In cooperation with the Whitefish Lake Conservation Organization (WILCO) through the Town of Wascott, Wisconsin

Water Quality and Hydrology of Whitefish (Bardon) Lake, Douglas County, Wisconsin, With Special Emphasis on Responses of an Oligotrophic Seepage Lake to Changes in Phosphorus Loading and Water Level

Scientific Investigations Report 2009–5089

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

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KEN SALAZAR, Secretary

U.S. Geological Survey
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Cover photos showing changes in the water level in Whitefish Lake. (All photos by Dale M. Robertson.)

Upper left–High water near the south shore of Whitefish Lake, August 2004.

Upper right–High water near the small peninsula in the South Basin of Whitefish Lake, August 2004; pictured are Fred Anderson, Sandra Anderson, and William Rose.

Lower left–Low water near the south shore of Whitefish Lake, May 2007.

Lower right–Low water near the small peninsula in the South Basin of Whitefish Lake, May 2007.

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Conversion Factors and Abbreviations

Multiply	By	To obtain
	Length	
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	0.004047	square kilometer (km^2)
	Volume	
acre-foot (acre-ft)	1,233.5	cubic meter (m^3)
	Mass and weight	
pound, avoirdupois (lb)	0.4536	kilogram (kg)
	Miscellaneous rates and yields	
inch per day (in/d)	25.4	millimeter per day (mm/d)
inch per year (in/yr)	25.4	millimeter per year (mm/yr)
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)
pound per year (lb/yr)	0.4536	kilogram per year (kg/yr)
pound per square mile (lb/mi^2)	0.1751	kilograms per square kilometer (kg/km^2)

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

$$°F=(1.8\times°C)+32$$

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).

*Hydraulic conductivity: The standard unit for hydraulic conductivity is cubic foot per day per square foot of aquifer cross-sectional area ($ft^3/d/ft^2$). In this report, the mathematically reduced form, foot per day (ft/d), is used for convenience.

Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29).

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Water Quality and Hydrology of Whitefish (Bardon) Lake, Douglas County, Wisconsin, With Special Emphasis on Responses of an Oligotrophic Seepage Lake to Changes in Phosphorus Loading and Water Level

By Dale M. Robertson, William J. Rose, and Paul F. Juckem

Abstract

Whitefish Lake, which is officially named Bardon Lake, is an oligotrophic, soft-water seepage lake in northwestern Wisconsin, and classified by the Wisconsin Department of Natural Resources as an Outstanding Resource Water. Ongoing monitoring of the lake demonstrated that its water quality began to degrade (increased phosphorus and chlorophyll *a* concentrations) around 2002 following a period of high water level. To provide a better understanding of what caused the degradation in water quality, and provide information to better understand the lake and protect it from future degradation, the U.S. Geological Survey did a detailed study from 2004 to 2008. The goals of the study were to describe the past and present water quality of the lake, quantify water and phosphorus budgets for the lake, simulate the potential effects of changes in phosphorus inputs on the lake's water quality, analyze changes in the water level in the lake since 1900, and relate the importance of changes in climate and changes in anthropogenic (human-induced) factors in the watershed to the water quality of the lake.

Since 1998, total phosphorus concentrations increased from near the 0.005-milligrams per liter (mg/L) detection limit to about 0.010 mg/L in 2006, and then decreased slightly in 2007–08. During this time, chlorophyll *a* concentrations and Secchi depths remained relatively stable at about 1.5 micrograms per liter (μg/L) and 26 feet, respectively. Whitefish Lake is typically classified as oligotrophic.

Because the productivity in Whitefish Lake is limited by phosphorus, phosphorus budgets were constructed for the lake. Because it was believed that much of its phosphorus comes from the atmosphere, phosphorus deposition was measured in this study. Phosphorus input from the atmosphere was greater than computed based on previously reported wetfall phosphorus concentrations. The concentrations and deposition rates can be used to estimate atmospheric loading in future lake studies. The average annual load of phosphorus to the lake was 232 pounds: 56 percent from precipitation, 27 percent from groundwater, and 16 percent from septic systems. During a series of dry years (low water levels) and wet years (high water levels), the inputs of water and phosphorus ranged by only 10–13 percent.

Results from the Canfield and Bachmann eutrophication model and Carlson trophic-state-index equations demonstrated that the lake directly responds to changes in external phosphorus loading, with percent change in chlorophyll *a* being similar to the percent change in loading and the change in total phosphorus and Secchi depth being slightly smaller. Therefore, changes in phosphorus loading should affect the water quality of the lake. Specific scenarios that simulated the effects of anthropogenic (human-induced) and climatic (water level) changes demonstrated that: surface-water inflow (runoff) based on current development has little effect on pelagic water quality, changes in the inputs from septic systems and development in the watershed could have a large effect on water quality, and decreases in water and phosphorus loading during periods of low water level had little effect on water quality. Sustained high water levels, resulting from several wet years with relatively high water and phosphorus input, however, could cause a small degradation in water quality. Although high water levels may be associated with a degradation in water quality, it appears that anthropogenic changes in the watershed may be more important in affecting the future water quality of the lake.

Fluctuations in water levels since 1998 are representative of what has occurred since 1900, with fluctuations of about 3 feet occurring about every 15 years. Based on total phosphorus concentrations inferred from sediment core analysis, there has been little long-term change in water quality and there has been a slight deterioration in water quality following most periods of high water levels. Therefore, the recent degradation in water quality appears to be consistent with what may have occurred several times in the past associated with high water levels.

Whitefish Lake is presently a relatively pristine lake and is classified as an Outstanding Resource Water primarily because of the small contributions of phosphorus from its watershed. Controls to minimize the phosphorus input to the lake are very important because what many would consider a small increase in phosphorus loading in most lakes could have a large effect on the phosphorus budget and water quality of Whitefish Lake.

Introduction

Whitefish Lake, which is officially named Bardon Lake, is a soft-water seepage lake in northwestern Wisconsin. The Wisconsin Department of Natural Resources (WDNR) has classified the lake as an Outstanding Resource Water because of the lake's excellent water quality, high recreational and aesthetic value, and high quality fishing. Outstanding Resource Waters are believed to be free from point-source and nonpoint-source pollution (Wisconsin Department of Natural Resources, 2008a). Based on data collected by the Citizen Lake Monitoring Network (Citizen Monitoring; Wisconsin Department of Natural Resources, 2008b), it appeared that the water quality of the lake was beginning to degrade: phosphorus and chlorophyll *a* (algae) concentrations measured in 2002–04 were higher than those measured in 1998–99. The higher phosphorus concentrations followed several years of relatively high water levels in the lake. Local residents were unsure whether the increased phosphorus concentrations were caused by natural, climatic effects associated with the high water levels or some anthropogenic factor(s) in the watershed. To better understand the cause of the apparent degradation in water quality, a detailed study was conducted from 2004 to 2008 by the U.S. Geological Survey (USGS). This study was done in cooperation with the Whitefish Lake Conservation Organization (WILCO; *http://www.uwsp.edu/cnr/landcenter/whitefish_lake/*) through the Town of Wascott, Wis., and was partially funded through the Lake Protection Grant Program of the WDNR and the Cooperative Water Program of the USGS. The results of this study are described in this report. Specifically, the water quality and hydrology of the lake are described, sources of phosphorus are quantified, and the effects of past and future changes in phosphorus inputs on the water quality of the lake are determined, especially with respect to changes in the water level of the lake. This study was part of an interdisciplinary project conducted on Whitefish Lake to provide information for local managers to develop a comprehensive lake-management plan. Other aspects of this project examined the lake sediments (paleolimnology), plankton, macrophytes, fish, and sociological information to improve the understanding and protection of the lake.

Whitefish Lake and Its Watershed

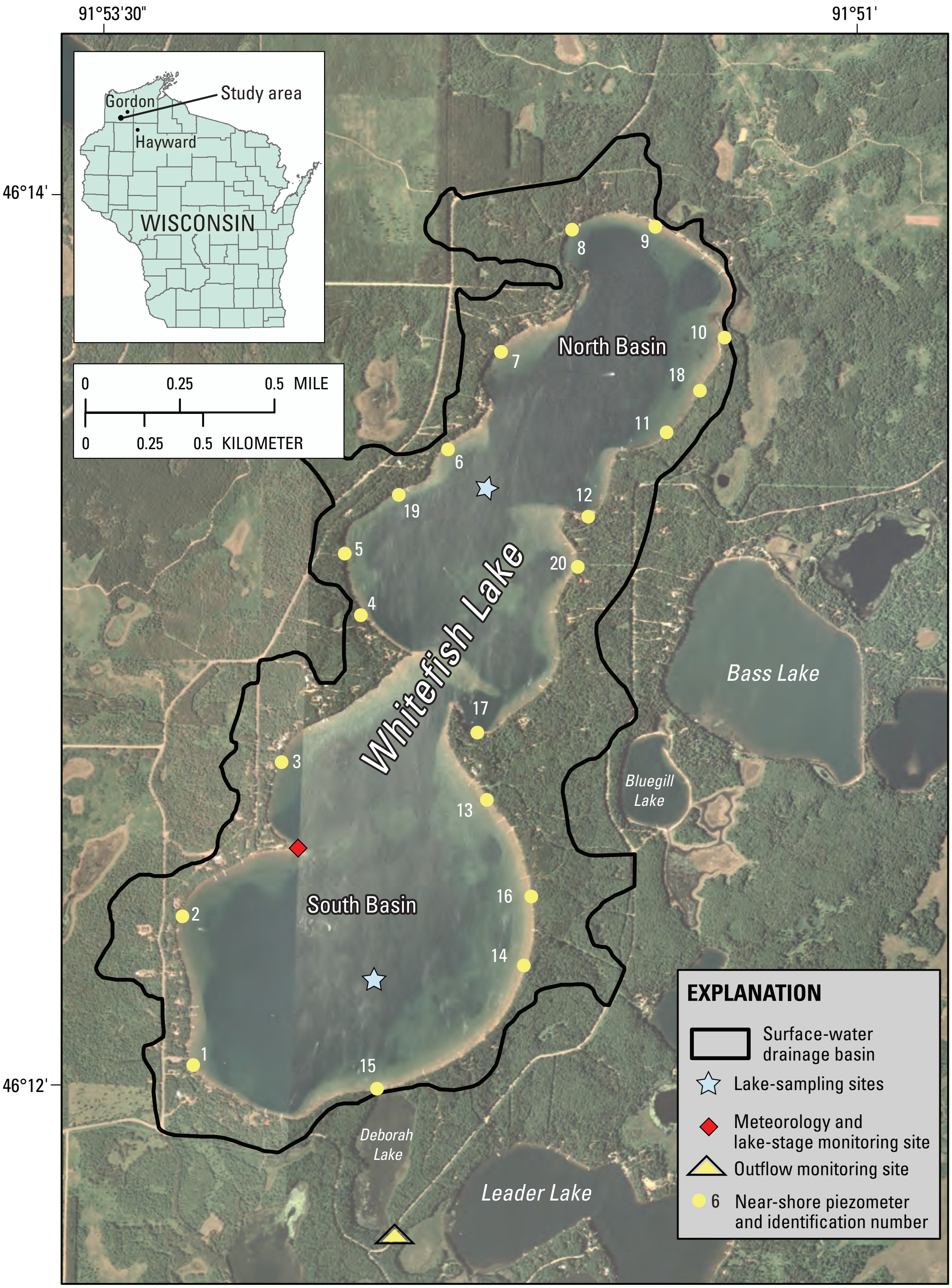
Whitefish Lake (fig. 1) is a natural seepage lake in Douglas County, in northwestern Wisconsin, which is about 25 mi northwest of Hayward, Wis., and 3 mi southwest of Gordon, Wis. Seepage lakes are closed-basin lakes that do not have an inlet or an outlet, and only occasionally overflow (Wisconsin Department of Natural Resources, 1999; Shaw and others, 2002). Whitefish Lake only rarely overflows into a small wetland (Deborah Lake) on the southeast side of lake. Because Whitefish Lake is primarily a landlocked waterbody, its principal sources of water are precipitation and groundwater, which may be supplemented by direct runoff from its small surrounding drainage area.

The drainage area of Whitefish Lake is 520 acres (fig. 1). The landcover in the watershed is primarily a mixture of deciduous hardwood (68 percent) and pine (30 percent) forest (Homer and others, 2007). The soil around Whitefish Lake consists primarily of Menahga sand with 0–12 percent slope (U.S. Department of Agriculture, 2006b). The Menahga series consists of very deep, well to excessively drained soils that formed in sandy glacial-outwash sediments on valley plains, moraines, and drumlins. These soils are very permeable, with minimum infiltration rates of 5–10 in/hr (Young and Hindall, 1973). Therefore, in the absence of anthropogenic compaction of the soils, very little surface runoff should reach the lake except during extremely large storms and, possibly, during spring runoff when the ground may be partially frozen.

The area and volume of Whitefish Lake were given on the 1967 Wisconsin Conservation Department lake survey map as 831.5 acres and 25,334 acre-ft, respectively. In this study, the morphometry of the lake was reevaluated based on an aerial image obtained in 2005 by the National Agricultural Imagery Program (U.S. Department of Agriculture, 2006a). In 2005, the lake was near full volume, similar to when the original map was constructed in 1967. The resulting surface area of the lake was determined to be 833 acres from the aerial image, which results in the drainage basin to lake area ratio of only 0.62. All bathymetric contours were then adjusted to the new surface area of the lake. The volume was then determined to be 24,184 acre-ft (table 1); slightly different values from the original estimates. All of the calculations in this study are based on the recently estimated morphometry. The lake consists of two basins (North and South Basins) connected by a narrows (fig. 1). The South Basin has a maximum depth of about 102 ft, whereas the maximum depth of the North Basin is about 60 ft (table 1).

Table 1. Morphometric characteristics of the basins in Whitefish (Bardon) Lake, Douglas County, Wis.

Basin	Area (acres)	Length (miles)	Width (miles)	Maximum depth (feet)	Mean depth (feet)	Volume (acre-feet)
North	376	1.42	0.46	60	26.12	9,616
South	457	1.26	.87	102	31.88	14,568
Entire lake	**833**	**2.55**	**.67**	**102**	**29.02**	**24,184**



Aerial image from U.S. Department of Agriculture, 2006a

Figure 1. Aerial photo of Whitefish (Bardon) Lake, Douglas County, Wis., with its watershed delineated. Surface-water and groundwater data-collection sites are identified.

Whitefish Lake is oligotrophic, meaning it does not contain an abundance of nutrients and is characterized by very clear, deep water (greater than 100 ft), and low primary productivity. Whitefish Lake has a two-story fishery with cold-water fish (primarily cisco and trout) and warm-water fish (primarily yellow perch and walleye) in the pelagic area of the lake (T. Hrabik, University of Minnesota, written commun., 2007). Oligotrophic seepage lakes are extremely sensitive to increases in phosphorus inputs. Only a small increase in phosphorus input can increase algal production and organic matter that when decomposed may be sufficient to deplete the dissolved oxygen in the deep waters of the lake (Newton and Jarrell, 1999).

Based on a recent survey of property owners around Whitefish Lake (Olson and others, 2006), most of the 100 property owners (77 percent) have one residential structure. The oldest reported residence dates back to 1910, shortly after the cutting of the pine forest in the region. A land economic inventory (E. Olson, University of Stevens Point, personal commun., 2008) found 30 structures on the lake in 1933. Based on a Wisconsin Conservation Department lake survey map, the number of cottages and permanent homes increased to 87 by 1967 (Sather and Johannes, 1973). The number of cottages and permanent homes has been relatively stable since the 1960s, increasing to about 100 in 2005 (Olson and others, 2006). Several of the residents, however, have replaced smaller seasonal structures with larger, year-round structures. Residents near Whitefish Lake anticipate further residential development near the lake in the coming decades.

In the interest of protecting the water quality of the lake, the Whitefish Lake Property Owners Association was formed in the 1940's. This group was originally involved in many zoning and political issues. The Association felt the water quality in the lake was good, but there was no standard by which to judge it or to make future comparisons. Therefore, to document the water quality of the lake and develop a baseline for future comparisons, the Association began a monitoring program with the USGS from 1998 to 2001; the monitoring program was partially funded through a WDNR Planning Grant and the USGS Cooperative Water Program. Results from that study (U.S. Geological Survey, Wisconsin District Lake Studies Team, 1999–2002) demonstrated that the lake had very low nutrient and chlorophyll *a* (algae) concentrations, had extremely good water clarity, and that near-surface water quality in the North Basin was similar to that in the South Basin. As part of political efforts to protect the lake, the Whitefish Lake Property Owners Association became a Non-profit Conservation Organization called the Whitefish Lake Conservation Organization (WILCO) in 2000. After the 1998–2001 USGS study, WILCO members continued to monitor the water quality of the lake as part of the WDNR's Wisconsin Citizen Lake Monitoring Network (Wisconsin Department of Natural Resources, 2008b). Results from the continued monitoring effort during 2001–04 documented degradation in water quality indicated by higher phosphorus and chlorophyll *a* concentrations, especially in 2002. During this time, the water level in Whitefish Lake, like that of many other lakes in the area (Robertson and others, 2009), was higher than normal. To determine if the change in water quality documented in Whitefish Lake was a natural phenomenon associated with the high water levels during the early 2000s or whether it was caused by changes in the watershed, WILCO organized a diverse task force of scientists to conduct an interdisciplinary examination of the lake. The investigation included an examination of the water quality and hydrology of the lake by the USGS, with emphasis on examining how changes in hydrology and water level affect the water quality of the lake.

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

Degradation in the water quality (trophic conditions) of many lakes is caused by an increase in nutrient loading from their watersheds. The effects of nutrient loading (input of nutrients over a specified period of time) are sufficiently understood, such that empirical eutrophication models have been developed that can predict in-lake phosphorus and 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 many lakes with very different nutrient loading rates and hydrologic conditions (for example, Vollenweider, 1975; Canfield and Bachmann, 1981); however, Lathrop and others (1998; 1999) also developed response models based on changes in nutrient loading to a specific lake—Lake Mendota, Wis.

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. Lathrop and others (1998) demonstrated that Lake Mendota, Wis., had a larger response to a series of wet or dry years than to a single unusual year. Therefore, it appears that it can take a few years for a lake to come in equilibrium with inputs from its watershed. After a series of wet years, it is also possible that new sources of nutrients develop, such as shoreline erosion caused by increased water level, increased input from wetland areas around a lake, increased nutrient release from nutrient-rich soils, or increased nutrient release from old septic systems flooded by rising groundwater levels in nearshore areas (Leira and Cantonati, 2008). Persistent fluctuations in precipitation and runoff can, therefore, result in changes in water level that can significantly affect the water quality of lakes.

Changes in water levels are natural phenomena for most lakes; however, it is believed that global climate change can and may have already affected the extent of water-level fluctuations in some lakes (Wantzen and others, 2008). Many scientists believe, based on results from General Circulation

Models, that climate change will either cause systematic decreases or increases in hydrologic budgets and 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). It is difficult to predict the effects of climate change on a particular lake from large global and regional models 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 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 circulation models indicate that seasonal precipitation patterns will change, resulting in more precipitation in winter and less in summer, which in turn may cause groundwater levels to increase and cause a rise in the water levels of lakes that have large groundwater inputs. Increases in air temperatures, however, may cause an increase in evapotranspiration and cause a decrease in water levels (Wittman, 2008).

Climate changes can cause increases in air temperatures that may affect lake water quality in other ways (Bates and others, 2008). Increases in air temperatures will increase water temperatures which in turn may stimulate algal blooms. Increases in air temperatures will also decrease the period of ice cover and, thus, increase the length of the growing season, which in turn may result in a longer period of algal and macrophyte growth. The longer growing season, however, may result in the available nutrients being depleted and fewer algal blooms later in the year. Only a few studies have examined how changes in water level affect the water quality of lakes (Leira and Cantonati, 2008), and these studies mostly examined changes in common ions and alkalinity (for example, Webster and others, 1996; White and others, 2008).

The recent degradation in water quality of Whitefish Lake that has occurred in conjunction with high water levels has led to several questions: Are the changes in the lake's water quality caused by changes in the watershed or are they caused by changes in water level and the factors causing the changes in water levels? Are the recent changes in water level caused by global climate change? Are the changes in water level and water quality indicative of what may occur in the future? It may be difficult to determine if climate change has caused the changes in water levels of Whitefish Lake or exactly what type of climatic changes may occur in the future; therefore, it is important to examine how conditions leading to lower and higher water levels affect the water quality of Whitefish Lake.

Several approaches can be used to predict how the water quality of a specific lake may respond to hydrologic changes and future climate change. One approach examines how the lake has responded to past changes in water level, and another approach uses numerical models to simulate how the lake should respond to specific climate scenarios. A third approach is to try to determine how the water quality of the lake responds to a range of conditions, such as conditions leading to low and high water levels.

Purpose and Scope

The primary purpose of this report is to provide a better understanding of what may be causing the apparent degradation in the water quality of Whitefish Lake detected by the ongoing Citizen Lake Monitoring Network and, therefore, provide information that can be used, in conjunction with information from other parts of the interdisciplinary project, to better understand and protect this relatively pristine lake in the face of increased developmental pressure and possible climatic changes. This report describes the water quality of Whitefish Lake, quantifies the water and phosphorus budgets for the lake based on data collected from November 2004 through October 2006, presents the results of model simulations that demonstrate the potential effects of changes in phosphorus inputs on the lake's water quality, reconstructs changes in the water level in the lake since 1900, and examines the relative importance of climatic changes (reflected as changes in water level and hydrologic changes in the watershed) and anthropogenic changes in the watershed to the water quality of the lake.

