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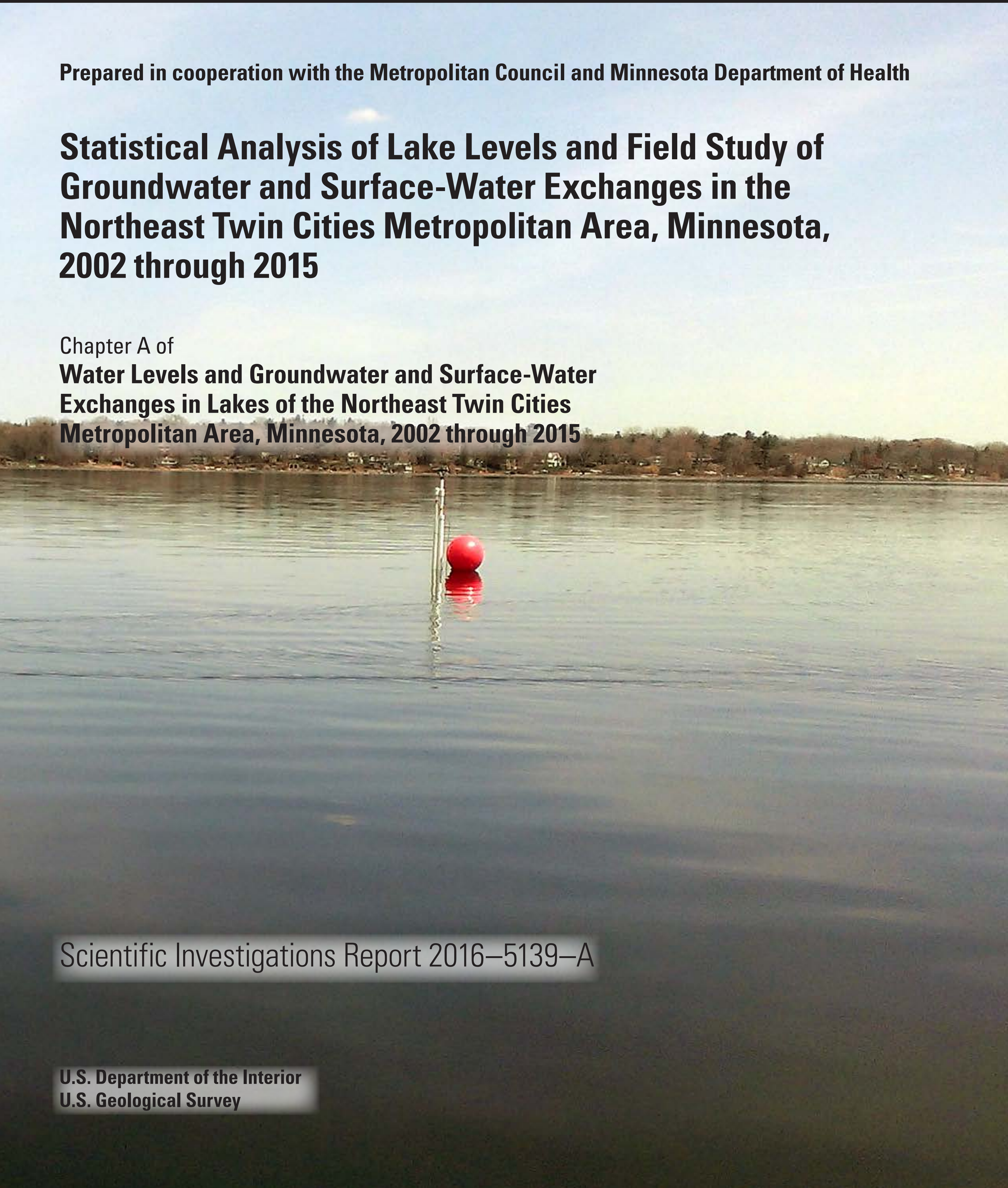
# Statistical Analysis of Lake Levels and Field Study of Groundwater and Surface-Water Exchanges in the Northeast Twin Cities Metropolitan Area, Minnesota, 2002 through 2015

Chapter A of
**Water Levels and Groundwater and Surface-Water Exchanges in Lakes of the Northeast Twin Cities Metropolitan Area, Minnesota, 2002 through 2015**



Scientific Investigations Report 2016–5139–A

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

**Cover photograph.** Deepwater piezometer in White Bear Lake, May 5, 2014 (photograph by Jared J. Trost, U.S. Geological Survey).

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By Perry M. Jones, Jared J. Trost, Aliesha L. Diekoff, Donald O. Rosenberry, Eric A. White, Melinda L. Erickson, Daniel L. Morel, and Jessica M. Heck

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SALLY JEWELL, Secretary

**U.S. Geological Survey**
Suzette M. Kimball, Director

U.S. Geological Survey, Reston, Virginia: 2016

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# Conversion Factors

U.S. customary units to International System of Units

| Multiply | By | To obtain |
|---|---|---|
| | Length | |
| inch (in.) | 2.54 | centimeter (cm) |
| inch (in.) | 25.4 | millimeter (mm) |
| foot (ft) | 0.3048 | meter (m) |
| mile (mi) | 1.609 | kilometer (km) |
| yard | 0.9144 | meter (m) |
| | Area | |
| acre | 4,047 | square meter ($m^2$) |
| acre | 0.4047 | hectare (ha) |
| acre | 0.004047 | square kilometer ($km^2$) |
| square foot ($ft^2$) | 929.0 | square centimeter ($cm^2$) |
| square foot ($ft^2$) | 0.09290 | square meter ($m^2$) |
| | Volume | |
| barrel (bbl; petroleum, 1 barrel=42 gal) | 0.1590 | cubic meter ($m^3$) |
| gallon (gal) | 3.785 | liter (L) |
| gallon (gal) | 0.003785 | cubic meter ($m^3$) |
| cubic foot ($ft^3$) | 0.02832 | cubic meter ($m^3$) |
| | Flow rate | |
| foot per day (ft/d) | 0.3048 | meter per day (m/d) |
| gallon per minute (gal/min) | 0.06309 | liter per second (L/s) |
| inch per day (in/d) | 0.0254 | meter per day (m/d) |
| inch per year (in/yr) | 25.4 | millimeter per year (mm/yr) |
| mile per hour (mi/h) | 1.609 | kilometer per hour (km/h) |
| | Pressure | |
| inch of mercury at 60 °F (in Hg) | 25.4 | mm of mercury at 60 °F (in Hg) |
| | Energy | |
| kilowatthour (kWh) | 3,600,000 | joule (J) |
| | Hydraulic conductivity | |
| foot per day (ft/d) | 0.3048 | meter per day (m/d) |

International System of Units to U.S. customary units

| Multiply | By | To obtain |
|---|---|---|
| | Volume | |
| liter (L) | 0.2642 | gallon (gal) |
| milliliter (mL) | 0.0338 | ounce, fluid (fl. oz) |

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

$$°F = (1.8 \times °C) + 32.$$

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

$$°C = (°F - 32) / 1.8.$$

## Datum

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

Vertical datums for lake levels and surface-water outlets varied between the lakes. The datums for each lake are provided in Minnesota Department of Natural Resources (2015).

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

## Supplemental Information

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

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

Chlorofluorocarbon (CFC) concentrations are given in units of picograms per kilogram (pg/kg) and picomoles per kilogram (pmol/kg). One picogram is $10^{-12}$ grams. One picomole is $10^{-12}$ moles. One mole contains $6.022 \times 10^{23}$ atoms or molecules of a substance. Sulfur hexafluoride ($SF_6$) concentrations are given in units of femtograms per kilogram (fg/kg) and femtomoles per kilogram (fmol/kg). One femtogram is $10^{-15}$ grams. One femtomole is $10^{-15}$ moles. CFC and $SF_6$ concentrations also are given in units of parts per trillion by volume (pptv), which represents the atmospheric concentration that would have yielded a measured aqueous concentration assuming equilibrium partitioning between atmosphere and water under the specified conditions (recharge temperature, recharge elevation, and excess air concentration).

Stable isotopes of oxygen and hydrogen in water are expressed in per mil, which is a unit expressing the ratio of stable-isotopic abundances of an element in a sample to those of a standard material. Per mil units are equivalent to parts per thousand. The Vienna Standard Mean Ocean Water-Standard Light Antarctic Precipitation was the standard material used.

## Abbreviations

| | |
|---|---|
| $\rho$ | rho |
| ANOVA | analysis of variance |
| Ar | argon gas |
| CFCs | chlorofluorocarbons |
| CFC–11 | trichlorofluoromethane ($CFCl_3$) |
| CFC–12 | dichlorodifluoromethane ($CF_2Cl_2$) |
| CFC–113 | trichlorotrifluoroethane ($C_2F_3Cl_3$) |
| CSP | continuous seismic-reflection profiling |
| CV | coefficient of variation |
| GIS | geographic information system |
| KGS | Kansas Geological Survey |

| | |
|---|---|
| LacCore | University of Minnesota's National Lacustrine Core Facility |
| LPMs | lumped parameter models |
| MGS | Minnesota Geological Survey |
| MNDNR | Minnesota Department of Natural Resources |
| $N_2$ | nitrogen gas |
| NAWQA | National Water-Quality Assessment |
| NLCD | National Land Cover database |
| NWIS | National Water Information System |
| PVC | polyvinyl chloride |
| RMSE | root-mean-square error |
| $SF_6$ | sulfur hexafluoride |
| SLAP | Vienna Standard Mean Ocean Water-Standard Light Antarctic Precipitation |
| SSURGO | Soil Survey Geographic database |
| SWB | Soil-Water Balance model |
| USGS | U.S. Geological Survey |

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# Statistical Analysis of Lake Levels and Field Study of Groundwater and Surface-Water Exchanges in the Northeast Twin Cities Metropolitan Area, Minnesota, 2002 through 2015

By Perry M. Jones,[1] Jared J. Trost,[1] Aliesha L. Diekoff,[1] Donald O. Rosenberry,[1] Eric A. White,[1] Melinda L. Erickson,[1] Daniel L. Morel,[1] and Jessica M. Heck[2]

## Abstract

Water levels declined from 2003 to 2011 in many lakes in Ramsey and Washington Counties in the northeast Twin Cities Metropolitan Area, Minnesota; however, water levels in other northeast Twin Cities Metropolitan Area lakes increased during the same period. Groundwater and surface-water exchanges can be important in determining lake levels where these exchanges are an important component of the water budget of a lake. An understanding of groundwater and surface-water exchanges in the northeast Twin Cities Metropolitan Area has been limited by the lack of hydrologic data. The U.S. Geological Survey, in cooperation with the Metropolitan Council and Minnesota Department of Health, completed a field and statistical study assessing lake-water levels and regional and local groundwater and surface-water exchanges near northeast Twin Cities Metropolitan Area lakes. This report documents the analysis of collected hydrologic, water-quality, and geophysical data; and existing hydrologic and geologic data to (1) assess the effect of physical setting and climate on lake-level fluctuations of selected lakes, (2) estimate potential percentages of surface-water contributions to well water across the northeast Twin Cities Metropolitan Area, (3) estimate general ages for waters extracted from the wells, and (4) assess groundwater inflow to lakes and lake-water outflow to aquifers downgradient from White Bear Lake.

Statistical analyses of lake levels during short-term (2002–10) and long-term (1925–2014) periods were completed to help understand lake-level changes across the northeast Twin Cities Metropolitan Area. Comparison of 2002–10 lake levels to several landscape and geologic characteristics explained variability in lake-level changes for 96 northeast Twin Cities Metropolitan Area lakes. Application of several statistical methods determined that (1) closed-basin lakes (without an active outlet) had larger lake-level declines than flow-through lakes with an outlet; (2) closed-basin lake-level changes reflected groundwater-level changes in the Quaternary, Prairie du Chien, and Jordan aquifers; (3) the installation of outlet-control structures, such as culverts and weirs, resulted in smaller multiyear lake-level changes than lakes without outlet-control structures; (4) water levels in lakes primarily overlying Superior Lobe deposits were significantly more variable than lakes primarily overlying Des Moines Lobe deposits; (5) lake-level declines were larger with increasing mean lake-level elevation; and (6) the frequency of some of these characteristics varies by landscape position. Flow-through lakes and lakes with outlet-control structures were more common in watersheds with more than 50 percent urban development compared to watersheds with less than 50 percent urban development. A comparison of two 35-year periods during 1925–2014 revealed that variability of annual mean lake levels in flow-through lakes increased when annual precipitation totals were more variable, whereas variability of annual mean lake levels in closed-basin lakes had the opposite pattern, being more variable when annual precipitation totals were less variable.

Oxygen-18/oxygen-16 and hydrogen-2/hydrogen-1 ratios for water samples from 40 wells indicated the well water was a mixture of surface water and groundwater in 31 wells, whereas ratios from water sampled from 9 other wells indicated that water from these wells receive no surface-water contribution. Of the 31 wells with a mixture of surface water and groundwater, 11 were downgradient from White Bear Lake, likely receiving water from deeper parts of the lake.

Age dating of water samples from wells indicated that the age of water in the Prairie du Chien and Jordan aquifers can vary widely across the northeast Twin Cities Metropolitan Area. Estimated ages of recharge for 9 of the 40 wells sampled for chlorofluorocarbon concentrations ranged widely from the early 1940s to mid-1970s. The wide range in estimated ages of recharge may have resulted from the wide range in the open-interval lengths and depths for the wells.

[1]U.S. Geological Survey.

[2]University of Minnesota.

Results from stable isotope analyses of water samples, lake-sediment coring, continuous seismic-reflection profiling, and water-level and flow monitoring indicated that there is groundwater inflow from nearshore sites and lake-water outflow from deep-water sites in White Bear Lake. Continuous seismic-reflection profiling indicated that deep sections of White Bear, Pleasant, Turtle, and Big Marine Lakes have few trapped gases and little organic material, which indicates where groundwater and lake-water exchanges are more likely. Water-level differences between White Bear Lake and piezometer and seepage measurements in deep waters of the lake indicate that groundwater and lake-water exchange is happening in deep waters, predominantly downgradient from the lake and into the lake sediment. Seepage fluxes measured in the nearshore sites of White Bear Lake generally were higher than seepage fluxes measured in the deep-water sites, which indicates that groundwater-inflow rates at most of the nearshore sites are higher than lake-water outflow from the deep-water sites.

# Introduction

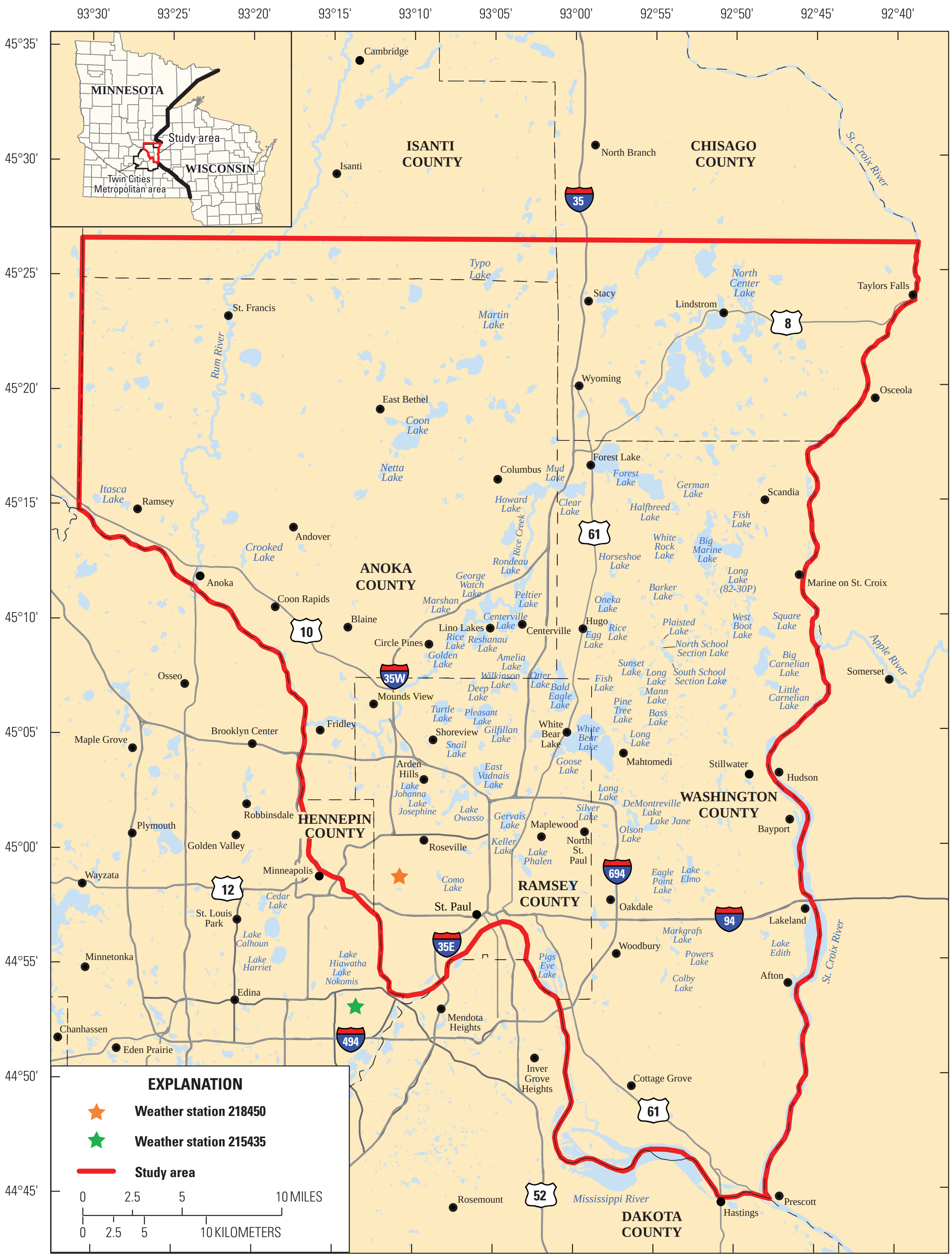
Recently (2003–13), water levels were historically low for several lakes in Ramsey and Washington Counties in the northeast Twin Cities Metropolitan Area in Minnesota (fig. 1). These lakes include White Bear, Turtle, and South School Section Lakes (fig. 1); and low water levels have limited access and recreational use of the lakes (that is, boating, fishing, and swimming). Water levels in some northeast Twin Cities Metropolitan Area lakes have recovered to near their ordinary high water levels, whereas others have not. The ordinary high water level is defined in Minnesota State Statutes as an elevation delineating the highest water level that has been maintained for a sufficient period to leave evidence on the landscape, commonly the point where the natural vegetation changes from predominantly aquatic to predominantly terrestrial (Minnesota Office of the Revisor of Statutes, 2016a). Periods of historically lower water levels in White Bear Lake and other northeast Twin Cities Metropolitan Area lakes correlate with periods of below-normal precipitation (Minnesota Department of Natural Resources, 1998).

Many hydrologic and physical characteristics can affect water levels in lakes. Seasonal and long-term (1925–2014) changes in precipitation and evaporation have resulted in lower lake levels, potentially reflecting weather variability or climate change (Williamson and others, 2009). Physical characteristics that can affect lake-water levels include the size of the lake, physical setting, and location relative to aquifers used for water supplies. For large lakes, groundwater and surface-water exchanges have less of an effect on seasonal or annual changes in water levels and storage than on smaller lakes; changes in large lakes generally are driven by differences in net water balances related to snowmelt, precipitation, and evaporation (Wilcox and others, 2007; Watras and others, 2013). Groundwater levels in aquifers can decline, which reduces water levels in lakes hydrologically connected to the aquifers (Zektser and others, 2005).

Lakes in the northeast Twin Cities Metropolitan Area with low water levels typically have no large surface-water inlets or outlets (rivers, streams, or ditches), and surface-water outflow happens only at high water levels. These closed-basin lakes tend to have smaller watersheds compared to lakes with more stable water levels. Under steady-state hydrologic conditions, water levels fluctuate more in closed-basin lakes than in flow-through lakes; even slight shifts in climatic or other hydrologic conditions can cause water-level changes of several feet (ft) in closed-basin lakes (Almendinger, 1990).

Water levels in closed-basin and other types of lakes reflect a balance of water inflows and outflows from a lake (Horne and Goldman, 1994). Water inflow to these lakes includes direct rainfall on the lake, surface runoff, and recharge to the local groundwater system that eventually seeps into the lake. Snowmelt contributes water to the lake through surface runoff and recharge to the local groundwater system. Water outflow from closed-basin lakes consists of evaporation from the lake surface and groundwater outflow. Low water levels in closed-basin lakes in the northeast Twin Cities Metropolitan Area can result from decreased precipitation, increased evapotranspiration, decreased groundwater inflow, or increased groundwater outflow from the lakes. Increased groundwater withdrawals from aquifers in the groundwater basin (which is the groundwater contributing area) of a lake could reduce groundwater and lake levels because of decreased amount of groundwater flowing into the lake or increased water outflow from the lake to aquifers. Routing of surface water out of the lake watershed could also reduce the amount of leakage or recharge to local groundwater systems, thereby reducing groundwater levels and potentially groundwater inflow to the lake.

Many lakes in the northeast Twin Cities Metropolitan Area have a long, complex history of modifications that address low or high lake levels, which often complicates the process of determining how lake levels respond to climate and water use in a watershed. One lake with a long, complex history of lake-level management is Lake Phalen (Bryan Murphy, St. Paul Parks and Recreation, written commun., April 26, 2016). In 1879, a maximum water-level elevation was established for the lake (Anklan, 1964). Between 1879 and 1950, Lake Phalen was dredged and connected to a chain of nearby lakes, and groundwater was pumped into the lake to stabilize lake levels (Anklen, 1964); however, water levels in the lake were still low in the 1950s and 1960s (Anklan, 1964), which generated public concern (St. Paul Dispatch, 1967; St. Paul Dispatch, 1968a; St. Paul Dispatch, 1968b). Another example of a lake with a complex history of lake-level management is White Bear Lake, which has had three different outflow elevations since 1925 to stabilize water levels (Jones and others, 2013). As another example, surface-water flow near Otter Lake has been altered drastically during 1925 through 2014 because of increased development and impervious surfaces, stormwater routing, ditching, and groundwater withdrawals (TKDA, 2005).




Base modified from Minnesota Department of Natural Resources digital data, 1:100,000
Minnesota Department of Transportation digital data, 1:100,000
Minnesota Department of Agriculture digital data, 1:100,000
ESRI digital data, 1:500,000, and U.S. Geological Survey digital data, 1:100,000
Universal Transverse Mercator projection, Zone 15
Horizontal coordinate information is referenced to
the North American Datum of 1983 (NAD 83)

**Figure 1.** Location of study area, lakes, and weather stations, northeast Twin Cities Metropolitan Area, Minnesota.

Recent urban expansion and associated land-use modifications in Ramsey and Washington Counties put into question whether or not recent (2003–11) water-level declines in White Bear Lake and other northeast Twin Cities Metropolitan Area lakes solely are due to precipitation declines. A U.S. Geological Survey (USGS) cooperative study for White Bear Lake determined that changes in other hydrologic variables besides precipitation were needed to explain the 2003–11 water-level decline (Jones and others, 2013). Analysis of the 2003–11 water-level decline in White Bear Lake indicated that a combination of decreased precipitation and increased groundwater withdrawals from the underlying Prairie du Chien-Jordan aquifer could explain the change in lake-level response to precipitation (Jones and others, 2013). The Prairie du Chien-Jordan aquifer is a major aquifer used for water supply in the northeast Twin Cities Metropolitan Area. Annual and summer groundwater withdrawals from the aquifer in the area have more than doubled from 1980 through 2010; groundwater withdrawals by municipalities account for most of the increase in groundwater withdrawals (Jones and others, 2013). Stable isotope analyses of well-water, precipitation, and lake-water samples collected by Jones and others (2013) indicated that water from White Bear Lake was flowing to the underlying aquifer and reaching wells that were open to the aquifer downgradient from the lake. Increases in water withdrawals from the aquifer and a hydraulic connection between the lake and the aquifer may result in an increased amount of water flowing from White Bear Lake to the aquifer, which could result in lower water levels in the lake.

Groundwater and surface-water exchanges can be an important factor in lake-level and ecosystem management and must be understood to assess the potential effects of management decisions on lake levels, particularly in closed-basin lakes. Two northeast Twin Cities Metropolitan Area lakes (Gilfillan and Snail Lakes) are augmented with surface water from the Mississippi River (fig. 1) (City of Shoreview, 2016). Other lakes (White Bear and Turtle Lakes) are being considered for lake-level augmentation (Minnesota Department of Natural Resources, 2016a; Short Elliott Hendrickson Inc., 2015). Quantification of the water balance components and groundwater and surface-water exchanges of these lakes would be needed to assess the effects of an augmentation plan. Consideration of groundwater and surface-water exchanges at deeper depths is warranted for water balances determined for deep lakes; however, exchanges are often calculated only between shallow groundwater systems and the lake. An understanding of groundwater and surface-water exchanges in the northeast Twin Cities Metropolitan Area has been limited by the lack of hydrologic data. To address this need, the U.S. Geological Survey, in cooperation with the Metropolitan Council and Minnesota Department of Health, completed a field and statistical study assessing lake-water levels and regional and local groundwater and surface-water exchanges near northeast Twin Cities Metropolitan Area lakes. The objectives of this study were to (1) identify northeast Twin Cities Metropolitan Area lakes (fig. 1) with low water levels and relatively large water-level fluctuations compared to other northeast Twin Cities Metropolitan Area lakes, (2) determine potential causes for these low and fluctuating water levels, and (3) assess groundwater and surface-water exchanges near selected northeast Twin Cities Metropolitan Area lakes.

A variety of hydrologic, water-quality, and geological techniques often are needed to assess groundwater and surface-water exchanges because they are complex (Rosenberry and LaBaugh, 2008). For this study, historical lake-level and hydrologic data from 1925 through 2014 were analyzed and compared to lake characteristics and physical settings to determine what factors might be affecting lake-level fluctuation. Stable isotope (oxygen and hydrogen) analyses and age dating of water samples were used to assess groundwater and surface-water exchanges across the northeast Twin Cities Metropolitan Area. Local groundwater and surface-water exchanges were assessed in the deep and shallow parts of White Bear Lake using four methods: (1) collection of continuous seismic-reflection profiling (CSP) data, (2) collection of lake-sediment cores, (3) measurement of water levels in piezometers, and (4) measurement of water flow in seepage meters. The CSP data were also collected in five other lakes (Big Marine Lake, Lake Elmo, Pleasant Lake, South School Section Lake, and Turtle Lake) to determine areas of these lakes where groundwater and surface-water exchange may happen. The hydrologic, statistical, and geologic techniques applied in this study have been successfully applied in other groundwater and surface-water interaction studies in Minnesota and around the United States (Dinçer, 1968; Jones, 2006; Jones and others, 2013; LaBaugh and others, 1997; Luukkonen and others, 2004; Sacks, 2002).

## Purpose and Scope

This report describes groundwater and surface-water exchanges and water levels in 96 lakes of the northeast Twin Cities Metropolitan Area, Minnesota, 2002 through 2015, based on results of a field and statistical study. This report also describes relations between precipitation and lake levels in 14 lakes during a long-term period (1925–2014). This report documents the analysis of collected hydrologic, water-quality, and geophysical data, and existing hydrologic data to (1) assess the effect of physical setting and climate on lake-level fluctuations of selected lakes, (2) estimate potential percentages of surface-water contributions to well water across the northeast Twin Cities Metropolitan Area, (3) estimate general ages for waters extracted from the wells, and (4) assess groundwater inflow to lakes and lake-water outflow to aquifers downgradient from White Bear Lake.

## Description of Study Area, Hydrology, and Hydrogeology

The northeast Twin Cities Metropolitan Area is in the gently rolling and flat glaciated landscapes of Ramsey, Washington, south Chisago, northeast Hennepin, and southeast Anoka Counties (fig. 1). About 300 lakes, and many attached and isolated wetlands, are present in the northeast Twin Cities Metropolitan Area. Some of the larger lakes include Bald Eagle, White Bear, Forest, and Big Marine Lakes (fig. 1). The larger lakes are used extensively for recreation, including fishing, boating, and swimming. Land use mainly consists of developed land in Ramsey, northeast Hennepin, and south Anoka Counties (fig. 2). The northern, eastern, and southeastern parts of the study area are a mixture of developed land, crop/pasture, forest, and wetlands (fig. 2).

The northeast Twin Cities Metropolitan Area is part of the St. Paul-Baldwin Plains and Moraines Ecological Subsection and southeast part of the Anoka Sand Plain Subsection of the Eastern Broadleaf Forest Province of Minnesota (Minnesota Department of Natural Resources, 2013a). The St. Paul-Baldwin Plains and Moraines Ecological Subsection, which makes up most of the study area, is dominated by the St. Croix Moraine, a Superior Lobe end moraine complex, and areas of glacial outwash. The Anoka Sand Plain is a broad and sandy lake plain, and covers the western and northern parts of the study area. It contains small dunes, kettle lakes, and tunnel valleys (Minnesota Department of Natural Resources, 2016b).

Climate in the northeast Twin Cities Metropolitan Area is continental, with cold winters and warm summers. The mean air temperature in July for the Minneapolis/St. Paul International Airport, Minnesota, is 73.8 degrees Fahrenheit (°F); the mean air temperature in January is 15.6 °F (National Climatic Data Center, 2016). Mean annual precipitation (1981–2010) is about 30.6 inches (in.) (National Climatic Data Center, 2016).

White Bear Lake, Turtle Lake, Big Marine Lake, Lake Elmo, and other lakes are parts of several chains of lakes that formed from the melting of glacial ice blocks lodged in bedrock valleys (Meyer and Swanson, 1992; Bauer, 2016). Many of these lakes are closed-basin lakes with no large natural surface-water inlets or outlets (rivers or streams). Surface-water outflow only happens from these closed-basin lakes during periods of high precipitation or snowmelt. The water level of many northeast Twin Cities Metropolitan Area closed-basin lakes were artificially maintained with groundwater augmentation for most years between the early 1900s and 1977 (Minnesota Department of Natural Resources, 1998).

White Bear Lake is one of the largest and deepest closed-basin lakes in the northeast Twin Cities Metropolitan Area, but its watershed is small. White Bear Lake covers about 2,100–3,100 acres, depending on the water level (Minnesota Department of Natural Resources, 1998). Recent (2011) estimates of the lake (open-water) area and watershed (drainage) area are 2,401 and 4,704 acres, respectively (Rice Creek Watershed District, 2011). The watershed-to-lake area ratio (about 2:1) is small compared to most lakes in Minnesota. White Bear Lake has a maximum depth of more than 80 ft, and the lake is deepest towards the southeast. The lake is a mesotrophic lake because it is moderately clear with an intermediate level of productivity (Minnesota Pollution Control Agency, 2016). In total, 37 storm sewer drains discharge water from local streets and wetlands into the lake.

