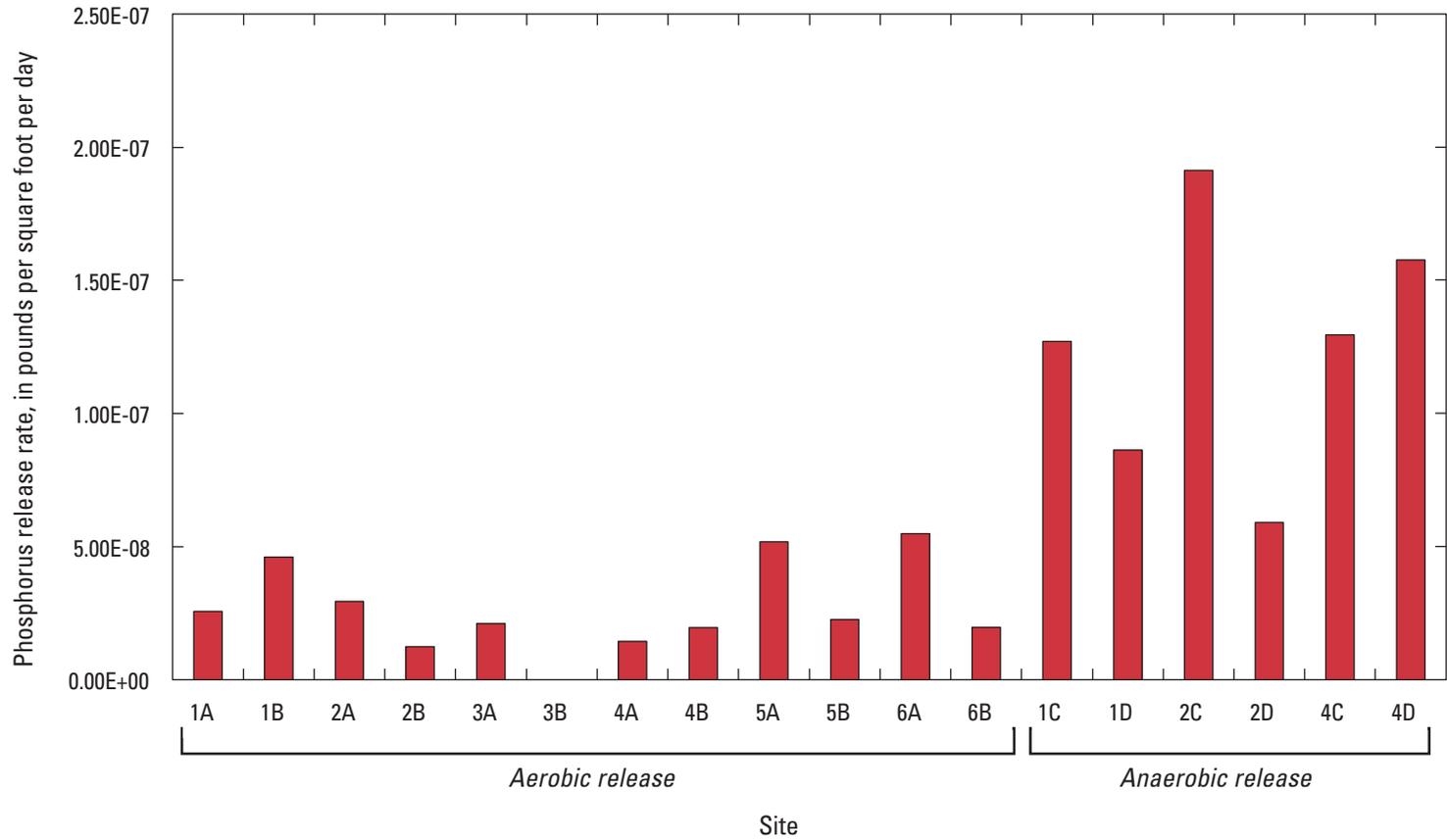


Figure 30. Phosphorus release rates from the sediments of Shell Lake, Wisconsin, 2009. Sites with an A or B represent estimated aerobic release rates, whereas sites with a C or D (1, 2, and 4) represent estimated anaerobic release rates. Location of sites are shown on figure 3.

Comparison With Estimated 1980 Phosphorus Loading

The first detailed water and phosphorus budgets for Shell Lake were constructed by WDNR for 1980 (Wisconsin Department of Natural Resources, 1982). WDNR estimated that the total annual phosphorus load was about 1,780 lb; of this total, precipitation amounted to 653 lb; total surface-water runoff from the entire watershed, 864 lb; groundwater, 185 lb; and septic systems, 82 lb. If the internal loading of 414 lb estimated in this current study is added to the total estimated by WDNR (1982), then the total phosphorus load would have been 2,220 lb. In comparison, the total annual phosphorus load estimated for 1980 as part of the current study (fig. 29) was 1,670 lb: precipitation amounted to 368 lb; surface-water runoff, 700 lb; groundwater, 44 lb; septic systems, 51 lb; and internal loading, 414 lb. The largest differences in phosphorus load estimates were for precipitation, which was estimated by the WDNR to be about twice that estimated in this study because of a higher estimated phosphorus concentration in precipitation, and groundwater, which was estimated to be about 4 times that estimated in this study because of the larger amount of groundwater discharge to the lake computed with the simpler gradient flux calculations.

Simulated Water Quality and Response to Changes in Phosphorus Loading

Empirical eutrophication models that relate phosphorus loading to specific water-quality characteristics can be used to determine how natural (weather-driven) and anthropogenic changes in phosphorus loading could modify the water quality of Shell Lake. These models were developed on the basis of comparisons between hydrologic and phosphorus loading determined for many different lake systems and specific measures describing their water quality, such as near-surface phosphorus and chlorophyll *a* concentrations and Secchi depth (Vollenweider, 1975; Canfield and Bachmann, 1981; Reckhow and Chapra, 1983; Cooke and others, 1993; Panuska and Kreider, 2003). One empirical model that has been shown to be good at predicting phosphorus concentrations in lakes of Wisconsin is the Canfield and Bachman (1981) natural-lake model (Robertson and others, 2002, 2005; Robertson and Rose, 2008; Robertson, Rose, and Juckem, 2009; Robertson, Rose, and Fitzpatrick, 2009; Robertson and others, 2012). Therefore, the Canfield and Bachman natural-lake model (1981; eq. 8) was applied to Shell Lake to determine how changes in phosphorus loading are expected to affect phosphorus concentrations in the lake:

$$\frac{\text{Total Phosphorus Concentration (mg/L)}}{L/1,000} = \frac{L/1,000}{Z(0.162 \times (L/Z)^{0.458} + 1/\tau)} \quad (8)$$

where L is the annual phosphorus-loading rate (in milligrams per square meter, or pounds per acre $\times 112.07$), Z is the mean depth of the lake (in meters, or feet $\times 0.305$), and

τ is the residence time of the water in the lake (in years). In this application, hydrology and phosphorus loadings for 2009 (tables 8 and 9) were used as inputs and represented the base case (referred to as “reference” hereafter) for comparisons with other basinwide changes in loadings.