The water-quality data and water and phosphorus budgets described in this report improve the understanding of the hydrologic system of Whitefish Lake and aid in the understanding of how changes in the water levels and nutrient loading affect the water quality of seepage lakes. Results of the study, in conjunction with that from other parts of the interdisciplinary project, should be useful to local managers in the preparation of a comprehensive lake-management plan. In addition, results from this study add to the database of lakes for which detailed hydrologic, phosphorus loading, and lake water-quality information are available to compare with other lakes in the region.

Study Methods and Sampling Sites

Personnel from the USGS collected lake, groundwater, and meteorological data from 2004 to 2006 as part of this study and water-quality data for the lake from 1998 to 2001 as part of an earlier study. In addition, water-quality data for the lake were collected by volunteers from the community as part of the WDNR's Wisconsin Citizen Lake Monitoring Network (Wisconsin Department of Natural Resources, 2008b) from 2000 to 2008, and by the WDNR from 1973 to 1975 as part of a state-wide assessment of lakes (Wisconsin Department of Natural Resources, 1974; 1975). Citizen Lake Monitoring Network data were obtained from the WDNR's Web site (Wisconsin Department of Natural Resources, 2008b). All available data, but primarily data on near-surface concentrations of total phosphorus and chlorophyll *a*, and Secchi depths were used to describe the water quality of the lake and characterize long-term changes in water quality; however, only data collected from November 1, 2004, to October 31, 2006, were used to describe the hydrology and phosphorus inputs to the lake. This latter period was divided into two monitoring years (MY): November 2004 through October 2005 (hereafter referred to as MY 2005) and November 2005 through October 2006 (hereafter referred to as MY 2006).

Lake-Stage and Water-Quality Monitoring

A continuously recording (at 15-minute intervals) lake-stage gage was installed in 2 ft of water just north of a small peninsula in the South Basin of Whitefish Lake (fig. 1) and operated for the 2-year monitoring period. Lake stage was monitored by means of a gas-purge system with a pressure transducer.

Water-quality data for the lake were collected quarterly from the South Basin by the WDNR from 1973–75 as part of a statewide assessment (Lillie and Mason, 1983). From 2000 to 2008, Citizen Lake Monitoring volunteers from the community intermittently collected water-quality data including total phosphorus and chlorophyll *a* concentrations, and Secchi depths from the South Basin and Secchi depths from the North Basin from 2000 to 2008, in accordance with protocols described by Betz and Howard (2005). The USGS collected data from the deepest locations in the North and South Basins (fig. 1) from 1998 to 2001 and from 2005 to 2006 as part of this study. From 1998 to 2001, the USGS sampled the lake in late winter, spring, and monthly during summer (May through September); however, in 2005 and 2006 the lake was not sampled during the winter. During each visit, profiles of water temperature, dissolved oxygen, specific conductance, and pH were collected with a multiparameter data sonde, and water clarity was measured with a standard 8-in.-diameter black and white Secchi disk (Secchi depth). Lake-water samples were collected with a Van Dorn sampler and near-surface samples were analyzed for total phosphorus and chlorophyll *a* concentrations. Near-bottom samples were collected about 1 m above the sediment-water interface and were analyzed for total phosphorus concentrations. Water samples collected during spring were analyzed for nutrients, common ions, and other characteristics such as color, turbidity, alkalinity, total dissolved solids, and silica.

The Lake Studies Team of the USGS Wisconsin Water Science Center implements a quality-assurance plan that involves collecting three types of samples, which include blanks, replicates, and spikes, from a subset of the lakes studied each year (U.S. Geological Survey, Wisconsin District Lake Studies Team, 2006; 2007). During the monitoring period, the replicates for total phosphorus were within 0.006 mg/L over a wide range in concentrations, except for one replicate from a eutrophic lake that was within 0.02 mg/L, and the replicates for chlorophyll *a* were within 18 percent of each other. Concentrations of total phosphorus and chlorophyll *a* in the blanks analyzed in 2005 and 2006 were below the laboratory detection limits of 0.005 mg/L for total phosphorus and 0.26 µg/L for chlorophyll *a*.

Lake-stage and water-quality data collected by the USGS were published in the annual data report series “Water Quality and Lake-Stage Data for Wisconsin Lakes,” water years 1998–2001 and 2005–06 (U.S. Geological Survey, Wisconsin District Lake Studies Team, 1999–2002 and 2006–07).

Lake Classification

One method of classifying the water quality of a lake is by computing trophic-state-index (TSI) values based on near-surface concentrations of total phosphorus and chlorophyll *a*, along with Secchi depths, as developed by Carlson (1977). TSI values were developed to place these three characteristics on similar scales to allow comparison among 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 average summer (May–September) TSI values:

$$TSI_P = 4.15 + 14.42 \text{ [ln total phosphorus (in micrograms per liter)]}, \quad (1)$$

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

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

Oligotrophic lakes have TSI values less than 40, a limited supply of nutrients, very high clarity, low algal populations, and low phosphorus concentrations, and they generally contain oxygen throughout the year in their deepest zones (Carlson, 1977). Mesotrophic lakes have TSI values between 40 and 50, a moderate supply of nutrients, moderate clarity, a tendency to produce moderate algal blooms, and oxygen is occasionally depleted in the deepest zones of the lake. Eutrophic lakes have TSI values greater than 50, a large supply of nutrients, and severe water-quality problems—such as low clarity and frequent seasonal algal blooms—and oxygen is typically depleted in the deeper zones of the lake.

Lake-Sediment Analyses

A separate paleoecological study of the lake-bottom sediments of Whitefish Lake was done by the WDNR, Bureau of Integrated Science Services, to determine the ages of the sediment from specific depths and to quantify historical changes in phosphorus concentrations and sedimentation rates in the lake (Garrison, 2006). Sediments from deep regions of the North and South Basin (fig. 1) were extracted by use of a piston coring device in October 2004 and then analyzed to reconstruct the water-quality (phosphorus concentration) history of the lake by examining changes in the diatom assemblages at different depths in the core. Only information from the core collected in the South Basin is discussed in this report.

Meteorology and Atmospheric Deposition Monitoring

A continuously recording (at 15-minute intervals) meteorological station (fig. 2) was installed on the small peninsula in the South Basin of Whitefish Lake (fig. 1), and operated for the 2-year monitoring period. The meteorological station was installed in a small clearing on the peninsula. The meteorological station was equipped with an anemometer, a wind direction vane, a total radiation pyranometer, a relative humidity sensor, an air-temperature sensor, a precipitation tipping bucket that was used during nonfreezing periods, and a data recorder. Most of the instruments were 10 to 15 ft off the ground. All of the meteorological data were summed into daily average or total values. The size of the clearing was large enough that all of the instruments, except the anemometer, were minimally obstructed. The anemometer was obstructed by trees to the west of the station; therefore, wind data from the Minong weather station, located 6 mi southwest of the lake, were used as the primary data source for wind velocities. The Minong weather station is operated by the WDNR (James Barnier, Division of Forestry, unpublished data; *http://www.dnr.state.wi.us/forestry/fire/weather/*). Data for periods when data were missing from the lake meteorological station were estimated from data collected at the Minong weather station, and missing wind data at the Minong station were estimated from data collected at the lake meteorological station. During winter, precipitation was estimated by averaging the daily data from the National Weather Service (NSW) stations (National Climatic Data Center, 2004–2006) at Gordon (about 3 mi northeast of the lake) and Solon Springs (about 9 mi north of the lake).

Phosphorus deposition on Whitefish Lake was determined from phosphorus concentrations measured in wetfall (rain and snow) and phosphorus deposition rates measured in dryfall, both of which were measured with a 0.95 ft diameter stainless-steel collection bowl mounted on a table about 30 in. above ground at the South Basin weather station (figs. 1 and 2). Netting was placed over the bowl to prevent contamination by birds. Local observers serviced the monitoring station and collected wet samples and dry samples. Wet samples were collected after a period with rainfall or snowfall. For wet samples, the bowl was deployed within a few hours of an imminent precipitation event, and the samples were collected as soon as possible after the event to reduce evaporation and other contamination. Dry samples were those collected after several days, usually 7–10 days without precipitation; these samples were taken from the bowl by rinsing it twice with 125 mL of distilled water. Both types of samples were acidified and chilled before they were sent to the laboratory for analysis of total phosphorus. The stainless-steel bowl was cleaned with phosphorus-free soap before each deployment. Phosphorus dryfall deposition rates were computed by determining the mass of phosphorus deposited (dry sample concentration x 0.25 L) per unit area per day of deployment. Samples that included insects and other nondepositional material were discarded.

Outlet Monitoring

There is no permanent outflow from Whitefish Lake; however, intermittent flow does occur into Deborah Lake, a small wetland on the southeast side of lake (fig. 1). The outlet from Deborah Lake then drains into Leader Lake. Water levels (stage) in the stream between Deborah and Leader Lakes at South Bass Lake Road were measured by the USGS and a local observer whenever flow was present and were used to determine flow by use of standard stage-discharge relations (Rantz and others, 1982). The phosphorus concentration of the water leaving the lake was assumed to be equal to that measured in the South Basin of the lake during the period with flow.

Surface-Water Inflow

Daily flows from the area surrounding the lake (surface-water inflow) were estimated from runoff coefficients of 0.0015 for grass and developed areas and 0.00015 for forested areas (Graczyk and others, 2003), by use of the precipitation estimated for the area, and by assuming that 10 percent of the watershed was grass and developed areas and 90 percent was forested. Phosphorus loads from this nearshore area were approximated by multiplying the estimated flow by a phosphorus concentration estimated for runoff from the area near Whitefish Lake. Because no runoff was observed, the concentration in the runoff was estimated from an experiment conducted with soils collected from near the public boat landing and from a nearby hillside. About 3 in. of surface soil was carefully placed into an 8-ft long eaves trough to resemble the area from which it was collected. Distilled water was then passed over the soil and allowed to seep through the soil. The water that passed over the soil and that flowed through the soil (interflow) was collected separately at the downstream end of the trough and analyzed for total and dissolved phosphorus. The overland flow was in contact with the soil for approximately 2 minutes and the interflow was in contact for at least 5 minutes. Total phosphorus concentrations were similar for both areas; however, overland flow had lower concentrations than interflow; the concentration in overland flow was 1.6 mg/L and interflow was 3.0 mg/L. Approximately 95 percent of the phosphorus was in the dissolved form for overland flow and interflow; therefore, the phosphorus concentrations were not significantly affected by particulate matter that was washed down the eaves trough.

Figure 2. Meteorological station installed on the southwest shore of Whitefish Lake, Douglas County, Wis. The location of the meteorological station is identified on figure 1. The atmospheric deposition collector is shown on the right side of the photo.

Groundwater Monitoring

Twenty nearshore piezometers (0.5-in. diameter observation wells) were installed around the periphery of the lake (fig. 1) to help define areas contributing groundwater to the lake, to determine the phosphorus concentration in the groundwater entering the lake, and to help quantify the phosphorus loading from groundwater. These piezometers were installed approximately 3 ft below the water table, to depths of 4 to 6 ft. Water levels in the piezometers were measured eight times (four times during MY 2005 and MY 2006) and used to determine groundwater gradients from the differences in water elevation in the piezometers and elevation of the lake surface. Dissolved phosphorus concentrations in the groundwater were measured once in 2004 (Nov.), twice in 2005 (May and Aug.), and once in 2006 (May) for each piezometer.

All lake, groundwater, and atmospheric-deposition samples were analyzed by the Wisconsin State Laboratory of Hygiene in accordance with standard analytical procedures described in the "Manual of Analytical Methods, Inorganic Chemistry Unit" (Wisconsin State Laboratory of Hygiene, 1993).

Simulation of Groundwater Flow near Whitefish Lake

The two-dimensional, analytic-element, steady-state, groundwater-flow modeling computer program, GFLOW (Haitjema, 1995) was used to estimate groundwater contributing areas around Whitefish Lake and to estimate exchanges of groundwater and lake water. A complete description of the use of analytic elements to simulate groundwater flow is beyond the scope of this report; however, a brief description is given below. A complete discussion of the methods for applying the model can be found in Strack (1989) and Haitjema (1995).

An infinite aquifer is assumed in analytic-element modeling. In GFLOW, the study area (domain) does not require a grid or involve interpolation between cells. When an analytic-element model is constructed, features that affect groundwater flow, such as surface-water bodies, aquifer characteristics, and recharge, are entered as mathematical elements or strings of elements. The amount of detail specified for the features depends on the distance from the area of interest. Each element is represented by an analytic solution. The effects of these individual solutions are added together to arrive at a solution for the groundwater-flow system. Because the solution is not confined to a grid, groundwater levels (heads) and flows can be computed anywhere in the study area without averaging values at specific locations in the model (nodal averaging). In the GFLOW model, the analytic elements are two dimensional and are used to simulate only steady-state conditions (no change in water levels over time). Natural groundwater-flow systems are three dimensional; however, two-dimensional models can provide reasonable approximations of groundwater flowlines when the lengths of the flowlines are long compared to the aquifer thickness (Haitjema, 1995, p. 23). In the study area, most groundwater was assumed to move through unconsolidated sand deposits that have a maximum saturated thickness of 300 ft or less. The lengths of the flowlines from recharge areas to discharge areas are typically several thousand feet or more in length.

The GFLOW model for the study area extended up to about 30 mi around Whitefish Lake (fig. 3). The geometry of the single-layer model included a bottom elevation set at 850 ft and an average aquifer thickness of about 200 ft, which were based on well logs for the study area and additional information from Clayton (1984). The sand deposits were assumed to have uniform hydraulic conductivity; however, a reduced transmissivity zone (transmissivity is the product of saturated thickness and hydraulic conductivity) was simulated along the St Croix River downstream of the St. Croix Flowage (fig. 3) where the aquifer is very thin and relatively impermeable crystalline bedrock is near the land surface (Clayton, 1984). In the model, the reduction in transmissivity was approximated by reducing the hydraulic conductivity in the zone instead of raising the base of the model to improve the model's solution stability. Hydraulic conductivity was adjusted as part of the calibration process. Surface-water features such as lakes and streams were simulated using several types of linesink elements. The model includes "far-field" and "near-field" sources and sinks of water, which are collectively referred to as "linesinks" (fig. 3). The far-field area surrounds the near-field area of interest. In the far field, streams are simulated as coarse linesinks having little or no resistance between the surface-water feature and the groundwater-flow system. The far field is simulated so that the model explicitly defines the regional groundwater-flow field near the area of interest. The near field represents the area of interest, and it includes Whitefish Lake and other adjacent lakes and streams that are in close hydraulic connection with the groundwater-flow system. Base flows measured in near-field streams were used to calibrate the model (flux calibration); flows in far-field streams were not simulated. Recharge was applied uniformly across the model area, while the recharge rate was adjusted during model calibration (described below). No high-capacity groundwater pumping wells were close enough to Whitefish Lake to have an expected influence on water levels near the lake; therefore, no pumping wells were simulated in the model. Domestic pumping wells were not included in the model because the withdrawal from these wells is widely distributed and the withdrawals are relatively small (especially where balanced by return flow from septic systems).

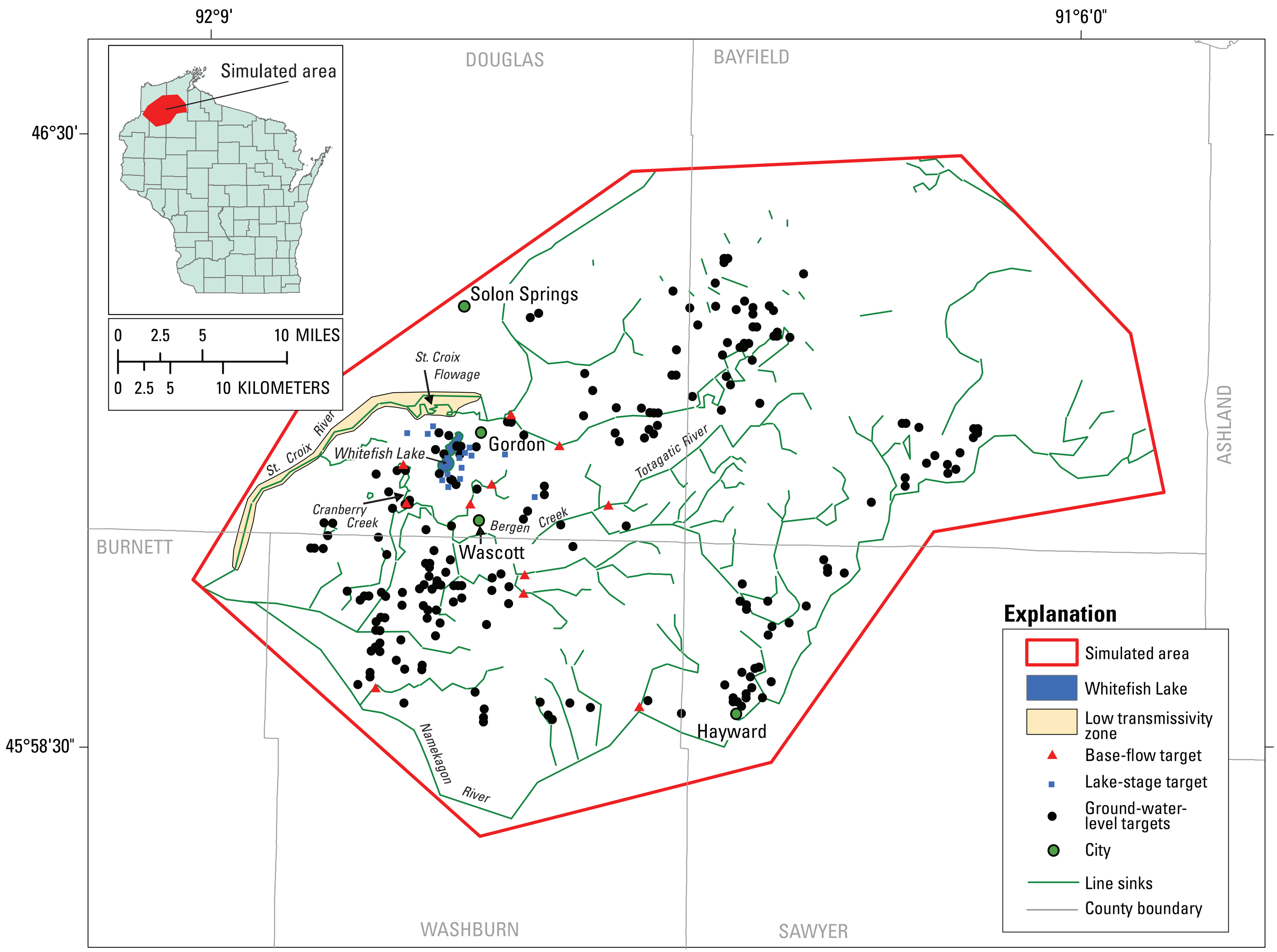


Figure 3. Hydrologic features simulated with the GFLOW groundwater-flow model near Whitefish Lake, Douglas County, Wis.

The stage of Whitefish Lake was simulated as a steady-state value, even though there was a decrease in stage of about 0.5 ft during the simulation period. Inputs to the model were determined for MY 2005, because the water level of the lake changed less dramatically in MY 2005 than in MY 2006 and the precipitation in MY 2005 was closer to the long-term average than in MY 2006. Small cyclic fluctuations tend to have minimal effect on regional groundwater flow patterns, and the net gains and losses to the groundwater-flow system immediately adjacent to the lake are expected to be approximated by steady-state conditions for a typical year. Whitefish Lake, itself, was simulated with a special type of linesink with resistance, called a “lake element,” by use of the methodology described by Hunt and others (2003). The lake element solves for lake stage based on estimated and simulated water inflows to the lake (precipitation and groundwater seepage) and outflows from the lake (evaporation and groundwater seepage). Small lakes adjacent to Whitefish Lake were simulated as high hydraulic conductivity features (Hunt and others, 2003). These representations of lakes allowed the model to simulate the stage of nearby lakes and both the stage and water balance of Whitefish Lake directly, which allowed for comparison with measured lake levels during calibration. Recharge to the high hydraulic conductivity lakes was estimated as total precipitation minus evaporation (approximately 5.4 in/yr) from November 2004 to October 2005 (estimated in the “Hydrology and Water Budget” section below). Streambed properties were represented by resistance along a linesink. Resistance is computed by dividing the bed-sediment thickness by the vertical hydraulic conductivity. For this model, resistance was set equal to 0.5 ft/(ft/d) for near-field streams and for Whitefish Lake. These values indicate a close hydraulic connection with the groundwater-flow system and are similar to sediment-resistance values used in other nearby GFLOW models (Juckem and Hunt, 2007; Hunt and others, 1998; Graczyk and others, 2003). The assigned stream widths ranged from 10 to 50 ft.