A strong correlation exists between temporal variations in water levels of White Bear Lake and local aquifers (Minnesota Department of Natural Resources, 1998), which indicates the local groundwater system is an important factor affecting lake levels. Groundwater and lake-water exchanges in White Bear Lake happen with glacial water-table and buried aquifers in Quaternary-age deposits (Jones and others, 2013). Local residents have indicated the presence of cooler waters in lake sediments at many locations along the shore and at deeper depths, indicating potential areas of groundwater inflow (Jones and others, 2013). Groundwater and lake-water exchanges commonly take place at shallow depths where organic sediment thicknesses are less than thicknesses at deeper depths (Winter and others, 1998). Water levels in wells near White Bear Lake indicate groundwater from the glacial deposits flows downward into the St. Peter Sandstone and Prairie du Chien Group, the uppermost bedrock units below the lake (Minnesota Department of Natural Resources, 1998; Setterholm, 1991).

The geology of the northeast Twin Cities Metropolitan Area consists of Precambrian and Paleozoic bedrock (fig. 3, table 1) underlying Quaternary glacial till and outwash of pre-Wisconsin and Wisconsin age (fig. 4; table 1). Jones and others (2013), Meyer and Swanson (1992), Swanson and Meyer (1990), and Bauer (2016) provide detailed descriptions of the geology of the northeast Twin Cities Metropolitan Area.

The surficial geology of the northeast Twin Cities Metropolitan Area mostly consists of glacial sandy lake sediments, outwash, and tills associated with the Grantsburg sublobe of the late-Wisconsin age Des Moines Lobe and glacial tills and outwash associated with the St. Croix end moraine deposited by the Wisconsin-age Superior Lobe (Meyer and Swanson, 1992; Swanson and Meyer, 1990; Bauer, 2016). Pre-Wisconsin age sands, gravels, and loams of the Pierce Formation are present as the uppermost glacial deposits in south Washington County (fig. 4). Organic materials (peat) associated with wetlands and fluvial deposits commonly are above the glacial deposits (fig. 4).

Sedimentary bedrock of Ordovician and Cambrian age lies under glacial deposits in most of the northeast Twin Cities Metropolitan Area, forming the northern part of the Twin Cities Basin (table 1, fig. 3; Bauer, 2016; Meyer and Swanson, 1992; Swanson and Meyer, 1990). These sedimentary bedrock units form a sequence of major bedrock aquifers, including the St. Peter, Prairie du Chien, and Jordan aquifers, used extensively for sources of municipal, commercial, and domestic water in southeast Minnesota (Runkel and others, 2003). The bedrock units are discontinuous, and buried bedrock valleys are present throughout the northeast Twin Cities Metropolitan Area. Buried bedrock valleys formed as glacial melt waters carved through older glacial deposits and bedrock surface, and these waters

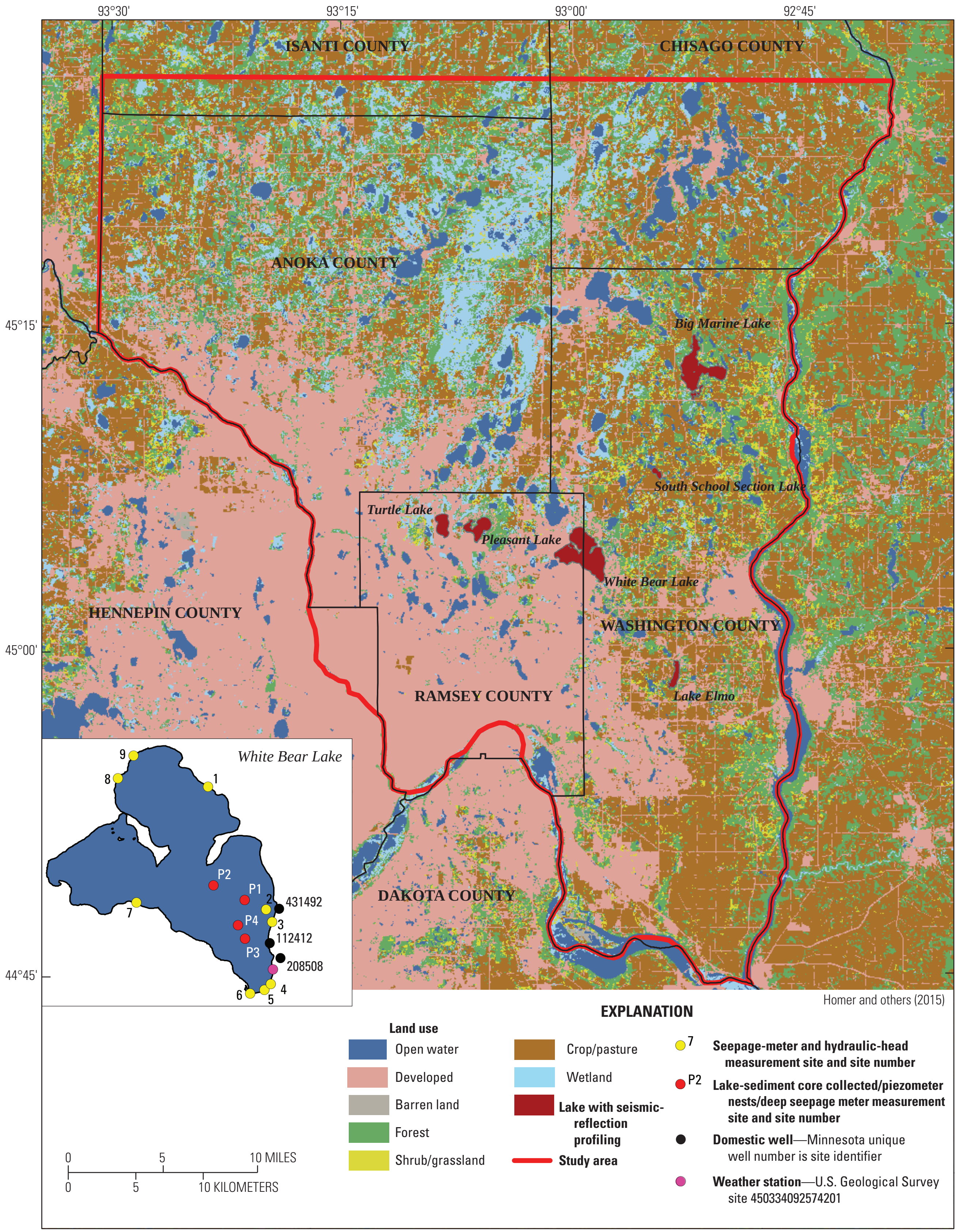



**Figure 2.** Land cover of study area and sample locations in White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.

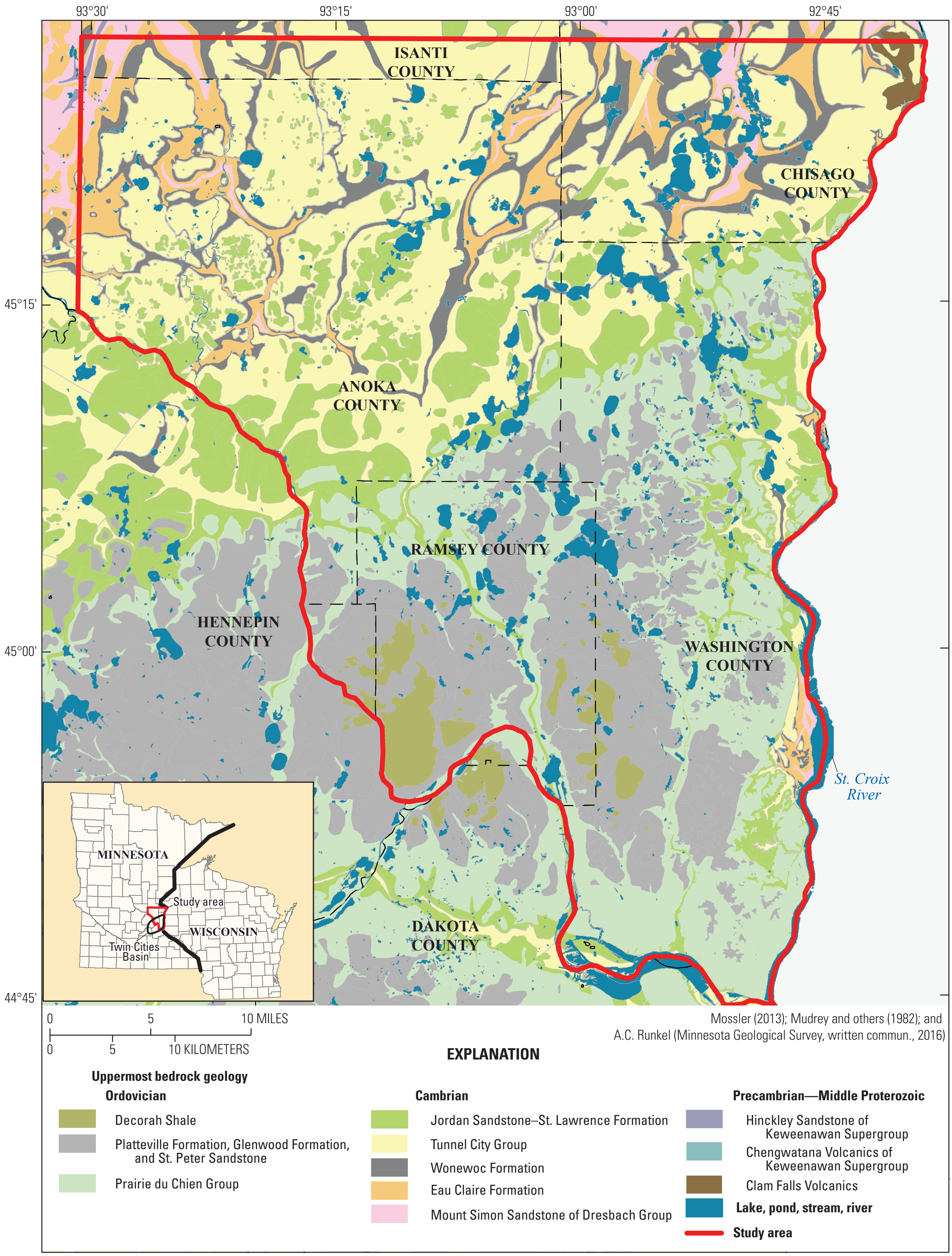



**Figure 3.** Uppermost bedrock geology of study area, northeast Twin Cities Metropolitan Area, Minnesota.

**Table 1.** Geologic units in northeast Twin Cities Metropolitan Area, Minnesota.

[Aquifer nomenclature follows the geologic nomenclature of the U.S. Geological Survey. ft, foot; ft/d, foot per day]

| System | Series | Geologic unit (aquifer) | Lithology | General thickness (ft) | Water-bearing characteristics |
|---|---|---|---|---|---|
| Quaternary | Pleistocene | Glacial lacustrine sediments (Des Moines Lobe deposits) | Fine to medium sand, silt, and clay[1,2] | | |
| Quaternary | Pleistocene | Ice-contact stratified deposits (Des Moines Lobe deposits) | Sand, loamy sand, and gravel, interbedded with silt and glacial till[1,2] | | |
| Quaternary | Pleistocene | Glacial till (Des Moines Lobe deposits) | Unsorted clay, loamy to sandy, sand and clay, gray, yellow brown, and reddish-brown, commonly mixed with Superior lobe till or sand[1,2] | | |
| Quaternary | Pleistocene | Glacial outwash (Superior Lobe deposits) | Sand, loamy sand and gravel, commonly overlain by loess[2] | | |
| Quaternary | Pleistocene | Glacial lacustrine sediments (Superior Lobe deposits) | Silt to medium-grained sand, interbedded with silty clay and gravelly sand[2] | Less than [1,2]50 to 250 | Groundwater extraction mainly by commercial and domestic wells; horizontal hydraulic conductivities for unconsolidated sediments in Metro model range from [3]23 to 240 ft/d, mean of 79 ft/d; vertical hydraulic conductivities for unconsolidated sediments in Metro model range from [3]21 to 88 ft/d, mean of 47 ft/d; horizontal hydraulic conductivities for till range from [3]3.3x$10^{-5}$ to 26 ft/d. |
| Quaternary | Pleistocene | Glacial till (Superior Lobe deposits) | Unsorted sandy-loam-textured sediments with pebbles, cobbles, and boulders, sand and gravel lenses are common, oxidized reddish brown above unoxidized reddish gray sediments[2] | | |
| Quaternary | Pleistocene | Glacial outwash and ice-contact deposits (pre- to late Wisconsin, Keewatin) | Sand, loamy sand, and gravel, some lacustrine silt and clay[2] | | |
| Quaternary | Pleistocene | Glacial till (pre- to late Wisconsin, Keewatin) | Unsorted sandy-loam-textured sediments with pebbles, cobbles, and boulders, oxidized yellowish to olive brown above unoxidized gray to dark gray[2] | | |
| Paleozoic | Ordovician (Middle) | Decorah Shale | Green calcareous shale interbedded with thin limestone[4] | [1,2]0–95 | Produces little water; horizontal hydraulic conductivities range from less than $10^{-6}$ ft/d (deep depths) to 60 ft/d (shallow depths).[4] |
| Paleozoic | Ordovician (Middle) | Platteville Formation | Fine-grained dolostone and limestone[4] | [1,2]25–33 | Produces little water; water flows mainly through bedding planes and vertical fractures. Horizontal hydraulic conductivities range from less than $10^{-2}$ ft/d (deep depths) to 98 ft/d (shallow depths).[4] |
| Paleozoic | Ordovician (Middle) | Glenwood Formation | Thin, green sandy shale[4] | [1,2]3–6 | Produces little water; horizontal hydraulic conductivity of $10^{-2}$ ft/d (shallow and deep depths).[4] |
| Paleozoic | Ordovician (Middle) | St. Peter Sandstone (St. Peter aquifer) | Fine- and medium-grained sandstone in the upper part; mudstone, siltstone, and shale interbedded with very coarse sandstone in the lower part[4] | [1,2]146–166 | Major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from $10^{-3}$ ft/d to greater than 49 ft/d; vertical hydraulic conductivities range from [3]2x$10^{-3}$ to 92 ft/d; effective porosity ranges from [3]0.28 to 0.3; storativity ranges from [3]9x$10^{-5}$ to 9.8x$10^{-3}$. |
| Paleozoic | Ordovician (Lower) | Prairie du Chien Group—Shakopee Formation (Prairie du Chien aquifer) | Thin to medium beds of dolostone, shale, and some siliciclastic sandstone[4] | [1,2]119–203 (Prairie du Chien Group) | Major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from [3,4]1.0 to 160 ft/d; vertical hydraulic conductivities range from [3,4]0.03 to 35 ft/d; effective porosity of [3]0.06; storativity ranges from [3]1.1x$10^{-5}$ to 3.4x$10^{-4}$. |
| Paleozoic | Ordovician (Lower) | Prairie du Chien Group—Oneota Dolomite (Prairie du Chien aquifer) | Thick beds of very fine-grained dolostone, fine and coarse clastic interbeds in the lower part of the formation[4] | | Part of major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from [4]1.5x$10^{-4}$ to 740 ft/d; vertical hydraulic conductivities range from [4]1.5x$10^{-4}$ to $10^{-3}$ ft/d; effective porosity of [3]0.06; storativity ranges from [3]1.1x$10^{-5}$ to 3.4x$10^{-4}$. |
| Paleozoic | Cambrian (Upper) | Jordan Sandstone (Jordan aquifer) | Coarse and fine clastic sandstone[4] | [1,2]66–101 | Major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from $10^{-2}$ to greater than [3,4]490 ft/d; vertical hydraulic conductivities range from [3,4]$10^{-4}$ to 47 ft/d; effective porosity of [3]0.32; storativity ranges from [3]4.9x$10^{-5}$ to 1.2x$10^{-4}$. |

**Table 1.** Geologic units in northeast Twin Cities Metropolitan Area, Minnesota.—Continued

[Aquifer nomenclature follows the geologic nomenclature of the U.S. Geological Survey. ft, foot; ft/d, foot per day]

| System | Series | Geologic unit (aquifer) | Lithology | General thickness (ft) | Water-bearing characteristics |
|---|---|---|---|---|---|
| Paleozoic | Cambrian (Upper) | St. Lawrence Formation | Interbedded fine clastic (sandstone, siltstone, shale) and carbonate (dolostone) rock[4] | [1,2]30–60 | Horizontal hydraulic conductivities range from less than [3,4]$10^{-2}$ to 46 ft/d; vertical hydraulic conductivities range from [3,4]$10^{-4}$ to 1.8 ft/d; effective porosity ranges from [3]0.15 to 0.20. |
| Paleozoic | Cambrian (Upper) | Tunnel City Group (formerly Franconia Formation[6]) | Shale, siltstone, and fine-grained sandstone with beds of limestone and dolostone[4] | [1,2]116–166 | Aquifer in southeast Minnesota; horizontal hydraulic conductivities range from less than [3,4]$10^{-3}$ to 98 ft/d; vertical hydraulic conductivities range from [3,4]$10^{-4}$ to 9.8 ft/d. |
| Paleozoic | Cambrian (Upper) | Wonewoc Formation (formerly the Ironton and Galesville Sandstone[6]) | Silty, fine- to coarse-grained poorly sorted sandstones in the upper part, fine- to medium-grained sandstone in the lower part[4] | [1,2]42–67 | Aquifer in southeast Minnesota; horizontal hydraulic conductivities range from [3,4]0.2 to 102 ft/d; vertical hydraulic conductivities range from [3,4]$10^{-3}$ to 8 ft/d; effective porosity of [3]0.25; storativity ranges from [3]$2.7\text{x}10^{-5}$ to $5.9\text{x}10^{-5}$. |
| Paleozoic | Cambrian (Upper) | Eau Claire Formation | Siltstone, fine- to medium-grained glauconitic sandstones, and shales[4] | [1,2]63–114 | Horizontal hydraulic conductivities range from less than [3,4]$10^{-3}$ to 0.3 ft/d; vertical hydraulic conductivities range from [3,4]$10^{-4}$ to $3\text{x}10^{-3}$ ft/d; effective porosity ranges from [3]0.28 to 0.35. |
| Paleozoic | Cambrian (Upper) | Mount Simon Sandstone | Fine- to coarse-grained quartzose sandstone, thin beds of siltstone, shale, and very fine-grained sandstone[4] | [1,2]160–336 | Major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from [3,4]$10^{-2}$ to 39 ft/d; vertical hydraulic conductivities range from [3,4]$10^{-4}$ to 14 ft/d; effective porosity of [3]0.23. |
| Precambrian | Proterozoic (Middle) | Hinckley Sandstone of Keweenawan Supergroup | Reddish-brown mudstones and siltstones, interbedded with reddish-brown feldspathic sandstone[5] | [7]more than 0.6 mile | Hydraulically connected to Mount Simon Sandstone forming a major aquifer; hydraulic conductivities and other hydrologic parameters unknown. |
| Precambrian | Proterozoic (Middle) | Chengwatana Volcanics of Keweenawan Supergroup | Basalt, porphyritic, interlayered with conglomeratic sedimentary rock[7,8] | Unknown | Generally not used as a source of water in the northeast Twin Cities Metropolitan Area; hydraulic conductivities and other hydrologic parameters unknown. |
| Precambrian | Proterozoic (Middle) | Solor Church Formation of Keweenawan Supergroup | Coarse-grained, ophitic basalt, with fine-grained, intergranlar basalt and porphyritic basalt[7] | Unknown | Generally not used as a source of water in the northeast Twin Cities Metropolitan Area; hydraulic conductivities and other hydrologic parameters unknown. |

[1]From Meyer and Swanson (1992).

[2]From Swanson and Meyer (1990).

[3]From Metropolitan Council (2016).

[4]From Runkel and others (2003).

[5]From Morey (1972).

[6]From Mossler (2008).

[7]From Setterholm (2010).

[8]From Boerboom (2001).

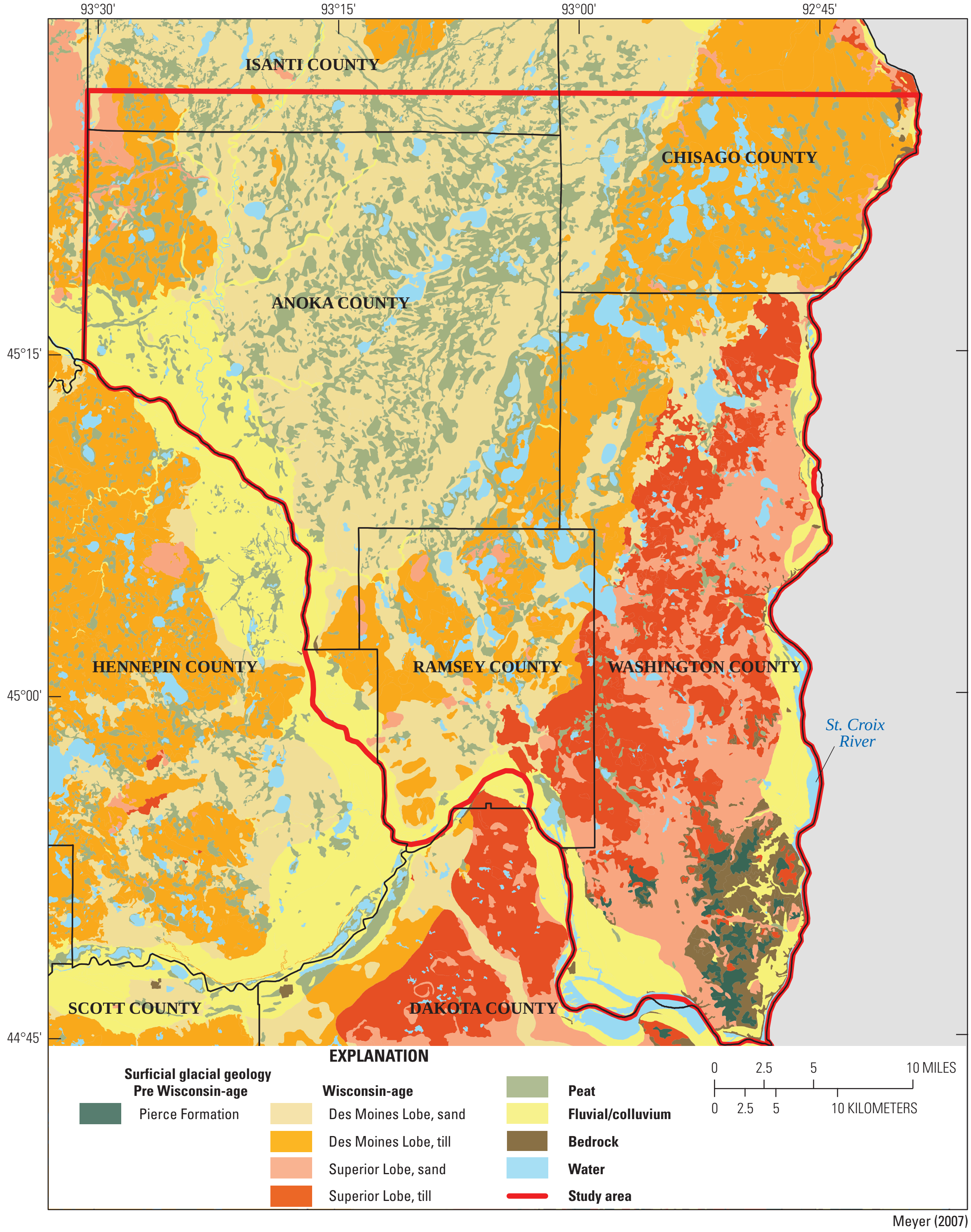



**Figure 4.** Surficial glacial geology of study area, northeast Twin Cities Metropolitan Area, Minnesota.

deposited glacial outwash and reworked till in the valleys. Precambrian volcanic bedrock underlies glacial deposits near the St. Croix River in Chisago County (fig. 3). Ordovician- and Cambrian-age bedrock only underlies soil or is exposed at the land surface in south Washington County (fig. 4).

## Previous Investigations

Previous hydrologic and hydrogeologic investigations have determined that many hydrologic, geologic, and environmental factors can affect lake levels in northeast Twin Cities Metropolitan Area lakes, including White Bear Lake (fig. 1). Coates (1924) completed one of the first evaluations of the hydrology of lakes in Ramsey County, Minn., studying the effects of low lake levels on development and recreational use in the 1920s. The first network of permanent water-level gages was established on Ramsey County lakes as part of this study, including White Bear Lake, Bald Eagle Lake, Turtle Lake, Snail Lake, Lake Johanna, Lake Josephine, Lake Owasso, and Gervais Lake (fig. 1). Coates (1924) also determined that decreases in groundwater levels resulting from groundwater withdrawals in and near the City of St. Paul had contributed to increases in water losses from many of the lakes. Brown (1985) investigated hydrologic factors affecting lake-level fluctuations in Big Marine Lake in Washington County, Minn., and defined exchanges between the lake and local groundwater systems. Hydrogeologic and geochemical data collected in the Brown (1985) study indicated lake-level fluctuations in the closed-basin lake were affected primarily by groundwater discharge (inflow) to the lake and water outflow from the lake to aquifers, and changes in the potentiometric surface of the bedrock aquifer had minor effects on lake-level changes. Barr Engineering (2010) determined that vulnerability of the northeast Twin Cities Metropolitan Area lakes and wetlands to groundwater withdrawals varied with surficial geology and the connection to groundwater.

Many hydrologic studies have been completed to assess low and fluctuating lake levels in White Bear Lake. Setterholm (1991) completed a hydrologic assessment of factors, including groundwater withdrawals, affecting lake levels on White Bear Lake for the White Bear Lake Conservation District. In the study, Setterholm recognized that glacial deposits surrounding the lake have the highest static water levels, and that successive bedrock aquifers below the glacial deposits, and the lake, have lower water levels; water from the glacial deposits recharges the underlying Prairie du Chien-Jordan aquifer in the White Bear Lake area. An historical assessment (1978–2011) of levels in White Bear Lake completed by Jones and others (2013) determined that the linear relation between annual lake-level change and annual precipitation shifted in 2003, thereby requiring a mean of 4 in. more of precipitation per year to maintain the lake level. A combination of decreased precipitation and increased groundwater withdrawals could explain the change in the lake-level response to precipitation. Almendinger (2014) discovered a strong correlation between lagged water levels in White Bear Lake and lagged streamflow in the Apple River in west Wisconsin (fig. 1), demonstrating a similar response of the lake-level and streamflow records to regional hydrologic influences, such as precipitation.

The effects of groundwater and surface-water exchanges on lake levels and lake-water budgets have been assessed in lakes across the northeast Twin Cities Metropolitan Area. Brown (1986) estimated the groundwater contribution in hydrologic and phosphorus budgets for seven lakes in the Twin Cities Metropolitan Area, including Square Lake, Eagle Point Lake, and Lake Elmo in the eastern part of the study area (fig. 1). Seasonal hydrologic budgets to Square Lake and Lake Elmo indicated a net groundwater inflow to the lakes during all seasons of the year; net annual groundwater inflows were about 39 and 69 million cubic feet ($ft^3$), respectively. Ruhl (1994) investigated groundwater and lake exchanges for East Vadnais Lake in north Ramsey County and determined that groundwater inflow and lake-water discharge to aquifers represented a small percentage of the total water budget for the lake. The Minnesota Department of Natural Resources (MNDNR) completed a lake and groundwater exchange study of White Bear Lake after a period of low lake-water levels associated with a drought during 1988 and 1989 (Minnesota Department of Natural Resources, 1998). Results from this study indicated net annual groundwater exchange with White Bear Lake between 1981 and 1990 ranged from an 11.4-in. loss to local aquifers to a 4.4-in. gain to the lake; the mean exchange was a 5-in. loss to local aquifers (Minnesota Department of Natural Resources, 1998). Coates (1924) estimated annual water outflow from White Bear Lake to aquifers ranged from 5.7 to 7.2 in. Alexander and others (2001) assessed groundwater flows to Big Marine, Big Carnelian, Square, and Little Carnelian Lakes (fig. 1) in the Carnelian-Marine Watershed District in Washington County. Alexander and others (2001) determined groundwater contribution to Big Marine and Big Carnelian Lakes was small because most water entering the lakes comes from precipitation; however, groundwater inflow was a major contributor to Square Lake and, to a lesser extent, Little Carnelian Lake. Maine on St. Croix Watershed Management Organization (2002) monitored groundwater inflow to Square Lake using seepage meters and determined that 70 percent of the total water inflow to the lake was coming from local shallow groundwater, whereas only 9 percent of the total water leaving the lake discharges to aquifers. Stable isotope analyses of well-water, precipitation, and lake-water samples done by Jones and others (2013) indicated wells downgradient from White Bear Lake screened in the glacial buried aquifer or open to the Prairie du Chien-Jordan aquifer receive a mixture of surface water and groundwater; the largest surface-water contributions are in wells closer to White Bear Lake. Winter and Pfannkuch (1976) characterized the hydrologic interconnections between lakes and lateral groundwater flow within shallow deposits of the Anoka Sand Plain and patterns of groundwater flow between surficial, valley fill, and bedrock aquifers in a buried valley near Lino Lakes, Minn. (fig. 1).

# Methods of Study

Hydrologic and water-quality data were collected and analyzed to understand the exchanges between aquifers and surface waters (lakes, wetlands, and creeks) in the northeast Twin Cities Metropolitan Area (fig. 1). Historical precipitation, groundwater-level, and lake-level data collected in the northeast Twin Cities Metropolitan Area from 1925 through 2014 were analyzed to identify which northeast Twin Cities Metropolitan Area lakes are more susceptible to low water levels and large fluctuations and determine why these lakes are more susceptible. Water-quality samples were collected from domestic, municipal, and commercial wells in the northeast Twin Cities Metropolitan Area to estimate the percentages of surface-water contributions to the wells and determine ages of waters entering wells.

A variety of physical properties and chemical constituents were measured in precipitation, surface water, pore water in lake sediments, and groundwater to determine flow directions at deep-water locations where groundwater and surface-water exchanges happen in White Bear Lake. Lake-sediment cores and CSPs were collected in White Bear Lake to characterize lake sediments and structures in the lake and estimate the thickness of organic sediments that may restrict the amount of water exchange between the lake and underlying aquifers. The CSPs also were collected in five other northeast Twin Cities Metropolitan Area lakes (Big Marine Lake, Pleasant Lake, Lake Elmo, South School Section Lake, and Turtle Lake). Piezometers were installed at four deep-water sites in White Bear Lake to measure water levels and complete slug tests in glacial sediments below the lake (sites P1–P4; fig. 2). The water levels were compared to measured lake levels in White Bear Lake to assess potential direction of groundwater and surface-water exchange in deeper waters. Slug tests were completed to determine horizontal hydraulic conductivity values for glacial sediments below the lake. Hydraulic conductivity refers to horizontal hydraulic conductivity in this report. Seepage meters were placed near the four deep-water (sites P1–P4) and nine shallow-water sites (sites 1–9) to measure flows at the water-sediment interface (fig. 2) and estimate vertical hydraulic conductivity.