Total phosphorus concentrations predicted with the Canfield and Bachman model were then used to predict chlorophyll *a* concentrations and Secchi depths through the use of Carlson’s (1977) TSI equations (eqs. 1–3). In other words, the chlorophyll *a* concentrations and Secchi depths were computed such that a similar TSI value as that computed from the predicted total phosphorus concentration was obtained. The total phosphorus concentrations were also used to predict chlorophyll *a* concentrations through the use of a relation found by Hickman (eq. 9; 1980):

$$\log(\text{Chl } a, \mu\text{g/L}) = 0.542 + 3.27 * \log(\log(\text{total phosphorus, mg/L} * 1,000)) \quad (9)$$

There are no true calibration factors when Carlson’s TSI or Hickman’s equations are used to estimate chlorophyll *a* concentrations and Secchi depths in this manner; therefore, the output was adjusted to account for model biases by adjusting the results by the percentage of bias in the results for the base-case (reference) scenario.

Modeling Approach

To estimate how the water quality of Shell Lake should respond to changes in phosphorus loading, 17 scenarios were simulated with the Canfield and Bachman model, the Carlson TSI equations, and the Hickman equation (table 10 and fig. 31). Scenario 1 simulated the growing-season-average conditions for 2009. Eight simulations (2–9) were used to determine the general response of the lake to basinwide changes in external phosphorus loading: simulated decreases in external phosphorus loading by 75, 50, and 25 percent and increases in external phosphorus loading by 25, 50, 75, 100, and 200 percent. External loading excludes phosphorus loading from the sediment. Internal phosphorus loading from the sediments is not expected to change quickly in response to changes in external loading and, therefore, was not modified in scenarios 2–9. Scenario 1 established a base condition (reference concentrations) to which results from the other changes can be compared.

Eight scenarios (10–17) were then used to determine how changes in phosphorus loading due to changes in natural (hydrologic) and anthropogenic environmental conditions could have modified water quality of the lake in the past or could modify it in the future. Scenario 10 was used to quantify the effects of internal phosphorus loading from the sediments by removing the phosphorus load estimated to come from the sediments. Scenario 11 was used to quantify the effects of inputs from septic systems by removing the phosphorus load from septic systems. Three scenarios (12–14) were used to simulate the response of the lake to estimated phosphorus

loading in 1980 from three methods: (1) as estimated by the WDNR, (2) as estimated by the WDNR but with internal loading computed by this study added to the WDNR estimated loads in order to account for all phosphorus sources, and (3) estimated by this study. In scenarios 15–17, responses of the lake to the phosphorus inputs and changes in lake morphometry were evaluated for hypothetical lake levels representing relatively shallow conditions (mean depth less than 25.2 ft), moderate conditions (mean depth between 25.2 and 26.1 ft), or deep conditions (mean depth greater than 26.1 ft).

Four types of data are required as input into the Canfield and Bachman model: water-quality data, morphometric data (mean depth and surface area), hydrologic data (residence time), and nutrient-loading data (table 10). The loading data used as input to this model is for an entire year, but the model simulates water quality only for the growing season (May–September).

Verification and Adjustments for Model Biases

The ability of the Canfield and Bachman (1981) model to simulate the changes in phosphorus concentrations in the lake (termed “in-lake concentrations”) was evaluated by inputting the morphometry of the lake and the hydrologic and phosphorus loading for 2009 into the model (eq. 8) and simulating the growing-season-average near-surface total phosphorus concentration. The simulated total phosphorus concentration for 2009 (0.017 mg/L; fig. 31A) is similar to that measured in the lake (0.018 mg/L; table 7). This similarity indicates that the model is expected to accurately simulate the growing-season-average phosphorus concentrations in Shell Lake in response to changes in phosphorus loading. Because of the small discrepancy in measured and simulated phosphorus concentrations in 2009, all of the simulated phosphorus concentrations were increased by 2 percent.

Determining how well Carlson (1977) TSI equations simulate chlorophyll *a* concentrations and Secchi depths in Shell Lake was accomplished by comparing the measured values with those estimated by using equations 2 and 3 with the TSI value based on the measured total phosphorus concentration. Given a TSI value of 45.5 (based on the measured phosphorus concentration of 0.018 mg/L), the unadjusted predicted chlorophyll *a* concentration was 4.6 $\mu\text{g/L}$ and the unadjusted predicted Secchi depth was 9.0 ft (fig. 31B and C). The measured chlorophyll *a* concentration in 2009 was 8.2 $\mu\text{g/L}$, which is more than that predicted with equation 2. The measured Secchi depth in 2009 was 11.1 ft, which is greater than that predicted with equation 3. The discrepancy in chlorophyll *a* concentrations and Secchi depths indicates that measured chlorophyll *a* concentrations are higher than expected, and the clarity measured in Shell Lake is better than would be expected given the measured phosphorus concentrations. These discrepancies may have been caused by specific characteristics in Shell Lake that differ from other lakes with similar phosphorus concentrations, such as the scarcity of macrophytes in the lake, or different structure of its food web.

Because of the relatively large discrepancies in measured and simulated chlorophyll *a* concentrations and Secchi depths, all of the simulated chlorophyll *a* concentrations from equation 2 were increased by 79 percent and the Secchi-depth values from equation 3 were increased by 24 percent.