Groundwater levels in 204 wells and elevations in 15 lakes in the area, base-flow measurements at 4 locations on Cranberry and Bergen Creeks, and historical base-flow estimates at 7 streamflow gaging stations in the near field (fig. 3) were used to calibrate the groundwater-flow model. In addition, measured hydraulic gradients from the 20 nearshore shallow piezometers around the lake were used to visually evaluate the extent of the simulated groundwater capture zone along the lake shoreline. The GFLOW model was calibrated by varying horizontal hydraulic conductivity and groundwater recharge within an expected range until there was a reasonable match between measured and simulated lake levels, groundwater levels, and base flows for streams in the near field (fig. 4). Areal recharge to the groundwater-flow system was estimated to be 11 in/yr, and horizontal hydraulic conductivity was estimated to be 45 ft/d for the calibrated model. These values are within the range of values used to simulate groundwater flow in Bayfield County (Lenz and others, 2003) and Sawyer County (Juckem and Hunt, 2007). The stage of Whitefish Lake was simulated within 0.5 ft of the average stage during MY 2005. The mean error (a measure of model bias) of all groundwater-level targets is -5 ft (a negative value indicates that measured values were less than simulated values); the mean absolute difference is 15 ft. These values represent about 1–3 percent of the total range in observed water levels across the modeled area. Simulated base flows were within 0.8 ft^3/s (8 percent) of four synoptic base-flow measurements made along Cranberry and Bergen Creeks during MY 2005. Simulated base flows were within 8 ft^3/s (71 percent) of the historical base-flow estimates at the seven near-field gaging stations.

Water Quality of Whitefish Lake

Temperature profiles collected in the North and South Basins demonstrated that strong thermal stratification developed during summer and thus limited the extent of vertical mixing during this period. In both basins, the maximum temperature in the upper layer of the lake (epilimnion) reached about 26–27°C in early August. Thermoclines in each basin developed in May and reached a depth of about 26 to 40 ft in July. During summer, near-bottom temperatures reached about 11–12°C in the North Basin and about 7–8°C in the South Basin. Cooler temperatures in the South Basin result from that basin stratifying earlier when the water temperatures are cooler, and temperatures increase at a slower rate because the South Basin is deeper and its hypolimnion has a larger volume than the North Basin.

Dissolved oxygen concentrations of about 10–11 mg/L were near saturation throughout the lake during nonstratified periods in spring and about 8 mg/L throughout the epilimnion during summer. Once stratification developed, dissolved oxygen concentrations in the lower layer of the lake (hypolimnion) began to decrease. Near-bottom concentrations reached 0 mg/L by early-to-mid July in both basins. The maximum extent of anoxic conditions (0 mg/L) occurred in September, when anoxia occurred from the bottom to about 36 ft from the surface in the North Basin and 60 ft from the surface in the South Basin. A larger part of the hypolimnion in the North Basin became anoxic than in the South Basin, which is a result of the smaller hypolimnetic volume in the North Basin.

Dissolved oxygen concentrations throughout the water column of the lake were measured in 1973–75, 1998–2001, and in 2005–06. Dissolved oxygen profiles collected during mid-to-late summer demonstrated interannual variability, but no long-term changes in the extent of anoxia in the bottom water since the 1970s.

Whitefish Lake is a soft-water lake, and its alkalinity is less than 20 mg/L as calcium carbonate. Specific conductance typically is about 35–40 µS/cm throughout the lake; however, when the deep areas became anoxic, specific conductance increased to about 100 µS/cm near the bottom of the lake. The pH in the lake typically was about 7 standard units.

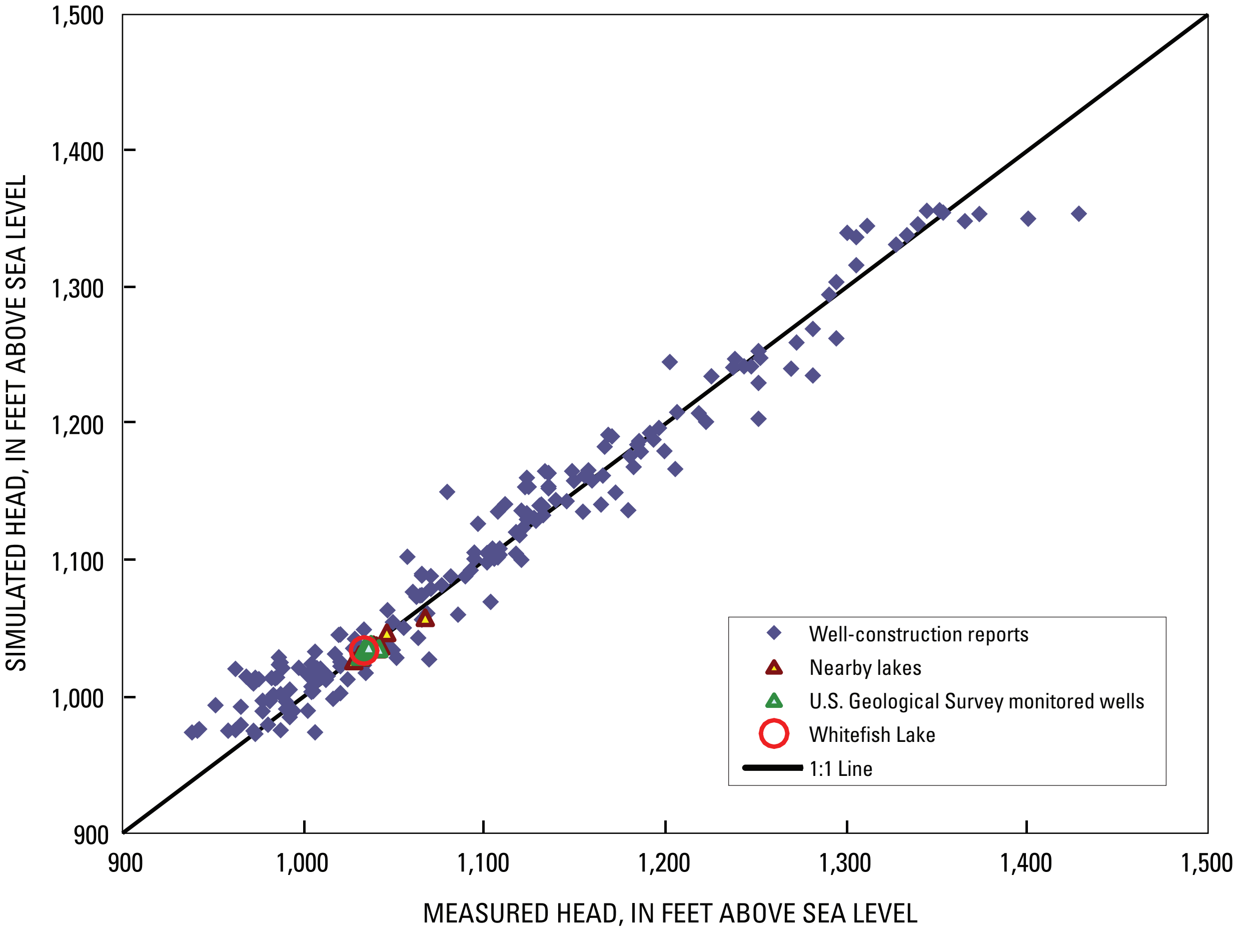


Figure 4. Measured and simulated heads for the calibrated GLFLOW model for the area near Whitefish Lake, Douglas County, Wis.

Water chemistry in Whitefish Lake is different from that in most northwestern Wisconsin lakes, as described by Lillie and Mason (1983); the Whitefish Lake has softer water and lower concentrations of most constituents than the other lakes. Lillie and Mason collected data from a random set of about 660 Wisconsin lakes, 282 of which were in northwestern Wisconsin. The average concentrations for the northwestern Wisconsin Lakes were 7 mg/L for calcium, 3 mg/L for magnesium, 4 mg/L for chloride, and 27 mg/L for alkalinity, compared to 4.9, 1.3, 0.9, and 17 mg/L, respectively, for Whitefish Lake. The water color in Whitefish Lake ranged from below detection to 10 platinum-cobalt units (based on two measurements), which is less colored than the average value of 30 for northwestern Wisconsin lakes.

Phosphorus and nitrogen are essential nutrients for plant growth, and they are the nutrients that usually limit algal growth in most lakes. High nutrient concentrations can cause high algal populations (blooms); therefore, increases in nutrient inputs can be the cause of accelerated eutrophication (that is, accelerated aging and increased productivity) of lakes. Near-surface concentrations of total phosphorus less than about 0.012 mg/L indicate oligotrophic conditions, whereas concentrations greater than about 0.024 mg/L indicate eutrophic conditions (Carlson, 1977). During MY 2005 and MY 2006, concentrations ranged from less than the 0.005-mg/L detection limit to 0.018 mg/L measured in October 2006 (fig. 5). Near-surface total phosphorus concentrations were similar throughout the lake, and they had little seasonality. The near-surface summer (May–September) average phosphorus concentration in the South Basin was 0.008 mg/L in MY 2005 and 0.010 in MY 2006 (table 2). Therefore, based on total phosphorus concentrations, Whitefish Lake would usually be classified as oligotrophic.

Since 1998, near-surface total phosphorus concentrations have ranged from less than the 0.005-mg/L detection limit to about 0.018 mg/L and were similar between the North and South Basins (fig. 5). Concentrations were consistently at or near the detection limit in 1998–99, but they increased in recent years to about 0.010–0.012 mg/L except for two higher concentrations measured in the late summer of 2006. Near-surface, summer average concentrations in both basins gradually increased from the detection limit to about 0.008–0.010 mg/L in 2006, before decreasing slightly in 2007 and 2008 (table 2).

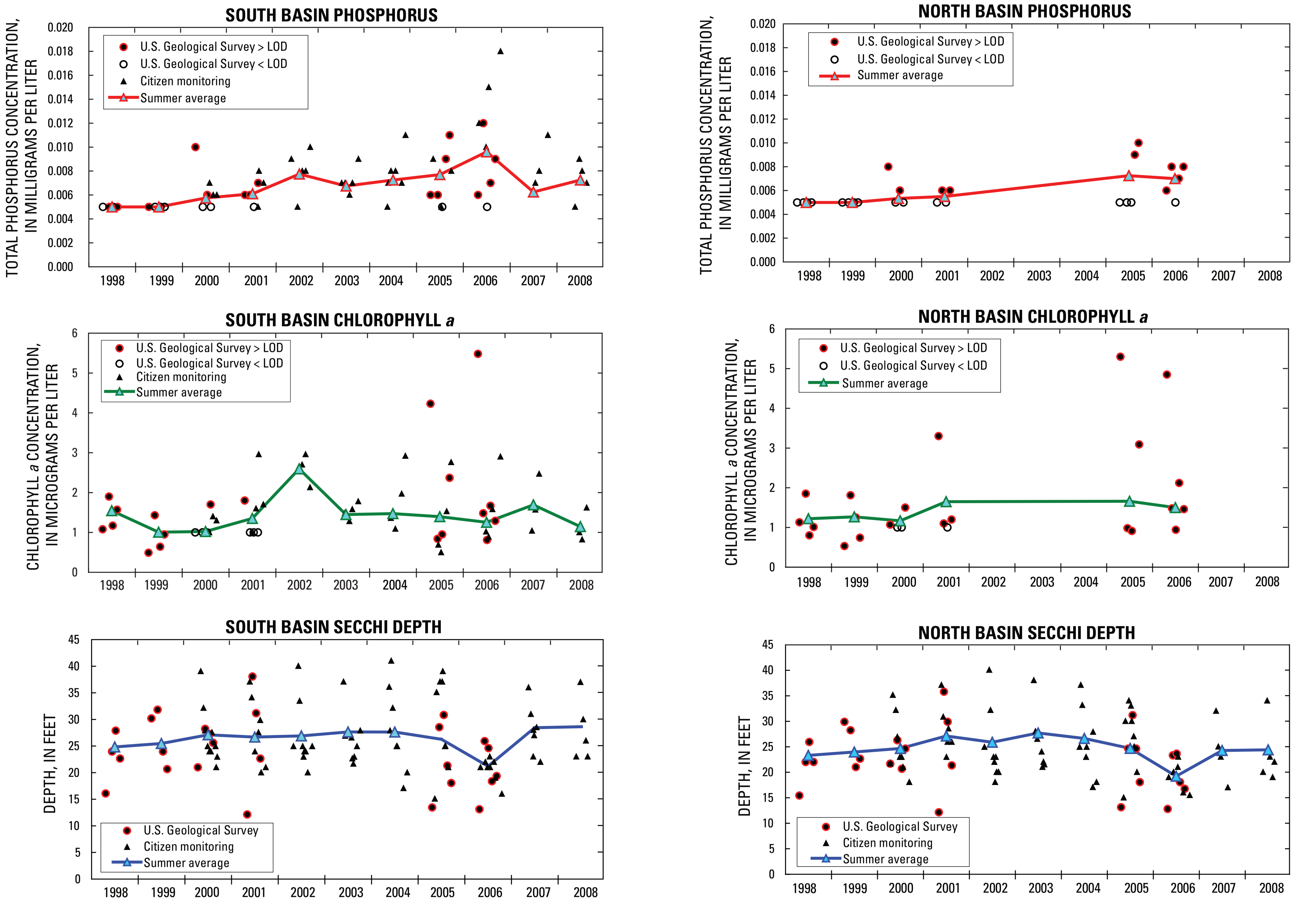


Figure 5. Near-surface total phosphorus concentrations, chlorophyll *a* concentrations, and Secchi depths in the South and North Basins of Whitefish Lake, Douglas County, Wis., 1998 to 2008.

Table 2. Near-surface summer average (May through September) water quality and trophic-state-index (TSI) values for Whitefish Lake, Douglas County, Wis.

[mg/L milligrams per liter; µg/L, micrograms per liter]

Year	Total phosphorus (mg/L)	Chlorophyll *a* (µg/L)	Secchi depth (feet)	Trophic-state-index values		
				Phosphorus	Chlorophyll *a*	Secchi depth
1973[a]	0.020	--	13.0	47	--	40
1974[b]	.013	--	13.0	37	--	40
1975[c]	.020	--	13.0	47	--	40
1998	.005	1.5	24.8	27	35	31
1999	.005	1.0	25.5	26	30	31
2000	.006	1.0	27.1	29	32	30
2001	.006	1.4	26.7	30	34	31
2002	.008	2.6	26.9	33	40	30
2003	.007	1.5	27.6	32	34	29
2004	.007	1.5	27.6	33	34	30
2005	.008	1.4	26.3	33	32	31
2006	.010	1.3	21.2	36	33	33
2007	.006	1.7	28.4	30	35	29
2008	.007	1.1	28.6	32	32	29
Average 1998–99	.0050	1.3	25.2	26	32	31
Average 2002–04	.0073	1.8	27.4	32	36	30
Average 2005–06	.0087	1.3	23.7	34	32	32
Average 2007–08	.0068	1.4	28.5	31	33	29
Average 1998–2007	.0067	1.5	26.2	31	34	30

[a]Averages are based on one sample date (Wisconsin Department of Natural Resources) in July (1973).

[b]Averages are based on two sampling dates (Wisconsin Department of Natural Resources) in May and August (1974).

[c]Averages are based on one sample date (Wisconsin Department of Natural Resources) in May (1975).

Slightly higher concentrations were measured in 2002 and 2006 compared to other years. In the early 1970s, summer average concentrations of 0.013 and 0.020 mg/L were determined from measurements by the WDNR (table 2); however, the accuracy of these data were unable to be determined.

Near-bottom total phosphorus concentrations increased after the near-bottom water became anoxic in late summer and late winter (fig. 6). Near-bottom concentrations were usually a little higher in the South Basin than in the North Basin. Maximum concentrations of about 0.15 to 0.20 mg/L were usually measured in late summer (September) and late winter (March). The increase in phosphorus concentrations just above the bottom indicates that there is a release of phosphorus from the deep sediments of the lake (internal phosphorus loading); however, because of the strong thermal stratification, this phosphorus in the deepest areas is not released into the surface water until late fall and near ice break up.

Maximum annual near-bottom total phosphorus concentrations have been relatively constant from 1998 to 2006, reaching about 0.15 to 0.20 mg/L in late summer and late winter. Near-bottom total phosphorus concentrations were not available in late summer for the 1973–75 samplings; however, the concentrations measured in July and early August were similar to those collected in similar months in 1998–2006. Therefore, there appears to be no long-term trend in near-bottom phosphorus concentrations or internal loading since 1973.

Near-surface total nitrogen concentrations (computed as the sum of Kjeldahl nitrogen and dissolved nitrite plus nitrate) were about 0.2 mg/L. Kjeldahl nitrogen represented most of the total nitrogen. These values may be a little lower than those measured between 1998 and 2001, but all concentrations were near the detection limit.

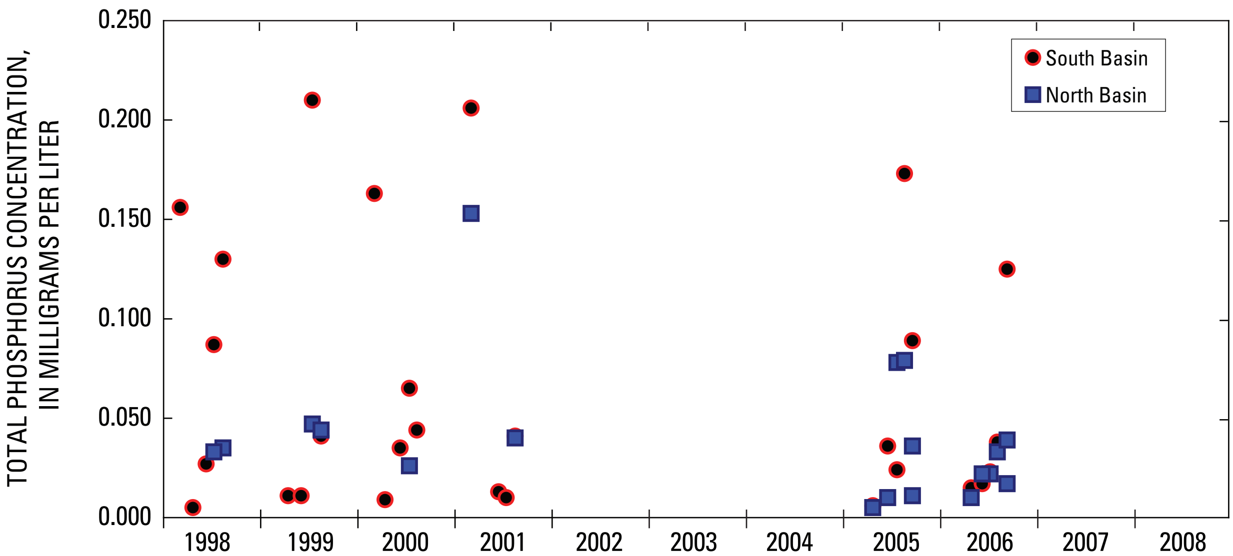


Figure 6. Near-bottom total phosphorus concentrations in the South and North Basins of Whitefish Lake, Douglas County, Wis., 1998 to 2006.

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 in a lake. The specific value of this ratio 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 the 1998–2001 data ranged from 45:1 to 68:1, and ratios for the 2005–06 data ranged from 32:1 to 40:1. This indicates that phosphorus should typically be the potentially limiting nutrient, and it should be the nutrient of concern when management efforts to improve or prevent degradation of the water quality in Whitefish Lake are considered.

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. An average concentration less than 2.6 µg/L indicates oligotrophic conditions, and an average concentration greater than about 7 µg/L indicates eutrophic conditions. During MY 2005 and MY 2006, concentrations ranged from less than 1 µg/L to 5–6 µg/L (fig. 5). The highest concentrations were measured in early spring; concentrations during summer were almost always less than 3 µg/L. The near-surface summer average chlorophyll *a* concentration in the South Basin was 1.4 µg/L in MY 2005 and 1.3 µg/L in MY 2006 (table 2). Based on chlorophyll *a* concentrations, Whitefish Lake would be classified as oligotrophic based on all June through August samples.

Since 1998, near-surface chlorophyll *a* concentrations have ranged from less than 1 µg/L to about 6 µg/L (fig. 5); however, concentrations seldom exceeded 3 µg/L during summer. Chlorophyll *a* concentrations may have increased slightly since 1998; no concentrations greater than 2 µg/L were measured until after 2001. Near-surface, summer average concentrations, however, have remained stable at about 1.5 µg/L except for the slightly higher concentration of 2.6 µg/L in 2002 (table 2).

During MY 2005 and MY 2006, Secchi depths in Whitefish Lake ranged from 13 ft in April 2005 to about 39 ft in July 2005 (fig. 5). Other than the poorer water clarity in spring, little consistent seasonal or spatial variability was apparent. The summer average Secchi depth in the South Basin was 26.3 ft in MY 2005 and 21.2 ft in MY 2006 (table 2). Since 1998, Secchi depths have changed very little. The poorest clarity was measured in 2006, when the summer average depth was 21.2 ft, and the best clarity was measured in 2007 and 2008, when the summer average depth was about 28.5 ft. Secchi depths measured in the lake during 1973–74, when the summer average depth was 13 ft, were much shallower than they were after 1998, which is consistent with the higher total phosphorus concentrations measured in the early 1970s. Secchi depths greater than 13 ft indicate oligotrophic conditions and those less than 6.6 ft indicate eutrophic conditions; therefore, based on Secchi depths, Whitefish Lake is classified as oligotrophic.

All three TSIs, based on near-surface concentrations of total phosphorus and chlorophyll *a*, and on Secchi depths, indicate that, in MY 2005 and MY 2006, Whitefish Lake was typically oligotrophic except during spring and fall when it was classified as mesotrophic based on chlorophyll *a* concentrations (fig. 7 and table 2). In general, TSI values were consistent among the three parameters except during spring and fall.