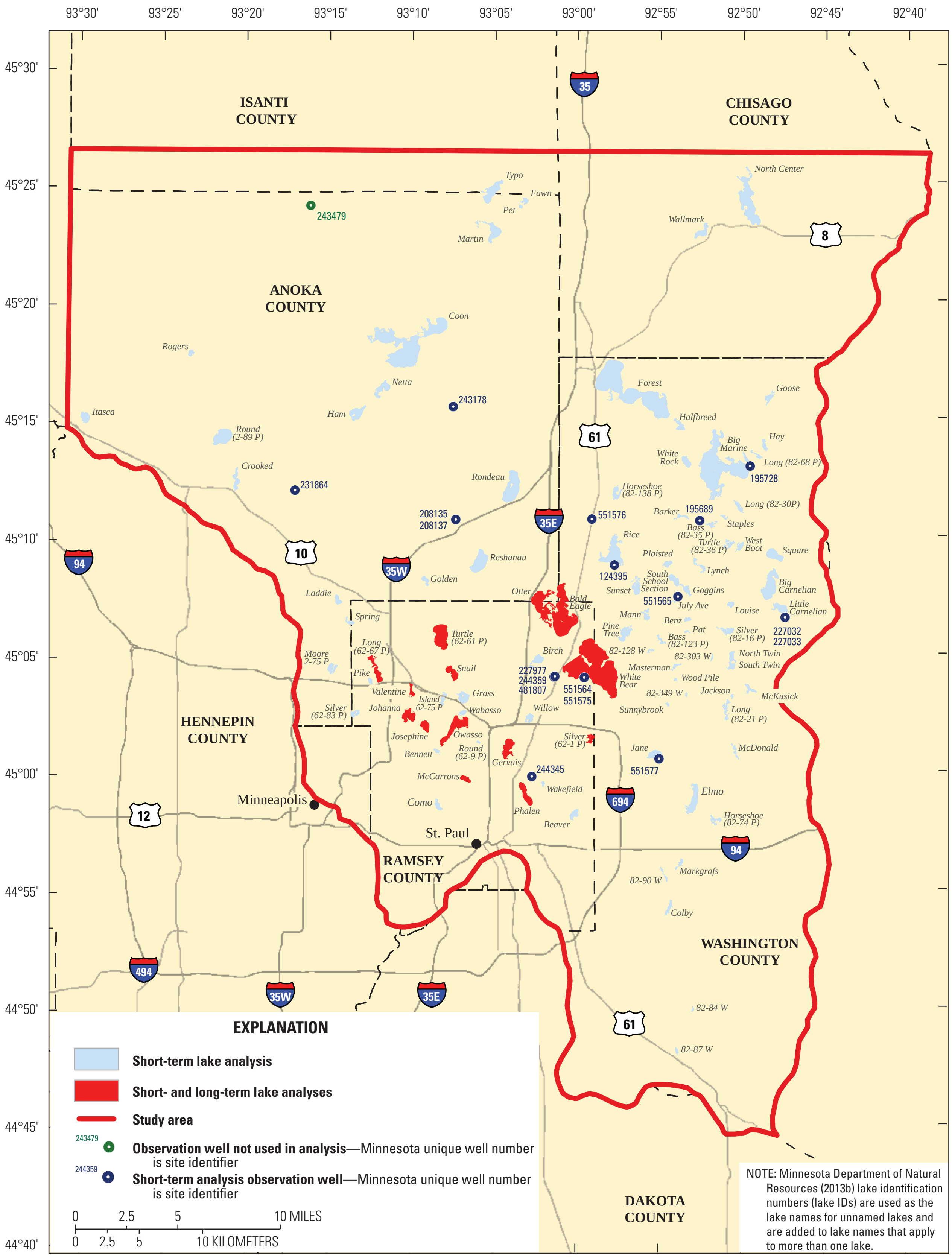
## Evaluation of Lake-Level Changes

Several datasets were processed and compiled to evaluate the lake-level changes that happened across the northeast Twin Cities Metropolitan Area. After data were compiled, a series of statistical summaries and tests were applied to the data.

### Hydrologic and Geographic Information System Data Sources and Processing

Statistical analyses were done on hydrologic and geologic data to assess lake-level fluctuations using a variety of geographic information system (GIS) data sources. Historical hydrologic and geologic data in the northeast Twin Cities Metropolitan Area (fig. 1) collected from 1925 through 2014 were compiled and analyzed. Two criteria were used to select lakes for inclusion in the statistical analyses. Lakes were required to (1) lie within the study area (fig. 5) and (2) have a complete record, defined as water-level measurements in at least 2 months for each year of analysis during open-water season (April–November). For each year of record, a monthly mean lake level was calculated for each month with data. The data were further summarized by computing an annual mean of the monthly means. These annual means for each year of complete data for each lake were used as the basis for all lake-level analyses. Two sets of statistical analyses were completed on lakes meeting these criteria. In the first set of analyses, lake-level data for 14 lakes were compared to precipitation data for 1925–2014 to assess the long-term relations between lake levels and precipitation (fig. 5). In the second set of analyses, lake-level data for 96 lakes were compared to climate data, lake characteristics, and watershed characteristics for 2002–10 (fig. 5). The goals of the short-term (2002–10) analyses were to assess the modern variability in lake-level fluctuations and identify any characteristics of lakes or their watersheds that are correlated with greater interannual lake-level fluctuations.

Water levels from 18 observation wells operated by the MNDNR were compared to the lake levels used for the short-term analyses to assess relations between lake levels and groundwater levels. The wells included in the short-term analyses were within the study area (fig. 5) and had (1) a complete data record for 1999–2014, defined as at least one water level per year in the months of January–May and November–December; (2) an open interval in glacial (Quaternary) deposits, the Prairie du Chien aquifer, or the Jordan aquifer; (3) an open interval starting less than 400 ft below land surface; and (4) a hydrograph from 1999 through 2014 that did not have any sudden, large water-level shifts. Several water-level records had a single water level that was more than 50 ft higher or lower than the rest of the record; these clear outliers were removed before analysis. Several wells also had seasonal drops in water levels caused by pumping. For this analysis, the static water-level condition was of interest. Only water levels in the months of January–May and November–December were used for the analysis because pumping typically caused drawdown during June–October each year. One well (MNDNR observation well 243479; fig. 5) met the criteria elements one through three but was excluded because it had a hydrograph with abrupt water-level changes, which indicated there is a strong local effect other than an annual pumping cycle. Well 243479 is a water-table well but had a substantial, sudden upward jump in its water-level record from 2009 through 2010 that was different than the hydrograph pattern in 2006–8 (Minnesota Department of Natural Resources, 2016c).




Base modified from Minnesota Department of Natural Resources digital data, 1:100,000
Minnesota Department of Transportation digital data, 1:100,000
Minnesota Department of Agriculture digital data, 1:100,000
U.S. Geological Survey digital data, 1:100,000
Universal Transverse Mercator projection, Zone 15
Horizontal coordinate information is referenced to
the North American Datum of 1983 (NAD 83)

**Figure 5.** Lakes and wells included in short-term (2002–10) and long-term (1925–2014) statistical analyses, northeast Twin Cities Metropolitan Area, Minnesota.

### Lake-Level, Groundwater-Level, and Precipitation Data Sources

The lake-level data were obtained from the MNDNR Lakefinder database (Minnesota Department of Natural Resources, 2015). Precipitation data for the short-term analysis were obtained from the Daymet dataset (Thornton and others, 2014). Groundwater levels were obtained from the MNDNR Cooperative Groundwater Monitoring Network Web service (Minnesota Department of Natural Resources, 2016c). Precipitation data for 1925–2014 were downloaded for a target location in Ramsey County, White Bear Township, township 30N, range 22W, section number 17, from the Precipitation Data Retrieval from a Gridded Database Web service (Minnesota Climatology Working Group, 2016a).

### Evaporation Estimates

Different equations and methods with differing data input requirements have been used to estimate open-water evaporation. Numerous publications have compared the accuracy of these methods (for example, Winter and others, 1995; Rosenberry and others, 2004). For this study, five methods and data sources were compared to determine the most appropriate method that could be applied across the study area (table 2). Detailed meteorological data were not available near every lake in the study area, so several temperature-based evaporation calculation methods were compared to field measurements made at White Bear Lake (fig. 1). The Daymet gridded dataset contains daily temperature data on a 0.6-mile (mi) (1-kilometer (km)) grid scale for the study area (Thornton and others, 2014). If a temperature-based method with Daymet data agreed with field measurements at White Bear Lake, it was assumed this method could be applied for each lake of interest.

The best-available field data for White Bear Lake (a combination of eddy-flux lake evaporation measurements and class A pan evaporation measurements in St. Paul, Minn.) initially were used to determine an evaporation reference for White Bear Lake. Second, a variety of calculation methods applied to different data sources were compared to the White Bear Lake reference on a monthly and annual basis. The method that “best reproduced” the reference evaporation estimates for White Bear Lake was then used to estimate open-water evaporation for the 96 lakes included in the statistical analysis. “Best reproduced” was defined as minimizing the root-mean-square error (RMSE) on an annual and monthly basis when compared to the reference method.

The equations, variables, and data sources for each evaporation equation are listed in table 2. Hourly climate data were downloaded from the Minneapolis St. Paul Airport weather station (KMSP, latitude: 44° 53’ 7” N, longitude: 93° 13’ 53” W, World Meteorological Organization station 72658, weather station 215435; fig. 1) and aggregated on a daily basis (Iowa State University of Science and Technology, 2015). The Daymet daily temperature data for the centroids of each of the 96 lakes were downloaded, and the appropriate variables were used as input to the various evaporation models (Thornton and others, 2014). Pan evaporation estimates were generally made between April 21 and October 10 of each year (Minnesota Climatology Working Group, 2016b). Evaporation calculations for all methods were constrained to the date ranges of the pan evaporation data to compare the methods.

The reference evaporation estimates for White Bear Lake were determined using the following procedure. Eddy-flux measurements of evaporation were made on White Bear Lake during July 24–October 31, 2014 (Tim Griffis, University of Minnesota, written commun., December 2, 2015). These data were compared to measurements of class A pan evaporation at the University of Minnesota’s St. Paul campus (Minnesota Climatology Working Group, 2016b). Total evaporation measured on White Bear Lake during this period was 69 percent of total pan evaporation; therefore, a coefficient of 0.69 was multiplied by monthly pan evaporation totals to estimate open-water evaporation for White Bear Lake for 1999–2014. The annual evaporation totals are illustrated in figure 6.

Five alternative methods were compared to the reference (pan) White Bear Lake evaporation: (1) a modified version of the FAO56 Penman-Montieth reference crop equation (Allen and others, 1998; State Climatology Office of North Carolina, 2016), (2) Hamon equation (Hamon, 1961), (3) Hargreaves-Samani method (Hargreaves and Samani, 1985; Samani, 2000), (4) Jensen-Haise equation (as in McGuinness and Bordne, 1972), and (5) the Papadakis method (as in Winter and others, 1995). A summary of the equations and data sources is in table 2.

The Hargreaves-Samani method produced the best overall annual and monthly evaporation estimates for White Bear Lake that minimized the RMSE on an annual and monthly basis compared to the other methods examined (tables 3, 4, 5). The Hargreaves-Samani method uses a *KT* coefficient (table 2) calculated from the monthly mean of daily temperature differences to estimate evaporation (Samani, 2000).

Evaporation was calculated for the open-water season for each lake using the Hargreaves-Samani method with lake-specific Daymet temperature data. Open-water season was determined from records of ice-out dates for White Bear Lake and Turtle Lake and records of ice-in dates for Turtle Lake (White Bear Lake Conservation District, 2016; Turtle Lake Homeowners Association, 2016). For each year, an annual mean ice-out date was determined for White Bear and Turtle Lakes. This date for each year was used as the ice-out date for all lakes of interest. The annual ice-in date recorded for Turtle Lake was applied to all lakes of interest. The daily evaporation estimates for each lake were aggregated on an annual basis for the statistical analyses (table 5).

**Table 2.** Summary of methods used to calculate lake evaporation.

[mm/d, millimeter per day; *PET*, potential evapotranspiration; Δ, slope vapour pressure curve (kilopascals per degree Celsius [kPa/°C]); $R_n$, net radiation flux at the crop or water surface (Megajoules per square meter per day); γ, psychrometric constant (kPa/°C); $T_{a2}$, air temperature at 2-meter (m) height (degrees Celsius [°C]); $u_2$, wind speed at 2-m height (meter per second); $e_s$, saturation vapour pressure (kilopascals [kPa]); $e_a$, actual vapour pressure (kPa); $R_s$, incident shortwave radiation, in calories per square centimeter per day; $T_{max}$, daily maximum air temperature in °C; $T_{min}$, daily minimum air temperature in °C; cm/d, centimeter per day; *D*, hours of daylight; *SVD*, saturated vapor density at mean air temperature (grams per cubic meter); --, not applicable; $R_a$, extraterrestrial, or solar radiation (cm/d); $T_{dd}$, daily temperature difference ($T_{max}$ - $T_{min}$) (°C); $T_a$, mean air temperature (°C); $T_{dm}$, monthly mean of daily temperature differences (°C); cm/mo, centimeter per month; $E0_{max}$ saturated vapor pressure at daily maximum air temperature (millibars); $E0_{min}$ saturated vapor pressure at daily minimum air temperature (millibars)]

| Equation number | Method (reference) | Units | Equation | Input data from Daymet[1] | Input data from KMSP weather station[2] |
|---|---|---|---|---|---|
| 1 | Penman-Monteith (Allen and others, 1998; State Climate Office of North Carolina, 2016) | mm/d | $PET = [(0.408*\Delta*R_n) + \gamma * (306/T_{a2} + 273) * u_2 * (e_s - e_a)]/(\Delta + \gamma)$ | $R_s$ | $T_{max}$, $T_{min}$, wind speed, relative humidity, barometric pressure. |
| 2 | Hamon, 1961 (as in Winter and others, 1995) | cm/d | $PET = 0.55 * 2.54 * (D/12)^2 * (SVD/100)$ | Day length, $T_{max}$, $T_{min}$ | -- |
| 3 | Hargreaves-Samani (Hargreaves and Samani, 1985; Samani, 2000) | cm/d | $PET = 0.0135 * KT * R_a * (T_{dd})0.5 * (T_a+17.8)$; $KT = 0.00185 * (T_{dm})^2$ | $T_{max}$, $T_{min}$ | -- |
| 4 | Jensen-Haise (as in McGuinness and Bordne, 1972) | cm/d | $PET = (0.014 * T_a - 0.37) * R_s * 0.000673 * 2.54$ | $R_s$, $T_{max}$, $T_{min}$ | -- |
| 5 | Papadakis (as in Winter and others, 1995) | cm/mo | $PET = 0.5625 * [E0_{max} - (E0_{min} - 2)]$ | $T_{max}$, $T_{minn}$ | -- |

[1]Thornton and others (2014).

[2]Iowa State University of Science and Technology (2015).

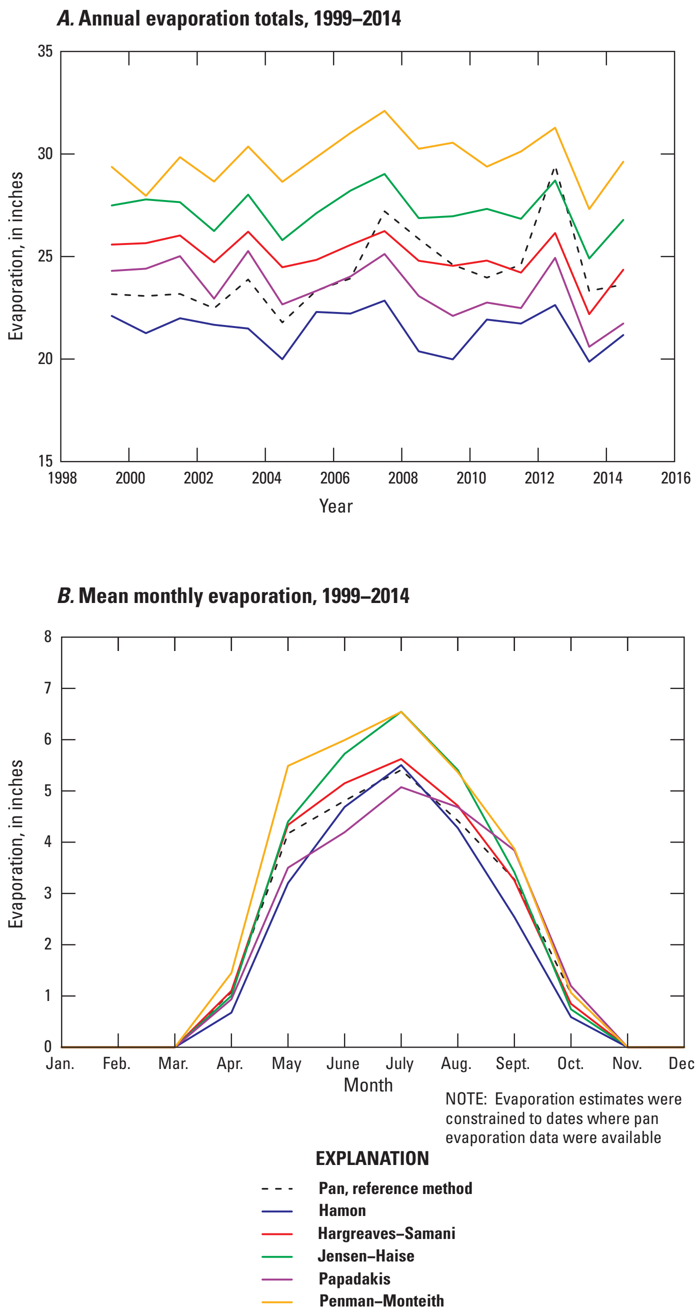



**Figure 6.** Annual and mean monthly evaporation totals calculated for White Bear Lake using six methods for 1999–2014, Minnesota.

## Geographic Information System Methodology

The GIS software (Esri, 2015) was used to determine lake hydrography, lake watershed boundaries, and many of the spatial characteristics to be investigated as potential explanatory variables for lake-level fluctuations in the northeast Twin Cities Metropolitan Area (fig. 1). Data for the 96 selected lakes are presented in appendix table 1–1. Some spatial characteristics were calculated based on the spatial extent of the lake watershed area, whereas others were based on the open-water area of the lake. The spatial characteristics can generally be divided into three categories: (1) spatial hydrography, (2) geology and soils, and (3) land cover and slope (appendix table 1–1).

### Lake Hydrography and Watershed Boundaries

Spatial extents for the 96 selected lakes and wetlands were obtained from the MNDNR Watershed Suite—Level 09 AutoCatchment Lakes (Minnesota Department of Natural Resources, 2013b). Watershed boundaries for the lakes were obtained from different water-management organizations in the northeast Twin Cities Metropolitan Area (fig. 1). The watersheds for lakes with an open-water area of 100 acres or greater were primarily obtained from the MNDNR Watershed Suite—Level 08 All Catchments (Minnesota Department of Natural Resources, 2013b). Of the 41 lakes of this size, 35 were extracted from MNDNR Level 08 Catchments using the Upstream Downstream Catchment program. Five lake-watershed boundaries (Lake Elmo, Lake Josephine, Square Lake, Turtle Lake, and White Bear Lake) were obtained from local lake-management organizations because these organizations incorporated more detailed information into their watershed delineations; for example, the Lake Elmo watershed boundary from the Valley Branch Watershed District accounts for a pipe that routes water underneath Lake Elmo from part of its watershed. The watershed for Round Lake in Andover, Minn., was extracted from the MNDNR Watershed Suite—Level 09 Automated Catchments (Minnesota Department of Natural Resources, 2013b) because a watershed for the lake was not available in the Level 08 database even though the lake has an open-water area greater than 100 acres. For lakes with a calculated open-water area of less than 100 acres (a total of 55 lakes), watershed boundaries were obtained from MNDNR Level 09 Catchments using the Upstream Downstream Catchment program, with the lake centroid as an input point. From these spatial coverages, lake open-water area (field LK_ACRES, appendix table 1–1) and watershed area (field WS_LkshedArea.ac, appendix table 1–1) were calculated using the calculate geometry tool within GIS software (Esri, 2015).

The maximum depth of each lake was obtained from watershed district, city, county, and state reports. If no maximum depth was present in any of these reports, the maximum depth was estimated to be the same as the closest similar-sized lake. The maximum lake depth is presented in field LK_MaxLkDepth.f of appendix tables 1–1 and 1–2. Lake perimeters (LK_PERIM_F) and water-body classifications of WETLAND or LAKE were obtained from the MNDNR Watershed Suite—Level 09 Automated Catchments (Minnesota Department of Natural Resources, 2013b). Mean annual lake elevation (LK_meanELEV in appendix table 1–1) is the mean of the 1999–2014 annual mean lake levels for each lake. The lake-level data were all obtained from the MNDNR Lakefinder database (Minnesota Department of Natural Resources, 2015).

**Table 3.** Mean monthly lake evaporation estimates calculated for White Bear Lake using six methods for 1999–2014, Minnesota.

| Month | Pan, reference method | Hamon | Hargreaves-Samani | Jensen-Haise | Papadakis | Penman-Monteith |
|---|---|---|---|---|---|---|
| | Evaporation estimates, in inches | | | | | |
| April | 1.1 | 0.7 | 1.1 | 1.0 | 0.9 | 1.4 |
| May | 4.2 | 3.2 | 4.3 | 4.4 | 3.5 | 5.5 |
| June | 4.8 | 4.7 | 5.1 | 5.7 | 4.2 | 6.0 |
| July | 5.4 | 5.5 | 5.6 | 6.5 | 5.1 | 6.5 |
| August | 4.4 | 4.3 | 4.7 | 5.4 | 4.7 | 5.4 |
| September | 3.3 | 2.5 | 3.3 | 3.4 | 3.8 | 3.9 |
| October | 1.1 | 0.6 | 0.8 | 0.7 | 1.2 | 1.1 |

**Table 4.** Root-mean-square error by month for lake evaporation methods compared to the pan coefficient method for 1999–2014 for White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.

| Month | Hamon | Hargreaves-Samani | Jensen-Haise | Papadakis | Penman-Monteith |
|---|---|---|---|---|---|
| | Root-mean-square error, in inches | | | | |
| April | 0.52 | 0.27 | 0.29 | 0.30 | 0.45 |
| May | 1.16 | 0.54 | 0.57 | 0.85 | 1.40 |
| June | 0.44 | 0.56 | 0.99 | 0.77 | 1.26 |
| July | 0.56 | 0.52 | 1.20 | 0.65 | 1.17 |
| August | 0.47 | 0.52 | 1.05 | 0.50 | 1.00 |
| September | 0.85 | 0.29 | 0.36 | 0.64 | 0.65 |
| October | 0.59 | 0.39 | 0.46 | 0.33 | 0.25 |

**Table 5.** Annual summary statistics of lake evaporation methods for 1999–2014 for White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.

[RMSE, root-mean-square error; --, no value]

| Statistic | Pan, reference method | Hamon | Hargreaves-Samani | Jensen-Haise | Papadakis | Penman-Monteith |
|---|---|---|---|---|---|---|
| | Evaporation, in inches | | | | | |
| Mean | 24.2 | 21.5 | 25.0 | 27.2 | 23.4 | 29.8 |
| Minimum | 21.8 | 19.9 | 22.2 | 24.9 | 20.6 | 27.3 |
| Maximum | 29.4 | 22.9 | 26.2 | 29.0 | 25.3 | 32.1 |
| RMSE | -- | 3.2 | 1.9 | 3.4 | 2.0 | 5.7 |

Elevation datums for the 96 lakes vary across the study area, and sometimes are unknown or have inaccurate datum conversions provided with the lake-level data. For this reason, all elevations are reported relative to the North American Vertical Datum of 1988 (NAVD 88) (unless otherwise noted) because the elevation differences potentially imposed by the different elevation datums is minimal compared to the range of mean lake-level elevations (771 ft to 1,005 ft) across the study area.

### Spatial Hydrography Characteristics

Hydrography coverage for each of the lakes and their watersheds was used to determine spatial hydrographic characteristics for the lakes. Values for these characteristics were entered under fields in appendix table 1–1. The hydrography coverage was used to calculate a ratio between lake open-water area and lake watershed area (watershed-to-lake area ratio), or the percentage of the lake watershed covered by the lake open-water area (field WStoLKArea, appendix table 1–1). A proximity analysis of the lakes to major rivers was used by determining the straight-line distance between lake centroids and river centerlines of four river systems in the study area: the Mississippi River (field LK_Mississippi, appendix table 1–1), St. Croix River (field LK_StCroix, appendix table 1–1), Rice Creek (field LK_RiceCrk, appendix table 1–1), and Rum River (field LK_Rum, appendix table 1–1) (fig. 1). For each lake, the minimum distance to the nearest of three of the river systems (Mississippi, St. Croix, and Rum Rivers) was determined (field LK_NearestMajRiv, appendix table 1–1).

Lake connectivity to other surface water was classified according to several datasets including the MNDNR Hydrography with Stream Types (Minnesota Department of Natural Resources, 2012), the U.S. Department of Agriculture Farm Services Agency Aerial Photography Field Office (2013), local lake-management reports, and consultations with local lake managers. Lake inflows are classified in the field LK_INLET, and lake outflows are classified in the field LK_OUTLET in appendix table 1–1. A “1” in the field LK_INLET indicates an active inlet was identified in at least one of the data sources, and a “0” in this field indicates an active inlet was not evident in any of the data sources. A “1” in the field LK_OUTLET indicates an active outlet was identified in at least one of the data sources, and a “0” in this field indicates an active outlet was not evident in any of the data sources. For this report, lakes without an active outlet were operationally defined as closed-basin lakes. All references to closed-basin lakes in this report are lakes with a “0” in the field LK_OUTLET.

Lake inlets and outlets were classified according to the following procedure. Perennial inflows and outflows according to the MNDNR Stream Type classification were considered active and assigned a “1” in the respective fields in appendix table 1–1. If an inflow or outflow was classified as intermittent or wetland connector by the MNDNR, then satellite imagery from the U.S. Department of Agriculture Farm Services Agency Aerial Photography Field Office (2013) was used to determine if a clear channel was visible. If a clear channel was visible on the aerial imagery, then it was classified as an active inlet or outlet, and a “1” was entered in the respective fields in appendix table 1–1. If no channels were visible, further investigations through local lake-management reports or consultations with local lake managers were done to determine if lakes had inlets or outlets. Lakes with reports that document underground outlets (such as culverts) and lake controls (that is, stop-log structures) were considered to have outlets and were noted in appendix table 1–2. If an inflow or outflow channel was not in the MNDNR Stream Type dataset (Minnesota Department of Natural Resources, 2012), or in the aerial photography, and no evidence from local sources supported the existence of an inflow or outflow, a “0” was entered in the respective fields in appendix table 1–1.

Classification of the lake connectivity data for some lakes proved to be difficult because the status of lake inlets and outlets varied for the lakes during the short-term analysis (2002–10); for example, White Bear Lake is classified in this report as a closed-basin lake even though it does have an outlet and is not classified as landlocked by the MNDNR. During 2002–10, the annual mean water level was above the outlet elevation for 2002 and 2003 but never above the outlet elevation for the remainder of 2004 through 2010; therefore, for most of the study period, lake levels were not responding as if in a flow-through condition but rather as a lake with no outlet. The definition of closed-basin in this report only refers to the period of interest and may differ from classifications made by the MNDNR or other organizations.

The presence of lake inlet- or outlet-control structures was determined through information from reports provided by and written or oral communication with northeast Twin Cities Metropolitan Area watershed districts, cities, and counties. The presence or absence of lake inlet- and outlet-control structures is presented in fields LK_InletControl and LK_OutletControl, respectively, in appendix tables 1–1 and 1–2. The value “0” in field LK_InletControl indicates there is no control structure present, the value “1” indicates there is a control structure, and the value “2” indicates an unknown presence of a control structure. In the field LK_OutletControl, the value “0” indicates there is no control structure present, and the value “1” indicates there is a control structure. Two lakes in the statistical analysis (Round [2–89 P] and Unnamed [82–303 W], appendix table 1–1) that have an unknown presence of an inlet-control structure (assigned a value of “2” in appendix table 1–1) were assigned a value of “0” (no presence) for the statistical analysis. Appendix table 1–2 provides descriptions of the inlet- and outlet-control structures. Common control structures include culverts and weirs. Sewer outfalls were not considered inlet-control structures in this study.

Groundwater withdrawals from 2002 to 2010 were incorporated into the statistical analysis as a spatial characteristic. The MNDNR monthly groundwater withdrawal data (Sean Hunt, Minnesota Department of Natural Resources, written commun., August 26, 2015) were aggregated by well number and year, and plotted in a GIS using the Universal Transverse Mercator coordinates of each well that were included in the data from the MNDNR. Total annual groundwater withdrawals were determined for two spatial extents: each lake watershed and a 2-mi buffer (Berg, 2014) around the open-water area of each lake. The total annual groundwater withdrawals for each geographic extent were divided by either the lake watershed or 2-mi buffer area to produce a volume of pumping per area per year (in cubic inches per square inch per year [$in^3/in^2 \cdot yr$ or inches per year [in/yr]). The mean volume of pumping per area per year within each of the lake watersheds is presented in field WS_AvgGWPump_m, and the mean pumping depth within a 2-mi buffer of lakes is presented in field BUF_AvgGWPump_m of appendix table 1–1.

Mean annual precipitation values from 2002 to 2010 (field LK_prcp_avg02.10, appendix table 1–1) were calculated from daily precipitation data retrieved for the Daymet grid cell (Thornton and others, 2014) at the centroid of each lake. Mean annual lake evaporation for each lake during the study period (field LK_evap_avg02.10, appendix table 1–1) was determined from annual lake evaporation estimates. These estimates were computed using Daymet temperature data and the Hargreaves-Samani method (Hargreaves and Samani, 1985; Samani, 2000) as described previously in the “Evaporation Estimates” section. The difference between mean annual precipitation depth and mean annual lake evaporation depth for each lake for the study period was calculated and is presented in field LK_pminuse_avg02.10 of appendix table 1–1.