Determining how well the Hickman (1980) equation simulates chlorophyll *a* concentrations was accomplished by comparing the average measured chlorophyll *a* concentration with that estimated with equation 9 when the measured total phosphorus concentration in the lake was used. The unadjusted predicted chlorophyll *a* concentration was 7.1 $\mu\text{g/L}$ (fig. 31B). The measured chlorophyll *a* concentration in 2009 was 8.2 $\mu\text{g/L}$, which is more than that predicted with equation 9, but much closer to the measured value than estimated from the TSI value based on the phosphorus concentration. Because of the discrepancies in measured and simulated chlorophyll *a* concentrations, all of the simulated chlorophyll *a* concentrations from equation 9 were increased by 15 percent.

Response in Water Quality to Basinwide Changes in Phosphorus Loading

On the basis of model simulations for basinwide changes in external phosphorus loading (scenarios 2–9; table 10), in-lake phosphorus concentrations should have a relatively linear response to a linear change in loading (fig. 31A); however, on a percentage basis, the changes in in-lake phosphorus concentrations are smaller than the changes in external phosphorus loadings. For example, a 50-percent change in external phosphorus loading should cause about a 22- to 26-percent change in in-lake phosphorus concentrations.

Based on Carlson TSI equations, simulated growing-season-average chlorophyll *a* concentrations had a relatively linear response to changes in external phosphorus loading (fig. 31B); however, on a percentage basis, changes in chlorophyll *a* concentrations were about 65–70 percent of the changes in external phosphorus loadings. For example, a 50-percent change in external phosphorus loading should cause about a 35-percent change in chlorophyll *a* concentrations. Based on the Hickman equation (eq. 9), simulated growing-season-average chlorophyll *a* concentrations were less responsive to increases in external phosphorus loading than to decreases in loading (fig. 31B); a 50-percent decrease in external phosphorus loading should cause about a 31-percent decrease in chlorophyll *a* concentrations, whereas a 50-percent increase in loading should cause about a 25-percent increase in concentrations. Both modeling approaches demonstrated relatively similar responses in chlorophyll *a* concentrations when external phosphorus loading was about 50 percent or less; however, with larger changes in external phosphorus loading (about 200 percent), the two approaches resulted in different expected chlorophyll *a* concentrations. Based on the Hickman equation, a 200-percent increase in external phosphorus loading should cause about an 80-percent increase in chlorophyll *a* concentrations.

Table 10. Estimated and measured water quality in Shell Lake, Wisconsin, in response to various phosphorus-loading scenarios based on the Canfield and Bachman natural-lake model and on Carlson trophic-state-index and Hickman equations

[ft, feet; mg/L, milligrams per liter; µg/L, micrograms per liter; DNR, Wisconsin Department of Natural Resources].

Simulations	Scenario number	Annual phosphorus load (pounds)	Water elevation	Mean depth (ft)	Residence time (years)	Total phosphorus concentration (mg/L)	Chlorophyll <i>a</i> concentration (µg/L)	Chlorophyll <i>a</i> concentration (µg/L)	Secchi depth, (ft)
General response									
Percent change in controllable load							TSI computed	Hickman computed	
-75%	2	744	1,217.7	24.8	9.3	0.010	3.7	4.1	19.2
-50%	3	1,073	1,217.7	24.8	9.3	0.013	5.2	5.7	15.1
-25%	4	1,403	1,217.7	24.8	9.3	0.015	6.7	7.0	12.7
0% - Reference	1	1,733	1,217.7	24.8	9.3	0.018	8.2	8.2	11.1
+ 25%	5	2,063	1,217.7	24.8	9.3	0.020	9.6	9.3	10.0
+ 50%	6	2,393	1,217.7	24.8	9.3	0.022	11.0	10.2	9.1
+ 75%	7	2,722	1,217.7	24.8	9.3	0.023	12.4	11.1	8.4
+ 100%	8	3,052	1,217.7	24.8	9.3	0.025	13.7	12.0	7.8
+ 200%	9	4,371	1,217.7	24.8	9.3	0.031	18.9	14.8	6.3
Specific anthropogenic and natural scenarios									
2009 No internal loading	10	1,319	1,217.7	24.8	9.3	0.015	6.4	6.6	13.2
2009 No septic input	11	1,682	1,217.7	24.8	9.3	0.017	8.0	7.9	11.3
1980 (DNR, no internal loading)	12	1,783	1,217.3	24.5	6.5	0.017	7.6	7.6	11.7
1980 (DNR, with internal loading)	13	2,197	1,217.3	24.5	6.5	0.019	9.3	8.9	10.2
1980 (USGS)	14	1,666	1,217.3	24.5	7.8	0.017	7.6	7.6	11.7
Shallow lake - mixed	15	1,790	1,217.6	24.7	8.2	0.018	8.2	8.0	11.1
Middle depth lake	16	1,864	1,219.2	25.6	8.1	0.017	8.0	7.9	11.3
Deep lake - stratified	17	1,835	1,221.4	27.0	8.4	0.016	7.4	7.4	12.0
Measured									
2009	NA	1,733	1,217.7	24.8	9.3	0.018	8.2	8.2	11.1
1980	NA	1,666	1,217.3	24.5	7.8	0.018	6.1	6.1	12.9
Shallow lake - mixed	NA	1,790	1,217.6	24.7	8.2	0.017	9.5	9.5	11.9 ¹
Middle depth lake	NA	1,864	1,219.2	25.6	8.1	0.016	7.7	7.7	12.5
Deep lake - stratified	NA	1,835	1,221.4	27.0	8.4	0.016	6.9	6.9	13.6

¹In 2011 (a shallow year), only one Secchi measurement was made in June, and the measurement was unusually deep (32 ft). This one unusual measurement was replaced with the average June value from other shallow years for computing the average shallow-lake Secchi depth. Had the unusual measurement not been replaced, the average shallow-lake Secchi depth would have been 12.7 ft.