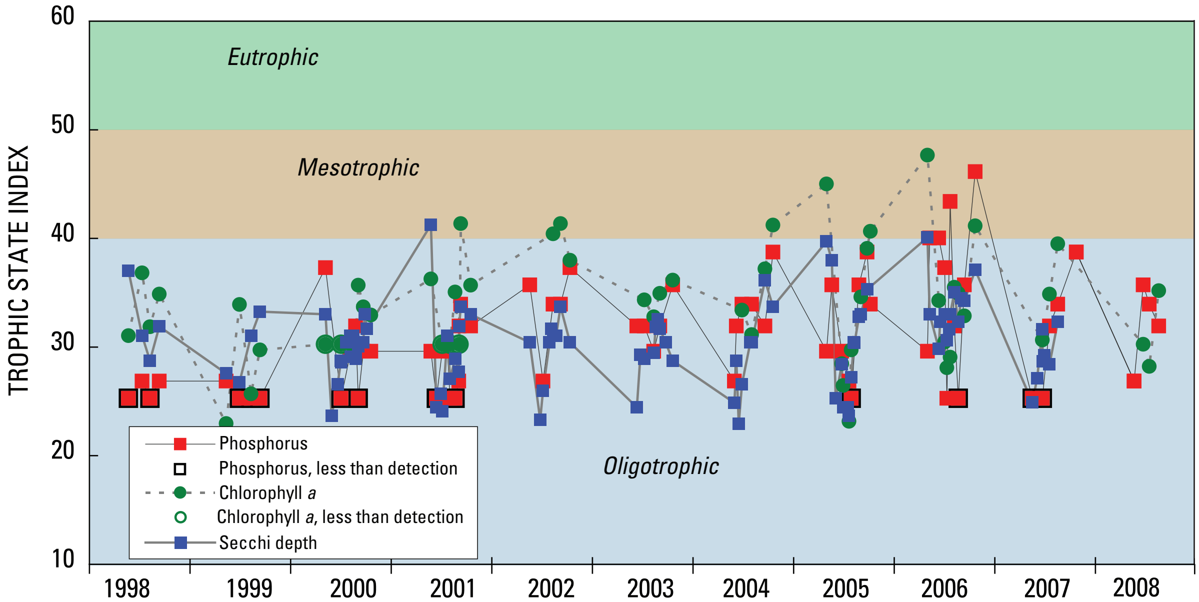


Figure 7. Trophic-state-index (TSI) values based on surface concentrations of total phosphorus and chlorophyll *a*, and Secchi depths, in the South Basin of Whitefish Lake, Douglas County, Wis., 1998 to 2008.

Inferred Lake Water Quality from Lake-Sediment Analyses

Results of analyses of the sediment cores extracted from the deep area of the South Basin in October 2004 were used to quantify changes in sedimentation rates and reconstruct historical changes in the water quality of Whitefish Lake (Garrison, 2006). Only data from the South Basin were used because data from the North Basin may have been affected by local changes in macrophyte densities (P. Garrison, Wisconsin Department of Natural Resources, personal commun., 2007). The average annual sedimentation rate for Whitefish Lake was 2.7 x 10^{-6} lb/in^2 (0.008 g/cm^2), which was the lowest rate measured in the 42 lakes examined by the author. The annual sedimentation rate steadily decreased from about 2.7 x 10^{-6} lb/in^2 (0.008 g/cm^2) in 1920 to about 1.7 x 10^{-6} lb/in^2 (0.005 g/cm^2) in 1976, and then increased back to 2.7 x 10^{-6} lb/in^2 near the top of the core (2002–04).

Changes in the diatom assemblages in the lake sediments were used to infer changes in the phosphorus concentrations in the water column of Whitefish Lake from 1800 to 2004 (fig. 8). The inferred value of 0.013 mg/L for 2002–04 at the top of the core overestimated the measured phosphorus concentration of 0.007 mg/L for this period (table 1); therefore, all of the values were adjusted for this bias by use of a constant percent-bias correction factor (fig. 8). On the basis of changes in the diatom assemblages, there has been very little long-term change in the phosphorus concentrations in the pelagic area of the lake. The adjusted in-lake phosphorus concentrations were relatively constant at about 0.007 mg/L (similar to the present value) from 1800 to 1880, and then increased slightly before decreasing to about 0.006 mg/L in 1910. Since 1910, phosphorus concentrations have fluctuated, but they have gradually increased to the current value of 0.007 mg/L, which is similar to that measured in 1800.

Hydrology and Water Budget

Because the productivity in Whitefish Lake should be limited or potentially limited by the input of phosphorus (based on N:P ratios), phosphorus should be the nutrient of concern when management efforts to improve or prevent degradation of the water quality of Whitefish Lake are considered. Most 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. Water budgets for the lake were quantified for each of the two monitoring years.

The hydrology of Whitefish Lake can be described in terms of components of its water budget. The water budget for a period of interest may be represented by

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

where ΔS is change in the volume of water stored in the lake, which 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}, limited runoff may occur from nearshore areas), and groundwater inflow (GW_{In}). Water leaves the lake through evaporation (E), groundwater outflow (GW_{Out}) and occasionally through surface-water outflow (SW_{Out}). Each term of the water budget was measured or estimated on a daily basis. Evaporation and groundwater often are minor components for many lakes;

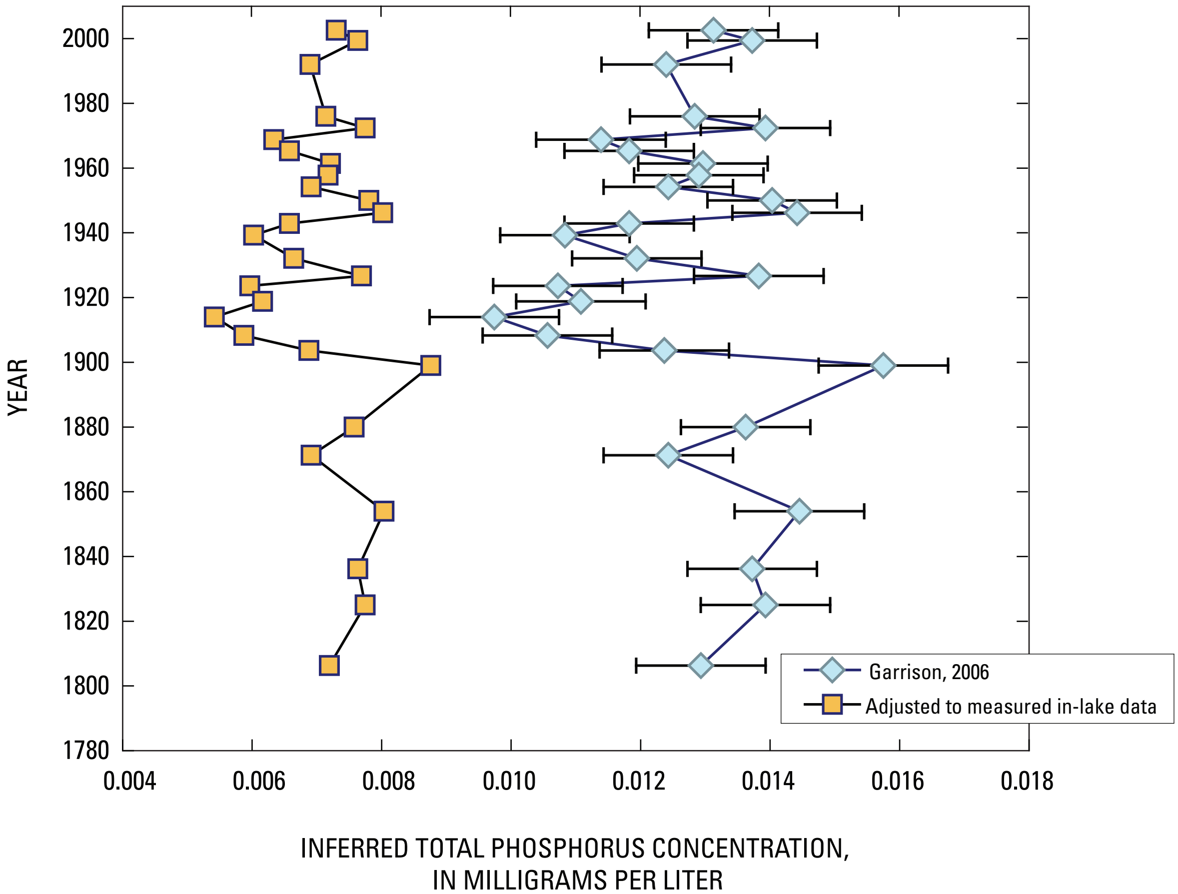


Figure 8. Estimated changes in summer total phosphorus concentrations based on changes in the diatom community in the sediment cores from Whitefish Lake, Douglas County, Wis., (Garrison, 2006), and inferred concentrations adjusted to that measured from 2002 to 2004.

however, they may be very important for a seepage lake, like Whitefish Lake. Therefore, evaporation and groundwater exchange were examined in detail with numerical models.

Change in Storage

Changes in the storage (volume) of the lake were determined from water-surface elevations measured continuously at the lake-stage gage near the southwest side of the lake (fig. 1). During the 2-year study period, lake stage declined (fig. 9) by 0.44 ft during MY 2005 and 1.10 ft during MY 2006. The highest stage recorded was 5.06 ft on June 14–15, 2005, and the lowest stage recorded was 3.15 ft on the last day of monitoring, October 31, 2006. Lake stage generally increased from late fall through early spring and declined during the summer months of both years. To simplify computations of the changes in lake volume, an average lake area of 833 acres was used in MY 2005 and 823 acres in MY 2006. The amount of water stored in the lake decreased by 314 acre-ft during MY 2005 and by 962 acre-ft during MY 2006 (table 3).

Precipitation

Precipitation on the lake surface during MY 2005 was 28.4 in. (1,970 acre-ft) and during MY 2006 was 21.8 in. (1,500 acre-ft; table 3). Precipitation was 91 percent of the long-term average for the period 1971–2000 at Gordon, Wis., during MY 2005 and was 70 percent of the long-term average during MY 2006. During MY 2005, precipitation totals for all months were below normal, except for December, June, September, and October (fig. 10). During MY 2006, totals for most months were below normal, except for November, December, August, and October.

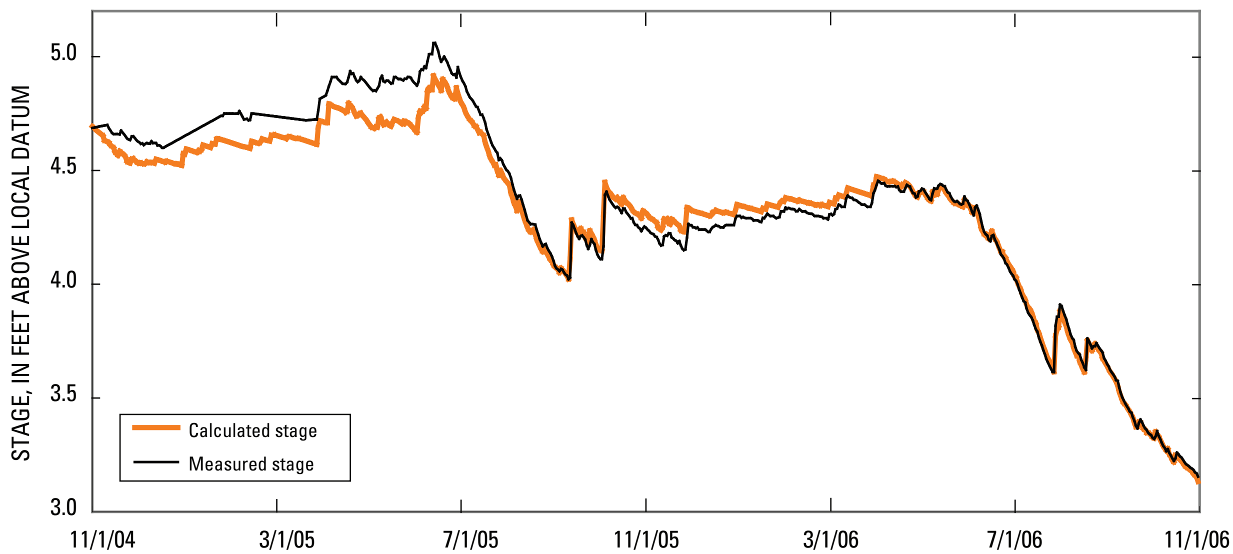


Figure 9. Measured and computed daily water levels (stages) from the full water budget of Whitefish Lake, Douglas County, Wis., November 1, 2004, to October 31, 2006. [Altitude of the local datum is 1,029.34 feet above the National Geodetic Vertical Datum of 1929.]

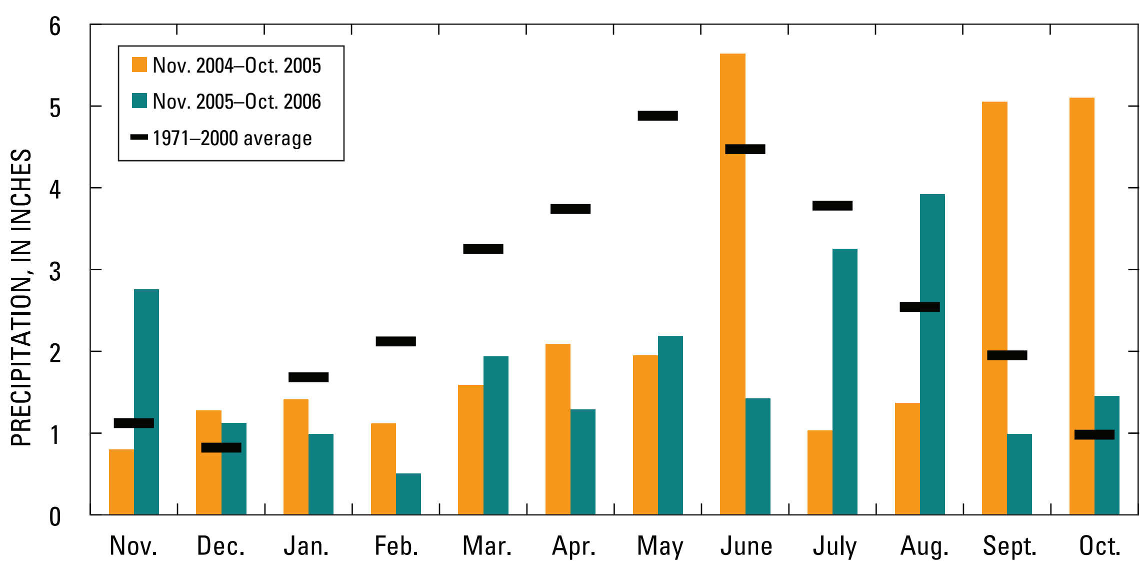


Figure 10. Precipitation at Whitefish Lake, Douglas County, Wis., November 2004 to October 2006, relative to long-term average precipitation at Gordon, Wis. [Altitude of the local datum is 1,029.34 feet above the National Geodetic Vertical Datum of 1929.]

Table 3. Water budget for Whitefish Lake, Douglas County, Wis., for monitoring years 2005 and 2006.

[MY, monitoring year, November–October; percent, percentage of total]

Budget component	MY 2005 (acre-feet)	MY 2005 (percent)	MY 2006 (acre-feet)	MY 2006 (percent)	Two-year average (acre-feet)	Two-year average (percent)
			Change in storage			
Change in storage	-314		-962		-638	
			Inputs to lake			
Precipitation	1,970	50.8	1,500	44.0	1,730	47.6
Surface-water inflow	.90	.02	.70	.02	.80	.02
Groundwater	1,910	49.2	1,900	56.0	1,910	52.4
Total input	**3,880**	**100**	**3,400**	**100**	**3,640**	**100**
			Outputs from lake			
Evaporation	1,600	38.0	1,800	41.2	1,700	39.7
Surface-water outflow	20.2	.48	.00	.00	10.1	.24
Groundwater	2,580	61.5	2,580	58.8	2,580	60.1
Total output	**4,200**	**100**	**4,380**	**100**	**4,290**	**100**

Evaporation and Net Groundwater Inflow

Because Whitefish Lake is larger and deeper than many nearby lakes, its water temperature increases more slowly in spring and decreases more slowly in fall than most other lakes; therefore, rates of evaporation from the surface of the lake may differ from those of other lakes. In addition, evaporation varies annually depending on meteorological conditions. Therefore, misleading information may result from the use of evaporation rates for other nearby lakes or estimation of evaporation from nearby evaporation-pan data that are typically used to estimate evaporation for small lakes. As an alternative for this study, daily evaporation was estimated for the 2-year detailed monitoring period by use of a one-dimensional hydrodynamic model, the Dynamic Lake Model (DLM; McCord and others, 2000; Robertson and others, 2002). In the process of simulating the hydrodynamics of the lake, the model computes detailed energy and hydrologic budgets, including the heat transfer associated with evaporation. Therefore, if the detailed energy budget is accurate, then the amount of evaporation should be correctly estimated. Net groundwater contributions to the lake also can be estimated by use of DLM by examining the discrepancies in the modeled and measured lake levels. Because Whitefish Lake consists of two basins, DLM was used to simulate the hydrodynamics of only the South Basin, but the results were extrapolated to the entire lake.

DLM is a 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 and have been successfully used to simulate vertical changes in water temperatures and changes in the timing of ice cover of many lakes and reservoirs around the world. DLM is 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 the DLM model and its application to estimate evaporation rates, see Robertson and others (2002).

Inputs to DLM included data describing the morphometry of the South Basin, meteorological conditions, average light extinction in the lake, and initial water temperature and salinity (conductivity) profiles in the lake; inflow and outflow were not included in the simulations for Whitefish Lake. Daily average air temperature, vapor pressure, wind speed, and inflow characteristics (temperature and specific conductivity/salinity) are used in the model, whereas daily total values for inflows, outflows, long-wave radiation (or average percentage of clear sky), short-wave radiation, rainfall, and snowfall are used. Meteorological data were obtained from the meteorological station in the South Basin and were supplemented with data from the stations operated by the WDNR and the NWS (see the "Meteorology and Atmospheric Deposition Monitoring" section). An average Secchi depth of 23 ft, the average during the open-water period during the 2-year simulation period, was used to estimate a light-extinction coefficient of 0.073 ft^{-1}. Water temperature and conductivity profiles measured in the South Basin on April 20, 2005, were used as the initial conditions for the simulation. Daily evaporation rates, prior to the start of the simulation, were assumed to be equal to the average of that estimated in 2005 and 2006 for each calendar day. It was assumed that no evaporation occurred when the lake was frozen.

To demonstrate that DLM accurately simulated the energy balance of the lake, the simulated thermal structure of the lake was compared with that measured (fig. 11). During the entire simulation period, the model accurately simulated the changes in the thermal structure of the lake, specifically the onset of stratification, surface and deep temperatures, and

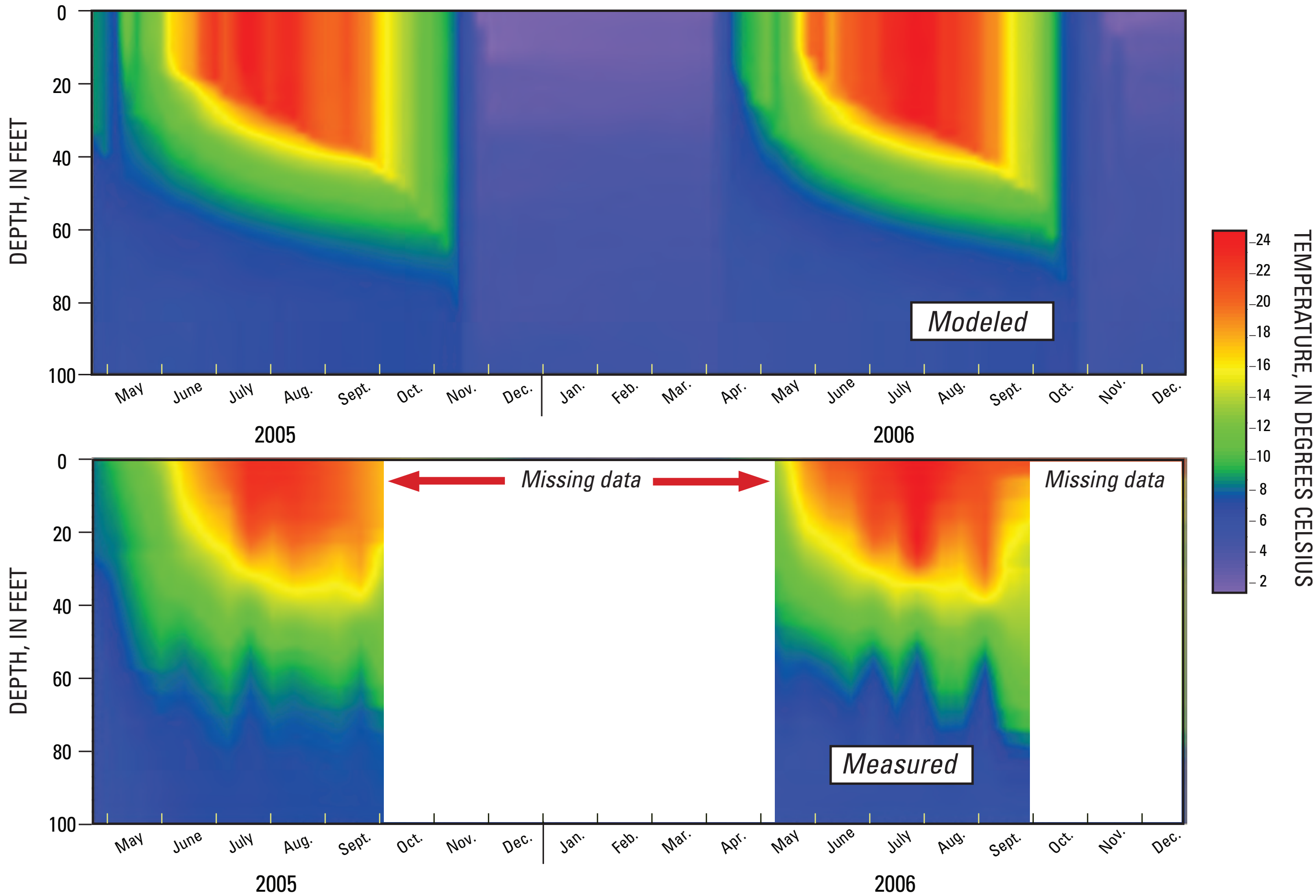


Figure 11. Measured and modeled (with the Dynamic Lake Model; McCord and others, 2000) distribution of water temperature, April 20, 2005, to December 31, 2006.

the depth of the thermocline; therefore, DLM should have correctly simulated the energy and hydrologic balances of the lake. Results from the model can also be used to describe changes in the thermal structure of the lake during periods when the lake was not monitored, such as late fall and winter. During the simulation period, the model required an additional loss of 0.92 ft^3/s (670 acre-ft/yr) of water to properly simulate the water level measured in the lake after the results were compensated for the small outflows that occurred in June and July 2005. The additional loss of water was assumed to be caused by the net loss of water from the lake to groundwater. The separation of net inflow into gross inflow and outflow is discussed in the “Groundwater Inflow and Outflow” section below.