Mean potential groundwater recharge from 2002 to 2010 was computed from annual potential recharge estimates produced from the Minnesota Soil-Water-Balance (SWB) model, which calculated recharge on a 1-km grid scale (Smith and Westenbroek, 2015). The model used land-cover and soil hydrologic characteristics along with daily precipitation data to estimate potential recharge. Annual potential groundwater recharge within the watershed of each lake was estimated by summing the annual potential groundwater recharge for all the SWB model grid cells within the watershed of each lake for each year from 2002 to 2010. The mean of the annual sums was computed and divided by the number of grid cells in each watershed to determine a mean value for the study period. These values are presented in field WS_mean_RCH_02.10 in appendix table 1–1.

### Geology and Soil Characteristics

Several datasets were used to calculate geologic characteristics underlying each lake and its watershed. Soil characteristics for the watershed of each lake were calculated using the Soil Survey Geographic (SSURGO) database (Soil Survey Staff, U.S. Department of Agriculture, and Natural Resources Conservation Service, 2016). These soil characteristics included mean hydraulic conductivity of saturated soils (field WS_K_Sat); mean percent organic matter (field WS_org_matter); and sand (field WS_pct_sand), silt (field WS_pct_silt), and clay (field WS_pct_clay) content of soils for each watershed. These data are presented in appendix table 1–1. The percentage of lake watershed area covered by hydrologic soil groups A (field WS_Soil_a), B (field WS_Soil_b), and C (field WS_Soil_c) also were determined and are presented in appendix table 1–1. Soils in hydrologic soil group D were rarely present in the study area and, therefore, were not included in the analysis.

The dominant surficial geology bordering each lake was derived from a Twin Cities Metropolitan Area surficial geology GIS dataset published by the Minnesota Geological Survey (MGS) (Meyer, 2007). The map units in this dataset were generalized to new categories as listed in appendix table 1–3. Map units from the Cromwell Formation were classified as Superior Lobe deposits. Map units from the New Ulm and New Brighton Formations and eolian sands were classified as Des Moines Lobe deposits. Lakes bordered predominantly by Superior Lobe deposits were coded as “1”, and those bordered predominantly by Des Moines Lobe deposits were coded as “0” in the field LK_GeoUnit in appendix table 1–1. No lakes were bordered by fluvial or pre-Wisconsin lobe deposits. Map units with till, clay, silt, or lake clay/silt in the map unit description were classified as till. Map units with till and sand in their description were classified as till. Map units with sand, ice-contact deposit, or outwash in the map unit description were classified as sand. Lakes bordered predominantly by sands were coded as “1”, and those bordered predominantly by tills were coded as “0” in the field LK_DepFunc in appendix table 1–1. The assignment of lakes to the generalized geologic units and textural classes was done visually in GIS.

The dominant bedrock unit underlying each lake (field LK_DOMBEDROCK, appendix table 1–1) was identified using a detailed bedrock GIS coverage of the Twin Cities Metropolitan Area (fig. 1) (Mossler, 2013). The vector dataset of this coverage was converted to a raster grid with a 328-ft (100-meter [m]) cell size. The raster grid was analyzed using the Feature Statistics to Table tool in the National Water-Quality Assessment (NAWQA) Area-Characterization Toolbox (Price and others, 2010) to determine the dominant upper bedrock unit underlying the open-water area of each lake.

The mean depth to bedrock underlying each lake (field LK_DPTHBDKAVM, appendix table 1–1) was determined using depth-to-bedrock GIS coverages from the MGS (Meyer and Swanson, 1992; Bauer, 2016). Depth-to-bedrock coverages also were used to determine whether or not the lake lies above a buried bedrock valley (field LK_BEDROCKVAL, appendix table 1–1). Buried bedrock valleys were identified from the depth-to-bedrock coverage as elongated channels with steep side slopes in the bedrock surface. The lake coverage was overlaid on the depth-to-bedrock coverage to determine whether or not each lake fell over a buried bedrock valley. Areas of large bedrock depths that did not form a channel on the depth-to-bedrock coverage were interpreted not to be buried bedrock valleys, and lakes overlying these areas were coded as “0” in field LK_BEDROCKVAL; whereas lakes overlying areas of large bedrock depths that did form a channel were coded as “1” in field LK_BEDROCKVAL in appendix table 1–1.

### Land Cover and Slope Characteristics

The 2006 National Land Cover Database (NLCD; Fry and others, 2011) was used to tabulate the percent area of land-use categories in the watershed of each lake. The land-cover categories were reclassified and grouped into the following six categories for the statistical analyses and included in appendix table 1–1: (1) NLCD class 11 to represent open water (field WS_PCT_11); (2) summation of NLCD classes 21, 22, 23, and 24 to represent developed space (field WS_SUM_DEV); (3) summation of NLCD classes 90 and 95 to represent wetlands (field WS_SUM_WETLND); (4) summation of NLCD classes 81 and 82 to represent agricultural lands (field WS_SUM_AGRICU); (5) summation of NLCD classes 41, 42, and 43 to represent forests (field WS_SUM_FOREST); and (6) summation of NLCD classes 52 and 71 to represent other vegetated lands (field WS_SUM_SHRUBGRASS). This reclassified raster dataset was used with the Tabulate Features to Percent tool in the NAWQA Area-Characterization Toolbox (Price and others, 2010) to calculate the percent area of each of the six land-cover categories in the watershed of each lake. A percent slope surface in the watershed of each lake was determined from a 98-ft (30-m) digital elevation model in the 3D Elevation Program (U.S. Geological Survey, 2015) using the ESRI ArcMap version 10.3 slope tool (Esri, 2015). The mean percent slope in each watershed was then calculated from the percent slope surface (field WS_Slope_pct, appendix table 1–1).

## Statistical Methodology

The statistical analyses were completed to help understand changes in lake levels across the northeast Twin Cities Metropolitan Area (fig. 1). Two statistical approaches were used to explore these relations: nonparametric correlations (Spearman’s rho [ρ]) and analysis of variance (ANOVA). Spatial patterns in lake-level changes for 96 lakes during a short-term period (2002–10) were evaluated and compared to climatic, landscape, and geologic characteristics of the lakes and their watersheds. Relations between precipitation and lake levels in 14 lakes during a long-term period (1925–2014) were evaluated.

### Short-Term Analyses

The objective of the short-term statistical analysis was to determine if certain climatic, landscape, or geologic characteristics of the region could explain the variability in lake-level changes during the 2002–10 study period; lake-level changes were based on the change in annual mean lake level from 2002 to 2010 (field LK_chnge02.10, appendix table 1–1). Lakes are defined in this analysis as any body of water with water-level records available through the MNDNR Lake Finder Web page (Minnesota Department of Natural Resources, 2015). All statistical analyses were completed in R version 3.2.3 (R Core Team, 2015).

A series of statistical analyses were used to explore which variables might explain the variability in lake-level change. The strength of association between LK_chnge02.10 and each of the continuous variables in appendix table 1–1 was examined using Spearman's ρ, a nonparametric rank correlation coefficient (Helsel and Hirsch, 2002). The correlations were computed with the smwrStats package, version 0.7.2 (Lorenz, 2015). Correlations were done using data from all 96 lakes and then done separately for the closed-basin (45 lakes) and flow-through lakes (51 lakes). The LK_chnge02.10 variable also was compared between different categorical variables in appendix table 1–1. An ANOVA was completed for each categorical variable to determine if the groupings explained a substantial amount of the variance in the lake-level change.

This analysis serves to examine the different multiyear hydrologic responses that happened throughout the study area during 2002–10. Statistical comparisons and models were used to make inferences and generate hypotheses about what physical processes might be affecting the system. Conclusions drawn from the various statistical methods are correlative and descriptive in nature and not causal but shed some insight on variability in lake-level change.

## Long-Term Analyses

The objective of the long-term analysis was to evaluate temporal relations between precipitation and lake levels from 1925 to 2014 in 14 lakes (fig. 5), complementing the short-term analyses that evaluated lake-level changes across the study area. Three separate evaluations were made with the long-term data: (1) a graphical evaluation of lake-level anomalies from 1925 to 2014, (2) an evaluation of precipitation and lake-level variability during two 35-year periods, 1943–78 and 1979–2014, and (3) an evaluation of lagged correlations between annual lake-level changes and precipitation. This analysis assumes that precipitation did not vary spatially among the 14 lakes during 1925–2014. All the lakes were within 12 mi of each other, and a single precipitation dataset was used for all the lakes. These evaluations were used to assess general relations between precipitation and lake levels in the northeast Twin Cities Metropolitan Area. The lake-level records for the 14 lakes were affected by a variety of anthropogenic factors, such as changing outlet elevations, storm-water routing, and augmentation with water from other sources, from 1925 to 2014 that were not directly incorporated in the analysis.

For the first evaluation, annual lake-level anomalies for each lake were computed to graphically compare lake levels through time on the same scale. Lake-level anomalies were calculated by subtracting the mean annual level (overall mean) for 1925–2014 from each annual mean level for each lake.

For the second comparison, variability in lake levels and precipitation was characterized using the coefficient of variation (CV). The CV is a measure of spread that describes the amount of variability relative to the mean. The CV was used for this analysis because it is unitless and can be compared across precipitation data and lake levels. A small CV indicates less dispersion relative to the mean, whereas a large CV indicates more dispersion. The CV of the annual mean lake levels and annual precipitation totals were calculated in R Stats package (version 3.2.3; R Core Team, 2015) according to the following equation:

$$CV_i = \frac{standard\ deviation\left(x_i\right)}{mean\left(x_i\right)} \times 100 \tag{1}$$

where

$CV$ is the coefficient of variation,
$i$ is the period of interest, and
$x$ is the annual lake-level or precipitation values.

The change in variability (as indicated by the CV) in annual precipitation totals and mean annual lake levels were determined for two 35-year periods: 1943–78 and 1979–2014. Three of the closed-basin lakes (Turtle, White Bear, and Snail Lakes, fig. 1) previously have been augmented with water from other sources throughout their history: White Bear Lake from the 1920s through 1978; Turtle Lake from 1923 through 1989; and Snail Lake up through the 1980s, 1993–2007, and 2009–present (2016). The histories for these three lakes indicate that augmentation was more commonly practiced before 1978, so the data were split into the two 35-year periods for comparison. In the analysis, it was assumed that the lake-level time series for each lake was stationary (having a constant mean through time) within each 35-year period. As a check on the assumption of stationarity, the data were detrended and the variance was calculated for precipitation and each lake for the two time periods. The lake-level data were detrended by a first-difference approach (as described in equation 2) and the precipitation data were detrended by removing the linear trend in the time series (Brockwell and Davis, 2002).

For the third evaluation of long-term data, lagged correlations between lake-level changes and precipitation were compared. Lagged correlations refer to the correlation between two time series shifted in time relative to one another. Correlation coefficients were calculated between the annual mean lake-level changes for each lake and the annual precipitation total from the year of the observed lake-level change (lag of 0), 1 year before (lag of 1 year), and up to 5 years before the year of observed lake-level change (lag of 5 years).

Because lake levels are highly autocorrelated through time, the first difference or change in mean lake level from year to year was calculated for each lake according to the following equation:

$$\Delta LL = L_t - L_{t-1} \tag{2}$$

where

$\Delta LL$ is the change in lake level, in feet;
$L_t$ is the mean annual lake level, in feet, in year $t$; and
$L_{t-1}$ is the mean annual lake level, in feet, in year $t$-1.

For each lake, the correlation between the annual lake-level changes (in feet) and total annual precipitation (in inches) at different time lags were computed using the ccf() function in the R Stats package version 3.2.3 (R Core Team, 2015). The sample correlations were used to identify lags of the total annual precipitation that were correlated with lake-level change.

## Water-Quality Sample Collection, Handling, Analysis, and Quality Control

Physical and chemical water-quality characteristics were measured in precipitation, lake-water, piezometer-water, and well-water samples to characterize and identify potential groundwater and lake-water exchanges in White Bear Lake and other northeast Twin Cities Metropolitan Area lakes (fig. 1). All precipitation, lake-water, and well-water samples were collected in 2014 following USGS protocols outlined in the USGS National Field Manual for the Collection of Water-Quality Data (U.S. Geological Survey, variously dated). All water-quality results were entered into the USGS National Water Information System (NWIS) database (U.S. Geological Survey, 2016a).

### Sample Collection

A series of environmental and replicate water samples were collected from precipitation stations, lakes, piezometers, and wells for analysis of oxygen-18/oxygen-16 ratios and hydrogen-2/hydrogen-1 (deuterium/protium) ratios to assess groundwater and surface-water exchange near northeast Twin Cities Metropolitan Area lakes and surface-water contribution to wells (fig. 1). These environmental and replicate samples consisted of (1) 10 bulk precipitation samples collected between May 27 and October 1, 2014; (2) 24 lake-water samples collected from four northeast Twin Cities Metropolitan Area lakes between May 21 and September 25, 2014; (3) 6 lake-water samples collected on September 12, 2014, at a single site on White Bear Lake at five different depths; (4) 7 piezometer-water samples collected from piezometers screened in lake and glacial sediments in White Bear Lake between September 12 and 17, 2016; and (5) 46 well-water samples collected from wells open to the Prairie du Chien, Jordan, or both aquifers between October 21 and 29, 2014. Unfiltered, unpreserved water samples were collected or transferred to a 60-milliliter (mL) clear glass bottle. The glass bottle was immediately capped with a polyseal cap, sealed by wrapping in electrical tape, and shipped to the USGS Reston Stable Isotope Laboratory in Reston, Virginia, for analyses.

Bulk precipitation samples were collected at a weather station on the east shore of White Bear Lake (USGS site 450334092574201) (fig. 2) to determine a local meteoric water line to be used in the analysis of oxygen-18/oxygen-16 ratios and deuterium/protium ratios for lake- and well-water samples. The meteoric water line is a linear relation between the deuterium/protium ratios and oxygen-18/oxygen-16 ratios for precipitation samples collected at a single or multiple locations (Gibson and others, 1993). The precipitation samples were collected in 4-liter (L) amber glass bottles through plastic funnels during storms on May 27, May 28, June 1–2, June 18, June 19, July 25, August 18–19, and September 30–October 1, 2014. Total precipitation amounts recorded at the nearest high-density observation site during these events were 0.46, 0.70, 2.23, 0.96, 1.79, 0.37 , 0.73, and 0.29 in., respectively (Minnesota Climatology Working Group, 2016c). The amber glass bottles used to collect the precipitation samples were deployed within 24 hours of the start of anticipated precipitation and were collected within 24 hours after the precipitation ended. Each precipitation sample was immediately poured from the amber glass collection bottle into a 60-mL clear glass bottle.

Oxygen-18/oxygen-16 ratios and deuterium/protium ratios for lake-water samples collected from four northeast Twin Cities Metropolitan Area lakes were used to assess evaporation rates and groundwater and surface-water exchanges on the lakes. Grab samples were collected between May and September 2014 from Bald Eagle Lake (seven samples), Big Marine Lake (six samples), Turtle Lake (six samples), and White Bear Lake (five samples). The lakes were selected based on their large area, their distribution in the northeast Twin Cities Metropolitan Area, the availability of water-level data, the occurrence of 2003–2010 declining water levels, and their accessibility. Grab samples were collected in 60-mL clear glass bottles from the littoral zone of each lake, either by wading into the water or from an existing dock.

Five lake-water samples were collected on September 12, 2014, from White Bear Lake to assess the variability of oxygen-18/oxygen-16 ratios and deuterium/protium ratios at the water surface and above and below the thermocline of the lake. The samples were collected from White Bear Lake near site P4 (fig. 2; White Bear Lake north of Birchwood near P4 site in table 6), which is one of the deeper parts of the lake. These five grab samples were collected in White Bear Lake from a boat at water depths ranging from 0.5 to 69 ft below the water surface (table 6). A 90-ft section of C–Flex® tubing connected to a low-flow peristaltic pump was used to collect the water samples. The tubing was strapped to the outside of a 0.25-in. polyvinyl chloride (PVC) pipe and lowered into the lake to the different sampling depths. Pumping rate during sampling was 0.06 gallon per minute (gal/min) and kept at a low rate to minimize mixing of the water column during sampling.

Piezometer-water samples were collected from 6 of 11 piezometers (P1–8.5, P1–16.5, P2–30, P3–9, P3–13.5, and P4–8; see “Deep-Water Piezometer and Lake Water-Level Differences at White Bear Lake” section) in the southeast bay of White Bear Lake between September 12 and 17, 2016, to assess oxygen-18/oxygen-16 ratios for water in deep-water lake and glacial sediments. One to two piezometers were sampled from each of the four sites (P1–P4; fig. 2). A 90-ft section of C–Flex® tubing connected to a peristaltic pump was used to collect the water samples. The tubing was lowered

**Table 6.** Water-quality data for precipitation, lake water, and groundwater collected in the northeast Twin Cities Metropolitan Area, Minnesota, May through October 2014.

[ft, foot; mmHg, millimeter of mercury; µS/cm at 25 ºC, microsiemens per centimeter at 25 degrees Celsius; ºC, degrees Celsius; mg/L, milligram per liter; per mil, part per thousand; --, no data; Wash. Co., Washington County]

| Water-quality sampling site | Site number | Date | Time (24 hour) | Sample type | Sampling lake-water depth (ft) | Barometric pressure, on site (mmHg) | Specific conductance, on site (µS/cm at 25ºC) | pH, on site (standard units) | Water temperature (ºC) | Dissolved oxygen, on site (mg/L) | Oxygen-18/ oxygen-16 ratio, unfiltered water (per mil) | Deuterium/ protium ratio, unfiltered water (per mil) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Precipitation | 450334092574201 | 5/27/2014 | 11:30 | Precipitation | -- | -- | -- | -- | -- | -- | -6.67 | -40.3 |
| Precipitation | 450334092574201 | 5/30/2014 | 06:50 | Precipitation | -- | -- | -- | -- | -- | -- | -7.12 | -43.0 |
| Precipitation | 450334092574201 | 6/2/2014 | 08:00 | Precipitation | -- | -- | -- | -- | -- | -- | -6.45 | -40.6 |
| Precipitation | 450334092574201 | 6/18/2014 | 08:40 | Precipitation | -- | -- | -- | -- | -- | -- | -3.72 | -17.5 |
| Precipitation | 450334092574201 | 6/19/2014 | 15:10 | Precipitation | -- | -- | -- | -- | -- | -- | -3.81 | -19.2 |
| Precipitation | 450334092574201 | 7/25/2014 | 07:20 | Precipitation | -- | -- | -- | -- | -- | -- | -7.64 | -51.1 |
| Precipitation | 450334092574201 | 8/19/2014 | 07:00 | Precipitation | -- | -- | -- | -- | -- | -- | -8.73 | -60.9 |
| Precipitation | 450334092574201 | 10/1/2014 | 16:40 | Precipitation | -- | -- | -- | -- | -- | -- | -13.74 | -97.3 |
| Bald Eagle Lake | 450650093000801 | 5/21/2014 | 11:50 | Lake | 0.5 | -- | -- | -- | -- | -- | -4.61 | -40.7 |
| Bald Eagle Lake | 450650093000801 | 6/25/2014 | 16:00 | Lake | 0.5 | -- | -- | -- | -- | -- | -4.44 | -39.4 |
| Bald Eagle Lake | 450650093000801 | 7/27/2014 | 06:10 | Lake | 0.5 | -- | -- | -- | -- | -- | -4.02 | -37.1 |
| Bald Eagle Lake | 450650093000801 | 8/19/2014 | 14:20 | Lake | 0.5 | -- | -- | -- | -- | -- | -3.72 | -36.2 |
| Bald Eagle Lake | 450650093000801 | 9/25/2014 | 16:20 | Lake | 0.5 | -- | -- | -- | -- | -- | -3.54 | -35.2 |
| Big Marine Lake | 451244092524301 | 5/21/2014 | 17:10 | Lake | 0.5 | -- | -- | -- | -- | -- | -3.21 | -32.8 |
| Big Marine Lake | 451244092524301 | 6/25/2014 | 16:30 | Lake | 0.5 | -- | -- | -- | -- | -- | -3.18 | -31.9 |
| Big Marine Lake | 451244092524301 | 7/27/2014 | 06:40 | Lake | 0.5 | -- | -- | -- | -- | -- | -3.00 | -32.3 |
| Big Marine Lake | 451244092524301 | 8/19/2014 | 14:50 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.70 | -31.3 |
| Big Marine Lake | 451244092524301 | 9/25/2014 | 15:40 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.68 | -30.8 |
| Turtle Lake | 450534093075101 | 5/21/2014 | 12:30 | Lake | 0.5 | -- | -- | -- | -- | -- | -3.21 | -33.1 |
| Turtle Lake | 450534093075101 | 6/25/2014 | 15:10 | Lake | 0.5 | -- | -- | -- | -- | -- | -3.08 | -31.9 |
| Turtle Lake | 450534093075101 | 7/27/2014 | 05:30 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.79 | -31.6 |
| Turtle Lake | 450534093075101 | 8/19/2014 | 13:30 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.78 | -32.1 |
| Turtle Lake | 450534093075101 | 9/25/2014 | 17:30 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.66 | -30.9 |
| White Bear Lake - West Bay | 450432093005101 | 5/21/2014 | 11:30 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.78 | -28.7 |
| White Bear Lake - West Bay | 450432093005101 | 6/25/2014 | 15:40 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.79 | -29.9 |
| White Bear Lake - West Bay | 450432093005101 | 7/27/2014 | 05:50 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.61 | -30.0 |
| White Bear Lake - West Bay | 450432093005101 | 8/19/2014 | 14:00 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.42 | -29.3 |
| White Bear Lake - West Bay | 450432093005101 | 9/25/2014 | 16:55 | Lake | 0.5 | -- | -- | -- | -- | -- | -2.41 | -28.6 |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:45 | Lake | 0.5 | -- | 301 | 8.4 | 18.9 | 8.4 | -2.51 | -28.7 |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:46 | Lake | 3.3 | -- | 301 | 8.4 | 18.9 | 8.4 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:47 | Lake | 6.6 | -- | 301 | 8.3 | 18.9 | 8.3 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:48 | Lake | 9.8 | -- | 301 | 8.2 | 18.9 | 8.3 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:49 | Lake | 13 | -- | 301 | 8.2 | 18.9 | 8.3 | -- | -- |

**Table 6.** Water-quality data for precipitation, lake water, and groundwater collected in the northeast Twin Cities Metropolitan Area, Minnesota, May through October 2014.—Continued

[ft, foot; mmHg, millimeter of mercury; µS/cm at 25 °C, microsiemens per centimeter at 25 degrees Celsius; °C, degrees Celsius; mg/L, milligram per liter; per mil, part per thousand; --, no data; Wash. Co., Washington County]

| Water-quality sampling site | Site number | Date | Time (24 hour) | Sample type | Sampling lake-water depth (ft) | Barometric pressure, on site (mmHg) | Specific conductance, on site (µS/cm at 25°C) | pH, on site (standard units) | Water temperature (°C) | Dissolved oxygen, on site (mg/L) | Oxygen-18/ oxygen-16 ratio, unfiltered water (per mil) | Deuterium/ protium ratio, unfiltered water (per mil) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:50 | Lake | 16 | -- | 301 | 8.1 | 18.8 | 8.2 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:51 | Lake | 20 | -- | 301 | 8.1 | 18.9 | 8.2 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:52 | Lake | 23 | -- | 301 | 8.1 | 18.8 | 8.0 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:53 | Lake | 26 | -- | 302 | 8.0 | 18.8 | 7.9 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:54 | Lake | 33 | -- | 302 | 8.0 | 18.6 | 8.1 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:56 | Lake | 36 | -- | 302 | 8.0 | 18.4 | 7.8 | -2.49 | -29.5 |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:57 | Lake | 39 | -- | 319 | 7.3 | 16.0 | 5.6 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 10:55 | Lake | 41 | -- | -- | -- | -- | -- | -2.70 | -29.6 |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:05 | Lake | 30 | -- | 301 | 8.0 | 18.7 | 8.1 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:10 | Lake | 43 | -- | 327 | 7.1 | 13.0 | 2.5 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:15 | Lake | 46 | -- | 323 | 6.9 | 11.8 | 2.1 | -2.74 | -30.9 |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:18 | Lake | 49 | -- | 324 | 6.9 | 11.3 | 1.9 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:21 | Lake | 52 | -- | 324 | 6.9 | 11.1 | 1.9 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:24 | Lake | 56 | -- | 325 | 6.9 | 10.9 | 1.7 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:27 | Lake | 59 | -- | 326 | 6.9 | 10.7 | 1.6 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:30 | Lake | 62 | -- | 327 | 6.9 | 10.6 | 1.7 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:35 | Lake | 66 | -- | 328 | 6.9 | 10.6 | 1.5 | -- | -- |
| White Bear Lake north of Birchwood near P4 | 450401092581301 | 9/12/2014 | 11:40 | Lake | 69 | -- | 329 | 6.9 | 10.6 | 1.6 | -2.67 | -30.9 |
| P4–8 White Bear Lake piezometer, Wash. Co. | 450401092581302 | 9/12/2014 | 13:00 | Piezometer | -- | -- | -- | -- | -- | -- | -2.38 | -28.6 |
| P1–8.5 White Bear Lake piezometer, Wash. Co. | 450417092580601 | 9/17/2014 | 14:10 | Piezometer | -- | -- | -- | -- | -- | -- | -2.62 | -29.2 |
| P1–16.5 White Bear Lake piezometer, Wash. Co. | 450417092580602 | 9/17/2014 | 15:00 | Piezometer | -- | -- | -- | -- | -- | -- | -2.46 | -28.6 |
| P2–30 White Bear Lake piezometer, Wash. Co. | 450426092583403 | 9/17/2014 | 09:35 | Piezometer | -- | -- | -- | -- | -- | -- | -2.28 | -26.5 |
| P3–9 White Bear Lake piezometer, Wash. Co. | 450353092580601 | 9/17/2014 | 17:15 | Piezometer | -- | -- | -- | -- | -- | -- | -2.39 | -27.8 |
| P3–13.5 White Bear Lake piezometer, Wash. Co. | 450353092580602 | 9/17/2014 | 16:35 | Piezometer | -- | -- | -- | -- | -- | -- | -2.39 | -27.4 |

into each piezometer below the water level in the piezometer, and at least three piezometer volumes of water (greater than 1.3 gallons [gal; 5 L]) were removed from the piezometer before sampling. The isotope ratios in these water samples were compared to oxygen-18/oxygen-16 ratios and deuterium/protium ratios from the lake-water samples to assess sediment-water and lake-water exchange in the lake at the sites.

Water samples were collected from municipal, commercial, irrigation, and domestic wells open to the Prairie du Chien, Jordan, or both aquifers between October 21 and 29, 2014, to estimate surface-water contributions to the wells and determine the age of recharge water for the well water. The samples were analyzed for field water-quality properties, oxygen-18/oxygen-16 ratios, deuterium/protium ratios, dissolved gases (nitrogen [$N_2$], argon [Ar], carbon dioxide, methane, and oxygen), chlorofluorocarbons (CFCs), and sulfur hexafluoride ($SF_6$) to assess groundwater and surface-water exchanges between lakes and wetlands near the wells and age date the well waters. All the water samples were collected following guidelines in the USGS National Field Manual (U.S. Geological Survey, variously dated), the USGS Reston Stable Isotope Laboratory (U.S. Geological Survey, 2016b), and the USGS Reston Groundwater Dating Laboratory in Reston, Va. (U.S. Geological Survey, 2016c, d, e). A total of 40 wells were sampled, and the wells were selected based on the aquifer(s) to which the well was open, the distribution of the well relative to other sampled wells in the northeast Twin Cities Metropolitan Area, the open interval length of the well, and the location of the well relative to lakes and wetlands of interest. Submersible pumps already in the wells were used to collect the water samples. At least three well volumes of water were pumped from the wells, and field water-quality properties had stabilized before collecting the water samples for laboratory analyses. The well-water samples were collected using Teflon tubing and a stainless-steel fitting connected to a faucet on the well. Unfiltered, unpreserved samples for determining oxygen-18/oxygen-16 ratios and deuterium/protium ratios were collected directly from the Teflon tubing into 60-mL clear glass bottles.

Unfiltered, unpreserved water samples were collected for dissolved gas analyses (U.S. Geological Survey, 2016c). A 5-ft section of C–Flex® tubing was connected to the Teflon tubing to collect water samples for dissolved gas analyses. The water samples were collected in 150-mL glass bottles capped with a rubber stopper. A 2-L plastic beaker was filled with well water from the C–Flex® tubing before sampling. A 150-mL sample bottle was placed in the beaker with the C–Flex® tubing placed in the bottom of the sample bottle. Well water was pumped into the bottle until the sample bottle was submerged with water in the beaker. Once the bottle was completely submerged, the bottle was capped with a rubber stopper with an inserted needle. The needle was inserted into the rubber stopper to allow trapped gases in the water to be removed from the sample bottle. While submerged, the bottle was shaken until all the trapped gases were removed from the sample water in the bottle. Once all gases were apparently removed from the bottle, the needle was removed from the stopper while the sample bottle was submerged. The cap was sealed with electrical tape to prevent air from entering the bottle between the cap and bottle. The water samples were chilled immediately after collection and shipped within 48 hours to the Reston Groundwater Dating Laboratory for analysis.

Unfiltered, unpreserved water samples were collected for CFC analyses (U.S. Geological Survey, 2016d). The CFCs are stable, synthetic, halogenated alkanes that were developed in the early 1930s as alternatives to ammonia and sulfur dioxide in refrigeration (Busenberg and Plummer, 1992; U.S. Geological Survey, 2016f). Concentrations of these compounds can be used to determine the age of water less than 50 years old. Water samples for CFC analyses were collected from the wells following procedures similar to those previously described for collecting the water samples for dissolved gas analyses. The tubing was inserted into the bottom of 125-mL boston round, clear glass bottles that were placed in a 2-L plastic beaker. The bottle and beaker were filled with sample water, which allowed at least 2 L of water to overflow from the beaker. The bottle was capped under water in the beaker to ensure no air bubbles were trapped in the bottle or cap, and electric tape was placed around the cap to prevent air from entering the water sample between the cap and bottle. The water samples were placed in coolers and shipped within 48 hours to the Reston Groundwater Dating Laboratory for analysis.