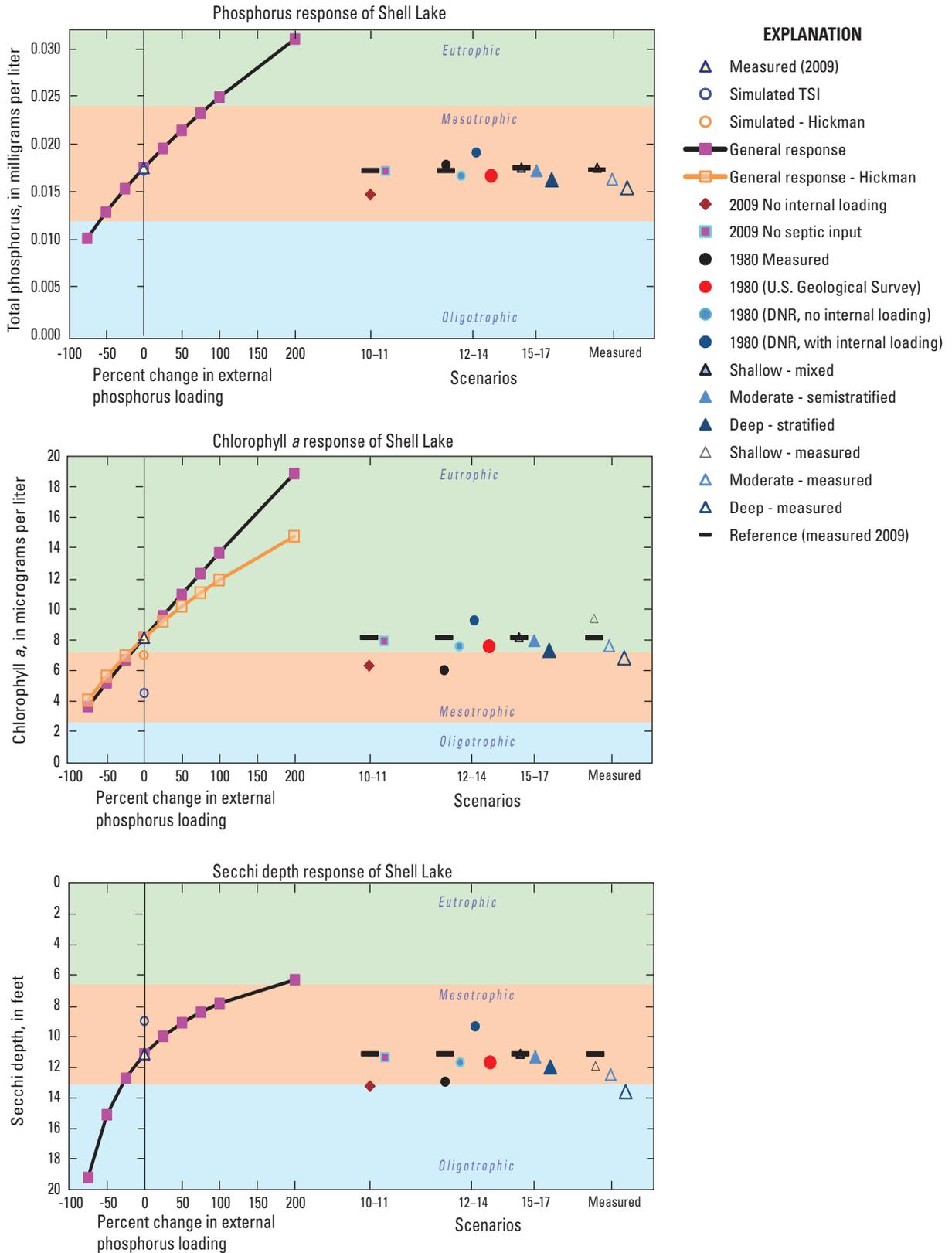


Figure 31. Measured and simulated growing-season-average near-surface concentrations of total phosphorus, chlorophyll *a*, and Secchi depths in Shell Lake, Wisconsin, in response to various phosphorus-loading scenarios. (DNR, Wisconsin Department of Natural Resources.)

Simulated growing-season-average Secchi depths had a nonlinear response to changes in external phosphorus loading, with a much larger response to decreases in loading than to increases in loading (fig. 31C). For example, a 50-percent decrease in external phosphorus loading should cause about a 35-percent increase in Secchi depths, whereas a 50-percent increase in external phosphorus loading should cause only about an 18-percent decrease in Secchi depths.

Quantifying the Effects of Internal Phosphorus Loading

Scenario 10, in comparison with scenario 1, was used to determine the effects of internal phosphorus loading from the sediments on the water quality of the lake. Complete removal of phosphorus loading from the sediments from scenario 1 (414 lb decrease; about 24 percent of the total phosphorus budget, table 10) resulted in simulated in-lake near-surface phosphorus concentrations decreasing by 0.003 mg/L. This decrease in phosphorus concentration resulted in a decrease in average chlorophyll *a* concentration (1.6–1.8 µg/L) and increase in average Secchi depth (2.1 ft) (table 10, fig. 31). Therefore, internal phosphorus loading in Shell Lake has measurable effect on the growing-season-average water quality in Shell Lake and could have an effect in the future if phosphorus loading to the lake is changed.

Quantifying Effects of Septic Input

Scenario 11, in comparison with scenario 1, was used to determine the effects of septic inputs on the water quality of the lake. Complete removal of the phosphorus loading from septic inputs from scenario 1 (51-lb decrease, about 3 percent of the total phosphorus budget; table 7) resulted in simulated in-lake near-surface phosphorus concentrations decreasing by 0.001 mg/L. This decrease in phosphorus concentration resulted in a decrease in average chlorophyll *a* concentration (0.2–0.3 µg/L) and an increase in average Secchi depth (0.2 ft) (table 8; fig. 31). Therefore, septic inputs to Shell Lake appear to have little effect on the growing-season-average water quality of Shell Lake, even if the number of capita years or soil retention (eq. 7) were to moderately change.