Daily evaporation, computed by DLM from November 1, 2004, through December 31, 2006, was variable, but it demonstrated a strong seasonal pattern of highest evaporation in summer, especially during July and August (fig. 12). The highest daily evaporation was about 0.4 in/d in early August. It was assumed that no evaporation occurred when the lake was frozen. The annual evaporation from the surface of the lake was 23.0 in. (1,600 acre-ft) in MY 2005 and 26.3 in. (1,800 acre-ft) in MY 2006 (table 3). These estimates are a little less than the average total annual free-surface evaporation estimated for the area of approximately 28 in. by Farnsworth and others (1982).

Surface-Water Inflow

There were no continuously flowing tributaries to the lake. Only one channel that could potentially convey surface runoff to the lake was identified. This channel was through a berm separating the lake from a marsh near the northeast side of the lake. No flow was identified through this channel during the 2-year study, and, therefore, this area was not included in the watershed delineated in figure 1. The only other source of surface runoff to the lake is from near-lake hill slopes. Annual precipitation and runoff coefficients from Graczyk and others (2003) were used to estimate the total surface-water inflow. Surface-water inflow is estimated to be 0.9 and 0.7 acre-ft in MY 2005 and 2006, respectively (table 3). Although these values are very small, the estimated flows may be higher than what actually occurred, given the very permeable soils around the lake.

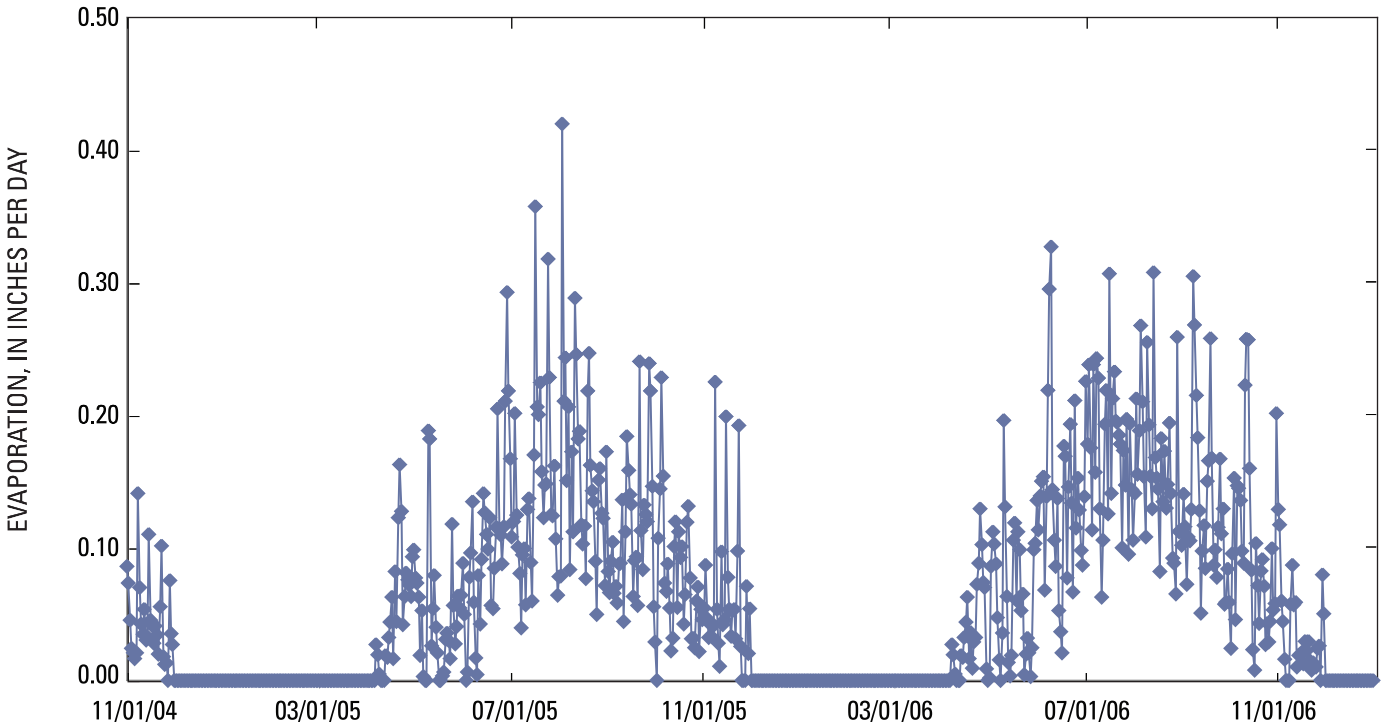


Figure 12. Daily evaporation from the surface of Whitefish Lake, Douglas County, Wis., November 2004 through December 2006, based on output from the Dynamic Lake Model.

Surface-Water Outflow

There are no continuously flowing outlets from the lake; however, flow intermittently occurred into a small channel to Deborah Lake and then into Leader Lake (fig. 1). Water levels (stages) in the stream (when water was present) between Deborah Lake and Leader Lake were used to estimate flow from the lake. Flow only occurred in this stream during June and July 2005. During these two months, it is estimated that 20.2 acre-ft of water left the lake (table 3).

Groundwater Inflow and Outflow

The calibrated GFLOW model was used to define the up-gradient groundwater-source areas around Whitefish Lake and to estimate the ratio of groundwater inflow and outflow. Whitefish Lake is a groundwater flow-through lake: groundwater enters from the east and west sides of the lake and water leaves the lake on the north and south ends (fig. 13). The groundwater contributing area extends about 0.5 mi to the northwest and about 1 mi to the east.

The total annual simulated groundwater flow into Whitefish Lake (GW_{In}) was 1.77 ft^3/s, and the total annual simulated loss from the lake to the groundwater-flow system (GW_{Out}) was 2.30 ft^3/s. Thus, the net steady-state loss of lake water to the groundwater-flow system was 0.53 ft^3/s. Although the results represent the expected average annual flow rates during steady-state conditions (no change in water levels), in actuality, the lake level dropped by 0.44 ft during MY 2005 and by 1.1 ft during MY 2006. To account for this change in water level, and the associated additional loss of lake water during the study period, an additional step was performed. That is, net annual lake-water outflow was estimated directly in the daily water budget calculation (equation 4) and in DLM to match the daily change in measured water levels. This updated net annual lake-water outflow (0.92 ft^3/s; see the “Evaporation and Net Groundwater Inflow” section) was then multiplied by the simulated steady-state ratio of groundwater inflow to outflow (0.77) from the GFLOW model to compute daily groundwater inflow to the lake (2.65 ft^3/s or 1,910 acre-ft) and outflow from the lake (3.58 ft^3/s or 2,580 acre-ft). This inflow rate was used to estimate the phosphorus loading from groundwater.

The difference between the groundwater inflow rates estimated by the GFLOW model and those estimated with DLM and the daily water budget model (equation 4) is attributed to calibration of the steady-state GFLOW model to a short calibration period (1 year), when the system was transitioning to a drier-than-average condition. That is, changes in lake-water storage caused by a declining lake level were incorporated into DLM and the water-budget calculation, but they were ignored in the GFLOW model. Nevertheless, the ratio of groundwater inflow to lake-water outflow from the GFLOW model is expected to remain relatively constant through time because groundwater and lake-water levels have similar annual fluctuations.

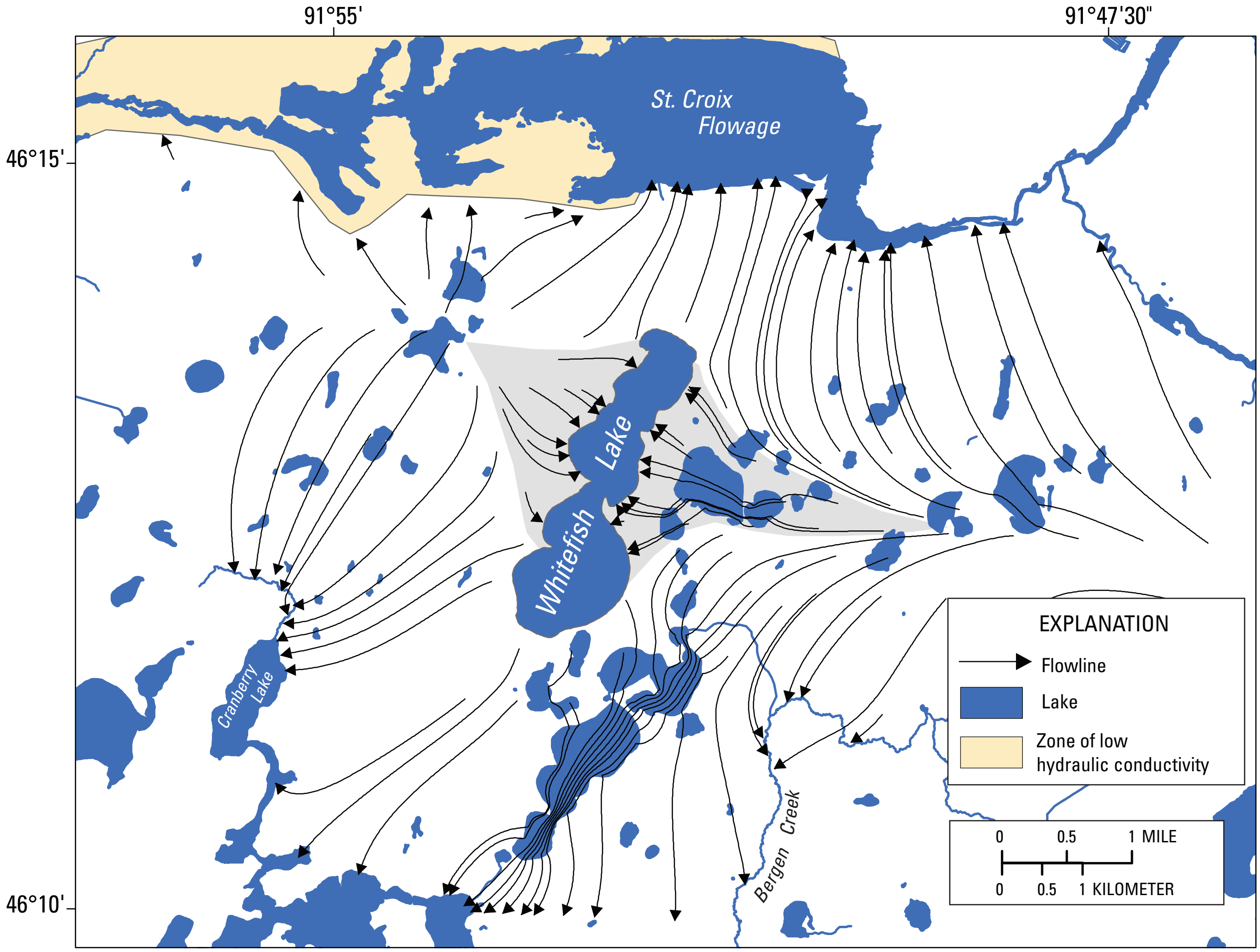


Figure 13. Simulated groundwater contributing area and flow directions near Whitefish Lake, Douglas County, Wis., from the GFLOW groundwater-flow model.

Although this model was considered reasonably calibrated and was sufficient for estimates of groundwater inflow to and outflow from Whitefish Lake, local heterogeneities that were not simulated with the model may locally alter the contributing area to a small degree. In addition, the model was calibrated primarily to MY 2005 data; groundwater/lake-water interactions for Whitefish Lake during other time periods may differ, and such evaluations would likely benefit from recalibration. Also, use of the model for evaluations beyond the stated purpose (to estimate groundwater contributing areas to the lake and groundwater exchanges with the lake) would require reevaluation of the model and likely require recalibration of the model to match new or existing data that correspond to the new purpose.

Water-Budget Summary

Precipitation and groundwater inflow were the main sources of water to Whitefish Lake, and each supplied about one-half of the total input during the 2-year study period (table 3; fig. 14). Surface runoff supplied less than 1 percent of water entering the lake. The total input of water in MY 2005, 3,880 acre-ft, was more than the 3,400 acre-ft in MY 2006, reflecting the near-normal precipitation during MY 2005 and much below-normal precipitation in MY 2006. The residence time, or the length of time required for water entering the lake to completely replace the volume of water in the lake, was 6.2 and 6.9 years on the basis of the total inflows for MY 2005 and MY 2006, respectively, or an average residence time of 6.6 years.

The total annual outputs of water from the lake were greater than total inputs, which resulted in a decrease in storage from the beginning to the end of each monitoring year. This amounted to a decrease in lake stage of 1.53 ft or about 1,290 acre-ft of storage in the lake over the 2-year period. On average, groundwater outflow accounted for about 60 percent of the total water output from the lake, and evaporation accounted for about 40 percent (table 3; fig. 14).

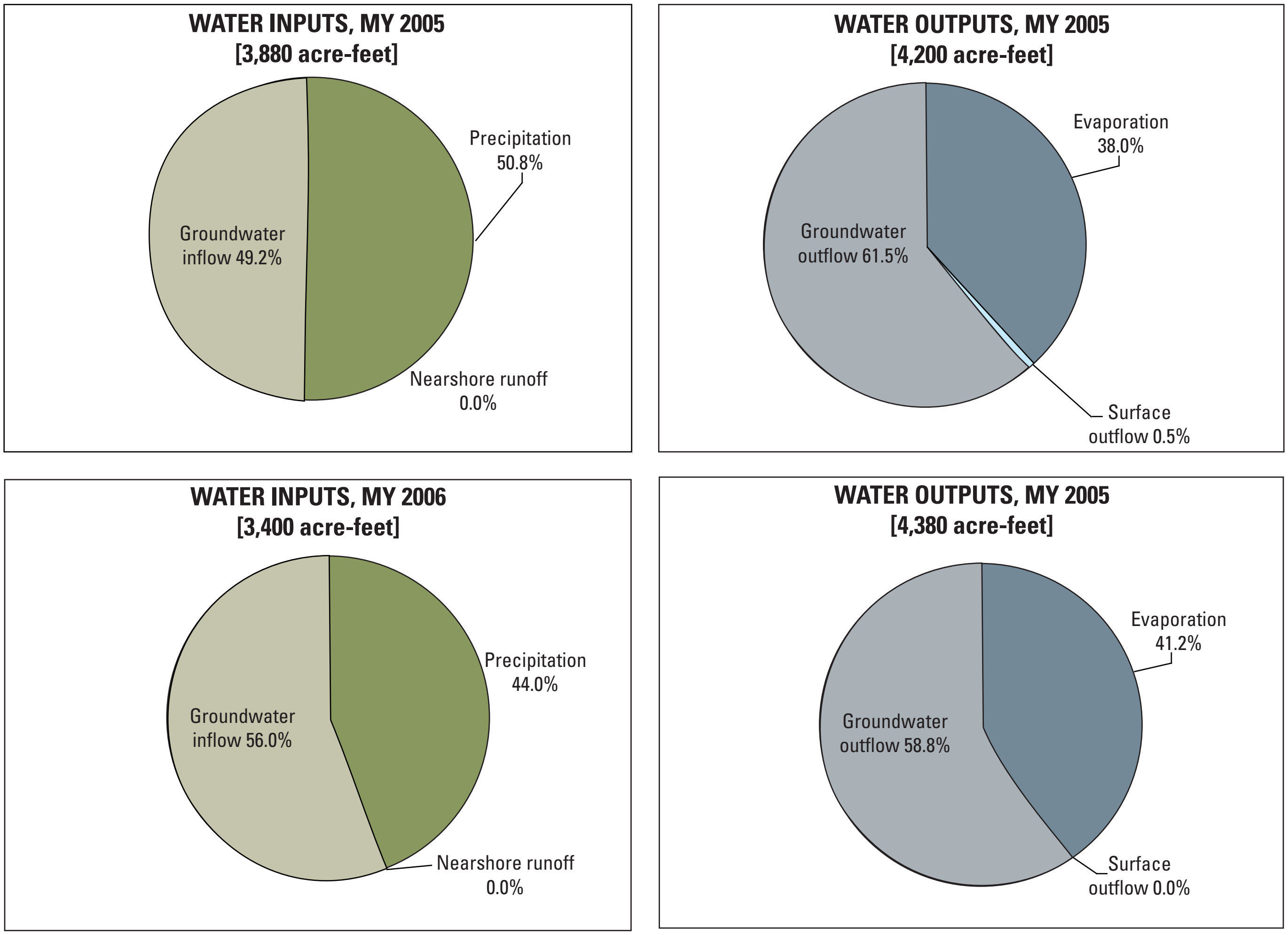


Figure 14. Water budgets for Whitefish Lake, Douglas County, Wis., for monitoring years (MY) 2005 and 2006. [%, percentage of total input or output]

The quality or accuracy of the water budget was evaluated by comparing the monthly sum of all inputs with the sum of all outputs plus the change in storage for the lake, expressed in acre-ft (fig. 15). The differences between calculated inputs and outputs reflect the cumulative errors in the estimates of all of the components in the water budget. The largest monthly differences occurred in November 2004 and June 2005, which, when expressed as percentages of the total volume of water exchanged, were about 16 and 7 percent, respectively. On an annual basis, the error was about 1 percent.

The minimal surface-water inflow into the lake was demonstrated by comparison of the measured lake stage with that estimated from a mass balance computation that did not include surface-water runoff during a 3.92 in. rain event during October 4–5, 2005 (fig. 16). If surface-water inflow from the shoreline areas was important, then the measured and computed lake stage would differ. During the rain event, the water level of the lake rose 0.310 ft (3.72 in.), which is close to that estimated when precipitation, groundwater, and evaporation were considered (0.305 ft or 3.67 in.). Therefore, even during large rain events, the very permeable soils limited the surface-water inflow to the lake.

Sources of Phosphorus

To help define where the phosphorus in Whitefish Lake originates and how much may be controllable, a detailed phosphorus budget was computed on a daily basis for each monitoring year. External sources of phosphorus to the lake include precipitation, surface-water and 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 (internal loading). Phosphorus is primarily removed from the lake through deposition to the lake sediments.

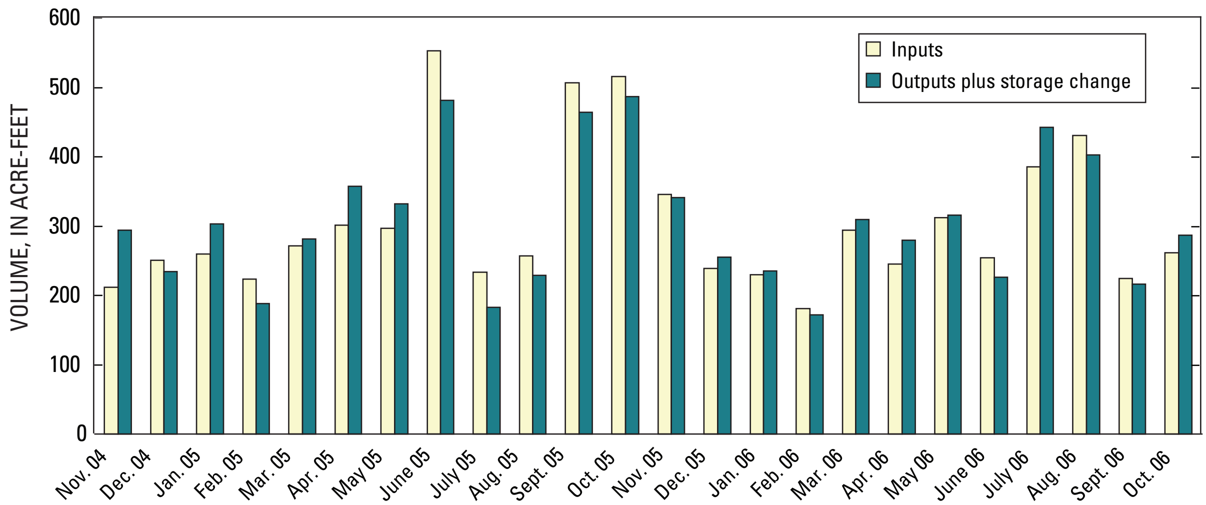


Figure 15. Monthly total water inputs and outputs plus storage change for Whitefish Lake, Douglas County, Wis., based on results from the water-balance model (equation 4). The difference between inputs and outputs indicates the cumulative error for all components of the water budget for each month.