Unfiltered, unpreserved water samples were collected for $SF_6$ analysis (U.S. Geological Survey, 2016e). The $SF_6$ gas is an atmospheric gas that is primarily anthropogenic in origin but also exists naturally in some minerals and igneous rocks (Busenberg and Plummer, 2000; U.S. Geological Survey, 2016g). Production of anthropogenic $SF_6$ began in the 1960s, and $SF_6$ concentrations in waters can be used to determine ages for waters that are younger than 1990 (U.S. Geological Survey, 2016g). Water samples were collected directly from the Teflon tubing connected to the well. The tubing was inserted into a 1-L, plastic-coated, amber glass bottle, and sample water was allowed to fill the bottle. At least 3 L of sample water was allowed to overflow from the bottle before removing the tubing from the bottle while it was overflowing. A polyseal, cone-lined cap was used to tightly cap the bottle, which allowed no air headspace in the bottle. Electrical tape was applied to the cap to prevent air from entering the bottle. The samples were stored in coolers and shipped within 48 hours to the Reston Groundwater Dating Laboratory for analysis (U.S. Geological Survey, 2016e).

## Sample Handling and Analysis

Field water-quality properties (water temperature, dissolved oxygen concentration, pH, and specific conductance) were measured for well-water, piezometer-water, and some lake-water samples with a YSI 6820 water-quality multiprobe meter before the collection of samples for laboratory analyses. The specific conductance, pH, and dissolved oxygen probes on the multiprobe meter were calibrated on each of the sampling dates before sampling.

Water samples for determining oxygen-18/oxygen-16 and deuterium/protium ratios were analyzed following methods outlined by the USGS Reston Stable Isotope Laboratory (U.S. Geological Survey, 2016h) and described by Révész and Coplen (2008). These isotopes are not measured directly because their concentrations are low but are expressed as values relative to the Vienna Standard Mean Ocean Water-Standard Light Antarctic Precipitation (SLAP) isotope scales (Joint Committee for Guides in Metrology, 2008). Results are reported in units of parts per thousand (per mil). The zero of the hydrogen-2 and oxygen-18 scales are normalized to the SLAP values of -428 and -55.5 per mil, respectively (Gonfiantini, 1978). The oxygen-18/oxygen-16 and deuterium/protium ratios present in water were used to distinguish sources of water when the degree of isotopic fractionation of the water is dissimilar for different sources of water. Hydrologic studies have used isotopic ratios in waters to identify groundwater discharge to lakes and sources of waters to wells (Dinçer, 1968; Sacks, 2002; Jones, 2006; Jones and others, 2013; Rosenberry and LaBaugh, 2008). These isotopic ratios are useful because they are part of the water and not solutes dissolved in the water. If the isotopic compositions of different sources of water are distinct, simple mixing models can be used to identify sources of water (Jones and others, 2013).

Well-water samples collected for dissolved gas concentrations were measured by gas chromatography and flame ionization detection following procedures outlined by the Reston Groundwater Dating Laboratory (U.S. Geological Survey, 2016i). Recharge temperatures and excess air concentrations were estimated for each water sample from the dissolved $N_2$ and Ar concentrations (U.S. Geological Survey, 2016j). Recharge temperatures for the water samples are the temperatures of waters when they reached the water table. These recharge temperatures are needed to establish CFC and $SF_6$ concentrations at the time of recharge, which are needed to determine an age date for the water. As recharge water enters an aquifer, the concentration of gases, including CFCs and $SF_6$, in the groundwater, is in equilibrium with the atmosphere at the temperature of recharge (Hunt and others, 2016); however, the groundwater also may have extra gases, including $N_2$ and Ar, from excess atmospheric gas trapped in pores after rapid rises in the water table (excess air). The extra gas can dissolve into the groundwater, resulting with $N_2$ and Ar in air being different from the equilibrium solubility ratio of $N_2$ to Ar in water (U.S. Geological Survey, 2016j). The excess air concentrations need to be known to correct the CFC and $SF_6$ concentrations for the presence of the excess air. Methane gas concentrations provide an indication of reducing conditions somewhere along the flow path of the well water (U.S. Geological Survey, 2016j).

Well-water samples for CFC analysis were analyzed for three different CFCs using gas chromatography following analytical procedures outlined by the Reston Groundwater Dating Laboratory (Busenberg and Plummer, 1992; U.S. Geological Survey, 2016k). The three CFCs were trichlorofluoromethane ($CFCl_3$ or CFC–11), dichlorodifluoromethane ($CF_2Cl_2$ or CFC–12), and trichlorotrifluoroethane ($C_2F_3Cl_3$ or CFC–113). Concentrations for the three CFCs in the water samples were determined using purge and trap gas chromatographic techniques with an electron-capture detector (Busenberg and Plummer, 1992; U.S. Geological Survey, 2016k). These concentrations can be used to determine age dates because (1) the mixing ratios of these compounds in the air are known or have been reconstructed during the past 50 years; (2) the Henry's law solubilities in water are known; and (3) CFC–11, CFC–12, and CFC–113 concentrations in air and young water are high and can be measured (U.S. Geological Survey, 2016f).

Well-water samples were analyzed for $SF_6$ concentrations following procedures outlined in Busenberg and Plummer (2000) and by the Reston Groundwater Dating Laboratory (U.S. Geological Survey, 2016l). The $SF_6$ concentrations were determined using a purge and trap gas chromatography procedure with an electron capture detector (U.S. Geological Survey, 2016l).

The USGS Tracer LPM program was used with CFC concentration data to interpret age distributions using lumped parameter models (LPMs) (Jurgens and others, 2012). These models are transport models based on simple aquifer geometry and flow configurations that account for effects of hydrodynamic dispersion or mixing within the aquifer, well bore, or discharge area (Jurgens and others, 2012). The age distribution for each well-water sample was investigated using five LPMs: piston-flow model, exponential mixing model, exponential piston-flow model, partial exponential model, and dispersion model. Binary mixing models that combined two or more of the five LPMs also were used in the investigation when the data analysis indicated a mixing model was appropriate. To determine an age for the water sample, an appropriate LPM is selected based on the conceptual model that is thought to best represent the groundwater-flow system. The selected LPM is then calibrated by adjusting parameter values for the selected model until simulated tracer (CFCs) concentrations best match measured concentrations from the well.

Initial LPM analyses of CFCs and $SF_6$ concentrations indicated that $SF_6$ concentrations in water samples from the 40 wells and CFC–12 concentrations in water samples from 16 of the 40 wells were too high to accurately fit any of the LPMs. The $SF_6$ concentrations were greater than 3.0 parts per trillion by volume (pptv) in all water samples collected from the 40 wells, with concentrations ranging from 319 to 1,910 femtograms per kilogram (fg/kg), or 3.4 to 20 pptv (table 7, available for download at http://dx.doi.org/10.3133/sir20165139A).

The $SF_6$ concentrations in water samples from 18 of the 40 wells were at levels that indicated the water samples were naturally elevated, potentially from exposure to terrigenic sources such as dolomites and limestones. Other studies have determined that elevated $SF_6$ concentrations may be the result of exposure to nonatmospheric sources (Busenberg and Plummer, 2000; Friedrich and others, 2013; Harnisch and others, 2000; Hunt and others, 2005). Groundwater typically can be dated using $SF_6$ concentrations in water if the concentrations in groundwater are in equilibrium with the atmospheric concentrations at the time of recharge and there are no other important sources of $SF_6$ (U.S. Geological Survey, 2016g). For Minnesota, the $SF_6$ concentrations in water that are in equilibrium with air at about 48 °F (9 degrees Celsius [°C]) and an elevation of about 900 ft relative to National Geodetic Vertical Datum of 1929 (NGVD 29), assuming no excess air, would be between 3.5 and 6 femtomoles per liter (fmol/L) (513 and 880 fg/kg, respectively) (Gerolamo Casile, U.S. Geological Survey, written commun., May 24, 2016). The $SF_6$ concentrations in air have increased from 2.34 pptv in 1990 to 8.5 pptv in 2014 (U.S. Geological Survey, 2016g). The source of $SF_6$ contamination in water samples collected from wells in the northeast Twin Cities Metropolitan Area (fig. 1) was not identified in this study, but Busenberg and Plummer (2000) determined that Paleozoic dolomitic bedrock could be a source of contamination. Many of the sample wells were open to the Prairie du Chien aquifer, which consists of dolomitic bedrock (Runkel and others, 2003).

The CFC–12 concentrations in water samples from 16 of the 40 wells were high enough to be considered contaminated by CFC sources besides the atmosphere (table 7). For Minnesota, the maximum anticipated CFC–12 concentration in water that is in equilibrium with air and no contamination would be 380 picograms per kilogram (pg/kg), assuming a recharge temperature of about 48 °F (9 °C), recharge elevation of about 900 ft relative to NGVD29, and no excess air (Gerolamo Casile, U.S. Geological Survey, written commun., May 24, 2016). Concentrations greater than 380 pg/kg were considered contaminated. The CFC–12 concentrations in water samples collected from 24 of the 40 wells that were determined not to be contaminated ranged from 3.1 to 355 pg/kg (table 7). Potential sources of CFC–12 contamination include local atmospheric enrichment, landfills, and industrial solvent spills (Bakwin and others, 1997; Höhener and others, 2003; Hurst and others, 1997), which commonly are present in urban settings such as the northeast Twin Cities Metropolitan Area.

The CFC–11 and CFC–113 concentrations in all the water samples were low enough to be used for estimating age dates for the waters. For Minnesota, the maximum anticipated CFC–11 and CFC–113 concentrations in water that is in equilibrium with air and no contamination would be 680 and 110 pg/kg, respectively, assuming a recharge temperature of about 48 °F (9 °C), recharge elevation of about 900 ft relative to NGVD29, and no excess air (Gerolamo Casile, U.S. Geological Survey, written commun., May 24, 2016). The CFC–11 and CFC–113 concentrations in the water samples from the 40 wells ranged from 5.3 to 336 pg/kg, and 2.3 to 43 pg/kg, respectively (table 7).

## Quality Assurance and Quality Control

Quality assurance and quality control are high priorities for USGS laboratories. The USGS Reston Stable Isotope Laboratory and USGS Reston Groundwater Dating Laboratory have internal quality-assurance policies, and both laboratories work directly with the USGS National Water-Quality Laboratory to ensure the quality of their analytical results (Révész and Coplen, 2008). All the water-quality data and sampling-site information, including quality-assurance and quality-control sample information, are stored in the USGS NWIS database (U.S. Geological Survey, 2016a). Electronic field sample forms were used to record field water-quality properties and water-quality instrument calibration data.

Sequential replicate samples were collected after the collection of six well-water samples, one piezometer-water sample, and five surface-water (lake) samples. A sequential replicate sample is collected consecutively after the collection of the environmental sample to assess variability among samples resulting from sample collection, processing, shipping, and laboratory procedures completed at different sampling times (U.S. Geological Survey, variously dated). The sequential-replicate samples were analyzed for oxygen-18/oxygen-16 and deuterium/protium ratios.

A split replicate sample was collected for two precipitation samples. A split replicate sample serves the same purpose as a sequential replicate sample but is split from the same water as the original sample rather than being collected after the original sample (U.S. Geological Survey, variously dated). By splitting the replicate sample from the same water as the environmental sample, the split replicate sample can be used to assess sample variability not associated with sample collection. The replicate samples were collected following protocols outlined in the USGS National Field Manual for the Collection of Water-Quality Data (U.S. Geological Survey, variously dated).

The differences between environmental and replicate samples for oxygen-18/oxygen-16 and deuterium/protium ratios indicated little variability resulting from sample collection, processing, shipping, and laboratory procedures completed at different sampling times. Differences for oxygen-18/oxygen-16 ratios in the collected water samples ranged from -0.07 to 0.16 per mil, and differences for deuterium/protium ratios in the collected water samples ranged from -0.4 to 1.5 per mil (table 8). The percentage differences between environmental and replicate samples for oxygen-18/oxygen-16 and deuterium/protium ratios were less than plus or minus (±) 5 percent (ranged from -1.00 to 5.6 percent) for all water samples, except for oxygen-18/oxygen-16 ratios for water samples collected from Turtle Lake on August 19, 2014, and deuterium/protium ratios for water samples collected from piezometer P4 on September 17, 2014 (table 8). Differences between environmental and replicate samples for oxygen-18/oxygen-16 and deuterium/protium ratios were all lower than the estimated expanded uncertainty of replicate measurements for oxygen-18/oxygen-16 and deuterium/protium ratios determined by the USGS Reston Stable Isotope Laboratory (±0.2 and ±2.0 per mil, respectively; U.S. Geological Survey, 2016b), which indicates that the differences were less than the uncertainty of the analyses. These estimated expanded uncertainties indicate that if a sample is reanalyzed, the new value would lie within the uncertainty bounds 95 percent of the time.

**Table 8.** Summary of quality-assurance data for sequential replicate samples collected from wells, piezometers, and lakes, and for split replicate samples collected from precipitation stations, northeast Twin Cities Metropolitan Area, Minnesota, May through October 2014.

[Values in parentheses are relative percentage differences (RDP) between concentrations in environmental samples and replicate samples where RPD = 100 x (environmental concentration - replicate concentration) / [(environmental concentration + replicate concentration)/2]. per mil, part per thousand]

| Site number | Date | Time (24 hour) | Sample type | Oxygen-18/ oxygen-16 ratio, unfiltered water (per mil) | Deuterium/ protium ratio, unfiltered water (per mil) | Difference between concentrations in environmental and replicate samples | |
|---|---|---|---|---|---|---|---|
| | | | | | | Oxygen-18/ oxygen-16 ratio, unfiltered water (per mil) | Deuterium/protium ratio, unfiltered water (per mil) |
| 451004093024401 | 10/29/2014 | 13:25 | Well | -8.44 (-0.59) | -60.9 (-0.7) | -0.05 | -0.4 |
| 450138092552001 | 10/29/2014 | 16:05 | Well | -7.73 (-0.13) | -57.8 (2.5) | -0.01 | 1.4 |
| 450251093022401 | 10/29/2014 | 10:25 | Well | -9.13 (-0.55) | -64.0 (0.3) | -0.05 | 0.3 |
| 450250093031601 | 10/29/2014 | 11:15 | Well | -7.71 (-0.39) | -56.7 (0.0) | -0.03 | 0.0 |
| 450346093051801 | 10/29/2014 | 09:35 | Well | -7.52 (-0.27) | -55.6 (2.0) | -0.02 | 1.1 |
| 450431093034701 | 10/29/2014 | 08:35 | Well | -6.99 (0.00) | -53.1 (-0.4) | 0.00 | -0.2 |
| 450417092580601 | 9/17/2014 | 14:11 | Piezometer | -2.65 (1.1) | -30.7 (5.0) | 0.03 | 1.4 |
| 450650093000801 | 6/25/2014 | 16:05 | Lake | -4.42 (-0.45) | -39.3 (-0.3) | -0.02 | -0.1 |
| 450650093000801 | 7/27/2014 | 06:15 | Lake | -3.98 (-1.0) | -37.5 (1.1) | -0.04 | 0.4 |
| 451244092524301 | 5/21/2014 | 17:15 | Lake | -3.22 (0.31) | -32.7 (-0.3) | 0.01 | -0.1 |
| 450534093075101 | 8/19/2014 | 13:35 | Lake | -2.94 (5.6) | -33.1 (3.1) | 0.16 | 1.0 |
| [1]450401092581301 | 9/12/2014 | 11:45 | Lake | -2.71 (1.5) | -30.7 (-0.6) | 0.04 | -0.2 |
| 450334092574201 | 5/27/2014 | 11:35 | Precipitation | -6.63 (-0.60) | -40.8 (1.2) | -0.04 | 0.5 |
| 450334092574201 | 7/25/2014 | 07:25 | Precipitation | -7.57 (-0.92) | -52.6 (2.9) | -0.07 | 1.5 |

[1]At 69-foot depth.

## Lake-Sediment Characteristics, Water-Level Differences, Hydraulic Conductivity, and Seepage Rates

The lithology and structure of lake sediments were characterized using collected lake-sediment cores and CSPs (see “Continuous Seismic-Reflection Profiling in Six Northeast Twin Cities Metropolitan Area Lakes” section). Four piezometer nests were installed in deep waters of White Bear Lake (P1–P4; fig. 2) where water exchange may happen across the water-sediment interface. Water levels in the piezometer nests installed in lake and glacial sediments were monitored between March and October 2014 and compared to monitored lake levels. Single well, falling-head and rising-head slug tests were completed in 5 of the 11 piezometers to estimate hydraulic conductivity of the lake and glacial sediments at the piezometer screens. Water flows across the water-sediment interface were measured using seepage meters in March and August 2014 and compared to measured water-level data and hydraulic conductivity estimates to assess groundwater and surface-water exchanges in 2014.

### Lake-Sediment Characteristics

Lake-sediment cores and CSPs were collected in White Bear Lake (fig. 1) to characterize lake sediments and their structure. CSPs also were collected in five other northeast Twin Cities Metropolitan Area lakes (Big Marine Lake, South School Section Lake, Lake Elmo, Pleasant Lake, and Turtle Lake) to characterize the sediment structure.

#### Lake-Sediment Coring in White Bear Lake

Lake-sediment cores were collected at four sites (P1–P4; fig. 2) in the southeast bay of White Bear Lake at locations coincident with the piezometer nests (see the “Deep-Water Piezometer and Lake Water-Level Differences at White Bear Lake” section). Cores were collected between March 4 and 7, 2014, by the University of Minnesota’s National Lacustrine Core Facility (LacCore), the MGS, and USGS. The lithologies of each core were described by limnologists and geologists at LacCore and the MGS. A detailed description of the coring process, photos of each lake-sediment core, and characterization of the core lithology are in Heck and others (2014).

The four sites were chosen to potentially represent a variety of lake- and pore-water exchange conditions in deep waters of White Bear Lake. The CSPs were assessed to determine the locations of the four sites (see the "Continuous Seismic-Reflection Profiling in Six Northeast Twin Cities Metropolitan Area Lakes" section). Three of the four sites were chosen for areas where the CSPs indicated little trapped gas, little to no organic material, and some structural changes happened in the lake sediments. These areas were thought to represent areas where water exchange between lake water and groundwater in lower sediments might happen in deep waters. The fourth site was selected in an area where trapped gases and thick deposits of organic materials were present, and represent areas with little to no water exchange between lake water and groundwater in lower sediments.

Multiple holes were used to collect overlapping drives of lake-sediment cores until sediment refusal happened at each site. The multiple cores were used to construct a composite stratigraphy at each site. The starting and ending depth of each core was determined from the lengths of core drives to account for any overpenetration or underpenetration on the drives. The starting depth of each core was determined by taking the starting depth of the previous drive, adding the length of the previous drive, and subtracting the previous core length. The ending depth of each drive was calculated by adding the drive length to the starting depth. This length is typically equal to the travel distance of the corer but can be more or less if there is underpenetration or overpenetration of the drive, respectively. The rods and coring tools were lowered through an aluminum casing that was placed from the ice surface down to the water-sediment interface. The casing was used to reduce flexing of the rods, which allowed maximum penetration of the corer.

The cores were collected using one of three types of piston corers: Griffith, Livingstone, and Bolivia corers (Heck and others, 2014). The Griffith corer is a single-drive corer that was initially used in the coring process to recover a single core of the water-sediment interface (University of Minnesota National Lacustrine Core Facility, 2010). The corer has a robust head that connects standard polycarbonate tubing to the drive rods. The Bolivia and Livingstone corers are repeat-drive corers that were used to collect multiple cores at depths below the water-sediment interface. The Bolivia corer is a modified Livingstone corer that is better at recovering lake sediments with high water content. The Bolivia corer is a square-rod piston corer that recovers a maximum of 4.92 ft of sediment with each drive. The corer accepts a 2.75-in. inside diameter, polycarbonate tube for core collection. When sediments become too compact or resistant for the Bolivia corer, the Livingstone corer typically is used to continue coring in the same hole (Myrbo and Wright, 2008). The Livingstone corer is a square-rod piston corer that can be used in water depths less than 100 ft to collect successive 3.28-ft drives of soft to consolidated lake sediment from a single hole. The corer uses a 2-in. inside diameter, steel barrel to decrease friction and increase the rigidity of the corer. The Livingstone corer is best used when coring more resistant material. All three corers are operated by pushing the core barrel, head, and rods into the sediment by hand, while the piston provides suction to recover and retain the core in the barrel (Heck and others, 2014). Each lake-sediment core was collected using LacCore standard sampling procedures (Myrbo and Wright, 2008).

Whole-core density, electrical resistivity, and magnetic susceptibility logging were completed on the core sections at LacCore. The core sections were then split lengthwise, imaged, and logged for high-resolution, magnetic susceptibility. A composite depth scale was constructed for the core sections by referencing field notes on the core intervals and correlating similar lithologies from the images of the overlapping sections of the cores (Heck and others, 2014). The cores, images, and analytical data were archived at LacCore. The lake-sediment cores, supplemented with the geologic descriptions made by the MGS and LacCore, were used to help describe the sediment lithology at each piezometer nest. A full description of the LacCore core analysis process is in Heck and others (2014).

### Continuous Seismic-Reflection Profiling in Six Northeast Twin Cities Metropolitan Area Lakes

The CSP surveys were completed in six lakes (White Bear Lake, Big Marine Lake, South School Section Lake, Lake Elmo, Pleasant Lake, and Turtle Lake) across the northeast Twin Cities Metropolitan Area in November 2013 to characterize lake sediments and their structure on the lake bottoms (fig. 2). The CSP is a water-borne surface geophysical technique that can be used in lacustrine and other open-water settings to interpret the lithologic character of lake sediments and aquifer materials below the lake sediments (Haeni, 1986). A detailed description of the method and theory of CSP, along with a discussion of the advantages and disadvantages of the method, is given in Gorin and Haeni (1989), Haeni (1986), Haeni and Placzek (1991), and Placzek and Haeni (1995). These lakes were selected based on either (1) recent (2003–13) lake-level declines, (2) deep lake depths, or (3) the presence of buried bedrock valleys below the lakes. The CSP survey results for White Bear Lake also were used to select four sites in deeper waters to collect lake-sediment cores, monitor water-levels in piezometers, and monitor flow rates across the water-sediment interface. The CSPs for path lines that run through the lake-sediment coring sites were compared to the lithologies of the cores to determine any relations between the seismic reflection responses and different types of lake sediments.

A CSP system consists of a sound source, receiver, personal computer, and control unit that transmits the outgoing sound wave, processes the received signals, and passes the digital data to the hard drive and monitor for near real-time display. The sound source and receiver are housed within the body of a 2.5-ft tow-vehicle, commonly referred to as a towfish, which was connected through electronic cabling to the control unit. A 2 kilowatt (kW) generator was used to power all the onboard and submerged equipment. The towfish was towed 3.3 ft (1 m) below the water surface and next to a 15.8-ft (4.8-m), aluminum-hull boat that was piloted by USGS personnel.

The sound source used for this investigation was a swept frequency system that sweeps through a range of frequencies in the kilohertz (kHz) range. The chirp acoustic source uses a digitally generated, frequency modulated (FM) pulse that sweeps over a user-selectable frequency range. A transmission pulse of 4 to 20 kHz was used to ensure adequate resolution of shallow features and sufficient depth of penetration was achieved. Transmission of the outgoing pulses was five times per second, and the recording length was 275 milliseconds, which is equivalent to a maximum record length depth of 451 ft (137.5 m) in freshwater and sediments. The frequency or energy of the outgoing pulse was adjusted to minimize attenuation because of spreading and improve penetration into the sediments. An increase in power may or may not improve the penetration, so a trial-and-error approach was used. In general, setting the power at about 75 percent of full scale maximized the signal-to-noise ratio. The optimal power setting was a lower value in the presence of strong water-bottom multiple signals reflected off a source recorded in the CSP.

The hydrophone arrays in the towfish are omnidirectional receivers used to measure the reflected acoustic waves. With a chirp system, the received signal is processed in a form of cross-correlation with the transmitted signal (Trabant, 1984). The output signal is received by the control unit, which correlates the output signal with a proprietary pulse (that is, a function of the output pulse frequency band); the resulting signal is then stored as data (Schock and LeBlanc, 1990).

The CSPs were collected between dawn and 5:00 p.m. Central Standard Time, Monday through Friday from November 12 to 21, 2013. Data collection focused on morning hours to reduce exposure to choppy water generated by wind and recreational boaters. The smoother water reduced potential noise in the geophysical data. Data were collected in path lines across the lake to maximize lake-surface coverage and cover the deep sections of the lakes. A zigzag pattern with straight-line segments was used over most of the lakes. The mean boat speed was about 2 knots (2 miles per hour [mi/h]).

Latitude and longitude for the CSP data were collected using a TRIMBLE PRO XRS global positioning system with datum referenced to the World Geodetic System of 1984 map datum. Latitude and longitude data were passed directly to the CSP acquisition software during data collection at a rate of once per second.

Depth of seismic signal penetration into the lake sediments will vary with the presence of gases in the sediment, geologic properties of the bottom sediments, and wind conditions. Trapped gases in organic-rich material can prevent signal penetration and cause multiple reflections that result in dark sections in the CSPs that cannot be interpreted (Johnson and White, 2007). Hard-packed sediments, cobbles, and boulders can reflect much of the seismic signal, thereby limiting signal penetration (Johnson and White, 2007; Banks and Johnson, 2011). Strong winds resulting in large fetches in the water can move the towfish and cause noise in the received signals.

## Deep-Water Piezometer and Lake Water-Level Differences at White Bear Lake

Piezometer nests (P1–P4) were installed at the four sites where lake-sediment cores were collected in the southeast bay of White Bear Lake to monitor water levels and estimate hydraulic conductivities in the lake and glacial sediments (fig. 2). Sites P1, P2, and P3 had three piezometers each; and site P4 had two piezometers (table 9, available for download at http://dx.doi.org/10.3133/sir20165139A). The depths of the piezometers ranged from 9 to 33 ft below land surface (lake bottom), and lake depths during piezometer installation ranged from 28.5 to 68.1 ft (table 9). The piezometers were installed from March 11 to 20, 2014, and removed from March 6 to 9, 2015.

The piezometers at each site were placed in a triangular formation about 6 ft from each other. Each piezometer at the sites was constructed to set the screens at depths that differed geologically and vertically from the other piezometers at the site, trying to prevent screen overlap between the piezometers at each site. At site P4, the screens overlapped by 1 ft because the galvanized pipe for piezometer P4–8 refused to be driven below 11 ft into the sediment (table 9). The piezometer naming convention was based on the site name (P1, P2, P3, or P4) and the depth, in feet, of the top of the screen below the lake bottom, with the exception of P1–8.5. The top of the screen for piezometer P1–8.5 was 7.5 ft below land surface (lake bottom).

The basic design of and materials used to construct each piezometer were similar (fig. 7). The upper part of each piezometer consisted of 1-in. inside diameter, schedule 40 PVC pipe and 2.9-ft hose. The length of the PVC casings ranged from 5.0 to 10.2 ft (table 9). The hose was used to provide flexibility needed to bend the PVC casing below the lake surface before the lake thawed in the spring and froze in late fall, which prevented damage to the PVC casing during ice melting and formation. The hose in each piezometer was attached to a lower 1-in. inside diameter, galvanized steel pipe. The length of the galvanized steel pipe of the piezometers ranged from 27 to 73 ft (table 9). The galvanized steel pipe of each piezometer was attached to a lower 10-slot (0.01 in.), 1-in. inside diameter, 3-ft long, wrapped, stainless steel screen.

A 10-ft long, 2-in. inside diameter PVC pipe was placed around the 1-in. PVC casing and hose of each piezometer when the piezometers were above the lake surface to (1) make the piezometers more visible to individuals driving boats, snowmobiles, and other recreational vehicles on the lake; and (2) keep the piezometers vertically straight above the lake surface in the summer and winter after the ice melted and froze. The galvanized pipe and screen were driven into the lake bottom using a jackhammer connected to a 2-in. diameter steel drive pipe that slid over the 1-in. diameter, galvanized steel pipe. The steel drive pipe was only used as a tool for installation and was not a part of the piezometer design.

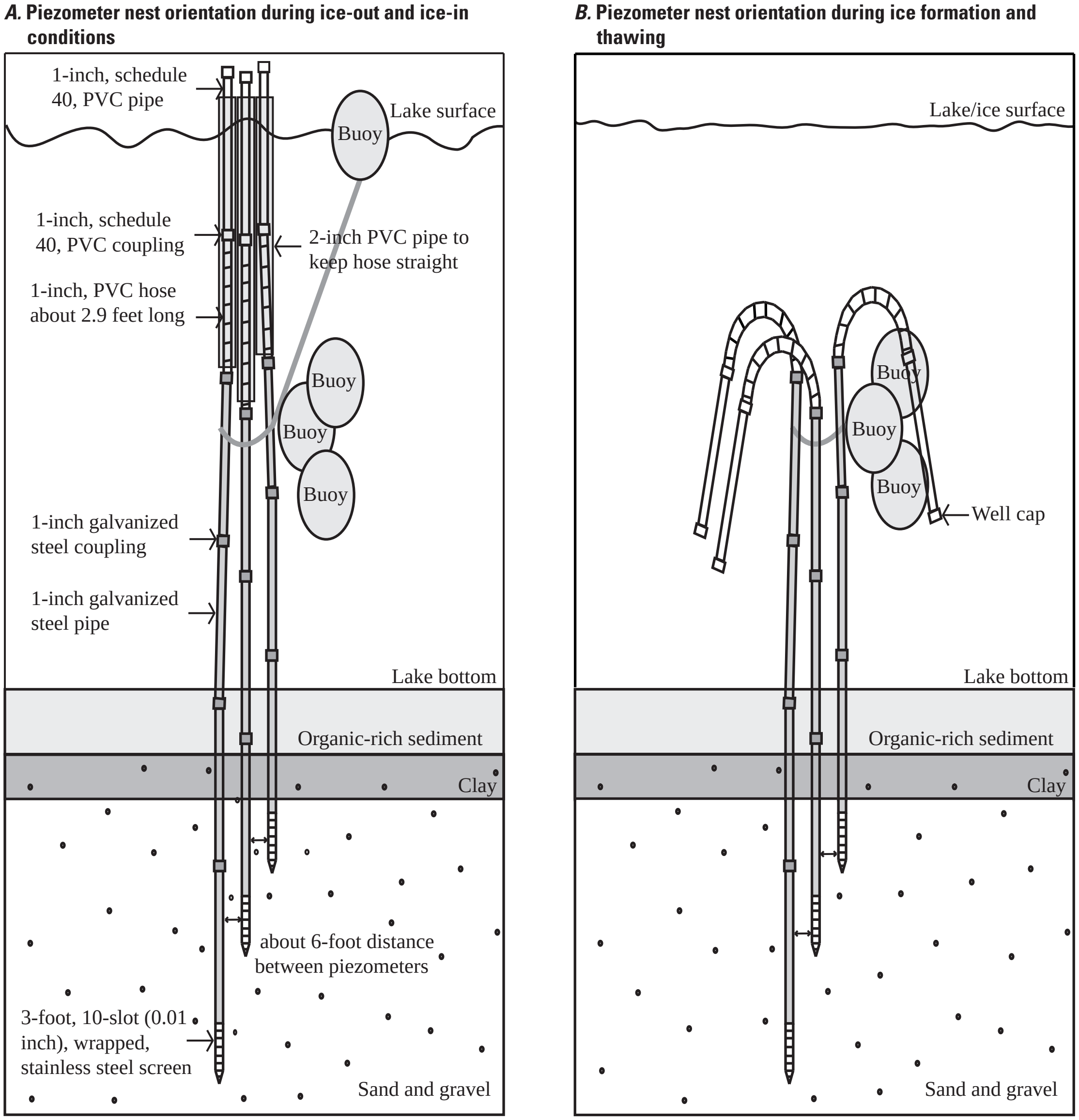



**Figure 7.** General construction of piezometer nests completed in sand and gravel deposits underlying the lake-bottom sediments of White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota. [PVC, polyvinyl chloride]

During the installation period, when White Bear Lake was not frozen, one marker buoy was connected to the piezometer nest beneath the PVC pipes and visible at the lake surface to aid in the piezometer nest visibility (fig. 7*A*). Three support buoys were connected to each piezometer nest just below the PVC protection pipes to help keep the casings straight during the installation period (March–November 2014) (fig. 7*B*).