Simulated Water Quality in 1980

Scenarios 12–14, in comparison with the detailed water-quality data collected in 1980, were used to help evaluate the phosphorus budgets constructed by the WDNR (Wisconsin Department of Natural Resources, 1982) and constructed as part of this study. The WDNR estimated the total annual phosphorus load to be 1,780 lb (2,190 lb if the internal loading estimated in this study is included), with a water residence time of 6.2 years, whereas the total annual load estimated in 1980 as part of this study was 1,640 lb, with a water residence

time of 7.6 years. During 1980, the growing-season-average measured near-surface total phosphorus concentration was 0.018 mg/L, chlorophyll *a* concentration was 6.1 µg/L, and Secchi depth was 12.9 ft (table 10; fig. 31). The predicted phosphorus concentrations associated with WDNR phosphorus loading estimates (without internal loading) and with loading estimates from this study were very similar as a result of the WDNR phosphorus loads being slightly higher and the residence time being slightly lower than estimated in this study. Both approaches slightly underestimated total phosphorus concentrations and Secchi depth but overestimated chlorophyll *a* concentrations (table 10; fig. 31). The predicted values with WDNR phosphorus loading (with internal loading added) overestimated total phosphorus concentrations and chlorophyll *a* concentrations and underestimated Secchi depth. The predicted values with WDNR phosphorus loading (without internal loading) and with loading from this study were most similar to those measured in 1980. Although both of these approaches had relatively similar total phosphorus loadings, the relative importance of the sources was quite different. The WDNR phosphorus-load estimates appeared to overestimate the loading from precipitation and groundwater and underestimate the input from the internal sediment loading because it was not attempted to be included in those estimates. If internal loading is added to the phosphorus loading estimated by the WDNR, the predicted water quality is worse than that measured in the lake.

Simulated Water Quality in Response to Changes in Water Level

In scenarios 15–17, the effects of changes in water level (and the accompanying changes in hydrology) were evaluated by examining the response of the lake with inputs and morphometry estimated when the lake was relatively shallow (years between 1949 and 2011, with water levels less than 1,218.5 ft and an average mean depth of 24.7 ft), when the lake was moderately deep (water levels between 1,218.5 ft and 1,220 ft and an average mean depth of 25.6 ft; referred to as “moderate”), and when the lake was relatively deep (greater than 1,220 ft and an average mean depth of 27.0 ft; referred to as “deep”). For these scenarios, depth for each of the years between 1949 and 2011 was classified as shallow, moderate, or deep on the basis of average water elevation for the year. Then the annual water and phosphorus loadings estimated from 1949–2011 were used as inputs into the water-quality models, and the annual results for each water level category were averaged. There was very little difference in total estimated loading to the lake among scenarios. Measured water-quality data for each of these years were collected with different frequencies. Therefore, to make the measured growing-season-average water-quality data (average of the average monthly data from May through September) more comparable among years the following selections were made: (1) only data from

1980 to 2011 were included because much of the monthly data in other years were missing, (2) all years used the average measured May and September conditions for 1980 to 2011 because only a few years had actual measured data in May and September, and (3) only years that had July and August data were included. For the years without June data, the average of the June values for the specific category were used in computing the summer average.

The changes in water level (scenarios 15–17; table 10; fig. 31) that have been observed in Shell Lake were simulated to have a modest effect on growing-season-average water quality. As the lake got deeper, growing-season-average phosphorus concentrations declined from 0.018 to 0.016 mg/L, growing-season-average chlorophyll *a* concentrations decreased from 8.0–8.2 to 7.4 µg/L, and growing-season-average Secchi depth increased from 11.1 to 12.0 ft. These small simulated changes were of similar magnitude to that measured in the lake (table 10; fig. 31). As the lake got deeper, measured phosphorus concentrations declined from 0.017 to 0.016 mg/L, chlorophyll *a* concentrations decreased from 9.5 to 6.9 µg/L, and Secchi depth increased from 12.7 to 13.6 ft. In 2011, the one Secchi depth measured in June was 32 ft. If this one unusual measurement was replaced with the average June value from other shallow years, then the Secchi depth would have increased from 11.9 to 13.6 ft. Therefore, although the relation between water quality and water level is small, growing-season-average water quality does appear to improve slightly as the depth of the lake increases.

Importance of Water Level to Lake Productivity

Based on measured and simulated conditions (table 10), the growing-season-average water quality in Shell Lake is slightly better (lower phosphorus and chlorophyll *a* concentrations and better water clarity) when the water level in the lake is high than when the water level is low. However, seasonal changes in water quality, with degrading water quality as summer progresses (fig. 18), are more apparent than the annual variability with changes in water level; that is, during years with the highest water levels (2000–2008), the water clarity in June was much better than in July and August compared with years with lower water levels.

It is believed that the water level affects the extent of stratification and the amount of deep mixing in the lake, which in turn affects amount of phosphorus from internal loading that reaches the epilimnion of the lake. To determine why the changes in water level affect the seasonality of water quality, the Dynamic Lake Model (DLM) was used to simulate changes in the stratification of the lake in response to changes in water level. DLM was first validated for 2009 conditions, when the majority of in-lake data were collected. DLM was then used to simulate the extent of stratification with water levels similar to what has been measured for the lake since 1936 (3.3 ft lower than 2009, similar to 2009, and 6.6 ft

deeper than 2009). Changes in water level in Shell Lake had a profound effect on the extent of stratification and deep mixing in the lake (fig. 32). When water levels were 3.3 ft lower than in 2009, the lake rarely stratified more than a few days at a time after the beginning of July. During 2009, the lake was stratified until early August; however, there were several periods of deep mixing during summer. This was also observed with the recording thermometers placed in the lake (fig. 17). When water levels were 6.6 ft higher than in 2009, the lake remained stratified throughout most of the summer.

Years with the highest water levels show the most seasonality in water quality, especially in Secchi depths (fig. 20). Differences in the amount of mixing (fig. 32) may be the reason for the differences in the amount of seasonality observed in water quality. High water levels result in relatively strong stratification in June through August (fig. 32C) and a very limited amount of deep mixing. Therefore, phosphorus released from the sediments may be trapped in the hypolimnion (deep area of the lake) and not available as a nutrient source for early summer productivity. This would result in low chlorophyll *a* concentrations (fig. 20B) and deep Secchi depths (fig. 20C). In years with lower water levels (fig. 32A), more phosphorus from internal loading would be available throughout summer, resulting in less seasonality. The increased seasonality in years with highest water levels, resulting from the greater Secchi depths early in the summer, may be the primary reason for the slightly better overall measured growing-season-average water clarity.

Although anaerobic conditions were not measured in the lake in 2009, it is possible that anaerobic conditions could develop during periods of persistent stratification. If anaerobic conditions developed, the rate of internal loading would be expected to increase rather dramatically (fig. 30). More internal loading would result in higher phosphorus concentrations developing near the bottom and more phosphorus being released in late summer, with possible higher chlorophyll *a* concentrations and poorer water clarity in late summer. High bottom phosphorus concentrations may have occurred when the water levels were very high, such as between 1995 and 2005 (fig. 21); however, little water-quality data were collected during this period.