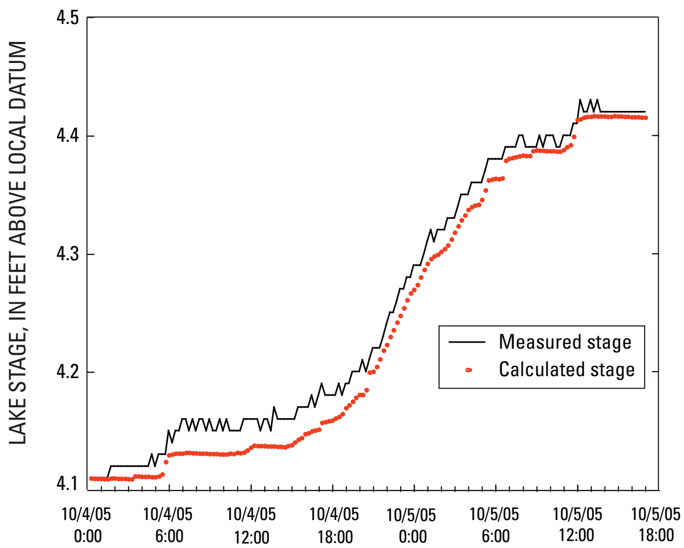


Figure 16. Measured stage in Whitefish Lake, Douglas County, Wis., compared to the estimated stage from the water-budget model with no overland flow from nearshore areas for the 2-day event occurring during October 4–5, 2005. [Altitude of the local datum is 1,029.34 feet above the National Geodetic Vertical Datum of 1929.]

Precipitation

Atmospheric deposition of phosphorus on Whitefish Lake was determined from phosphorus concentrations measured in wetfall (rain and snow) and phosphorus deposition rates in dryfall measured at the weather station installed on the shore of the South Basin of the lake (figs. 1 and 2). Phosphorus concentrations measured in 29 precipitation events from both years were combined by calendar day (fig. 17). Phosphorus concentrations in the precipitation had distinct seasonality with lowest concentrations in winter (near the 0.005-mg/L detection limit) and highest in early-to-mid summer and late fall (near 0.03 mg/L). Monthly average concentrations were computed from the individual measurements collected during 2004–06. Total monthly wetfall deposition was then computed by multiplying the monthly average phosphorus concentrations by the estimated precipitation on the lake (fig. 18 and table 4). Wetfall deposition was highest during summer when concentrations were highest and precipitation was greatest. The total phosphorus contributed by precipitation was 86.6 lb in MY 2005 and 72.7 lb in MY 2006. From these data, an average concentration of 0.009 mg/L was estimated for snowfall in winter (December–March) and 0.017 mg/L for rainfall in nonwinter months (April–November). The overall average annual concentration was 0.016 mg/L. This annual concentration is about twice that previously estimated by Rose (0.007 mg/L; 1993) for northern Wisconsin.

Phosphorus deposition rates during 21 periods without precipitation (table 4) also had distinct seasonality. The lowest daily deposition rates, usually less than 0.1 lb/mi^2, occurred in August–March, and the highest rates occurred during April–July; the highest daily rate of 1.2 lb/mi^2 was in June. The highest rates coincided with the release of pollen from conifers around the lake. Pollen was observed as distinct patches on the shoreline during these periods. The individual deposition rates were then combined and used to estimate daily dryfall deposition rates for each month. Total monthly dryfall deposition (fig. 18) was then determined from the number of days without measureable precipitation in each month. Because pollen deposition should decrease with the distance from the shore, it was assumed that only 33 percent of the large increase during March–June actually occurs over the entire surface of the lake (this equates to an approximately exponential decay to the center of the lake). Therefore, approximately 51 lb/yr of phosphorus are deposited on Whitefish Lake during periods with no precipitation.

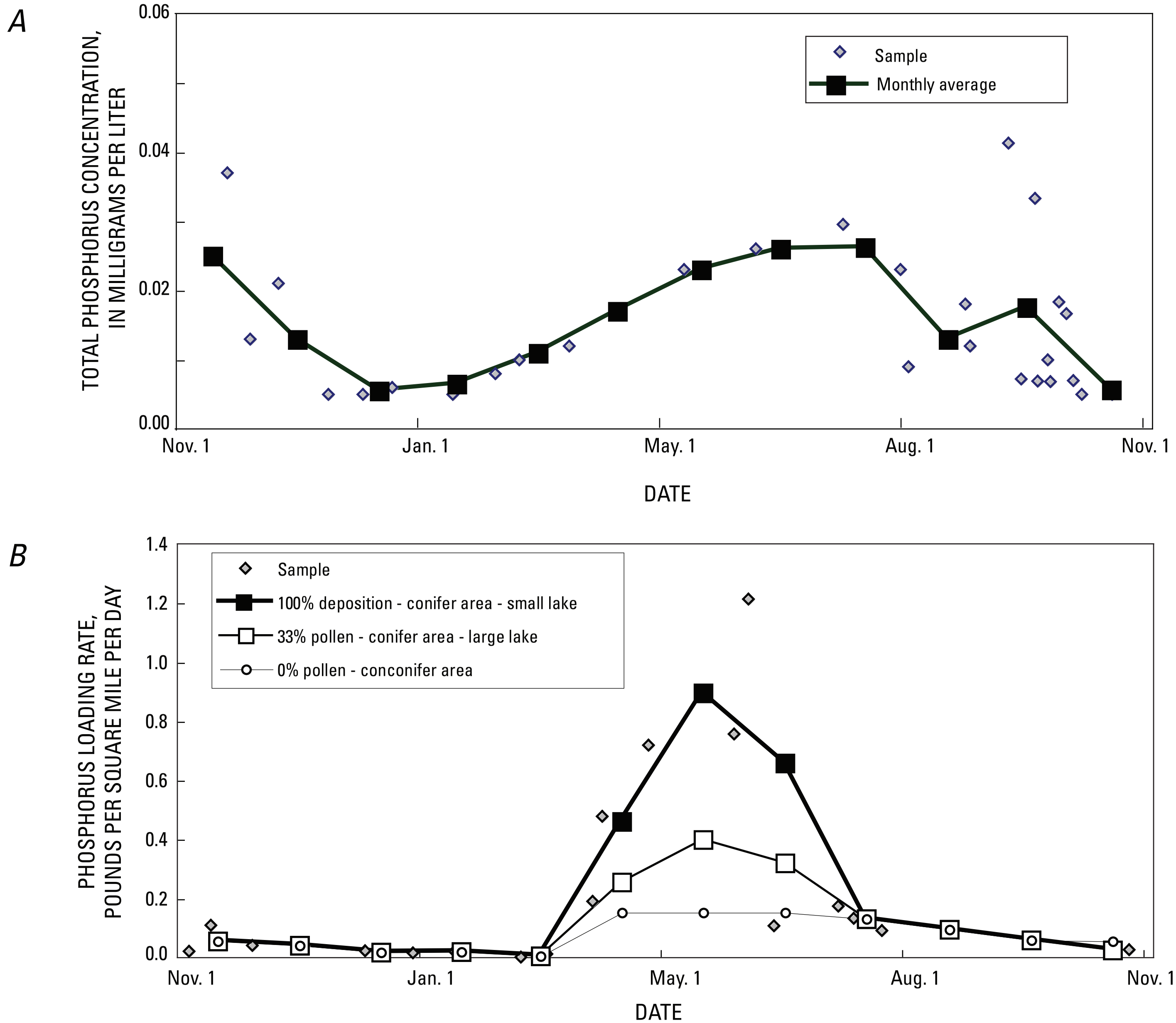


Figure 17. *A*, total phosphorus concentrations measured during wetfall events and *B*, daily dryfall deposition rates measured during periods with no precipitation in monitoring years 2005 and 2006, near Whitefish Lake, Douglas County, Wis. Data from both years were combined to estimate the changes in concentrations and loading rates.

Table 4. Total phosphorus deposition rates for Whitefish Lake, Douglas County, Wis., and rates for other types of land cover. (Rates are provided to extrapolate results to other areas of northern Wisconsin.)

[mg/L, milligrams per liter; lb, pound; lb/mi²/d, pounds per square mile per day; %, percent]

Month/ period	Wetfall			
	Number of samples into average[a]	Average concentration (mg/L)	Precipitation (inches)	Whitefish Lake deposition (lb)
Nov. 2004	2	0.025	0.80	3.8
Dec. 2004	3	.013	1.28	3.1
Jan. 2005	2	.006	1.41	1.5
Feb. 2005	2	.007	1.12	1.4
Mar. 2005	2	.011	1.59	3.3
Apr. 2005	0	.017	2.09	6.7
May 2005	1	.023	1.95	8.5
June 2005	1	.026	5.64	27.7
July 2005	2	.026	1.03	5.1
Aug. 2005	3	.013	1.37	3.4
Sept. 2005	7	.018	5.05	16.7
Oct. 2005	4	.006	5.10	5.5
Total	**29**	**--**	**28.42**	**86.6**
Nov. 2005	2	.025	2.76	13.0
Dec. 2005	3	.013	1.12	2.7
Jan. 2006	2	.006	.99	1.0
Feb. 2006	2	.007	.50	.6
Mar. 2006	2	.011	1.94	4.0
Apr. 2006	0	.017	1.29	4.1
May 2006	1	.023	2.19	9.5
June 2006	1	.026	1.43	7.0
July 2006	2	.026	3.25	16.1
Aug. 2006	3	.013	3.92	9.6
Sept. 2006	7	.018	.99	3.3
Oct. 2006	4	.006	1.45	1.6
Total	**29**	**--**	**21.83**	**72.7**
Winter	9	.009	9.94	17.7
Nonwinter	20	.017	40.31	141.6
Annual Total	--	--	--	--
Annual Average	**29**	**.016**	**25.12**	**7.8**

Table 4. Total phosphorus deposition rates for Whitefish Lake, Douglas County, Wis., and rates for other types of land covers. (Rates are provided to extrapolate results to other areas of northern Wisconsin.)—Continued

[mg/L, milligrams per liter; lb, pound; lb/mi²/d, pounds per square mile per day; %, percent]

Month/ period	Dryfall						Total
	Number of samples into average[a]	Number of dry days	Conifer areas (small lake) full deposition rate (lb/mi²/d)	Conifer areas (large lake) deposition rate 33% of pollen (lb/mi²/d)	Nonconifer areas deposition rate no pollen (lb/mi²/d)	Whitefish Lake dryfall deposition (lb)	Whitefish Lake total deposition (lb)
Nov. 2004	3	28	0.058	0.058	0.058	2.10	5.88
Dec. 2004	1	29	.042	.042	.042	1.60	4.74
Jan. 2005	2	27	.020	.020	.020	.69	2.16
Feb. 2005	1	26	.022	.022	.022	.73	2.10
Mar. 2005	2	29	.007	.007	.007	.28	3.57
Apr. 2005	3	28	.462	.257	.154	9.34	16.05
May 2005	3	29	.896	.401	.154	15.12	23.58
June 2005	2	22	.660	.323	.154	9.22	36.90
July 2005	3	29	.133	.133	.133	5.00	10.11
Aug. 2005	0	28	.097	.097	.097	3.54	6.90
Sept. 2005	0	25	.062	.062	.062	2.01	18.76
Oct. 2005	1	27	.027	.027	.058	.93	6.39
Total	**21**	**327**	**--**	**--**	**--**	**50.57**	**137.13**
Nov. 2005	3	26	.058	.058	.058	1.93	14.96
Dec. 2005	1	29	.042	.042	.042	1.58	4.33
Jan. 2006	2	29	.020	.020	.020	.74	1.76
Feb. 2006	1	27	.022	.022	.022	.75	1.37
Mar. 2006	2	25	.007	.007	.007	.24	4.25
Apr. 2006	3	27	.462	.257	.154	8.91	13.05
May 2006	3	28	.896	.401	.154	14.44	23.95
June 2006	2	27	.660	.323	.154	11.20	18.19
July 2006	3	28	.133	.133	.133	4.78	20.92
Aug. 2006	0	26	.097	.097	.097	3.25	12.87
Sept. 2006	0	29	.062	.062	.062	2.31	5.58
Oct. 2006	1	28	.027	.027	.058	.96	2.51
Total	**21**	**329**	**--**	**--**	**--**	**51.08**	**123.75**
Winter	6	110.5	.023	.023	.023	3.30	20.97
Nonwinter	15	217.5	.299	.169	.109	47.52	189.08
Annual Total	--	--	67.6	39.4	26.1	--	--
Annual Average	**21**	**328**	**.206**	**.120**	**.103**	**5.82**	**121.60**

[a] The number of samples during each month over the 2-year period.

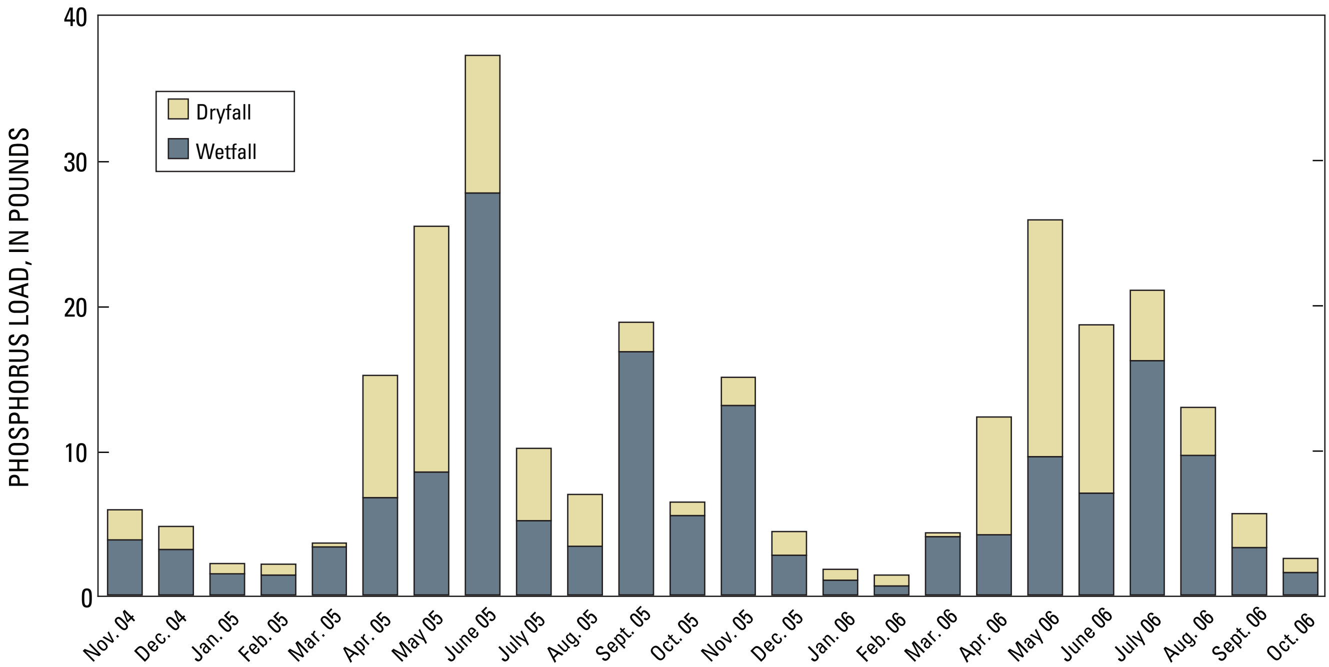


Figure 18. Monthly atmospheric wetfall and dryfall of total phosphorus into Whitefish Lake, Douglas County, Wis., during monitoring years (MY) 2005 and 2006.

Most of the dryfall deposition on the lake in April–June is associated with pollen being released from conifers (fig. 18). Therefore, it is estimated that the total phosphorus deposited on the lake from the atmosphere (wetfall plus dryfall) was 137 lb in MY 2005 and 124 lb in MY 2006 (tables 4 and 5), which is 3.4 times as much as would have been estimated on the basis of using only a wetfall concentration of 0.007 mg/L from Rose (1993).

The phosphorus concentrations in wetfall and the dryfall deposition rates measured at the meteorological station near Whitefish Lake can be used to estimate phosphorus deposition onto other lakes in northern Wisconsin. Phosphorus in rain and snow can be estimated from seasonal average concentrations of 0.009 mg/L measured for snow and 0.017 mg/L measured for rain if total snowfall and rainfall are measured, or from an overall annual average concentration of 0.016 mg/L if just total precipitation is measured. The dryfall deposition rates can be used to estimate annual phosphorus deposition on small lakes where 30–50 percent of the nearby shoreline is surrounded by conifers by using the full rate of about 67.6 lb/mi^2 (fig. 17 and table 4). For large lakes that are similar in size to Whitefish Lake and that also are surrounded by a high percentage of conifers (about 30–50 percent of the nearby shoreline), the rates should be reduced during early summer to 33 percent pollen deposition, which results in an annual rate of approximately 39.4 lb/mi^2. For lakes with few or no conifers surrounding the lake, the effects of pollen on the rates in April–July should be removed; therefore, the lower curve in fig. 17 (0 percent pollen) should be used, which results in an annual dryfall deposition rate of 26.1 lb/mi^2 (table 4).

Surface-Water Inflow

The only source of surface inflow is from the nearshore areas around the lake, although little signs of overland flow were detected except by the public boat landing. The total volume of surface runoff to the lake was estimated from daily precipitation and published runoff coefficients from Graczyk and others (2003). Because no runoff was observed, the concentration in the runoff was estimated from an experiment conducted with soils collected near the public boat landing and from a nearby hillside. Total phosphorus concentration in overland flow was estimated to be 1.6 mg/L and in water moving through the upper few inches of the soil was 3.0 mg/L. If the maximum concentration of 3.0 mg/L were assumed to be the concentration in surface-water inflow, the total phosphorus input from the nearshore areas is estimated to be about 2–3 lb in MY 2005 and 2006 (table 5). This estimate, although very small, is expected to be higher than actual, because little runoff would flow horizontally through the upper few inches of the soil and very few signs of direct runoff were observed except near the boat landing on the southwest shore of the lake.

Groundwater

Dissolved phosphorus concentrations in water collected from the nearshore piezometers (fig. 1) were used to estimate the phosphorus concentration in groundwater entering the lake. The concentrations ranged from near the 0.005-mg/L detection limit in piezometers 15 and 18 to 0.08 mg/L

Table 5. Phosphorus budget for Whitefish Lake, Douglas County, Wis., for monitoring years 2005 and 2006, and for a series of dry and wet years.

[MY, monitoring year, November–October; all data in pounds or percentage of total]

Budget component	MY 2005 (pounds)	MY 2005 (percent)	MY 2006 (pounds)	MY 2006 (percent)	2-year average (pounds)	2-year average (percent)	Series of dry years (pounds)	Series of wet years (pounds)
				Inputs to lake				
Precipitation	137	57.5	124	55.1	131	56.4	117	162
Surface-water inflow	2.9	1.2	2.3	1.0	2.6	1.1	2.6	2.6
Groundwater inflow	62.4	26.2	63.0	27.9	62.7	27.0	62.7	62.7
Septic systems	35.9	15.1	35.9	15.9	35.9	15.5	35.9	35.9
Total input	**238**	**100**	**225**	**100**	**232**	**100**	**218**	**264**
				Outputs from lake				
Surface-water outflow	.3	100	.0	100	.2	100	.2	.2
Total output	**.3**	**100**	**.0**	**100**	**.2**	**100**	**.2**	**.2**

in piezometer 14. The median concentration of all of the samples was 0.012 mg/L and the average concentration was 0.019 mg/L. To estimate the phosphorus input to the lake by groundwater, it was assumed that the concentration of groundwater entering the lake was equal to the median concentration measured in the piezometers (0.012 mg/L). An average concentration was not used because a few locations may be influenced by septic systems, and input from septic systems was estimated separately. Thus, the annual input from groundwater was about 63 lb of phosphorus (table 5).

It was estimated from the nearshore runoff experiments that the phosphorus concentration of water seeping through the upper few inches of soil quickly increased to about 3 mg/L compared to only 0.012 mg/L measured in the groundwater. Therefore, most of this phosphorus must be removed by the soil in the unsaturated vadose zone prior to reaching the shallow groundwater and being transported to the lake.

Septic Systems

The input of phosphorus from septic systems (M) was estimated by use of equation 5 (Reckhow and others, 1980):

$$M = E_s * (\text{number of capita years}) * (1 - S_R), \qquad (5)$$

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). In applying equation 5, it was assumed that the most likely value for E_s was 1.5 lb of phosphorus per capita year. Typical export coefficients range from 1.1 lb per capita year (Reckhow and others, 1980; Panuska and Kreider, 2003) to 1.8 lb per capita year (Garn and others, 1996). Olson and others (2006) estimated that 100 properties were around the lake, of which 89 had septic systems. They also estimated that 25 percent of the properties had, on average, 2.5 year-round residents and that 75 percent of the properties had, on average, 3.5 three-month residents. Therefore, the total number of capita years was estimated to be 114. A value of 0.7 (70-percent retention by the soil) was used for S_R. Based on the above assumptions, the annual input of phosphorus from septic systems was about 51 lb; however, not all of the area around the lake contributes groundwater to the lake. It is estimated that 70 percent of the properties are in areas that contribute groundwater to the lake (fig. 13); therefore, the total estimated input from septic systems is about 36 lb (table 5). Because inputs from septic systems were not directly measured, low and high estimates for septic inputs were obtained by applying high and low estimates for S_R (0.9 and 0.5) in equation 5. The low and high estimates for phosphorus inputs from septic systems were 12 and 60 lb, respectively.