Water levels were measured in the piezometers on three to six dates from May 30 to October 16, 2014 (table 10). The water levels for all four piezometer nests were monitored from a boat. On some dates, wave action was too high at some piezometers to get accurate water-level measurements. To minimize errors caused by wave action, tape-down measurements on dates when there was wind or when lake levels were not stable were not used to determine water levels in the piezometers. Water levels were measured relative to an established measuring point on the top of the 1-in. PVC casing in each piezometer using a steel or electrical tape to the nearest 0.01 ft. These tape-down measurements were made when there was no ice in the piezometers. The top of the 1-in. PVC casings were surveyed into local benchmarks to obtain measuring point elevations relative to the North American Vertical Datum of 1988 (NAVD 88).

Continuous water levels were monitored in the lake and piezometers P3–9, P3–18.5, and P4–8 between May 30 and November 7, 2014. These sites were chosen for continuous water-level monitoring because (1) lake-sediment coring and

**Table 10.** Piezometer tape-down, water-level measurements and associated errors, White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.

[ft, foot; NAVD 88, North American Vertical Datum of 1988; ± plus or minus]

| Piezometer number (figs. 22, 23) | Date | Water-level elevation (ft above NAVD 88) | Water level (ft above lake surface [positive value] or ft below lake surface [negative value]) | Water-level error (±ft) |
|---|---|---|---|---|
| P1–8.5 | 5/30/2014 | 920.95 | -0.30 | 0.10 |
| | 6/20/2014 | 921.88 | -0.16 | |
| | 8/14/2014 | 921.52 | -0.20 | |
| | 8/20/2014 | 921.47 | -0.16 | |
| | 9/17/2014 | 921.46 | -0.14 | |
| | 10/16/2014 | 921.15 | -0.30 | |
| P1–16.5 | 5/30/2014 | 920.93 | -0.32 | 0.08 |
| | 6/20/2014 | 921.83 | -0.21 | |
| | 8/14/2014 | 921.49 | -0.20 | |
| | 8/20/2014 | 921.50 | -0.13 | |
| | 9/17/2014 | 921.46 | -0.13 | |
| | 10/16/2014 | 921.32 | -0.13 | |
| P1–19.5 | 5/30/2014 | 920.94 | -0.31 | 0.12 |
| | 6/20/2014 | 921.71 | -0.33 | |
| | 8/14/2014 | 921.38 | -0.31 | |
| | 8/20/2014 | 921.42 | -0.21 | |
| | 9/17/2014 | 921.44 | -0.15 | |
| | 10/16/2014 | 921.19 | -0.24 | |
| P2–17.5 | 5/30/2014 | 921.25 | 0.00 | 0.05 |
| | 6/20/2014 | 922.11 | 0.11 | |
| | 8/14/2014 | 921.74 | 0.05 | |
| | 8/20/2014 | 921.64 | 0.02 | |
| | 9/17/2014 | 921.68 | 0.07 | |
| P2–23 | 5/30/2014 | 921.05 | -0.20 | 0.04 |
| | 6/20/2014 | 921.8 | -0.20 | |
| | 8/14/2014 | 921.43 | -0.26 | |
| | 8/20/2014 | 921.42 | -0.20 | |
| | 9/17/2014 | 921.40 | -0.21 | |
| P2–30 | 5/30/2014 | 920.80 | -0.29 | 0.06 |
| | 6/20/2014 | 921.71 | -0.29 | |
| | 8/14/2014 | 921.32 | -0.37 | |
| | 8/20/2014 | 921.36 | -0.26 | |
| | 9/17/2014 | 921.35 | -0.26 | |
| P3–9 | 5/30/2014 | 921.08 | -0.17 | 0.07 |
| | 6/20/2014 | 921.81 | -0.23 | |
| | 8/14/2014 | 921.62 | -0.11 | |
| | 8/20/2014 | 921.45 | -0.17 | |
| | 9/17/2014 | 921.42 | -0.16 | |
| | 10/16/2014 | 921.28 | -0.19 | |

**Table 10.** Piezometer tape-down, water-level measurements and associated errors, White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.—Continued

[ft, foot; NAVD 88, North American Vertical Datum of 1988; ± plus or minus]

| Piezometer number (figs. 22, 23) | Date | Water-level elevation (ft above NAVD 88) | Water level (ft above lake surface [positive value] or ft below lake surface [negative value]) | Water-level error (±ft) |
|---|---|---|---|---|
| P3–13.5 | 5/30/2014 | 921.09 | -0.16 | 0.11 |
| | 6/20/2014 | 921.85 | -0.19 | |
| | 8/14/2014 | 921.70 | -0.03 | |
| | 8/20/2014 | 921.48 | -0.14 | |
| | 9/17/2014 | 921.42 | -0.16 | |
| | 10/16/2014 | 921.32 | -0.15 | |
| P3–18.5 | 5/30/2014 | 920.92 | -0.33 | 0.07 |
| | 6/20/2014 | 921.75 | -0.29 | |
| | 8/14/2014 | 921.39 | -0.34 | |
| | 8/20/2014 | 921.30 | -0.31 | |
| | 9/17/2014 | 921.30 | -0.28 | |
| | 10/16/2014 | 921.20 | -0.27 | |
| P4–6 | 5/30/2014 | 921.26 | 0.01 | 0.08 |
| | 6/20/2014 | 921.94 | -0.10 | |
| | 9/12/2014 | 921.65 | 0.00 | |
| P4–8 | 5/30/2014 | 921.11 | -0.14 | 0.10 |
| | 6/20/2014 | 921.93 | -0.11 | |
| | 9/12/2014 | 921.66 | -0.02 | |

CSPs indicated organic-rich materials were less than 3 ft thick, and little to no trapped gases were present at the sites; (2) P3–9 and P3–18.5 were the shallowest and deepest piezometers, respectively, at site P3, which could potentially provide the largest water-level differences at the site; and (3) P4–8 is the deepest piezometer at site P4, which could potentially result in the largest water-level difference between the piezometer and the lake at the site. Unvented pressure transducers were used in the lake and piezometers P3–9, P3–18.5, and P4–8 to record water pressure. The pressure transducers in P3–9, P3–18.5, and P4–8 were placed about 10 ft below the top of each piezometer casing. The pressure transducer used to monitor water levels in White Bear Lake was placed 4 ft below the top of a 5-ft long, 2-in. inside diameter, screened PVC tube connected to piezometers at site P2. The tube was placed 2 ft above the lake surface during installation. Each pressure transducer was set to record hourly water levels.

Barometric (air) pressures were monitored at the lake to subtract out the effects of barometric pressure changes on the pressure data collected from the unvented transducers installed in the piezometers and lake. An unvented pressure transducer was installed near the weather station (USGS site

450334092574201) along the southeast shore of White Bear Lake (fig. 2) to monitor hourly barometric pressure at the lake. Subtracting the measured barometric pressure from the measured pressures in the piezometers and lake resulted in pressure data that represented actual water-level changes in the piezometers and lake.

All the pressure transducers used to monitor water levels and barometric pressure were calibrated before installation, and the calibrations were checked after removal from the sites. The pressure transducers were calibrated following the manufacturer's instructions (Solinst Canada Ltd., 2015) and procedures outlined by Cunningham and Schalk (2011). A water bath and barometer were used to determine a linear relation between pressure and depth for each pressure transducer. The linear relation between pressure and depth was programmed into the data loggers associated with each pressure transducer.

The pressure transducer measurements were checked against the tape-down, water-level measurements made from May 30 to October 16, 2014, to determine drift in the transducer readings (table 10). The continuous lake levels were checked against the MNDNR lake-level data (Minnesota Department of Natural Resources, 2015). The measuring point error associated with the tape-down, water-level measurements in the piezometers was calculated based on the mean difference between the measured and calculated stickup values of each tape-down, water-level measurement (Cunningham and Schalk, 2011). The measured stickup for these piezometers is the length of casing above the lake bottom recorded during each tape-down, water-level measurement. The calculated stickup is based on a surveyed stickup elevation for the piezometers measured when the ice was frozen and shortly after the piezometer was installed. All the water-level and barometric pressure data are stored in the USGS NWIS database (U.S. Geological Survey, 2016a), and can be retrieved using the site numbers in table 9.

## Hydraulic Conductivities of White Bear Lake Sediments

Single well, falling- and rising-head slug tests were completed on 5 of the 11 installed deep piezometers: P1–16.5, P2–30, P3–9, P3–18.5 and P4–8 (table 9). The tests were completed to estimate hydraulic conductivity of the glacial material beneath the lake bottom of White Bear Lake (fig. 2). Three falling-head and three rising-head slug tests were completed in piezometers P1–16.5, P2–30, and P4–8; and two falling-head and two rising-head slug tests were completed in piezometers P3–9 and P3–18.5. The slug tests were completed in February 2015 when ice was present on the lake surface. Immediately after the piezometers were pulled above the ice surface in mid-January 2015, a heated cable, traditionally used for preventing pipes from freezing, was placed inside each piezometer through the depth of potential ice formation. The water inside the piezometers would freeze to a depth similar to the ice thickness on the lake if the heating cable was not used. Two hours before completing a slug test, the piezometer's heating cable was plugged into a generator to thaw the water in the piezometer. Once the piezometers were fully thawed, a portable ice house heated with a portable heater was placed over the piezometer nest to prevent the piezometers from freezing again once the heating cable was removed to complete the slug tests. Before the heating cable was removed, a static water level was measured using a steel tape. A nonvented pressure transducer, set to record water pressure every 0.5 seconds, was immediately submerged 8 ft below the measuring point in the piezometer being tested after the heating cable was removed. A separate pressure transducer was used to measure the barometric pressure during the tests. Before the test was started, the water level of the piezometer nest was allowed to go back to static.

The solid slugs used during the slug tests were based on the recommended design presented in the USGS Groundwater Technical Procedure Document 17 for completing slug tests (Cunningham and Schalk, 2011).The length of the solid slug used in piezometers P1–16.5, P2–30, P3–9, and P3–18.5 was 2.04 ft and expected to displace 0.96 ft of water, whereas the length of the solid slug used in piezometer P4–8 was 3.04 ft and expected to displace 1.44 ft of water. A 30-pound, braided fishing line was tied to the top of each slug to set and remove the slug from the piezometer. The slug was placed 1 ft above the static water level before the slug was introduced and was set 1 ft below the static water level after the slug was introduced to ensure a clean and instantaneous water-level change. To ensure the completion of each slug test, the water level was measured using a steel tape to check the water level was back to static before an additional slug test was completed in the well.

The Aqtesolv Pro Program, version 4.5 (Duffield, 2007), was used to analyze the water-level recoveries provided by the slug tests. The Kansas Geological Survey (KGS) method was used to analyze these water-level recoveries to produce estimates of hydraulic conductivity (Hyder and others, 1994). This method implies a curved solution to declining or rising water-level data collected during a single-well slug test in an unconfined or a confined aquifer with a completely or partially penetrating well. The KGS method assumes the following:

1. the unconfined or confined aquifer is infinite in extent, homogeneous, and of uniform thickness;
2. the potentiometric surface of the aquifer is initially horizontal;
3. the slug is introduced or removed instantaneously to and from the well;
4. head losses during the test are negligible;
5. the water-level response from the slug test is classified as unsteady or overdamped (nonoscillating);
6. water is released instantaneously from storage with decline of hydraulic head; and
7. the wells are fully or partially penetrating.

The KGS method provides corrections for low permeability material around the well screen, such as mud residue from well installation, and takes anisotropy into account (U.S. Environmental Protection Agency, 2014).

Because the collected lake coring data were not available through all screened intervals of the piezometer, nearby domestic well logs were used to help estimate the aquifer thickness for all the piezometers. The domestic wells that were chosen were the closest and deepest well logs available to the piezometer nests installed for this study and near the southeast shoreline of White Bear Lake (Minnesota Unique Wells 431492, 208508, and 112412; fig. 2) provided by the Minnesota County Well Index (Minnesota Department of Health and Minnesota Geological Survey, 2016). The glacial sediments were estimated between 873 and 863 ft (NAVD 88) at piezometer nest P1, 840 to 824 ft (NAVD 88) at P2 and P4, and 864 to 844 ft (NAVD 88) at P3. The saturated thickness of the sand-and-gravel aquifer near each piezometer was estimated to be 16 ft based on the geologic logs from three nearby domestic wells (Minnesota Department of Health and Minnesota Geological Survey, 2016).

Based on the lithologic descriptions provided by LacCore (Heck and others, 2014) and the geologic well logs near the piezometers, the piezometer screens were assumed to be set in sand and gravel material. The hydraulic conductivity anisotropy, or relation between vertical and horizontal hydraulic conductivities, was assumed to be 1 based on the assumption that the vertical and horizontal hydraulic conductivities of the sand and gravel material is similar.

## Mini-Piezometer and Seepage-Meter Surveys in White Bear Lake

Mini-piezometers were used with manometers to measure hydraulic-head differences between pore waters in lake sediments and lake water (Rosenberry and LaBaugh, 2008; Jones and others, 2013). The mini-piezometers provide a comparison between the stage of a surface-water body, such as a lake, and the hydraulic head (water-level elevation) beneath the surface-water body at the depth to which the screen at the end of the probe is driven (Winter and others, 1988). The difference in hydraulic head divided by the distance between the screen and water-sediment interface is a measurement of the vertical hydraulic-head gradient. Mini-piezometers do not provide a direct indication of seepage flux; however, hydraulic-head measurements from mini-piezometers can be used in combination with water-flux measurements from a seepage meter to yield information about the vertical hydraulic conductivity of the sediments (Kelly and Murdoch, 2003; Zamora, 2006).

Hydraulic-head differences and seepage-meter measurements were made at nine sites along the lakeshore of White Bear Lake from August 20 through 23, 2014 (fig. 2). Multiple measurements of hydraulic-head differences were made at sites where multiple seepage-meter measurements were made. For each measurement, the probe was inserted by hand into the lake sediments as deep as possible and connected to a manometer. The manometer measured small hydraulic-head differences between the pore and lake water. Probe insertion depths into the lake sediments ranged from 0.59 to 3.0 ft; the deeper insertion depths were at locations with thick deposits of organic materials. Once the probe was pushed beneath the water-sediment interface, the outer pipe of the mini-piezometer was retracted to expose the screen. A plastic tube was inserted into the top of the mini-piezometer to connect the probe to one end of a manometer. Another plastic tube connected to the manometer was placed in the lake. A vacuum pump was connected to the manometer to fill the tubing with water from the probe and surface water. The tubing to the lake was clamped before the measurements to develop sufficient suction to pull water through the mini-piezometer screen and tubing. Air bubbles trapped in water in the tubing were removed by physically moving the tubing to aid in bubble release through buoyancy.

Once the tubing was full of water and free of bubbles, air was bled into the top of the manometer through the tubing until the menisci were visible in the tubing on both sides of the manometer. The difference in height of the menisci was recorded. This difference equals the difference between hydraulic head in the sediment pore water at the screen and the stage of the lake.

The flux of water across the water-sediment interface was measured directly using half-barrel, seepage meters from August 20 to 23, 2014, at nine sites along the shore of White Bear Lake (fig. 2; table 11). Seepage meters are devices that isolate a small area of the bed of a surface-water body and measure the flow of water across that area (Rosenberry and LaBaugh, 2008). A half-barrel, seepage meter consists of a cut-off end of a 55-gallon (gal) steel (or plastic) storage drum to which a plastic bag is attached to register the change in water volume during the time of bag attachment (Lee, 1977; Lee and Cherry, 1978). At each site, one to six seepage meters were submerged in the lake and placed in the sediment to contain the water flow that crosses that part of the water-sediment interface. A bag containing a known volume of water was attached to the submerged drum for a measured amount of time, after which the bag was removed and the volume of water contained in the bag was measured again. The change in volume during the time the bag was attached to the drum represented the volumetric rate of flow through the part of the bed covered by the drum (volume per time). The volumetric rate of flow was divided by the 2.7-square-foot ($ft^2$) area covered by the chamber to yield seepage as a flux velocity (distance per time). Flux velocity normalizes the area covered by the seepage meter and allows comparisons of results with other studies (and other sizes of seepage meters). One to three seepage meters were installed at each of the nine sites in White Bear Lake. Multiple measurements were made at each seepage meter to obtain a mean seepage flux rate for each site. A total of 63 seepage-meter measurements were made along the shore of White Bear Lake (table 11). A median flux was calculated for each meter location (table 11). At the sites where seepage

**Table 11.** Seepage-flux measurements and hydraulic-head measurements on the shore of White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota, August 2014.

[deg:min:sec, degree-minute-second; ft/d, foot per day; ft, foot; *K*, vertical hydraulic conductivity, from equation 3; --, no data]

| Seepage meter site name | Seepage meter site number (fig. 2) | Seepage meter number | Latitude (deg:min:sec) | Longitude (deg:min:sec) | Start date of seepage meter measurement | Mid time of seepage meter measurement (24 hour) | Duration of measurement (minutes) | Seepage flux (ft/d to the lake [positive value] or ft/d out of the lake [negative value]) | Median flux (ft/d to the lake [positive value] or ft/d out of the lake [negative value]) | Date of head measurement | Time of head measurement (24 hour) | Head-difference measurement (ft to the lake [positive value] or ft out of the lake [negative value]) | Mini-piezometer insertion depth (ft) | *K* values (ft/d) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Northeast Shore | 1 | 1 | 45° 05’ 26.0” | 92° 58’ 37.3” | 8/20/2014 | 16:54 | 15 | 1.09 | 0.94 | 8/20/2014 | 19:28 | 0.03 | 1.02 | 36 |
| Northeast Shore | | | | | | 17:12 | 17 | 0.95 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 18:43 | 24 | 0.94 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 19:11 | 25 | 0.93 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 19:42 | 14 | 0.94 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | 1 | 2 | 45° 05’ 25.7” | 92° 58’ 37.3” | 8/20/2014 | 17:00 | 17 | 0.77 | 0.71 | 8/20/2014 | 19:28 | 0.03 | 1.02 | 27 |
| Northeast Shore | | | | | | 17:21 | 18 | 0.75 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 18:45 | 27 | 0.71 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 19:16 | 27 | 0.57 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 19:47 | 12 | 0.63 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | 1 | 3 | 45° 05’ 25.6” | 92° 58’ 37.2” | 8/20/2014 | 16:57 | 17 | 0.30 | 0.14 | 8/20/2014 | 19:28 | 0.03 | 1.02 | 5.6 |
| Northeast Shore | | | | | | 17:16 | 19 | 0.18 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 18:47 | 30 | 0.14 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 19:22 | 25 | 0.08 | -- | -- | -- | -- | -- | -- |
| Northeast Shore | | | | | | 19:50 | 17 | 0.13 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | 2 | 1 | 45° 04’ 9.7” | 92° 57’ 40.7” | 8/20/2014 | 15:07 | 179 | 0.15 | 0.18 | 8/20/2014 | 20:40 | 0.15 | 1.56 | 1.9 |
| Mahtomedi Beach | | | | | | 18:09 | 149 | 0.21 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | | | | | 8/21/2014 | 09:03 | 19 | 0.48 | 0.50 | 8/21/2014 | 10:28 | 0.33 | 0.92 | 1.4 |
| Mahtomedi Beach | | | | | | 09:33 | 27 | 0.51 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | | | | | | 10:08 | 26 | 0.50 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | 2 | 2 | 45° 04’ 9.5” | 92° 57’ 40.7” | 8/20/2014 | 14:47 | 192 | 0.10 | 0.15 | 8/20/2014 | 20:40 | 0.15 | 1.56 | 1.5 |
| Mahtomedi Beach | | | | | | 18:05 | 147 | 0.19 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | | | | | 8/21/2014 | 08:59 | 21 | 0.13 | 0.12 | 8/21/2014 | 10:28 | 0.33 | 0.92 | 0.33 |
| Mahtomedi Beach | | | | | | 09:27 | 31 | 0.11 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | | | | | | 10:02 | 35 | 0.12 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | 2 | 3 | 45° 04’ 9.8” | 92° 57’ 40.8” | 8/20/2014 | 14:58 | 193 | 0.05 | 0.06 | 8/20/2014 | 20:40 | 0.15 | 1.56 | 0.60 |
| Mahtomedi Beach | | | | | | 18:17 | 150 | 0.06 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | | | | | 8/21/2014 | 09:09 | 26 | 0.06 | 0.05 | 8/21/2014 | 10:28 | 0.33 | 0.92 | 0.14 |
| Mahtomedi Beach | | | | | | 09:40 | 53 | 0.04 | -- | -- | -- | -- | -- | -- |
| Mahtomedi Beach | | | | | | 10:40 | 51 | 0.05 | -- | -- | -- | -- | -- | -- |
| South of Mahtomedi Beach | 3 | 1 | 45° 04’ 1.4” | 92° 57’ 44.1” | 8/21/2014 | 15:47 | 18 | 0.48 | 0.50 | 8/21/2014 | 16:15 | 0.04 | 0.99 | 12 |
| South of Mahtomedi Beach | | | | | | 16:11 | 33 | 0.51 | -- | -- | -- | -- | -- | -- |
| South of Mahtomedi Beach | | | | | | 16:51 | 21 | 0.50 | -- | -- | -- | -- | -- | -- |

**Table 11.** Seepage-flux measurements and hydraulic-head measurements on the shore of White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota, August 2014.—Continued

[deg:min:sec, degree-minute-second; ft/d, foot per day; ft, foot; *K*, vertical hydraulic conductivity, from equation 3; --, no data]

| Seepage meter site name | Seepage meter site number (fig. 2) | Seepage meter number | Latitude (deg:min:sec) | Longitude (deg:min:sec) | Start date of seepage meter measurement | Mid time of seepage meter measurement (24 hour) | Duration of measurement (minutes) | Seepage flux (ft/d to the lake [positive value] or ft/d out of the lake [negative value]) | Median flux (ft/d to the lake [positive value] or ft/d out of the lake [negative value]) | Date of head measurement | Time of head measurement (24 hour) | Head-difference measurement (ft to the lake [positive value] or ft out of the lake [negative value]) | Mini-piezometer insertion depth (ft) | *K* values (ft/d) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| South of Mahtomedi Beach | 3 | 2 | 45° 04’ 1.4” | 92° 57’ 44.1” | 8/21/2014 | 15:45 | 18 | 0.81 | 0.81 | 8/21/2014 | 16:15 | 0.04 | 1.17 | 22 |
| South of Mahtomedi Beach | | | | | | 16:14 | 22 | 0.81 | -- | -- | -- | -- | -- | -- |
| South of Mahtomedi Beach | | | | | | 16:43 | 18 | 0.79 | -- | -- | -- | -- | -- | -- |
| South of Mahtomedi Beach | 3 | 3 | 45° 04’ 1.3” | 92° 57’ 43.9” | 8/21/2014 | 15:46 | 20 | 0.19 | 0.21 | 8/21/2014 | 17:41 | 0.06 | 1.17 | 4.4 |
| South of Mahtomedi Beach | | | | | | 16:10 | 20 | 0.26 | -- | -- | -- | -- | -- | -- |
| South of Mahtomedi Beach | | | | | | 16:35 | 22 | 0.21 | -- | -- | -- | -- | -- | -- |
| South, Southeast Shore | 6 | 1 | 45° 03’ 19.7” | 92° 58’ 00.4” | 8/22/2014 | 13:18 | 68 | 0.01 | 0.01 | 8/22/2014 | 13:37 | 0.04 | 3.00 | 0.49 |
| South, Southeast Shore | | | | | | 14:29 | 232 | 0.00 | -- | -- | -- | -- | -- | -- |
| Southeast Shore | 5 | 2 | 45° 03’ 21.8” | 92° 57’ 49.2” | 8/22/2014 | 11:49 | 29 | 0.14 | 0.07 | 8/22/2014 | 11:52 | 0.07 | 1.45 | 1.5 |
| Southeast Shore | | | | | | 12:22 | 34 | 0.09 | -- | -- | -- | -- | -- | -- |
| Southeast Shore | | | | | | 12:59 | 34 | 0.05 | -- | -- | -- | -- | -- | -- |
| Southeast Shore | | | | | | 13:37 | 33 | 0.06 | -- | -- | -- | -- | -- | -- |
| East, Southeast Shore | 4 | 1 | 45° 03’ 23.2” | 92° 57’ 46.8” | 8/22/2014 | 10:48 | 43 | 0.44 | 0.44 | 8/22/2014 | 11:42 | 0.09 | 1.35 | 6.7 |
| East, Southeast Shore | | | | | | 11:35 | 30 | 0.50 | -- | -- | -- | -- | -- | -- |
| East, Southeast Shore | | | | | | 12:10 | 29 | 0.41 | -- | -- | -- | -- | -- | -- |
| Bellaire Beach | 7 | 1 | 45° 04’ 16.3” | 92° 59’ 43.3” | 8/22/2014 | 09:13 | 239 | 0.03 | 0.04 | 8/22/2014 | 10:05 | 0.01 | 1.05 | 6.6 |
| Bellaire Beach | | | | | | 13:14 | 460 | 0.04 | -- | -- | -- | -- | -- | -- |
| Bellaire Beach | | | | | | 21:09 | 622 | 0.05 | -- | -- | -- | -- | -- | -- |
| Bellaire Beach | 7 | 2 | 45° 04’ 16.4” | 92° 59’ 43.4” | 8/22/2014 | 09:17 | 238 | 0.01 | 0.01 | 8/22/2014 | 10:05 | 0.01 | 1.05 | 2.1 |
| Bellaire Beach | | | | | | 13:18 | 460 | 0.01 | -- | -- | -- | -- | -- | -- |
| Bellaire Beach | | | | | | 21:02 | 632 | 0.03 | -- | -- | -- | -- | -- | -- |
| Bellaire Beach | 7 | 3 | 45° 04’ 16.6” | 92° 59’ 43.3” | 8/22/2014 | 09:23 | 236 | -0.00 | 0.01 | 8/22/2014 | 10:05 | 0.01 | 1.05 | 1.9 |
| Bellaire Beach | | | | | | 13:22 | 461 | 0.01 | -- | -- | -- | -- | -- | -- |
| Bellaire Beach | | | | | | 21:08 | 630 | 0.01 | -- | -- | -- | -- | -- | -- |
| Ramsey County Beach | 9 | 1 | 45° 05’ 37.7” | 92° 59’ 44.6” | 8/23/2014 | 12:02 | 152 | -0.00 | 0.01 | 8/23/2014 | 12:00 | 0.00 | 2.30 | 0.00 |
| Ramsey County Beach | | | | | | 14:45 | 194 | 0.01 | -- | -- | -- | -- | -- | -- |
| West Ramsey County Beach | 8 | 1 | 45° 05’ 28.5” | 92° 59’ 54.2” | 8/23/2014 | 12:28 | 148 | -0.00 | 0.00 | 8/23/2014 | 12:38 | 0.00 | 0.59 | 0.00 |
| West Ramsey County Beach | | | | | | 15:00 | 164 | -0.00 | -- | -- | -- | -- | -- | -- |
| West Ramsey County Beach | 8 | 2 | 45° 05’ 28.4” | 92° 59’ 53.5” | 8/23/2014 | 12:27 | 154 | 0.06 | 0.05 | 8/23/2014 | 12:54 | 0.00 | 0.72 | 0.00 |
| West Ramsey County Beach | | | | | | 15:16 | 151 | 0.05 | -- | -- | -- | -- | -- | -- |

fluxes and hydraulic-head differences were measured, the values of vertical hydraulic conductivity were calculated by applying the following version of Darcy's law:

$$K_v = v \div (H \div d) \quad (3)$$

where

$K_v$ is the vertical hydraulic conductivity, in feet per day;

$v$ is the seepage-meter median flux velocity, in feet per day;

$H$ is the hydraulic-head difference, in feet; and

$d$ is the mini-piezometer insertion depth, in feet.

The flux of water across the water-sediment interface also was measured directly at the four deep-water sites (P1, P2, P3, and P4; fig. 2) using an automated seepage meter (Rosenberry and Morin, 2004; Waldrop and Swarzenski, 2006) between March 11 and 19, 2014. The 3.56-ft diameter seepage meter consisted of an aluminum ring, 6-in. tall, welded to an aluminum dome that extended the total height of the seepage meter to 1.5 ft. An electromagnetic flowmeter mounted at the top of the dome measured flow through a 0.5-in. diameter opening; upward flow generated positive values in millivolts (mV), and downward flow created negative values. Output was converted to cubic feet per day ($ft^3$/day), which was divided by the surface area covered by the seepage meter (9.95 $ft^2$) and multiplied by 1,440 minutes per day (min/d) to convert volumetric output to a flux velocity in inches per day (in/d). The seepage meter was lowered slowly through holes cut in the lake ice until the bottom edge of the seepage meter was about 4 in. below the water-sediment interface. Three underwater cameras mounted to the seepage meter facilitated depth of installation and ensured that the meters did not move during deployment. The meter was connected to a braided steel cable and suspended from the ice surface. Output from the flowmeter was scanned at 5-second intervals, and 1-minute, mean values were stored by a digital datalogger. Because flow was expected to be slow, and it was suspected that the lake-ice surface might move slightly, measurements were averaged during more than 12 hours at each measurement site. Sediments commonly are disturbed and can be compressed during meter installation, thereby resulting in collection of erroneous data immediately after meter installation (Rosenberry and LaBaugh, 2008). Time-series plots were generated and early deployment data were discarded until output was deemed to be at a steady state; equilibration generally took 2–3 hours.