Improved water quality in years with high water levels is the opposite of what may be expected to occur in most lakes (Robertson and Rose, 2011). In most cases, phosphorus concentrations and phosphorus loading from surface-water runoff increase during wet years with high surface-water runoff because of increased erosion. However, total phosphorus concentrations in the surface-water runoff to Shell Lake decreased during high runoff, possibly a result of the increased flow rate through upstream wetlands. It is hypothesized that the faster water moves through the wetlands, the less phosphorus is released from their bottom sediments. This would result in very little increase in loading to Shell Lake during wet years, and the change in water quality being driven primarily by differences in the intensity and duration of stratification.

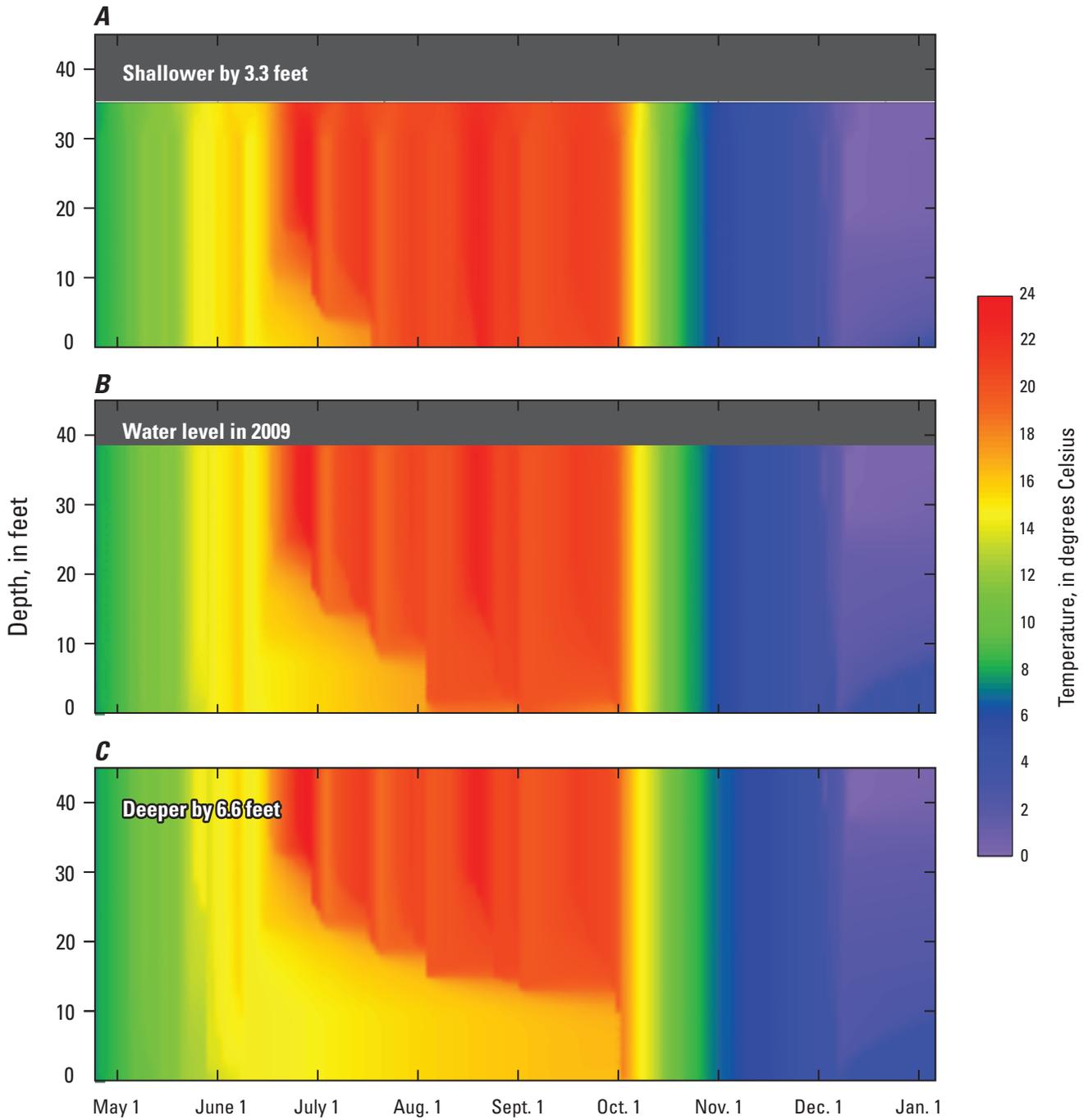


Figure 32. Simulated (with the Dynamic Lake Model) water temperature distribution during open-water periods in Shell Lake, Washburn County, Wisconsin, when the water levels are *A*, 3.3 feet shallower than 2009, *B*, similar to 2009, and *C*, 6.6 feet deeper than 2009. The gray areas at the top of *A* and *B* reflect lower water levels than in *C*.

Importance of Internal Phosphorus Loading to Lake Productivity

Typically, in deep dimictic (stratified) lakes, phosphorus from internal loading primarily occurs when the bottom sediment becomes anaerobic. This phosphorus is trapped in the hypolimnion during most of the summer and released into the shallow water (epilimnion) primarily at fall overturn. In shallow polymictic lakes like Shell Lake, however, the phosphorus released from the sediments is not trapped in the bottom of the lake but is intermittently released to the shallow water throughout the summer. These mixing events resulted in increases in the near-surface phosphorus concentrations in Shell Lake as summer progressed. This is the primary reason why the additional internal phosphorus load was quantified in this study. In most lakes, there is very little phosphorus release from the sediments if the surface of the sediments remain aerobic. Sediment core analyses demonstrated that the release of phosphorus from the sediments in Shell Lake would be higher if they went anaerobic; however, there was still considerable release from the aerobic sediments in the lake. It was estimated that internal loading from the aerobic sediments of Shell Lake added about 25 percent of the total annual phosphorus load to the lake and increased the phosphorus concentrations in the lake, especially as summer progressed—accentuating seasonality in the water quality of the lake.