Surface-Water Outflow

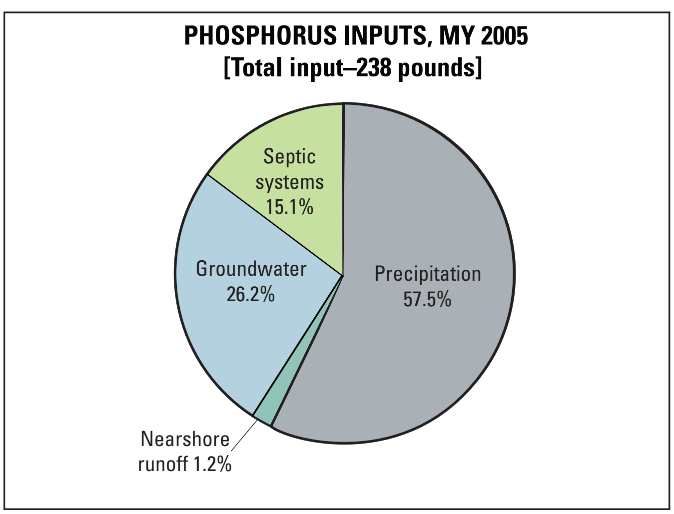
The total phosphorus discharged from Whitefish Lake into Deborah Lake through the small outlet on the southeast side of the lake was estimated by multiplying the discharge at South Bass Lake Road by the average lake phosphorus concentration during the period of flow (0.006 mg/L). Therefore, it is estimated that 0.3 lb of phosphorus was discharged from the lake in MY 2005 and no phosphorus was discharged in MY 2006 (table 5).

Phosphorus-Budget Summary

The average annual load of phosphorus to the lake was 232 lb, of which 131 lb came from atmospheric deposition (56 percent), 63 lb (27 percent) came from groundwater, and 36 lb (16 percent) came from septic systems (table 5; fig. 19). If the high estimates for phosphorus contributions from septic systems (60 lb) were used in the budget, then inputs from septic systems would increase to about 23 percent of the total external phosphorus load, whereas if the low estimate were used (12 lb), inputs from septic systems would decrease to about 2 percent. The total phosphorus load to the lake ranged from 238 lb in MY 2005 to 225 lb in MY 2006.

In addition to the external sources of phosphorus, the increase in phosphorus concentrations just above the bottom of the lake (fig. 6) indicates that there is internal phosphorus loading to the lake during periods of anoxia. The relatively strong thermal stratification that exists when phosphorus is released from the deep bottom sediments, however, prevents this phosphorus from being released into the surface water until late fall and near the time of ice break-up in spring. It appears that most of this dissolved phosphorus is rapidly consumed by the biota in the fall or oxidized and removed from the water column early in the spring because dissolved and total phosphorus concentrations in the lake in April/May are usually at or below their respective detection limits, 0.002 and 0.005 mg/L. Therefore, this additional phosphorus does not appear to be available to drive summer productivity and was not considered in the phosphorus budget for the lake. There appears to be no long-term trend in near-bottom phosphorus concentrations; therefore, internal loading of phosphorus does not appear to have changed during the period since the lake has been monitored.

To determine how the hydrologic and phosphorus loading to the lake should vary during a series of dry years and a series of wet years, the water and total phosphorus loading to the lake was estimated for the 3-year period with the lowest precipitation (1974–76, average precipitation of 22.3 in.) and the 3-year period with the highest precipitation (1994–96, average precipitation of 37.3 in.) since 1915. During these years, phosphorus from wetfall was estimated by multiplying the total volume of precipitation by an average phosphorus concentration of 0.016 mg/L, and phosphorus from dryfall was assumed to be a constant value of 50.8 lb (table 4). The assumption of a constant input from dryfall may be biased slightly high in wet years and low in dry years. The total phosphorus from precipitation during the dry years was 117 lb and during the wet years was 162 lb. During these years, water and phosphorus from the other sources were assumed to be constant: phosphorus from surface-water inflow was 2.6 lb, groundwater contributed 1,910 acre-ft of water and 62.7 lb of phosphorus, and septic input was constant at 35.9 lb of phosphorus (table 5). During the dry years, the average water loading was 3,450 acre-ft and phosphorus loading was 218 lb. During the wet years, the average water loading to the lake was 4,500 acre-ft and phosphorus loading was 264 lb. The total water loading to the lake, therefore, varies by about 13 percent and the total phosphorus loading varies by about 10 percent. If it was assumed that the water level of the lake was similar to that at the end of the two periods, then the residence time of water in the lake was 6.8 years during the dry years and 5.5 years during the wet years.



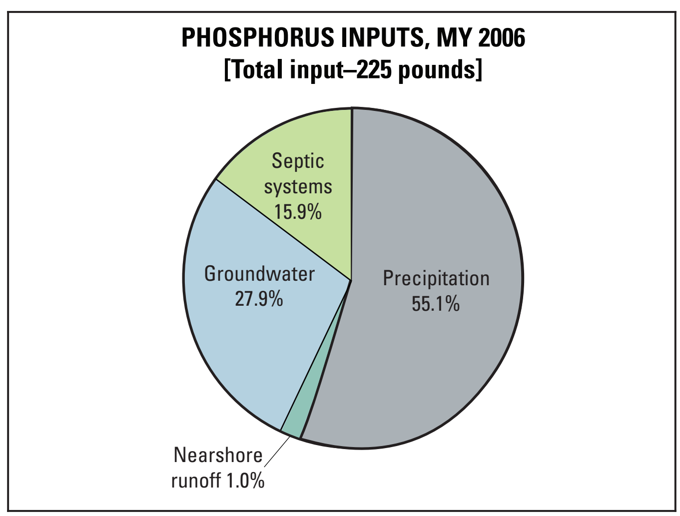


Figure 19. Phosphorus budgets for Whitefish Lake, Douglas County, Wis., for monitoring years (MY) 2005 and 2006. [%, percentage of total input.]

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

Empirical eutrophication models that relate phosphorus loading to specific water-quality characteristics can be used to determine how natural and anthropogenic changes in phosphorus loading could modify the water quality of Whitefish Lake. These models were developed on the basis of comparisons of hydrologic and phosphorus loading 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 (Reckhow and Chapra, 1983; Cooke and others, 1993; Panuska and Kreider, 2003). Most of the phosphorus models were derived from analyses of many lakes with very different phosphorus-loading rates and hydrologic conditions (for example, Vollenweider, 1975; Canfield and Bachmann, 1981). Most empirical models that predict phosphorus concentrations are very sensitive to the residence time of water in the lake, and they are not capable of accurately simulating water quality in closed-basin seepage lakes—lakes without continuous outlets and, therefore, with very long residence times. One empirical model that has been shown by Robertson and Schladow (2008) to be relatively insensitive to residence time is the Canfield and Bachmann (1981) natural-lake model. This model has also been shown to be one of the best empirical models for predicting the phosphorus concentrations of lakes in Wisconsin (Robertson and others, 2002; 2005; 2009; Robertson and Rose, 2008). Therefore, the Canfield and Bachmann natural-lake model (1981; equation 6) was applied to Whitefish Lake to determine how changes in phosphorus loading should affect phosphorus concentrations in the lake:

$$\text{Total phosphorus concentration } (\mu g/L) = \frac{L}{0.305 * Z\,(1.62 \text{ x } (L/Z)^{0.458} + 1/\tau)}, \qquad (6)$$

where

- L is the annual phosphorus-loading rate (in mg/m^2 = lb/acre x 112.07),
- Z is the mean depth of the lake (in meters = ft x 3.2808), and
- τ is the residence time of the water in the lake (in years).

In this application, the hydrology and phosphorus loading measured in MY 2005 was used as a base case for other comparisons because the water level of the lake changed less dramatically than in MY 2006 (fig. 9) and the precipitation in MY 2005 was closer to normal than in MY 2006 (table 3).

The total phosphorus concentrations predicted with the Canfield and Bachmann model were then used to predict chlorophyll *a* concentrations and Secchi depths through the use of Carlson's (1977) TSI equations (equations 1–3). In other words, the chlorophyll *a* concentrations and Secchi depths were computed that yielded a similar TSI value as that computed from the predicted total phosphorus concentration. There are no true calibration factors when Carlson's TSI equations are used to estimate chlorophyll *a* concentrations and Secchi depths; however, the output can be adjusted to account for model biases by adjusting the results by the percentage of bias in the results for the base scenario.

Modeling Approach

To estimate how the water quality of Whitefish Lake should respond to changes in phosphorus loading, 20 scenarios were simulated with the Canfield and Bachmann model and Carlson TSI equations (table 6 and fig. 20). Scenario 1 simulated the conditions for MY 2005 and established a base condition (concentration) to which results from the other simulations/scenarios can be compared. Eight simulations (2–9) were used to determine the general response of the lake to basinwide changes in phosphorus loading: simulated decreases in controllable external phosphorus loading by 75, 50, and 25 percent and increases in controllable external phosphorus loading by 25, 50, 75, 100, and 200 percent. Controllable external loading includes phosphorus from septic systems and surface-water inflow, and it represents only a small part of the total phosphorus load to the lake. Two scenarios (10 and 11) were then used to evaluate the effects on model results of the two estimates made in the phosphorus-budget computations, the soil retention estimate used in the computation of inputs from septic systems and the percentage of pollen reaching the lake in atmospheric dryfall.

Seven scenarios (12–18) were then used to determine how changes in phosphorus loading caused by anthropogenic and natural (climatic) changes associated with changes in water level could have modified or could modify the water quality of the lake in the future. Four scenarios (12–15) were used to simulate the effects of anthropogenic changes in the watershed: removal of all surface-water inflow, removal of all septic inputs, doubling of all septic inputs, and uncontrolled development in the watershed. To simulate the effects of uncontrolled development in the watershed, it was assumed that 25 percent of the watershed would be converted to residential areas without stormwater and runoff controls. A phosphorus export rate of 0.45 lb/acre was used for this development, which is typical for residential areas (Panuska and Lillie, 1995). Uncontrolled development would increase the phosphorus loading to the lake by 51 lb. Three scenarios (16–18) were then used to simulate the effects of natural changes (changes in phosphorus loading associated with water level/climatic): a decrease in water level by 5 ft, a series of three dry years (described earlier), and a series of three wet years (described earlier).

Four types of data are required as input into the Canfield and Bachmann model: water-quality data (table 2), morphometric data (mean depth), hydrologic data (residence time), and nutrient-loading data (table 6). Although loading data are summarized for the entire year, this model and the Carlson TSI equations simulate water quality only for the growing season (May–September); therefore, the water-quality data in table 2 were used for comparisons.

Table 6. Estimated water quality in Whitefish Lake, Douglas County, Wis., in response to various phosphorus-loading scenarios based on the Canfield and Bachmann natural-lake model (1981) and Carlson (1977) trophic-state-index equations.

[mg/L, milligrams per liter; µg/L, micrograms per liter; %, percent]

Simulations	Scenario number	Annual phosphorus load (pounds)	Water elevation (feet)	Surface area (acres)	Mean depth feet)	Residence time (years)	Predicted total phosphorus concentration (mg/L)	Predicted chlorophyll a concentration (µg/L)	Predicted Secchi depth (feet)
General response									
Percent change in controllable load									
- 75%	2	209	4.55	833	29.0	6.1	0.0072	1.2	28.8
- 50%	3	219	4.55	833	29.0	6.1	.0075	1.3	27.9
- 25%	4	229	4.55	833	29.0	6.1	.0077	1.4	27.1
0% - Base	1	**238**	**4.55**	**833**	**29.0**	**6.1**	**.0079**	**1.4**	**26.3**
+ 25%	5	248	4.55	833	29.0	6.1	.0082	1.5	25.5
+ 50%	6	258	4.55	833	29.0	6.1	.0084	1.5	24.9
+ 75%	7	268	4.55	833	29.0	6.1	.0086	1.6	24.2
+ 100%	8	277	4.55	833	29.0	6.1	.0088	1.7	23.6
+ 200%	9	316	4.55	833	29.0	6.1	.0097	1.9	21.6
Sensitivity									
Less soil retention	10	262	4.55	833	29.0	6.1	.0085	1.6	24.6
100% dryfall	11	274	4.55	833	29.0	6.1	.0088	1.6	23.8
Specific anthropogenic and natural scenarios									
No surface-water inflow	12	235	4.55	833	29.0	6.1	.0079	1.4	26.5
No septic	13	203	4.55	833	29.0	6.1	.0071	1.2	29.4
Double septic	14	275	4.55	833	29.0	6.1	.0088	1.7	23.7
Uncontrolled buildout (25 percent of watershed)	15	289	4.55	833	29.0	6.1	.0091	1.7	22.9
Shallower by 5 feet	16	218	-.45	726	27.9	5.1	.0079	1.4	26.4
Series of dry years (ending in 1976)	17	218	3.57	820	28.5	6.8	.0079	1.4	26.3
Series of wet years (ending in 1996)	18	264	5.29	837	29.6	5.5	.0081	1.5	25.8

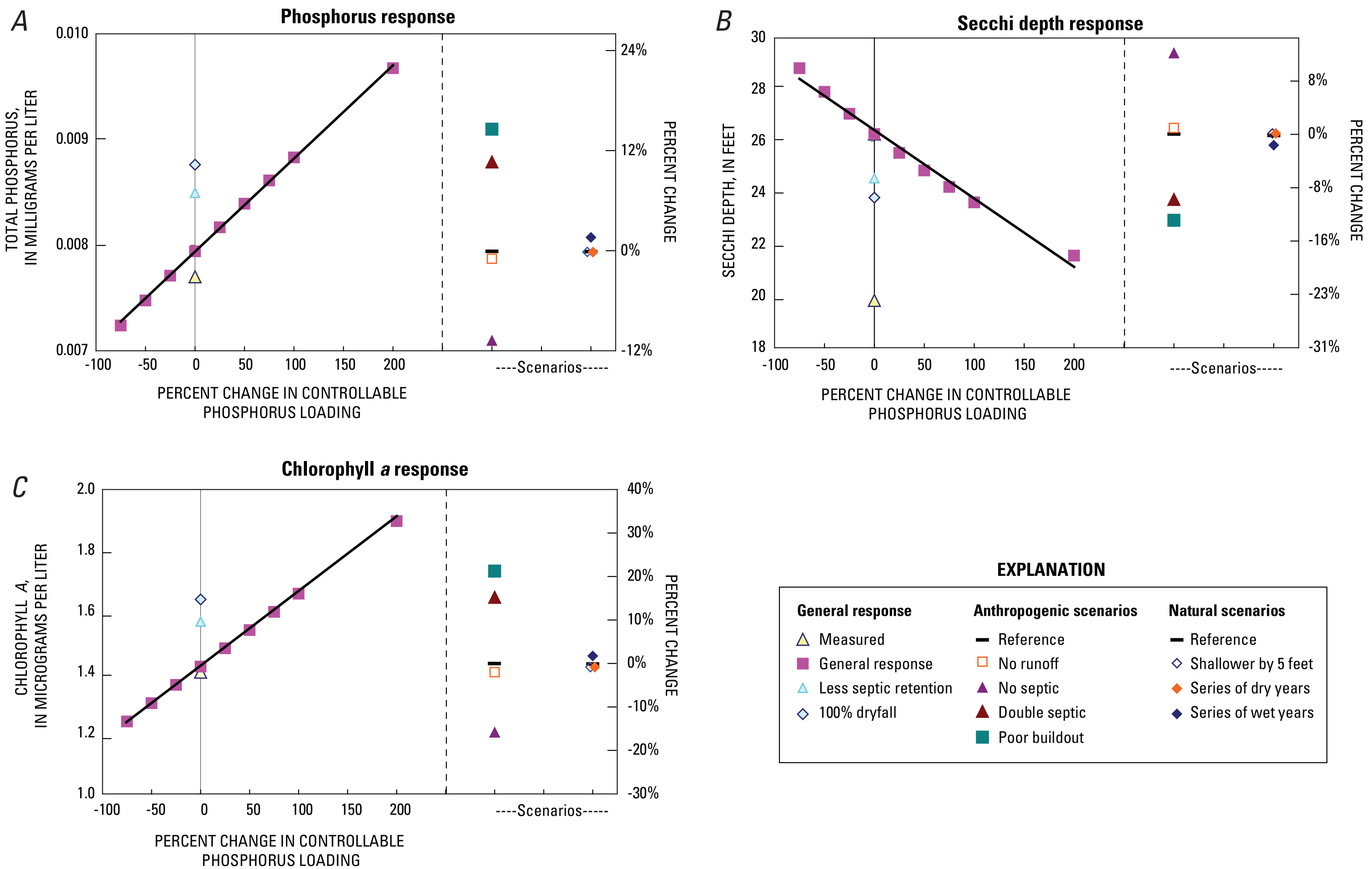


Figure 20. Measured and simulated average summer near-surface concentrations of *A*, total phosphorus, *B*, Secchi depths, and *C*, chlorophyll *a* in Whitefish Lake, in response to various phosphorus-loading scenarios. Phosphorus concentrations were simulated with the Canfield and Bachmann natural-lake model (1981) and chlorophyll *a* concentrations and Secchi depths were simulated with Carlson (1977) trophic-state-index equations. (See table 6 for a description of each scenario.)

Verification and Adjustments for Model Biases

The capability of the Canfield and Bachmann (1981) model at simulating phosphorus concentrations in this closed-basin seepage lake was evaluated by inputting the morphometry of the lake and the hydrologic and phosphorus loadings for MY 2005 into the model (equation 6) and simulating near-surface total phosphorus concentration. The simulated total phosphorus concentration for MY 2005 (0.0079 mg/L; fig. 20*A* and scenario 1 in table 6) is similar to that measured in the lake (0.008 mg/L; table 2). This similarity confirms that the model is expected to accurately simulate changes in Whitefish Lake in response to changes in phosphorus loading.

Determining how well the Carlson (1977) TSI equations simulate chlorophyll *a* concentrations and Secchi depths in Whitefish Lake was accomplished by using the TSI value based on the simulated total phosphorus concentration to compute the chlorophyll *a* concentration and Secchi depth in the lake with equations 2 and 3. Given a TSI value of 34.0 (from equation 1 with the simulated phosphorus concentration of 0.0079 mg/L), the predicted chlorophyll *a* concentration was 1.42 µg/L and unadjusted predicted Secchi depth was 19.9 ft (fig. 20*B* and 20*C*). The measured chlorophyll *a* concentration in MY 2005 was 1.4 µg/L, which is similar to that simulated. The measured Secchi depth in MY 2005 was 26.3 ft, which is 32 percent greater than that predicted with equation 3. The discrepancy in Secchi depths indicates that the clarity measured in Whitefish Lake is better than would be expected given the measured phosphorus and chlorophyll *a* concentrations. This descrepency may have been caused by the structure of the food web and pelagic fish that are present in Whitefish Lake (T. Hrabik, University of Minnesota, written commun., 2007) in comparison with other lakes with similar phosphorus and chlorophyll *a* concentrations. Because of the large discrepancy in measured and simulated Secchi-depth values, all of the simulated Secchi-depth values from equation 3 were increased by 32 percent.

Sensitivity of Model Predictions to Assumptions in Load Estimations

The effects of the assumption made in the estimate of phosphorus retention by the soil in the computation of septic-system inputs and the assumption made in the percentage of pollen reaching the lake in the estimate of atmospheric dryfall were evaluated in scenarios 10 and 11 (table 6): in scenario 10, the retention coefficient in equation 5 was set to 0.5, and in scenario 11, 100 percent of the pollen inputs were used. A decrease in the soil retention coefficient from 0.7 (scenario 1) to 0.5 (scenario 10) increased the load from septic systems from about 36 to 60 lb and increased the overall load by about 10 percent. This increase in loading resulted in an increase in the simulated phosphorus concentration from 0.0079 to 0.0085 mg/L, an increase in chlorophyll *a* concentrations from 1.4 to 1.6 µg/L, and a decrease in the predicted Secchi depth from 26.3 to 24.6 ft. An increase in the amount of phosphorus in dryfall associated with pollen deposition from 33 percent (scenario 1) to 100 percent (scenario 11) increased the phosphorus load from dryfall from 50.6 to 86.4 lb and increased the overall load by about 15 percent. This increase in loading resulted in an increase in the simulated phosphorus concentration from 0.0079 to 0.0088 mg/L, an increase in the simulated chlorophyll *a* concentrations from 1.4 to 1.6 µg/L, and a decrease in the predicted Secchi depth from 26.3 to 23.8 ft. Both of these scenarios indicate that the assumptions made in constructing the phosphorus budget had a small affect on the model predictions. If the higher estimates of loading were used, then all of the phosphorus results would be increased by only 0.0005 mg/L, chlorophyll *a* concentrations would be increased by about 0.2 µg/L and Secchi depths would be decreased by about 2 ft; however, the relative changes from the base case would remain the same.

Response in Water Quality to Basinwide Changes in Phosphorus Loading

On the basis of model simulations for basinwide changes in controllable phosphorus loading (scenarios 1–9), phosphorus concentrations in the lake should have a linear response to a linear change in loading (fig. 20*A*; table 6). The changes in in-lake phosphorus concentrations, on a percentage basis, are much smaller than the changes in controllable external phosphorus loadings. Changes in in-lake phosphorus concentrations are only about 10 percent of the changes in phosphorus loading. For example, a 50-percent change in controllable external phosphorus loading should cause a 5-percent change in in-lake phosphorus concentrations.