The electromagnetic seepage meter was calibrated and tested in the laboratory before and after field measurements, and zero-flow offset values were determined at the beginning and end of each deployment. Flowmeter output was adjusted to zero-flow values in the field by suspending the seepage meter above the water-sediment interface for about 15 minutes before and immediately after insertion into the sediment; however, because flow was slow and the hypolimnetic water was cold, heat generated by the electromagnet warmed the water inside the flowmeter and created upward flow through the seepage meter because of buoyancy of the warmed water. In addition to determining a value while the meter was suspended above the sediment bed, a rubber stopper was placed in the flowmeter orifice to stop flow while sensor output was recorded to determine actual zero-flow values. Field-determined offsets were applied to field data to compensate for zero-flow sensor drift and the buoyancy effect. The buoyancy effect was further studied in the laboratory, resulting in a second, larger offset; therefore, two mean seepage values are presented in the "Field Study of Groundwater and Surface-Water Exchanges, Mini-Piezometer and Seepage Meter" section, one based on the field-determined buoyancy offset and the other based on the laboratory-determined buoyancy offset.

# Statistical Analysis of Lake Levels

The statistical analyses for the short-term (2002–10) and long-term (1925–2014) periods were completed to help understand changes in lake levels across the northeast Twin Cities Metropolitan Area (fig. 1). Short-term (2002–10) lake-level analyses indicated that multiple physical variables can explain lake-level changes that occurred in northeast Twin Cities Metropolitan Area lakes. Different long-term (1925–2014) lake-level responses occurred in 14 lakes to precipitation, with lake levels in White Bear Lake having the most variability over the period.

## Short-Term Statistical Analysis—Relations Between Lake-Level Changes and Environmental Variables for 2002–10

During 1999–2014, most of the 96 lakes with complete records within the study area (fig. 5) experienced extended periods of lake-level declines (figs. 8, 9). Of the 96 lakes, water levels declined in 88 lakes during 2002–10; this period will be the focus of the statistical analyses presented in this section of the report. Lake-level changes from 2002 to 2010 ranged from 0.77 to -9.90 ft (field LK_chnge02.10, appendix table 1–1). The lakes with largest declines were generally in a region extending from White Bear Lake northeast to Goose Lake (fig. 5).

The years 2002–10 represent a period of declining water storage in the northeast Twin Cities Metropolitan Area. The mean precipitation in Ramsey County in 2002 was 42.2 in. (Minnesota Climatology Working Group, 2016d), which is well above the 1981–2010 mean of about 30.6 inches per year (in/yr) (National Climatic Data Center, 2016). This precipitation caused most lakes in the study area to have higher levels in 2002 compared to 1999 (fig. 8) and caused increases in groundwater levels in many observation wells (Jones and others, 2013). Potential groundwater recharge in the study area was higher in 2002 compared to 2003–10 (Smith and Westenbroek, 2015). After 2002, annual precipitation was lower, so the water stored in the system slowly declined after 2002.

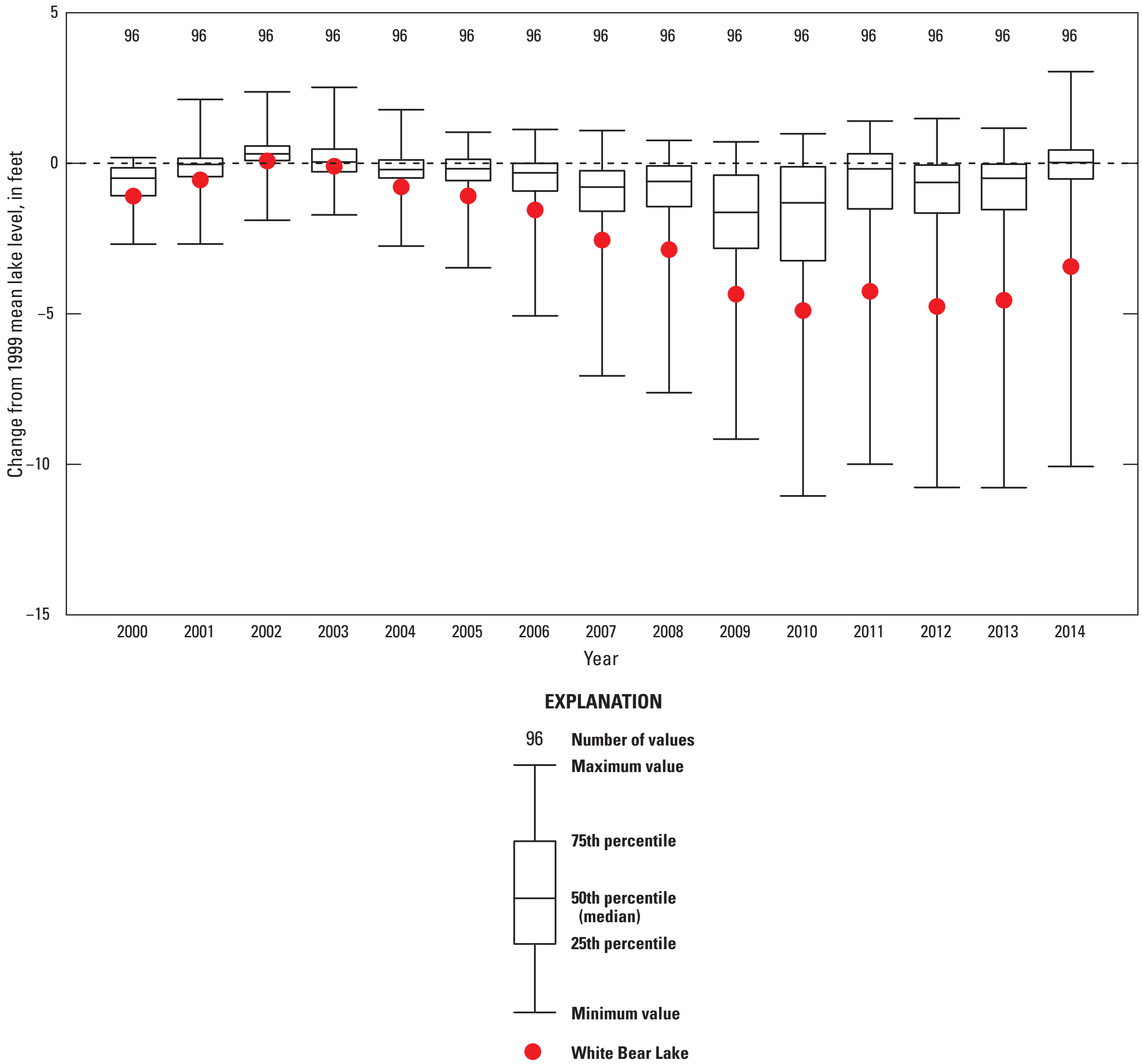



**Figure 8.** Change from 1999 mean lake level for 96 lakes for each year during 2000–14, northeast Twin Cities Metropolitan Area, Minnesota.

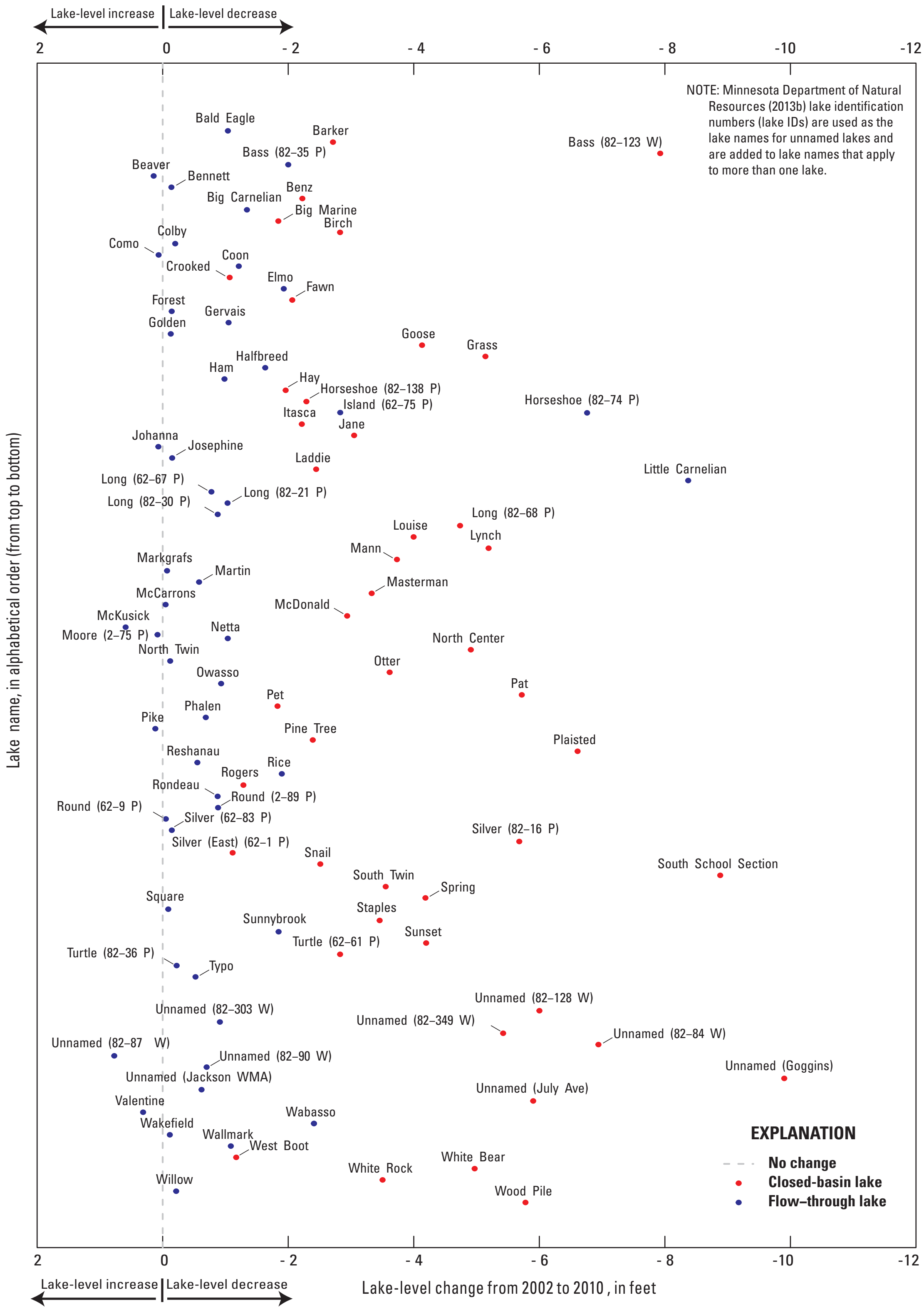



**Figure 9.** Lake-level changes from 2002 to 2010 in closed basin and flow-through lakes, northeast Twin Cities Metropolitan Area, Minnesota.

Results from two sets of statistical evaluations of lake-level change from 2002 to 2010 are described in this section: (1) Spearman's ρ correlations among lake-level change and landscape, geologic, and climatic variables; and (2) ANOVA on groups of lakes. Spearman's ρ correlations are summarized in table 12. Lakes were classified according to several categorical variables to explore how lake-level changes from 2002 to 2010 differed between groups. The ANOVA results of the comparisons are given in table 13. Results from the statistical analyses of lake levels are described relative to (1) watershed size and lake outflow, (2) groundwater and landscape position, (3) geology, (4) precipitation and evaporation, (5) land cover, and (6) groundwater withdrawals.

## Watershed Size and Lake Outflow

Annual mean lake levels changed little from 2002 to 2010 if the watershed of a lake was large and connected enough to support regular outflow from the lake. Grouping lakes as either closed-basin (no active outlet) or flow-through (with an active outlet) lakes (fig. 10) was the most significant classification of lakes according to ANOVA analyses (table 13). Flow-through lakes (lakes with an active outlet) generally had small or slightly positive changes in lake level, whereas closed-basin lakes without an active outlet typically had large negative changes in lake level (table 13; figs. 10*A*, *B*, *C*) from 2002 to 2010. Flow-through lakes with an active outlet generally had larger watersheds than closed-basin lakes (fig. 11*A*). The median watershed area for flow-through lakes was 2,763 acres compared to only 816 acres for closed-basin lakes.

Lake-level changes from 2002 to 2010 in closed-basin lakes without a regular surface-water outflow were more negative (representing a decline in lake level) with increasing watershed area (fig. 11*A*; variable WS_LkshedArea.ac, table 12). An even stronger negative correlation exists between lake-level change and the watershed-to-lake area ratio for closed-basin lakes (variable WStoLKArea, table 12). This relation may be negative because the surface-water storage capacity (such as isolated wetlands) within the lake watersheds and the actual watershed areas to lakes may have varied from 2002 to 2010.

The assumption of a constant watershed area throughout 2002–10 is probably reasonable for flow-through lakes with consistent outflow but is probably not reasonable for closed-basin lakes in watersheds with substantial wetland cover. Many of the lake watersheds within the northeast Twin Cities Metropolitan Area (fig. 1) contain isolated wetlands (fig. 2), which store water and may prevent water from reaching the lake for which the watershed was delineated. The degree of connectivity between these wetlands and the lake changes with variations in the water level of the wetlands and lake. Generally, as water levels decline, the likelihood for wetlands to become isolated from the lake of interest increases. In flow-through lakes, the percentage of wetland cover within the lake watershed (variable WS_SUM_WETLND, table 12) in the statistical analysis was negatively correlated with lake-level changes. In flow-through and closed-basin lakes separately, the wetland-to-lake-area ratio (variable WETtoLK_Area, table 12) was negatively correlated with lake-level changes. These negative correlations indicate that as wetland area increases, lake-level changes tended to be more negative, and water in wetlands was not consistently routed to the lakes of interest. As water levels declined after 2002, connections between parts of these watersheds may have ceased, and contributing areas to closed-basin lakes may have become much smaller than what is represented in appendix table 1–1 and figure 11*A*.

Plaisted, North School Section, and South School Section Lakes (fig. 1) are examples of lakes with varying watershed areas during 2002–10. Light detection and ranging data collected in 2011 indicate a clear drainage path from Plaisted Lake to North School Section Lake with a high point of about 967 ft (Minnesota Department of Natural Resources and Minnesota Geospatial Information Office, 2011). Aerial imagery from 2003 confirms that all three of these lakes, Plaisted, North School Section, and South School Section Lakes, became one lake at high water levels (U.S. Department of Agriculture Farm Services Agency Aerial Photography Field Office, 2003). At high water levels, when the lakes are all connected, the total watershed area for South School Section Lake listed in appendix table 1–1 (1,995 acres) is probably reasonable; however, at low water levels, aerial imagery (U.S. Department of Agriculture Farm Services Agency Aerial Photography Field Office, 2010) indicates that North and South School Section Lakes are typically separated by land with no obvious drainage features or culverts (Jim Solstad, Hydrologist, Minnesota Department of Natural Resources, written commun., August 23, 2016). During periods of low water levels, a more suitable watershed area for South School Section Lake would be its direct catchment area of 287 acres, or a nearly 10-fold decrease in watershed area (Jim Solstad, Hydrologist, Minnesota Department of Natural Resources, written commun., August 23, 2016). Uncertainty in high water levels and watershed area with time in small lakes is acknowledged in the metadata for the Minnesota Department of Natural Resources Watershed Suite (Minnesota Department of Natural Resources, 2013b).

## Groundwater and Landscape Position

Lake-level changes in closed-basin lakes were of similar magnitude to groundwater-level changes in the northeast Twin Cities Metropolitan Area (fig. 1) from 2002 to 2010. The lakes and wells with the largest declines were generally in a region extending from White Bear Lake northeast to Goose Lake (figs. 5, 12). This region contains the highest land-surface elevations in the study area (fig. 12) and a regional groundwater divide (fig. 10 of Jones and others, 2013). Potentiometric surfaces of the water-table aquifer for the region indicate that groundwater from the divide flows east and southeast to the St. Croix River and flows west and southwest towards the Mississippi River (fig. 1) (Jones and others, 2013).

**Table 12.** Correlations of continuous variables with the change in annual mean lake level from 2002 to 2010 for 96 lakes, northeast Twin Cities Metropolitan Area, Minnesota.

[*n*, number of lakes; ρ, Spearman's correlation coefficient; Sig, statistical significance level; NLCD, National Land Cover Database; <, less than; ****, less than 0.0001; ***, between 0.0001 and 0.001; **, between 0.001 and 0.01; *, between 0.01 and 0.05; +, positive correlation with lake-level change; NS, not significant; -, negative correlation with lake-level change]

| Variable name | Variable description | All lakes (*n*=96) | | | | Closed basin (*n*=45) | | | | Flow through (*n*=51) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | *ρ* | *p*-value | Sig | Sign | *ρ* | *p*-value | Sig | Sign | *ρ* | *p*-value | Sig | Sign |
| WS_SUM_DEV | Percentage of developed land cover within lake watershed (sum of NLCD classes 21, 22, 23, and 24) | 0.50 | <0.0001 | **** | + | 0.26 | 0.0843 | NS | NS | 0.52 | 0.0001 | *** | + |
| BUF_AvgGWPump_m | Mean groundwater withdrawals from 2002 to 2010 within 2-mile buffer of lake edge | 0.34 | 0.0008 | *** | + | 0.09 | 0.5416 | NS | NS | 0.33 | 0.0196 | * | + |
| LK_StCroix | Straight-line distance from lake to the St. Croix River | 0.33 | 0.0013 | ** | + | 0.29 | 0.0575 | NS | NS | 0.18 | 0.1965 | NS | NS |
| LK_DPTHBDKAVM | Mean depth to bedrock underneath the lake footprint | 0.26 | 0.0114 | * | + | 0.09 | 0.5754 | NS | NS | -0.00 | 0.9836 | NS | NS |
| WS_AvgGWPump_m | Mean groundwater withdrawals from 2002 to 2010 within lake watershed | 0.25 | 0.0140 | * | + | -0.02 | 0.9004 | NS | NS | 0.23 | 0.1029 | NS | NS |
| WS_SUM_AGRICU | Percentage of agricultural land cover within lake watershed (sum of NLCD classes 81 and 82) | -0.54 | <0.0001 | **** | - | -0.40 | 0.0062 | ** | - | -0.46 | 0.0007 | *** | - |
| LK_meanELEV | Mean 1999–2014 lake level minus 771.49 feet (the minimum mean of all lakes) | -0.49 | <0.0001 | **** | - | -0.31 | 0.0410 | * | - | -0.28 | 0.0503 | NS | NS |
| WS_SUM_SHRUBGRASS | Percentage of shrub and grass land cover within lake watershed (sum of NLCD classes 52 and 71) | -0.45 | <0.0001 | **** | - | -0.41 | 0.0048 | ** | - | -0.44 | 0.0012 | ** | - |
| LK_Mississippi | Straight-line distance from lake to the Mississippi River | -0.37 | 0.0003 | *** | - | -0.05 | 0.7511 | NS | NS | -0.46 | 0.0007 | *** | - |
| WS_pct_sand | Area and depth-weighted mean percentage of sand in watershed soils | -0.34 | 0.0008 | *** | - | -0.10 | 0.5279 | NS | NS | -0.30 | 0.0333 | * | - |
| WS_Soil_c | Percentage of watershed area classified as hydrologic soil group C | -0.30 | 0.0028 | ** | - | -0.39 | 0.0081 | ** | - | -0.10 | 0.4975 | NS | NS |
| WS_pct_clay | Area and depth-weighted mean percentage of clay in watershed soils | -0.28 | 0.0055 | ** | - | -0.38 | 0.0098 | ** | - | -0.24 | 0.0943 | NS | NS |
| WS_Soil_b | Percentage of watershed area classified as hydrologic soil group B | -0.26 | 0.0103 | * | - | -0.37 | 0.0117 | * | - | -0.12 | 0.4179 | NS | NS |
| WS_SUM_FOREST | Percentage of forest land cover within lake watershed (sum of NLCD classes 41, 42, 43) | -0.23 | 0.0243 | * | - | -0.01 | 0.9708 | NS | NS | -0.45 | 0.0009 | *** | - |
| LK_Rum | Straight-line distance from lake to the Rum River | -0.22 | 0.0325 | * | - | -0.33 | 0.0280 | * | - | -0.07 | 0.6243 | NS | NS |
| LK_pminuse_avg02.10 | Mean of the 2002 to 2010 annual precipitation minus evaporation totals at lake centroid | 0.19 | 0.0571 | NS | NS | 0.17 | 0.2744 | NS | NS | 0.24 | 0.0955 | NS | NS |
| WS_mean_RCH_02.10 | Mean recharge within lake watershed for 2002 to 2010 | 0.18 | 0.0876 | NS | NS | 0.40 | 0.0070 | ** | + | 0.07 | 0.6367 | NS | NS |
| LK_prcp_avg02.10 | Mean of the 2002 to 2010 annual precipitation totals at lake centroid | 0.16 | 0.1182 | NS | NS | 0.15 | 0.3391 | NS | NS | 0.20 | 0.1560 | NS | NS |
| WS_LkshedArea.ac | Area of lake watershed | 0.14 | 0.1645 | NS | NS | -0.31 | 0.0395 | * | - | -0.25 | 0.0771 | NS | NS |
| LK_ACRES | Lake surface area | 0.09 | 0.4069 | NS | NS | 0.21 | 0.1646 | NS | NS | -0.32 | 0.0240 | * | - |
| WStoLKArea | Watershed-to-lake area ratio | 0.08 | 0.4097 | NS | NS | -0.48 | 0.0011 | ** | - | -0.00 | 0.9806 | NS | NS |
| WS_org_matter | Area and depth-weighted mean percentage of organic matter in watershed soils | 0.04 | 0.6998 | NS | NS | 0.30 | 0.0429 | * | + | -0.36 | 0.0094 | ** | - |
| LK_PERIM_F | Length of lake periemeter | 0.00 | 0.9932 | NS | NS | 0.10 | 0.4955 | NS | NS | -0.43 | 0.0018 | ** | - |
| WS_K_Sat | Area and depth-weighted mean saturated hydraulic conductivity of soils | -0.19 | 0.0712 | NS | NS | 0.21 | 0.1622 | NS | NS | -0.35 | 0.0111 | * | - |
| LK_NearestMajRiv | Minimum of the straight-line distances from a lake to the Mississippi, St. Croix, and Rum Rivers | -0.17 | 0.1020 | NS | NS | -0.09 | 0.5353 | NS | NS | -0.31 | 0.0266 | * | - |
| WS_PCT_11 | Percentage of land use 11 (open water) within lake watershed | -0.17 | 0.1044 | NS | NS | 0.38 | 0.0105 | * | + | -0.39 | 0.0055 | ** | - |
| WS_pct_silt | Area and depth-weighted mean percent silt in watershed soils | -0.16 | 0.1270 | NS | NS | -0.31 | 0.0356 | * | - | -0.26 | 0.0642 | NS | NS |
| WS_Soil_a | Percentage of watershed area classified as hydrologic soil group A | -0.13 | 0.1936 | NS | NS | 0.41 | 0.0055 | ** | + | -0.42 | 0.0020 | ** | - |
| WS_SUM_WETLND | Percentage of wetland cover within lake watershed (sum of NLCD classes 90 and 95) | -0.12 | 0.2273 | NS | NS | 0.17 | 0.2516 | NS | NS | -0.45 | 0.0010 | ** | - |
| LK_evap_avg02.10 | Mean of the 2002 to 2010 annual evaporation totals at lake centroid | -0.11 | 0.2710 | NS | NS | 0.39 | 0.0091 | ** | + | -0.37 | 0.0079 | ** | - |
| LK_MaxLkDepth.f | Maximum lake depth | -0.06 | 0.5499 | NS | NS | -0.11 | 0.4871 | NS | NS | -0.16 | 0.2707 | NS | NS |
| WETtoLKArea | Wetland-to-lake area ratio | -0.08 | 0.4267 | NS | NS | -0.34 | 0.0240 | * | - | -0.37 | 0.0068 | ** | - |
| WS_Slope_pct | Mean percent slope in lake watershed | -0.04 | 0.7258 | NS | NS | -0.26 | 0.0790 | NS | NS | 0.11 | 0.4415 | NS | NS |

**Table 13.** Analysis of variance results for the change in annual mean lake level from 2002 to 2010 for 96 lakes grouped by different variables, northeast Twin Cities Metropolitan Area, Minnesota.

[Variables are defined in appendix table 1–1. Sig, statistical significance level; NS, not significant; <, less than; ****, less than 0.0001; ***, between 0.0001 and 0.001; **, between 0.001 and 0.01; *, between 0.01 and 0.05]

| Variable[1] | Group degrees of freedom | Residuals degrees of freedom | Group sum of squares | Residuals sum of squares | Group mean squared error | Residuals mean squared error | *F* statistic | *p*-value | Sig |
|---|---|---|---|---|---|---|---|---|---|
| LK_InletControl | 1 | 94 | 12.25 | 516.85 | 12.25 | 5.50 | 2.23 | 0.1389 | NS |
| LK_OutletControl | 1 | 94 | 146.97 | 382.13 | 146.97 | 4.07 | 36.15 | <0.0001 | **** |
| LK_BEDROCKVAL | 1 | 94 | 26.75 | 502.35 | 26.75 | 5.34 | 5.01 | 0.0276 | * |
| LK_INLET | 1 | 94 | 53.56 | 475.54 | 53.56 | 5.06 | 10.59 | 0.0016 | ** |
| LK_OUTLET | 1 | 94 | 220.46 | 308.64 | 220.46 | 3.28 | 67.14 | <0.0001 | **** |
| LK_DOMBEDROCK | 6 | 89 | 61.96 | 467.13 | 10.33 | 5.25 | 1.97 | 0.0787 | NS |
| LK_WB_TYPE | 1 | 94 | 35.83 | 493.27 | 35.83 | 5.25 | 6.83 | 0.0104 | * |
| LK_GeoUnit | 1 | 94 | 74.03 | 455.07 | 74.03 | 4.84 | 15.29 | 0.0002 | *** |
| LK_DepFunc | 1 | 94 | 9.25 | 519.85 | 9.25 | 5.53 | 1.67 | 0.1990 | NS |

[1]LK_InletControl, flag that indicates the presence of an inlet control on a lake; LK_OutletControl, flag that indicates the presence of an outlet control on a lake; LK_BEDROCKVAL, flag that indicates the presence of a bedrock valley underlying the lake; LK_INLET, flag that indicates the presence of a surface-water inlet to a lake; LK_OUTLET, flag that indicates the presence of a surface-water outlet from a lake; LK_DOMBEDROCK, uppermost bedrock unit with the largest spatial extent under a lake; LK_WB_TYPE, Minnesota Department of Natural Resources' classification of the water body type; LK_GeoUnit, lobe of most of the geologic units bordering a lake; LK_DepFunc, textural description of most of the geologic units bordering a lake.

Changes in groundwater levels and closed-basin lake levels from 2002 to 2010 were negatively correlated with mean water-level elevation (figs. 11*B*, 11*C*; table 12), which indicates that relative water-level position affects closed-basin lake-level. Groundwater levels (excluding two outliers—227033 and 227032 in fig. 11*B*) and closed-basin lake levels had significant negative correlations with their respective mean water-level elevations (groundwater—Spearman's $\rho$=-0.55, *p*-value=0.03; closed-basin lakes—Spearman's $\rho$=-0.31, *p*-value=0.04). Two lakes, Horseshoe and Little Carnelian Lakes, were extreme outliers in that they had large negative changes in lake level but also had large watersheds and outflow for most of the study period (fig. 11*A*). These two outliers are described in the "Major Anomalies" section of this report. Water balances in closed-basin lakes in the glacial terrain generally are controlled by the exchange of water with the atmosphere (precipitation and evaporation) and groundwater (Winter and others, 1998). Previous studies have demonstrated that closed-basin lake levels reflect groundwater levels of adjacent aquifers (Watras and others, 2013; fig. 8 of Jones and others, 2013), and groundwater fluxes into and out of a lake in a low landscape position are more stable than fluxes in lakes farther upgradient (Cheng and Anderson, 1994). More stable groundwater fluxes at low elevations may explain greater stability of closed-basin lake levels at low elevations compared to high elevations because changes in closed-basin lake levels tend to be similar to changes in local groundwater levels.

## Geology

Lakes bordered primarily by Superior Lobe deposits (sand and till) had significantly more negative lake-level changes (mean change of -3.3 ft) than lakes bordered by Des Moines Lobe deposits (mean change of -1.5 ft) (variable LK_GeoUnit, p-value=0.0002 in table 13; figs. 10*E*, 13). Of the 96 lakes, 51 were in Des Moines Lobe deposits and 45 were in Superior Lobe deposits (fig. 10*E*). Differences in several parameters, including lake connectivity, landscape position, soil composition, and degree of urban development, can explain this lake-level change and geology relation. Lakes generally were less connected to other surface-water features in the Superior Lobe deposits compared to Des Moines Lobe deposits. A larger proportion of lakes included in this analysis on the Superior Lobe deposits were closed-basin (26 of 45 lakes) compared to Des Moines Lobe deposits (19 of 51 lakes). Lakes in the Superior Lobe deposits were generally at higher elevations compared to lakes in Des Moines Lobe deposits (figs. 12, 13). Lake-level changes were more negative at higher elevations compared to lower elevations in the northeast Twin Cities Metropolitan Area, particularly for closed-basin lakes (fig. 11*B*).