Internal phosphorus loading can limit the response of a lake to reductions in external phosphorus loading and could limit the response of Shell Lake to phosphorus-load reductions in its watershed. The impact of internal phosphorus loading in delaying the response in a lake's productivity to decreases in external nutrient loading has been documented in several lakes (Larsen and Mercier 1976; Larsen and others, 1979 and 1981; Robertson and Rose, 2008). This conclusion also is in agreement with the belief that once a polymictic lake becomes eutrophic, it is difficult to alter its trophic state because the high internal phosphorus loading usually does not quickly respond to reductions in external phosphorus loading (Newton and Jarrell, 1999; Nurnberg, 1998; Welch and Cooke, 1995). However, if the reduction in the external load to Shell Lake were to persist for a long period of time, the internal phosphorus release rate should gradually decrease and come to equilibrium with the new external loading.

Relative Importance of Natural and Anthropogenic Changes in the Watershed to Productivity in Shell Lake

One purpose of this study was to understand how changes in water quality of Shell Lake caused by natural changes in hydrology and water level compared with changes in water quality caused by anthropogenic modifications to the watershed. Based on Garrison's (2012) reconstructed changes in water quality from 1860 to 2010 derived from changes in the

diatom communities archived in the sediments and also based on Williamson's results from changes in Chironomid (aquatic insects) communities (Jeremy Williamson, Polk County Wisconsin Land and Water Resources, written commun., 2012), anthropogenic changes in the watershed had a much larger effect than the natural changes associated with fluctuations in water level. As a result of anthropogenic changes in the watershed (building of the sawmill in 1881; the establishment of the village of Shell Lake; and land-use changes in the 1920s, likely increased agriculture) and increased phosphorus loading to the lake, Garrison estimated that growing-season-average phosphorus concentrations increased from about 0.012 mg/L in the mid-1800s to about 0.018 mg/L around 2010. These changes in phosphorus concentrations agree with what could occur with relatively large changes in loading (fig. 31). Modeling results indicate that a 50-percent reduction in external loading from that measured in 2009 would be required to change phosphorus concentrations from 0.018 mg/L to 0.012 mg/L.

Although the effects of natural changes in water level (caused by meteorological and climate variability) on water quality appear to be small, based on measured data (1980–2011) and model simulations (1949–2011), changes in water level do have a modest effect on growing-season-average water quality: small water-quality improvements during higher water levels. These small improvements in water quality were also observed in Garrison's reconstructed phosphorus concentrations (fig. 23), with relatively low phosphorus concentrations observed around 2000 when highest water levels occurred in the lake. The fluctuations in water level, however, have a larger effect on the seasonality in water quality, with better water quality, especially increased Secchi depths, in early summer during years with high water levels. This seasonal improvement in water quality, however, is small when the entire summer is considered, as quantified in Garrison's study.

Summary and Conclusions

Shell Lake is a relatively shallow, soft-water, terminal lake (closed-basin lake with inlets but no outlet) in Washburn County, Wisconsin. Since water level measurements began in 1936, the water level of Shell Lake has fluctuated by about 10 ft (about 30 percent of its maximum depth). Starting around 1990, the lake level rose above average conditions, and in the early 2000s, numerous homes and municipal buildings were flooded. In an effort to reduce flooding, the City of Shell Lake sought and received a permit from the Wisconsin Department of Natural Resources (WDNR) to divert water out of the lake to the nearby Yellow River. A cooperative project between the City of Shell Lake and the U.S. Geological Survey (USGS) was initiated in 2006 to satisfy a requirement of the permit: to evaluate the hydrology of the lake, its interaction with the groundwater-flow system, and the effects of the diversion on the water level of Shell Lake and nearby upgradient lakes.

Concerns over deteriorating water quality in the lake, especially associated with changes in water level and possibly associated with the water diversions, prompted an additional cooperative project between the City of Shell Lake and the USGS. As part of this water-quality project, further monitoring of the lake and groundwater system were conducted, primarily in 2009, to understand how changes in water level affect the water and nutrient input into the lake and, ultimately, the water quality of the lake. This report presents the results of both studies.

USGS personnel collected lake, stream, and groundwater-quality data from December 2006 to November 2011. Groundwater levels in four nested wells (two wells of differing depth in each nest) were monitored from November 2006 to April 2010, and groundwater-quality samples were collected annually from 2007 to 2009. Six minipiezometers were also installed in July 2009 and sampled twice during 2009 for phosphorus concentrations in groundwater. Lake and stream data were collected primarily during open-water periods from February to November 2009. Owing to below-average precipitation during 2008–9, additional streamflow and water-quality data were collected through November 2011. The USGS also collected continuous water temperature data throughout the water column at two sites in the lake during 2009 to describe the extent of vertical stratification and help describe variability in dissolved oxygen concentrations in the lake. In addition to USGS data, historical water-quality data for Shell Lake were collected by the WDNR and volunteers from the City of Shell Lake.

Numerical model simulations were performed to improve understanding of the hydrology and water quality of Shell Lake and to evaluate how the hydrology and water quality are expected to respond to future changes that may occur in the watershed. The hydrologic system was simulated with the groundwater-flow model MODFLOW-2005 to simulate groundwater movement in the area around Shell Lake, including groundwater/surface-water interactions. The MODFLOW model was first calibrated for steady state conditions, or “average” conditions corresponding to January 1, 2007, to December 31, 2008, when the majority of hydrologic data were collected. The steady-state simulation was performed to estimate spatial hydrogeologic properties, such as aquifer and confining-unit permeability. Simulated results from the MODFLOW model indicate that groundwater flows generally north in the area around Shell Lake, with flow locally converging toward the lake. However, total groundwater inflow to Shell Lake is small (approximately 5 percent of the water budget) compared with water entering the lake from precipitation (83 percent) and surface-water runoff (13 percent). The limited interaction between Shell Lake and the groundwater system is likely associated with the pervasiveness of shallow clay in the area around Shell Lake.

Following the steady-state calibration, the MODFLOW model was applied in transient mode, enabling annual average hydrologic conditions to be simulated for each year from 1949 to 2009. To directly evaluate the effect of the diversion on water levels, the diversion was removed from the transient

simulation, and the resulting simulated water levels were compared with water levels from the calibrated model. The maximum difference in simulated annual average water level for Shell Lake between simulations with and without the diversion was approximately 3.3 feet (ft) and occurred in 2005 at the end of the diversion process. The results also illustrate that although the water level of the lake continued to decline through 2009 in response to local weather patterns, the direct effect of the diversion decreased after the diversion ceased. Over the 4 years of simulated recovery (2006–9), drawdown attributable to the diversion rebounded by about 0.6 ft because of increased groundwater inflow and decreased lake-water outflow due to the lowering of the lake relative to the water table.