Simulated summer average chlorophyll *a* concentrations also had a linear response to changes in controllable external phosphorus loading (fig. 20*B*). The changes in chlorophyll *a* concentrations, on a percentage basis, are smaller than the changes in external phosphorus loadings. For example, a 50-percent change in controllable external phosphorus loading causes about a 10-percent change in chlorophyll *a* concentrations.

Simulated summer average Secchi depths had a slightly less linear response to changes in controllable external phosphorus loading than did total phosphorus and chlorophyll *a* concentrations and had a slightly larger response to reductions in loading than to increases in loading (fig. 20*C*). The changes in Secchi depths, on a percentage basis, were smaller than the changes in external phosphorus loadings. For example, a 50-percent change in controllable external phosphorus loading causes about a 5-percent change in Secchi depths.

The relatively small response of each of these constituents to changes in controllable phosphorus loading is primarily because most of the phosphorus presently input into Whitefish Lake is from uncontrollable sources, not that Whitefish Lake is unresponsive to external loading. In reality, a 100-percent increase in controllable phosphorus loading results in an increase in the total phosphorus load of 16.4 percent (from 238 lb for the base case to 277 lb) and should cause an increase in the total phosphorus concentrations in the lake of 11.4 percent, an increase in chlorophyll *a* of 16.9 percent, and a decrease in Secchi depths of 10.1 percent. Therefore, changes in phosphorus loading should affect the water quality of Whitefish Lake.

Response in Water Quality to Specific Anthropogenic and Natural Scenarios

The eutrophication models were used to simulate how the summer average concentrations of total phosphorus and chlorophyll *a*, and Secchi depths in Whitefish Lake should respond to seven specific anthropogenic and natural scenarios with different annual phosphorus loadings and residence times of water in the lake (scenarios 12–18 in table 6). The phosphorus concentrations were first estimated with the Canfield and Bachmann model and the results were then used to compute Carlson TSI values using equation 1. The predicted TSI values were then input into equations 2 and 3, and chlorophyll *a* concentrations and Secchi depths were computed. The simulated Secchi depths were then adjusted to account for the original 32-percent discrepancy in the relation (fig. 18*C*).

Four scenarios (12–15) were used to simulate the effects of anthropogenic changes in the watershed. Complete removal of the phosphorus loading from surface-water runoff (scenario 12) had very little affect on the lake's water quality (fig. 20; table 6) because this source represented only about 1 percent of the annual phosphorus loading. Complete removal of the phosphorus loading from septic systems (scenario 13) decreased the phosphorus concentration by 0.0008 mg/L, decreased chlorophyll *a* concentrations by 0.2 µg/L, and increased the Secchi depth by 3.1 ft, whereas doubling the input from septic systems (scenario 14) increased the phosphorus concentration by 0.0008 mg/L, increased chlorophyll *a* concentrations by 0.2 µg/L, and decreased the Secchi depth by 2.6 ft. Uncontrolled development of about 25 percent of the watershed (scenario 15), which includes typical losses from septic systems, had the most dramatic affect on water quality of the scenarios tested and increased the phosphorus concentration by 0.0012 mg/L, increased chlorophyll *a* concentrations by 0.3 µg/L, and decreased Secchi depths by 3.4 ft.

Three scenarios (16–18) were used to determine how natural/climatic changes, such as changes in phosphorus loading associated with changes in water level, should affect water quality (fig. 20; table 6). A decrease in water level by 5 ft from that in MY 2005, combined with decreases in phosphorus loading caused by precipitation and deposition falling on a smaller surface area (scenario 16), had almost no impact on the water quality in the lake. This minimal change was caused by less phosphorus loading (218 lb) being input into a smaller lake (726 acres). Sustained low water levels of about 1-ft lower than in MY 2005 caused by a series of dry years with less precipitation (and less phosphorus input) on a smaller surface area (scenario 17), again had almost no affect on water quality. Sustained high water levels of about 0.8 ft higher than in 2005 caused by a series of wet years with more precipitation (and more phosphorus input) on a larger surface area (scenario 18), however, increased the phosphorus concentrations by 0.0002 mg/L, increased chlorophyll *a* concentrations by 0.1 µg/L, and decreased Secchi depths by 0.5 ft. The relatively small degradation in the water quality in the lake following a series of wet years was caused by increased precipitation causing an increase in the phosphorus loading by about 10 percent, but the higher loads were input into a larger lake. Therefore, although high water levels may cause higher phosphorus concentrations in the lake, it appears that anthropogenic changes in the watershed could be more important than natural/climatic changes in affecting the future water quality of the lake.

TSI values based on near-surface concentrations of total phosphorus and chlorophyll *a,* and on Secchi depths indicate that Whitefish Lake is usually oligotrophic (fig. 7). On the basis of the model simulations, however, the lake is susceptible to degradation if phosphorus loading increases. Although results of the simulations in table 6 indicate that the lake will remain oligotrophic under the conditions of all of the scenarios tested, results from the eutrophication models also indicate that with an increase in phosphorus loading of only 180 lb/yr, the lake could become mesotrophic.

Historical Changes in Water Levels and Water Quality in Whitefish Lake

Detailed historical water-level (stage) measurements are available for only a few lakes in Wisconsin, and they have only been made in Whitefish Lake since 1998 except for one measurement in 1967 (fig. 21). Since 1998, the water level decreased about 2.4 ft, before rising slightly in 2008. Data were available, however, to estimate water levels from 1972 to 2007 in nearby Bluegill Lake, where distances from a reference to the edge of the lake were measured by William and Martha Marshand (written commun., 2007). Bluegill Lake is a small lake about 0.25 mi east of Whitefish Lake, and it is in the same groundwater flow system (figs. 1 and 13). Measurements of distances to the edge of the lake were converted to water-level elevations (fig. 21) based on ground-level elevations surveyed in the summer of 2006. Since 1972, the water level of Bluegill Lake has gone through at least three cycles of rising and falling. The water level in Bluegill Lake tracked the changes in water level measured in Whitefish Lake from 1998 to 2007 (fig. 21); therefore, a numerical relation was used to predict historical changes in Whitefish Lake from 1985 (the year in which continuous annual measurements began at Bluegill Lake) until 2003 (when detailed measurement began in Whitefish Lake).

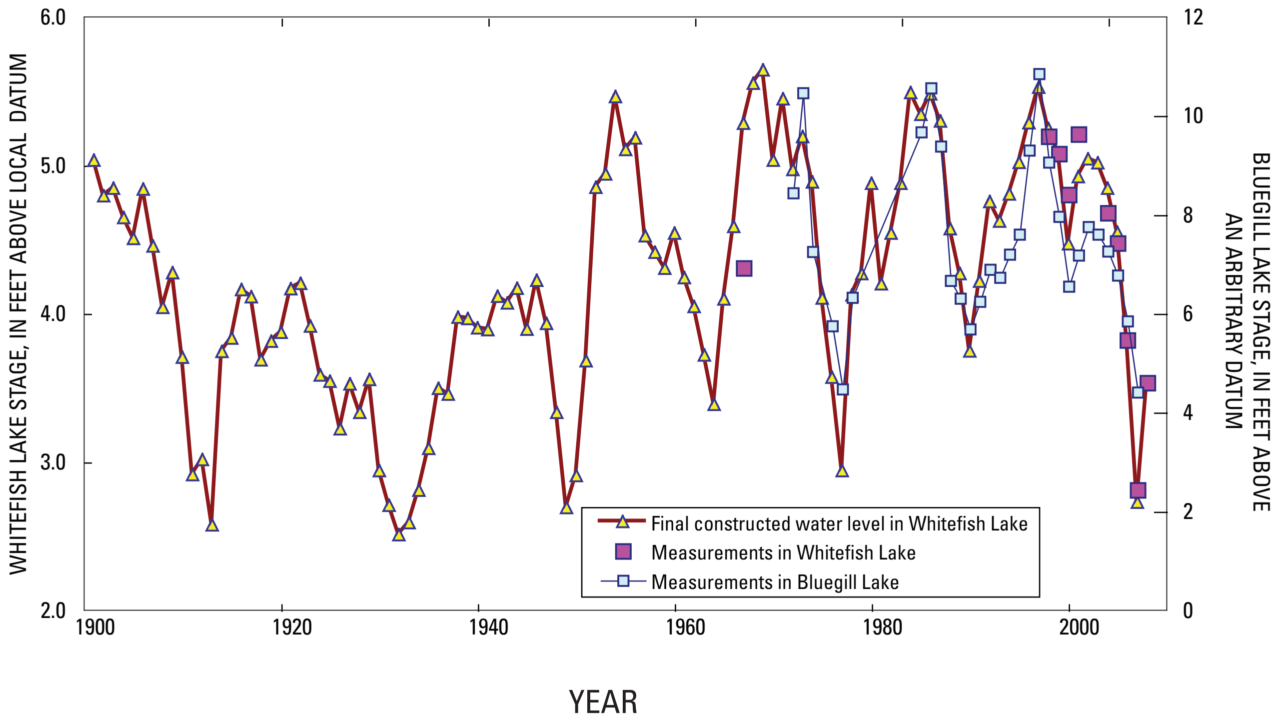


Figure 21. Estimated and measured lake stage (water level) in Whitefish Lake, Wis., and Bluegill Lake, Douglas County, Wis., 1900 to 2008. [Altitude of the local datum is 1,029.34 feet above the National Geodetic Vertical Datum of 1929.]

To estimate how the water level of Whitefish Lake changed prior to 1985, a multiple-regression time-series relation was developed between the estimated water levels in Whitefish Lake from 1985 to 2007 and precipitation measured at Gordon, Wis., approximately 3 mi northeast of Whitefish Lake. This relation explained 74 percent of the variability in the estimated water levels in Whitefish Lake. The water level of Whitefish Lake from 1900 to 1984 was then estimated by use of the time-series model that related the water level in Whitefish Lake to the total annual precipitation at Gordon for the present year, the previous 2 years, and the previous 4 years. From 1900 to 1954, precipitation measured at Spooner, Wis., approximately 25 mi south of Whitefish Lake, was used to fill in any missing precipitation data in the Gordon records. The final constructed water levels for Whitefish Lake indicate that the water level has experienced about eight cycles of rising and falling since 1900 (fig. 21). The changes in water level after 1998 are representative of the fluctuations that should have occurred since 1900, with fluctuations of about 3 ft occurring about every 15 years. Large fluctuations in water level have been known to occur in Whitefish Lake and have affected the area and the behavior of people around the lake. The prolonged low-water levels in the early 1930s (fig. 21) resulted in the growth of small trees and the use of a road on the present-day sand beaches in the southern part of the lake (Kinnie Smith, personal commun., 2008). The trees and road were later eroded by the higher water levels in the late 1930s and 1940s.

Relative Importance of Anthropogenic Changes in the Watershed and Changes in Water Level

One purpose of this study was to understand how natural changes in the water level and associated changes in nutrient loading affect the water quality of Whitefish Lake in comparison with those caused by anthropogenic changes in the watershed. Based on the changes in water level and water quality since 1998, it appears that part of the apparent deterioration in water quality between 2002 and 2006 may have been caused by the sustained high-water level (and associated changes in phosphorus input) of Whitefish Lake because after the extended period of high water, phosphorus concentrations increased by about 0.003 mg/L and water clarity decreased by about 4 to 5 ft. Then, while water levels declined (and phosphorus input declined) in 2007–08, phosphorus concentrations decreased and water clarity increased to conditions close to those measured around 2000. The changes in measured water quality since 1998 are slightly larger than those predicted by the eutrophication models if changes only in water level and the associated changes in phosphorus loading occurred.

Therefore, other factors may have contributed to the decrease in water quality, such as increased erosion from flooding of additional shoreline during the highest water levels that was not accounted for in the phosphorus budget, an increase in atmospheric deposition, anthropogenic changes in the watershed, or possibly minor changes in the procedures used by the laboratory where the analyses were conducted. The improvements in water quality in 2007–08 (fig. 5) that occurred with declining water levels indicate that most of the deterioration in water quality may have been caused by natural processes.

The constructed water levels (fig. 21) were compared to the adjusted inferred total phosphorus concentrations obtained from the sediment core analysis (fig. 8) to determine if the small long-term changes in water quality and slightly deteriorated water quality following several years of high water levels found in this study are consistent with previous changes that are archived in the sediments of the lake (fig. 22). Both the current monitoring and modeling study and the results of the sediment-core analyses indicate that there has been very little long-term change in the water quality of the lake. Results from the phosphorus budget indicated that about 85 percent of the phosphorus loading to the lake was from natural sources and, therefore, little long-term change would be expected. The timing of slightly deteriorated water quality following several years of high water levels is also roughly consistent with results found in the sediment-core study. High total phosphorus concentrations, although they were only slightly higher than those during periods with low water levels, were found near the end of extended periods of high water levels from 1900 to 2004. Toward the end of each period of high water level, the inferred phosphorus concentration increased by about 0.001 mg/L except toward the end of the high water levels around 1985; however, no data were available from the sediment core for the period from 1984 to 1992. Therefore, it appears that the recent degradation in water quality is consistent with high phosphorus concentrations associated with sustained high water levels (and increased phosphorus input) that may have occurred several times in the past.

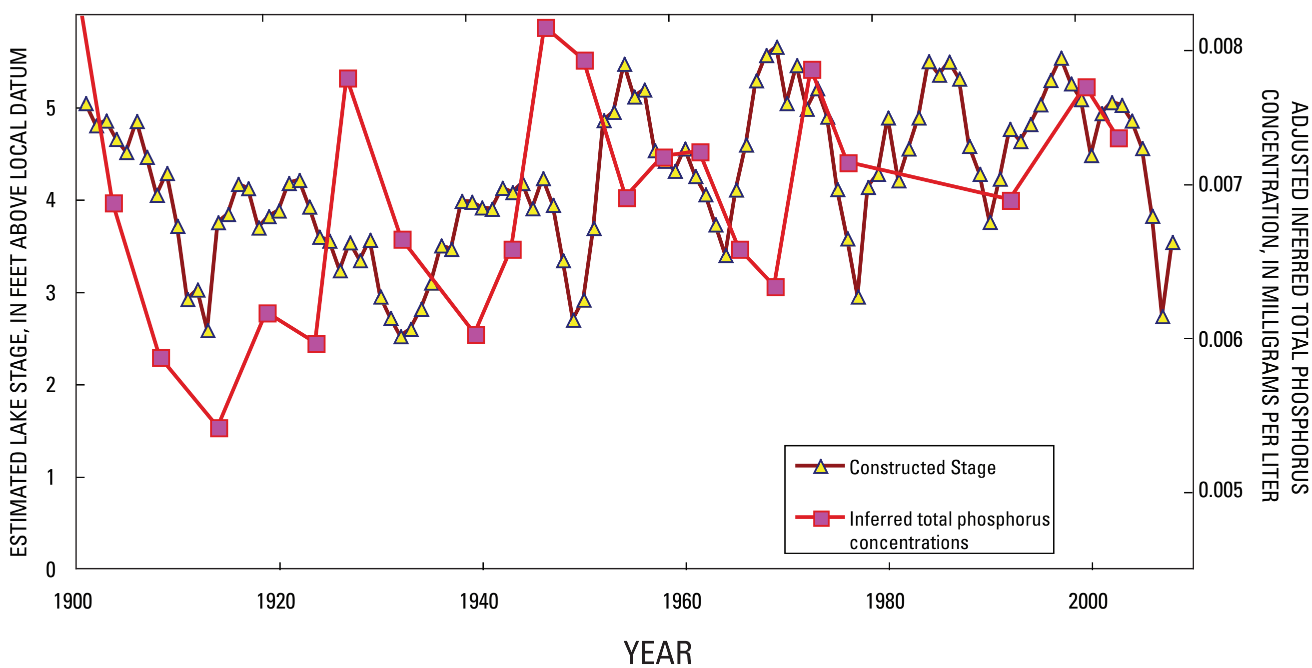


Figure 22. Adjusted inferred total phosphorus concentrations in the South Basin of Whitefish Lake, Douglas County, Wis., compared to the estimated lake stage (water level) in Whitefish Lake, 1900 to 2008. [Altitude of the local datum is 1,029.34 feet above the National Geodetic Vertical Datum of 1929.]

Summary and Conclusions

Whitefish Lake is an oligotrophic, soft-water, seepage lake in northwestern Wisconsin that is classified by the WDNR as an Outstanding Resource Water because of its clear waters and diversity of plants and wildlife. Data collected as part of the ongoing monitoring program conducted by the local community demonstrated that the pelagic water quality of the lake began to degrade, shown by increases in phosphorus and chlorophyll *a* concentrations, around 2000–02. The decrease in water quality, however, coincided with relatively high water levels in the lake. To provide a better understanding of what may have been causing the apparent degradation in water quality, and therefore, provide information for management of the lake should development increase or the climate change, the USGS conducted a detailed study during 2004–08. The goals of the study were to describe the lake's past and present water quality, construct water and phosphorus budgets, simulate the potential effects of changes in phosphorus inputs on the lake's water quality, analyze changes in the water level in the lake since 1900, and relate the importance of changes in water level (and associated changes in phosphorus input) and anthropogenic changes in the watershed to the water quality of the lake.

Since 1998, summer average total phosphorus concentrations have increased from near the 0.005-mg/L detection limit to about 0.010 mg/L in 2006, before decreasing slightly in 2007–08. From 1998 to 2008, near-surface chlorophyll *a* concentrations and Secchi depths have remained relatively stable at about 1.5 µg/L and 26 ft, respectively. Therefore, Whitefish Lake would typically be classified as oligotrophic.

Because the productivity in Whitefish Lake should be limited by phosphorus, it should be the nutrient of concern when management efforts to improve or prevent degradation in the lake's water quality are considered. Most phosphorus enters the lake with water inputs; therefore, both water and phosphorus budgets were constructed. Precipitation and groundwater inflow were the predominant sources of water, each supplying about one-half of the total input, whereas surface-water inflow supplied less than 1 percent of total input. Because Whitefish Lake has little contributions from the watershed, it was believed that much of the phosphorus reaching the lake came from the atmosphere; therefore, wetfall and dryfall phosphorus input from the atmosphere were measured in this study. The amount of phosphorus input from the atmosphere was about 3 times higher than that computed based on previous wetfall phosphorus concentrations. The concentrations and deposition rates estimated at Whitefish Lake may be used to estimate atmospheric loading in future lake studies. The average annual load of phosphorus to the lake was 232 lb, of which 131 lb (56 percent) came from atmospheric deposition, 63 lb (27 percent) came from groundwater, and about 36 lb (16 percent) came from septic systems. To determine how the hydrologic and phosphorus loading to the lake have varied with fluctuations in water level, water and phosphorus budgets were constructed for a series of dry years (low water levels) and wet years (high water levels). The total water input into the lake ranged by only 13 percent and the total phosphorus loading ranged by 10 percent.

Results from the Canfield and Bachmann eutrophication model and Carlson TSI equations demonstrated that changes in controllable phosphorus loading have a linear relationship with phosphorus and chlorophyll *a* concentrations and Secchi depths in the lake; however, the changes should be only 10 to 20 percent of the changes in controllable phosphorus loading. In other words, a 50-percent increase in controllable external phosphorus loading should cause a 5- to 10-percent change in water quality. The response is small not because the lake is unresponsive to external loading, but because most of the phosphorus loading to the lake comes from uncontrollable sources. A 16.4-percent increase in total phosphorus loading caused total phosphorus concentrations to increase by 11.4 percent, chlorophyll *a* concentrations to increase by 16.9 percent, and the Secchi depths to decrease by 10.1 percent. Therefore, changes in phosphorus loading should affect the water quality of the lake.

Seven scenarios were used to simulate the effects of anthropogenic and natural/climatic (water level and the associated changes in phosphorus input) changes on water quality, which demonstrated that: surface-water inflow (runoff) presently has little effect on the lake, changes in the inputs from septic systems and potential development in the watershed could have a large effect on water quality, decreases in water level (and associated decreases in phosphorus input) has little effect on water quality, and sustained high water levels (and associated increases in phosphorus input) following several wet years could cause a small degradation in water quality. Although increases in phosphorus input associated with high water levels may degrade water quality, it appears that anthropogenic changes in the watershed may be more important than natural/climatic changes in affecting the future water quality of the lake.

Reconstructed water levels of Whitefish Lake indicated that the changes in water level since 1998 are representative of fluctuations that have occurred since 1900, with fluctuations of about 3 ft occurring about every 15 years. A comparison of constructed water levels with the inferred total phosphorus concentrations obtained from sediment core analyses indicated that there has been very little long-term change in water quality and that there is a slight deterioration in water quality following sustained high water levels. Therefore, it appears that the recent degradation in water quality is consistent with what may have occurred several times in the past associated with sustained high water levels.

Whitefish Lake is presently a relatively pristine lake, and it is classified by the WDNR as an Outstanding Resource Water primarily because of the small input of phosphorus from its watershed. Although the recent degradation in water quality appears to be associated primarily with increases in phosphorus input associated with high water levels, the lake is still susceptible to degradation associated with small changes in its watershed. Because of the small amount of phosphorus that is presently input into the lake, what many would consider a small increase in phosphorus input could have a large effect on the lake's water quality. Therefore, management actions to minimize future phosphorus input into this lake are important for maintaining the quality of its water.

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