A delayed response in drawdown associated with the Shell Lake water diversion was transmitted through the groundwater-flow system to upgradient lakes. Simulated maximum drawdown in the aquifer due to the diversion occurred from zero to more than 4 years after the peak drawdown occurred in Shell Lake and ranged from less than 0.5 ft to more than 2 ft. Simulated drawdown in upgradient lakes was further delayed, and 4 years after the diversion ceased, simulated drawdown due to the diversion was less than 0.5 ft in all upgradient lakes and diminished with distance from Shell Lake.

The MODFLOW model accurately simulates groundwater movement in the area around Shell Lake, but it has limited capability to simulate more detailed surface-water runoff to the lake, as needed for phosphorus load calculations. Therefore, an empirical runoff model was developed and used to estimate the total annual runoff and phosphorus loading to Shell Lake.

Data collected in Shell Lake indicated that it was a polymictic lake (characterized by frequent deep mixing) and that its productivity was limited by the amount of phosphorus input to the lake. The lake was typically classified as oligotrophic-mesotrophic in June, mesotrophic in July, and mesotrophic-eutrophic in August, indicating that water quality deteriorates as summer progresses. In polymictic lakes like Shell Lake, phosphorus released from the sediments (internal phosphorus loading) is not trapped in the bottom of the lake but is intermittently released to the shallow water throughout the summer, resulting in deteriorating water quality as summer progresses.

Because the productivity of Shell Lake was typically limited by phosphorus, the sources of phosphorus to the lake were quantified, including phosphorus contributed from the bottom sediments. During 2009, the total annual input of phosphorus to Shell Lake was 1,730 pounds (lb), of which 1,320 lb came from external sources (76 percent) and 414 lb came from the sediments in the lake (24 percent). The largest external source of phosphorus was from surface-water runoff, which delivered 907 lb or about 52 percent of the total phosphorus load compared with about 13 percent of the water input. The second largest contribution was from precipitation, which delivered about 19 percent of the

phosphorus load compared to about 83 percent of the water input. Contributions from septic systems and groundwater accounted for about 3 and 2 percent of the phosphorus load, respectively. Because 2009 followed a series of dry years, phosphorus loading from each component of the budget was estimated for each year from 1949 to 2011. Since 1949, the average annual phosphorus loading to the lake has been 1,830 lb, but annual loads have varied from 1,400 lb in 1976 to 2,420 lb in 1977. Surface-water runoff was always the major contributor of phosphorus to the lake.

Water quality in the lake was simulated with empirical eutrophication models that relate hydrology and phosphorus loading to specific water-quality characteristics in the lake. The Canfield and Bachman natural-lake model, Carlson's TSI equations, and Hickman's chlorophyll *a* model were used to describe how changes in phosphorus loading caused by natural and anthropogenic changes in the watershed affect the total phosphorus and chlorophyll *a* concentrations and Secchi depth in the lake. The eutrophication models were used to evaluate effects of increases and decreases in phosphorus loading on historical or potential future water quality in Shell Lake. Simulating a range in external phosphorus loading indicated that Shell Lake water quality (phosphorus and chlorophyll *a* concentrations and Secchi depths) is slightly more sensitive to reductions in phosphorus loading than increases in external loading. Results also indicated that a 50-percent reduction in external loading from that measured in 2009 would be required to change phosphorus concentrations from 0.018 mg/L (measured in 2009) to 0.012 mg/L (estimated for the mid-1800s from analysis of diatoms in sediment cores). Model simulations indicate that septic input may increase phosphorus concentrations by 0.001 mg/L, increase average chlorophyll *a* concentrations by 0.2–0.3 µg/L, and decrease the average Secchi depth by 0.2 ft. Model simulations also indicate that internal phosphorus loading from sediments may increase phosphorus concentrations by 0.003 mg/L, increase chlorophyll *a* concentrations by 1.6–1.8 µg/L, and decrease growing-season-average Secchi depth by 2.1 ft. Therefore, internal phosphorus loading in Shell Lake has measurable effect on the growing-season-average water quality in Shell Lake and may continue to effect the lake in the future even if phosphorus loading to the lake is changed. Therefore, to reduce phosphorus concentration in the lake more than by 0.003 mg/L requires decreasing phosphorus loading from surface-water contributions, primarily runoff to the lake. Potential improvements in the water quality of Shell Lake will be directly related to the reduction in phosphorus loading to the lake. Some of the ways in which phosphorus loading to the lake can be reduced is by reducing flows or phosphorus concentrations added to the lake by ephemeral streams, wetlands, and nearshore developed areas, and by reducing inputs from septic systems.

One purpose of this study was to understand how changes in water quality of Shell Lake caused by natural changes in hydrology and water level compared with changes caused by anthropogenic modifications to the watershed. Reconstructed

changes in water quality from 1860 to 2010, based on changes in the diatom and Chironomid communities archived in the sediments and eutrophication model simulations, suggest that anthropogenic changes in the watershed (building of the sawmill in 1881, the establishment of the village of Shell Lake, and land-use changes in the 1920s (likely, increased agriculture and increased residential and urban development around the lake)) had a much larger effect on water quality than the natural changes associated with fluctuations in water level. Although the effects of natural changes in water level (caused by meteorological and climate variability) on water quality appear to be small, changes in water level do have a modest effect on growing-season-average water quality—with small water-quality improvements during periods with higher water levels. The fluctuations in water level, however, affect the degree of seasonal variability in water quality—with better water quality, especially increased clarity (deeper Secchi depths), in early summer during years with high water levels.

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Appendix 1

Results for basic water chemistry samples from eight nested wells around Shell Lake.
Download at http://pubs.usgs.gov/sir/2013/5181/ShellLake_Juckem_appendix1.pdf.

Appendix 2

Results for heavy metals and volatile organic compounds from nested wells around Shell Lake.
Download at http://pubs.usgs.gov/sir/2013/5181/ShellLake_Juckem_appendix2.pdf.