Texture and hydrologic characteristics of watershed soils were correlated with lake-level changes in closed-basin lakes. Closed-basin lake-level changes were negatively correlated with the percent clay content (variable WS_pct_clay), the percent silt

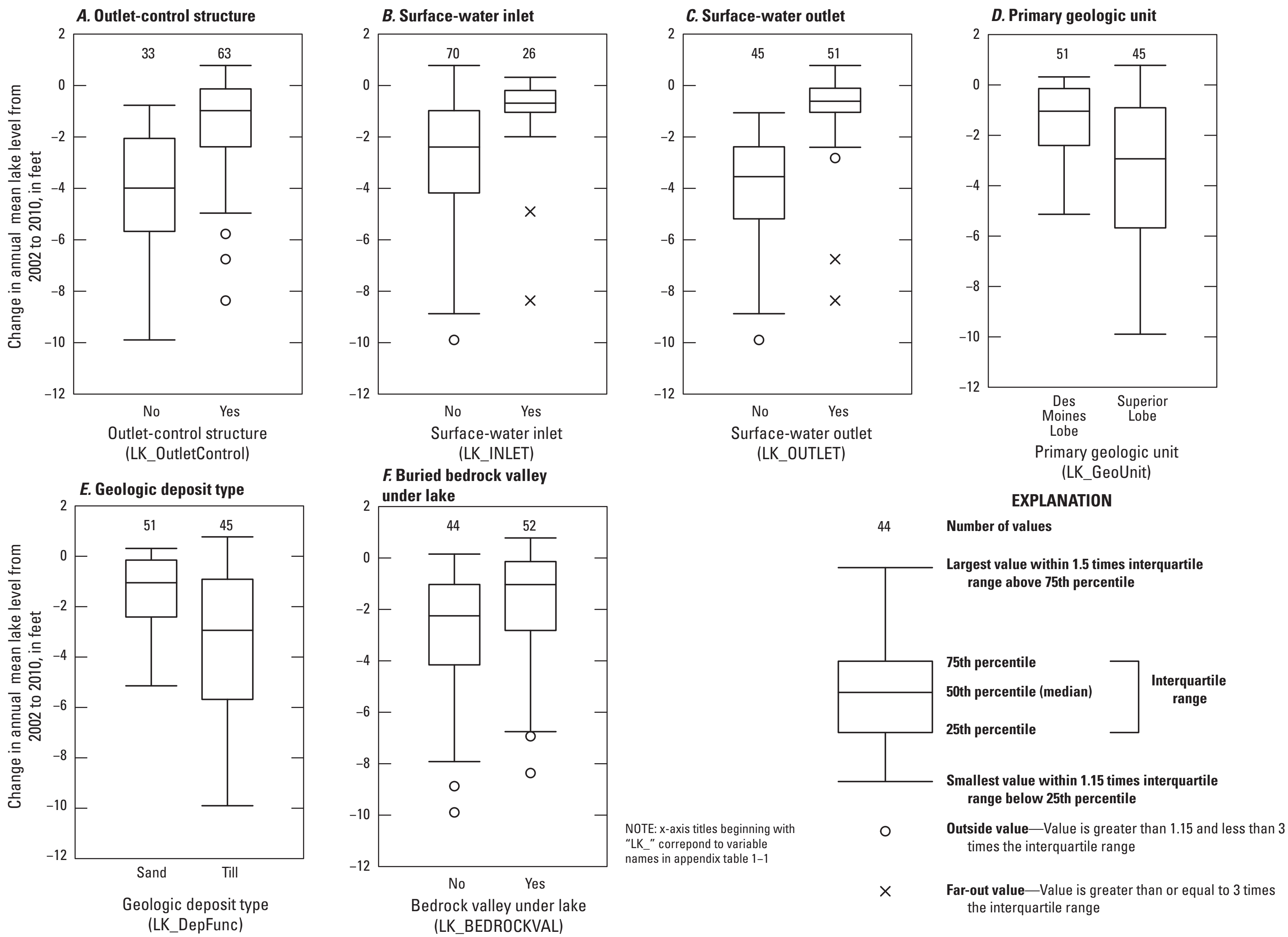



**Figure 10.** Change in annual mean lake levels from 2002 to 2010 for 96 lakes by different variables, northeast Twin Cities Metropolitan Area, Minnesota.

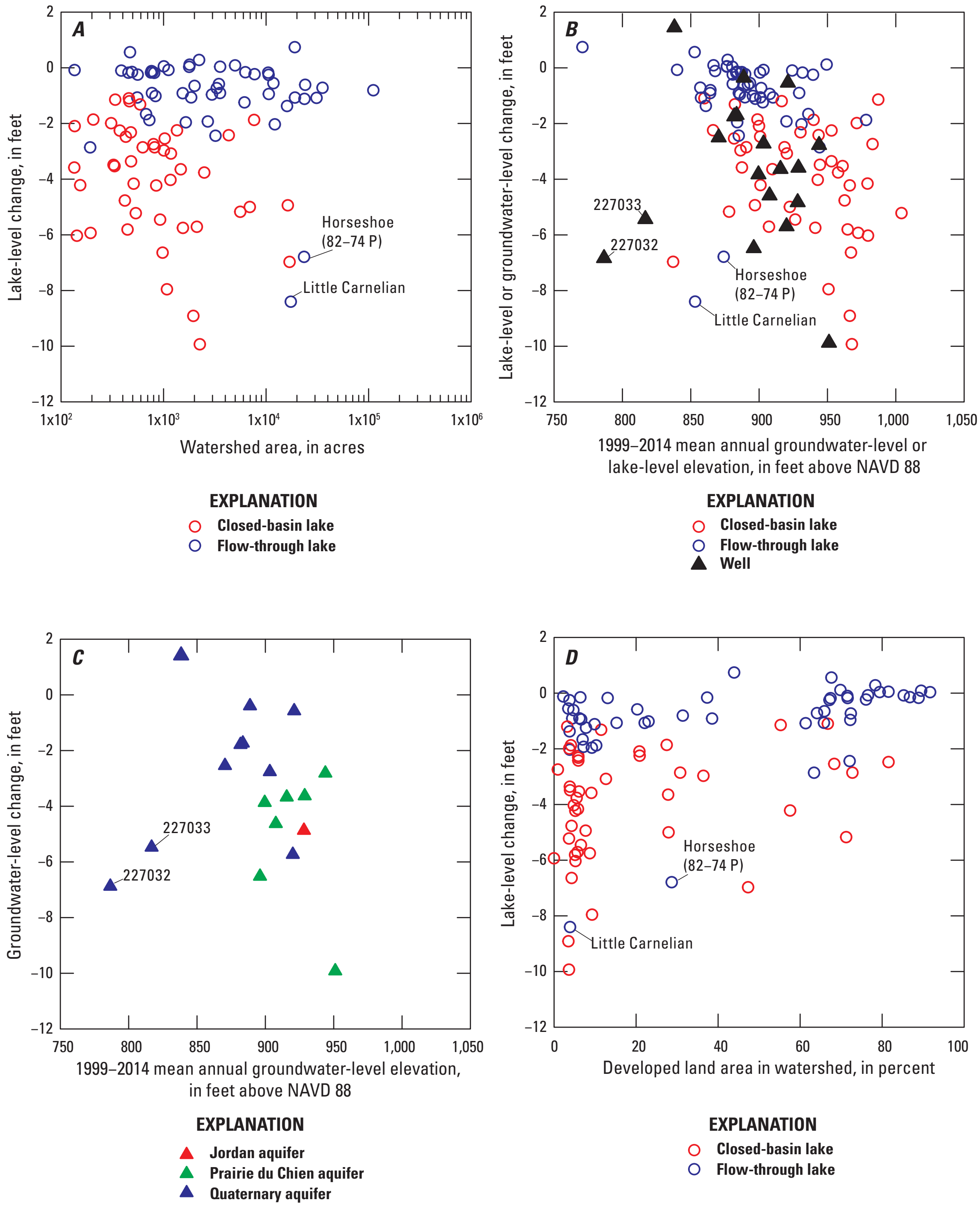



**Figure 11.** Relations between changes in annual mean lake levels and groundwater levels from 2002 to 2010 and selected variables, northeast Twin Cities Metropolitan Area, Minnesota. *A*, annual mean lake-level change compared to watershed area; *B*, annual mean lake-level change and groundwater-level change compared to mean annual lake-level elevation or groundwater-level elevation; *C*, annual mean groundwater-level change compared to mean annual groundwater-level elevation; *D*, annual mean lake-level change compared to percent developed land area in watershed.

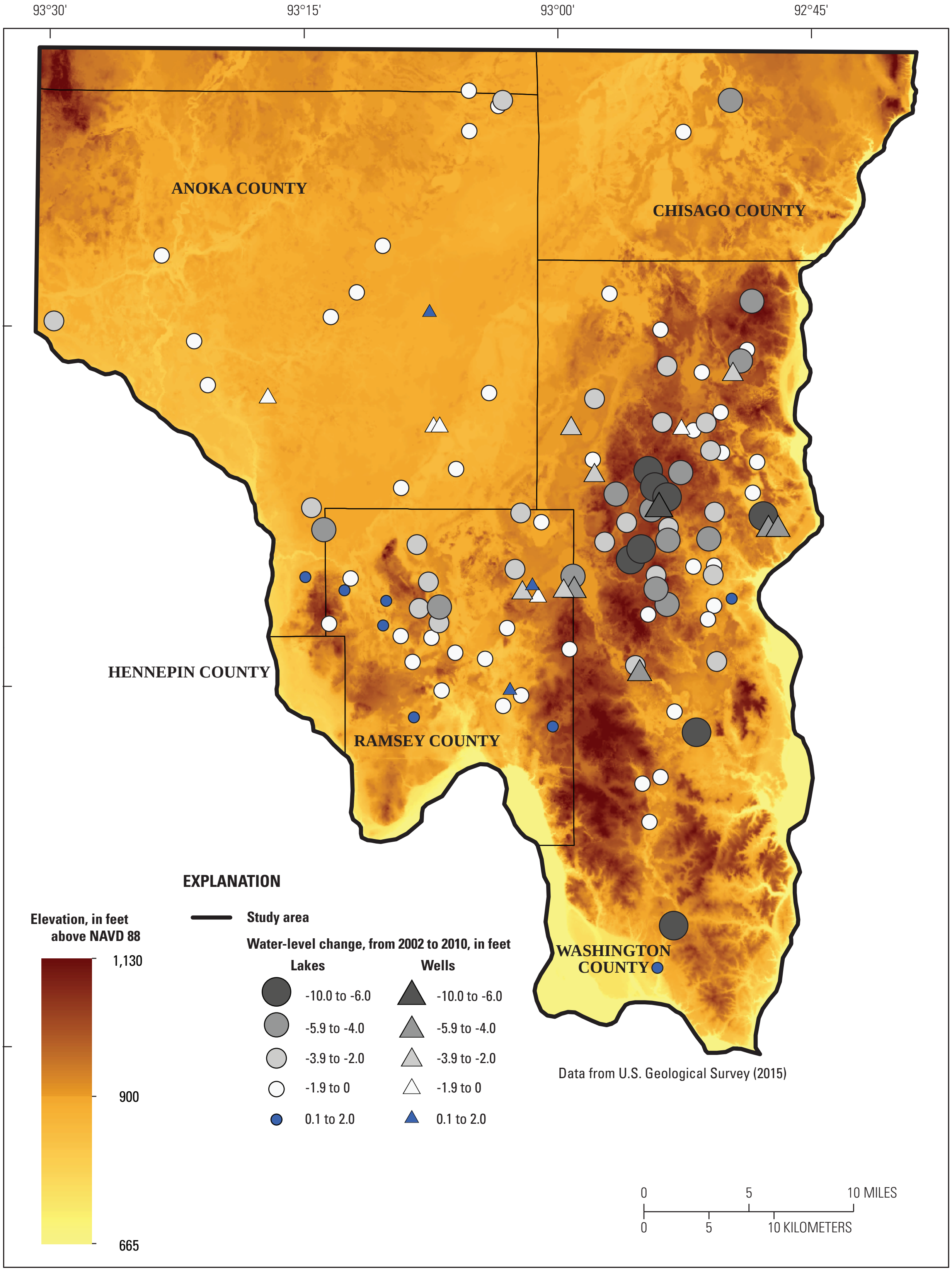



**Figure 12.** Land-surface elevation and annual mean water-level change from 2002 to 2010 in lakes and wells, northeast Twin Cities Metropolitan Area, Minnesota.

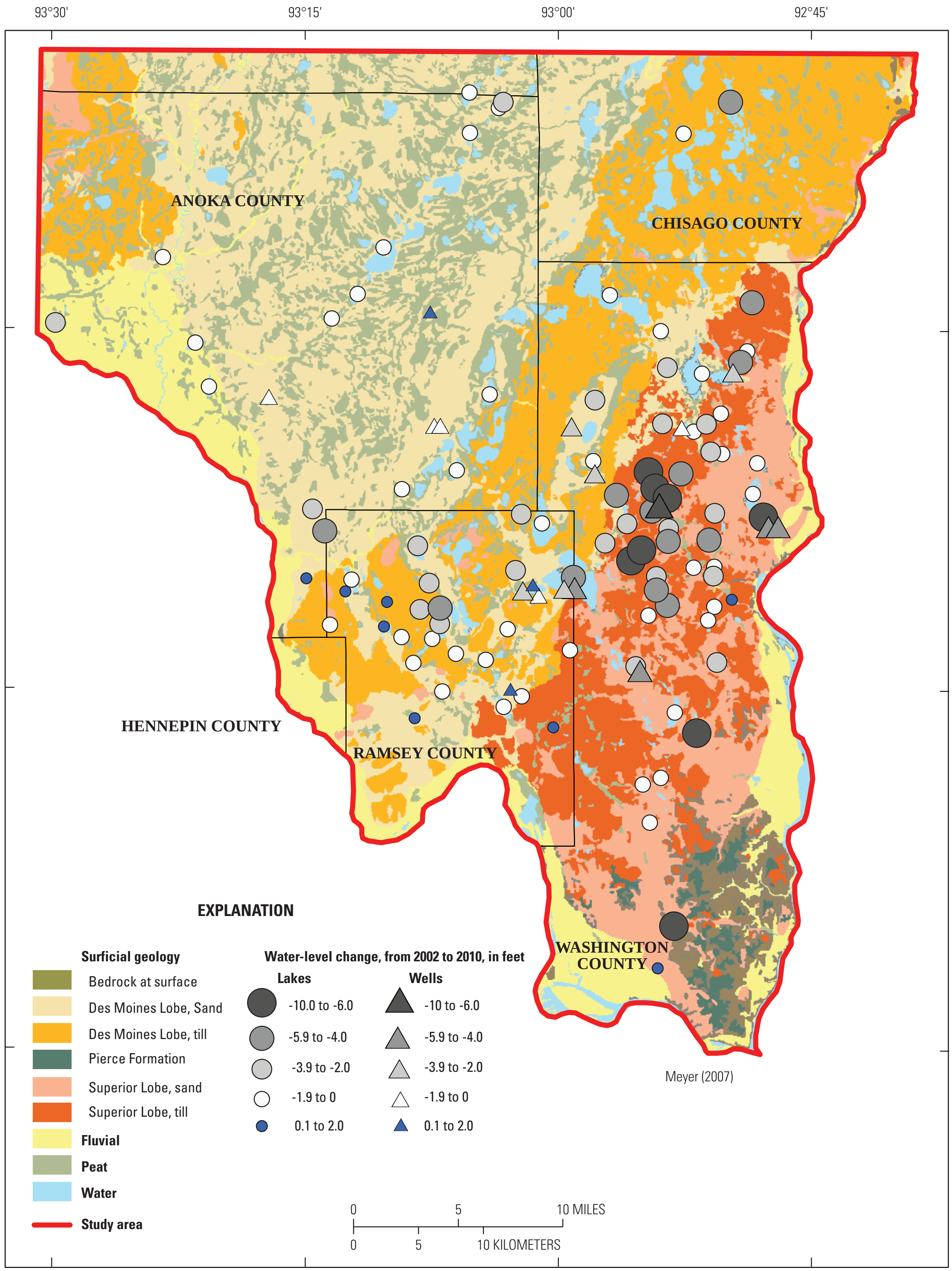



**Figure 13.** Surficial geology and annual mean water-level change from 2002 to 2010 in lakes and wells, northeast Twin Cities Metropolitan Area, Minnesota.

content (variable WS_pct_silt), and the percentage of watershed soils in hydrologic soil groups B (variable WS_Soil_b) and C (variable WS_Soil_c) (table 12). Closed-basin lake-level changes were positively correlated with the percentage of watershed soils in hydrologic soil group A (variable WS_Soil_a). Soils classified as hydrologic soil group A have higher infiltration capacities than group B and C soils. Although not statistically significant, lake-level changes in all lakes bordered by till deposits were slightly more negative than lake-level changes in all lakes bordered by sandy deposits (fig. 10*E*). Specific yield is defined as the ratio of the volume of water that a saturated rock or soil will yield to the total volume of the rock or soil (Johnson, 1967). Clay and silty soils have low specific yields (mean of 6 percent) compared to soils with more sand (mean of 16 percent) (Morris and Johnson, 1967). A larger water-level change in lakes that have watersheds with clayey and silty sediments generally is necessary to produce the same volume of water to that of lakes that have watersheds with sandy sediments because of differences in soil specific yields. Using the means listed above for a specific volume of soil, it would take a water-level decline of 2.7 ft for lakes that have watersheds with clayey and silty sediments to produce the same volume of water from a 1-ft decline in lakes that have watersheds with sandy sediments. If two lakes were losing similar volumes of water to evaporation, the lake that has a watershed with higher silt and clay content soils would experience larger water-level declines because the volume of water released per unit depth is smaller than lakes that have watersheds with sandy soils.

Lake levels changed slightly less over buried bedrock valleys compared to lakes not overlying a buried bedrock valley (fig. 10*F*). The depth of most buried bedrock valleys in the study area ranged from 200 to 500 ft (Meyer and Swanson, 1992; Bauer, 2016). In the southeast part of the study area, depth to bedrock in the buried bedrock valleys is shallower and ranges from 130 to 250 ft (Bauer, 2016). Mean lake-level elevation covaries with this classification grouping. In total, 75 percent of the lakes overlying a buried bedrock valley had mean lake levels below 900 ft (NAVD 88), whereas most of the lakes (75 percent) that did not overlie a buried bedrock valley had mean lake-level elevations above 900 ft (NAVD 88).

Lakes were classified as having the Prairie du Chien Group, Tunnel City Group, Eau Claire Formation, Wonewoc Sandstone, St. Lawrence Formation, St. Peter Sandstone, or the Jordan Sandstone as the dominant uppermost bedrock unit underlying the lake extent. The dominant uppermost bedrock unit under the lakes (LK_DOMBEDROCK) was not a significant factor for explaining lake-level declines (table 13).

## Precipitation, Evaporation, and Groundwater Recharge

Spatial variability in mean annual precipitation (variable LK_prcp_avg02.10) across the northeast Twin Cities Metropolitan Area (fig. 1) was not correlated with the magnitude of lake-level changes from 2002 to 2010 for either flow-through lakes or closed-basin lakes (table 12). This lack of correlation indicates that spatial variability in precipitation during 2002–10 was less important for explaining the magnitude of long-term lake-level changes than other factors that affect how water moves into or across the landscape including watershed size, soil hydrologic characteristics, and elevation.

Lake evaporation was positively correlated with lake-level changes in closed-basin lakes and negatively correlated with lake-level changes in flow-through lakes (variable LK_evap_avg02.10, table 12). The positive relation with closed-basin lakes seems contrary to the expectation that lake levels would decrease more with greater evaporative loss; however, the range of the mean annual evaporation estimates (about 1.2 in.) was small compared to the overall magnitude of evaporation (28.3 to 29.5 in., variable LK_evap_avg02.10, appendix table 1–1). The full range of evaporation estimates is only about 4 percent of the overall mean, well within the error of the Hargreaves-Samani method used for estimating lake evaporation.

Closed-basin lake levels also were more stable with higher rates of potential groundwater recharge (variable WS_mean_RCH_02.10, table 12). The lack of a strong correlation between precipitation and closed-basin lake-level change (table 12) indicates that the ability of precipitation to infiltrate to groundwater was the underlying mechanism for the correlation between recharge and lake-level changes in closed-basin lakes. This corroborates the relation of closed-basin lake levels varying less with increasing percentages of hydrologic soil group A.

## Land Cover

Between 2002 and 2010, lakes in watersheds with large percentages of developed land use tended to have small increases to modest decreases in lake level, whereas lakes in more rural settings tended to have large lake-level declines (table 12; figs. 11*D*, 14). The strongest correlations were between lake-level change and percent areas of some land-cover classes (table 12). Considering all lakes together, the variable that had the most significant negative correlation (Spearman’s $\rho$=-0.54) with the lake-level change from 2002 to 2010 was the percentage of agriculture in the watershed (variable WS_SUM_AGRICU; table 12). Conversely, the variable that had the most significant positive correlation (Spearman’s $\rho$=0.50) with lake-level change was the percentage of developed area in the watershed (variable WS_SUM_DEV; table 12; fig. 11*D*). Flow-through lake-level changes had a strong positive correlation with the percentage of developed area (Spearman’s $\rho$=0.52), whereas closed-basin lake-level changes had a nonsignificant, weak positive correlation (Spearman’s $\rho$=0.26) with the percentage of developed area (table 12). Flow-through lake-level changes were significantly and negatively correlated with the percentages of agricultural (variable WS_SUM_AGRICU), shrub and grass (variable WS_SUM_SHRUBGRASS), forest

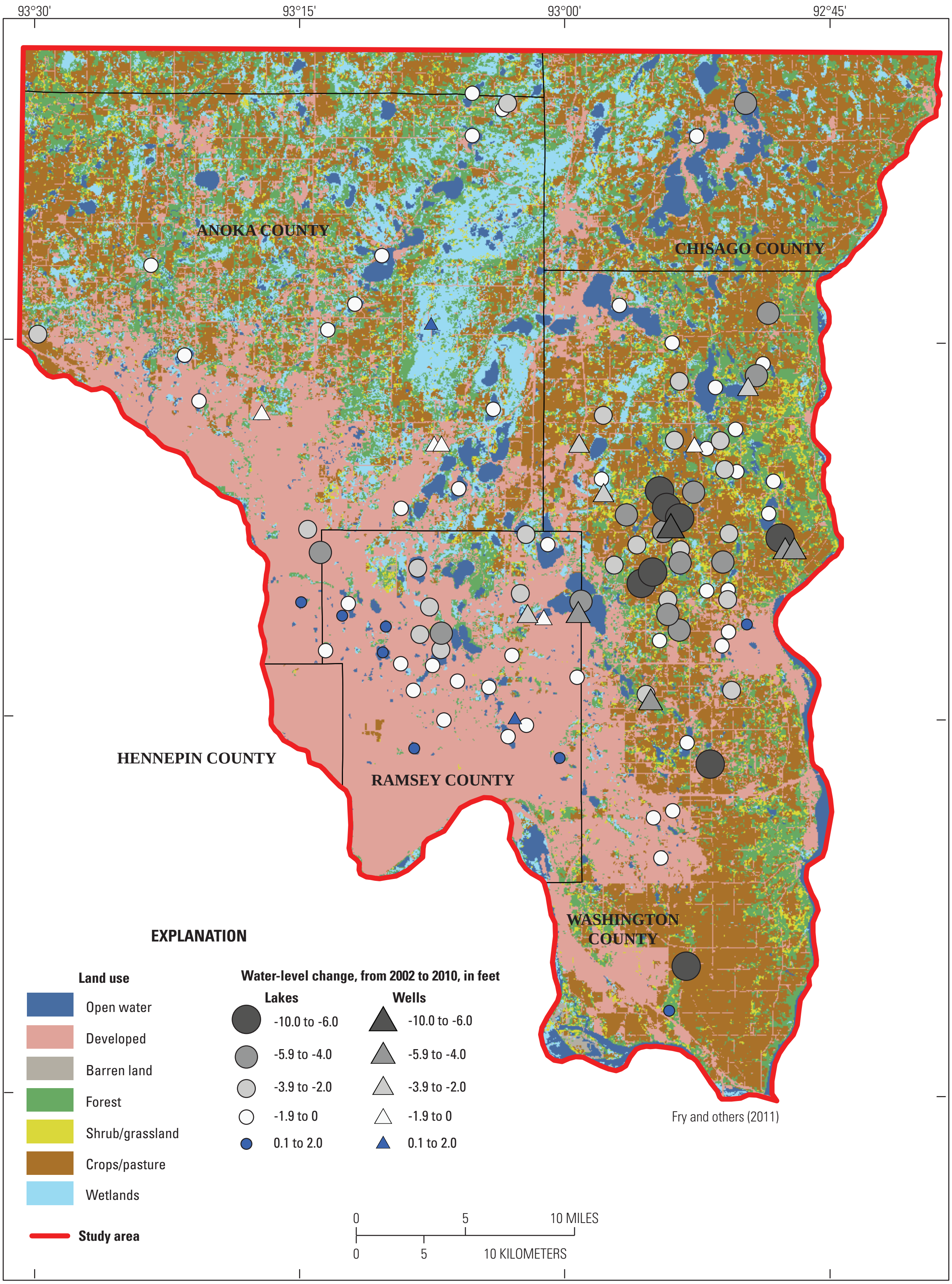



**Figure 14.** 2006 land cover and annual mean water-level change from 2002 to 2010 in lakes and wells, northeast Twin Cities Metropolitan Area, Minnesota.

(variable WS_SUM_FOREST), wetland (variable WS_SUM_WETLND), and total open water (variable WS_PCT_11) cover in watersheds (table 12). Closed-basin lake-level changes were significantly and negatively correlated with the percentage of agricultural (variable WS_SUM_AGRICU) and shrub and grass (variable WS_SUM_SHRUBGRASS) cover in watersheds (table 12).

The hydrologic changes introduced in urbanized areas that serve to manage and route water may explain the relative stability of lake levels in urban areas. In the northeast Twin Cities Metropolitan Area, surface-water control structures used to manage lake levels are more common in urbanized settings compared to rural settings. For this report, urbanized watersheds are defined as those that have at least 50 percent developed land cover (appendix table 1–1, variable WS_SUM_DEV), and rural watersheds have less than 50 percent developed land cover. Urbanized watersheds of the northeast Twin Cities Metropolitan Area, compared to rural watersheds, had (1) a greater proportion of lakes with active outflows and (2) more frequent occurrence of lake outflow-control structures. Most urban lakes (24 of 31) were flow-through lakes with active outlets, whereas most rural lakes (38 of 65) were closed-basin lakes. Only 52 percent of the lakes in rural watersheds had some sort of outflow-control structure in place compared to 94 percent in urbanized watersheds.

Outlet-control structures and inlets significantly reduced lake-level change from 2002 to 2010 (fig. 10*A*, *B*, *C*; table 13). Most of the flow-through lakes had some kind of outflow-control structure, such as a weir, culvert, or stop-log structure, to manage lake levels. The presence of an outflow-control structure significantly reduced lake-level change (F-statistic=36.15, table 13), though it was not as significant as just having an active outlet (F-statistic=67.14, table 13). Many of the lake-level control structures, such as culverts and weirs, used in the northeast Twin Cities Metropolitan Area function to keep water levels below a certain flood stage and maintain natural flow conditions (Minnesota Office of the Revisor of Statutes, 2016b). Lakes with a surface-water inlet had less variable lake levels than lakes with no surface-water inlet (table 13; fig. 10*C*).

Urbanized lakes in the northeast Twin Cities Metropolitan Area tended to be in landscape positions of lower elevations (figs. 12, 14), which may also contribute to the relative stability of urbanized lakes. This stabilization of lake levels is of particular interest for closed-basin lakes. Levels in urban closed-basin lakes generally changed less (mean of -2.8 ft) compared to rural closed-basin lakes (mean of -4.2 ft).

## Groundwater Withdrawals

Urbanized watersheds had higher rates of groundwater withdrawals and more stable lake levels compared to rural watersheds from 2002 to 2010. Mean annual groundwater use (variable WS_AvgGWPump_m, appendix table 1–1) within lake watersheds from 2002 to 2010 was positively correlated with the percentage of the developed watershed (Spearman's $\rho$=0.52, *p*-value less than 0.0001). Other studies have determined that groundwater withdrawals can deplete surface-water resources (Barlow and Leake, 2012), but groundwater withdrawals do not seem to be the most significant factor in explaining lake-level changes across the northeast Twin Cities Metropolitan Area (fig. 1). Closed-basin lake-level changes were not correlated with mean groundwater withdrawals within their watershed (variable WS_AvgGWPump_m) or within a 2-mi buffer around the lakes (variable BUF_AvgGWPump_m) (table 12). A moderate positive correlation exists between flow-through lake-level changes and groundwater withdrawals within a 2-mi buffer of lakes (table 12). Other hydrologic factors associated with lakes in urbanized settings, such as connections to other water bodies, stormflow routing, and outflow-control structures, may mask effects from groundwater withdrawals. This hypothesis is supported by the stability of lake levels in urbanized settings described in the "Land Cover" section.

The statistical analysis in this report identified different variables important for lake-level change than the temporal analysis done for White Bear Lake by Jones and others (2013). Jones and others (2013) presented a regression model, which indicated that recent (2002–10) lake-level declines in White Bear Lake could be explained by annual groundwater withdrawals, precipitation, and evaporation; however, the analysis in this report, which considers a broader spatial extent than Jones and others (2013), indicates that physiographic factors are more strongly correlated with lake-level changes than groundwater withdrawals within 2 mi of lakes. Lake levels declined at some lakes with little to no permitted groundwater withdrawals within 2 mi of their shore.

The statistical analysis in this report addresses a different set of questions at a much broader scale than Jones and others (2013) and is not well-suited to draw detailed conclusions about the effects of groundwater withdrawals on specific lakes. Jones and others (2013) addressed factors varying through time near White Bear Lake that could explain the lake-level decline from 2003 to 2010. The present analysis addresses factors varying spatially that could explain lake-level changes across the northeast Twin Cities Metropolitan Area. The present analysis points out sensitive lakes and some common characteristics that they share. A detailed water-budget analysis for each of the 96 lakes was beyond the scope of this report. A groundwater-flow model could examine the effects of groundwater withdrawals on lakes in the northeast Twin Cities Metropolitan Area.

## Major Anomalies

Two lakes (Horseshoe and Little Carnelian Lakes; fig. 1) did not fit the general conceptual understanding developed through the statistical analyses. Both lakes are classified as flow-through lakes at low elevations but still had large lake-level declines after 2006 (figs. 9, 15). Both lakes have large watersheds (Horseshoe Lake, 24,000 acres; Little Carnelian, 17,800 acres), but it is unclear how frequently the lakes were receiving surface-water inputs from 2002 to 2010. In the statistical analysis, watershed area was assumed to be constant for all lakes throughout 2002–10. Horseshoe and Little Carnelian Lakes are on sandy Cromwell Formation outwash, a glacial deposit that contains lakes with substantial interaction with groundwater (Alexander and others, 2001; Jones and others, 2013).

The abrupt lake-level declines that happened in Horseshoe and Little Carnelian Lakes from 2006 to 2010 (fig. 15) indicate that the lakes may have changed from flow-through lakes in 2002–5 to closed-basin lakes in 2006–10. As water levels declined, surface-water connections between parts of watersheds could have become disconnected, isolating the lakes from other surface-water bodies. As a result, Little Carnelian and Horseshoe Lakes may have had vastly different watershed areas for 2002–6 compared to 2006–10. Variability in lake levels has been observed in Little Carnelian Lake since the installation of the fixed-head weir outlet at Big Carnelian Lake and a surface-water channel connected to Little Carnelian Lake (Jim Shaver, Carnelian-Marine-St. Croix Watershed District , written commun., November 17, 2015). The watershed delineations for both lakes used for this analysis assumed all catchments were always connected and contributing to the lakes (that is, there are no isolated or noncontributing catchments) (Minnesota Department of Natural Resources, 2013b). These abrupt changes in lake levels highlight the nature of some lakes and the thresholds after which lake-level behavior changes.

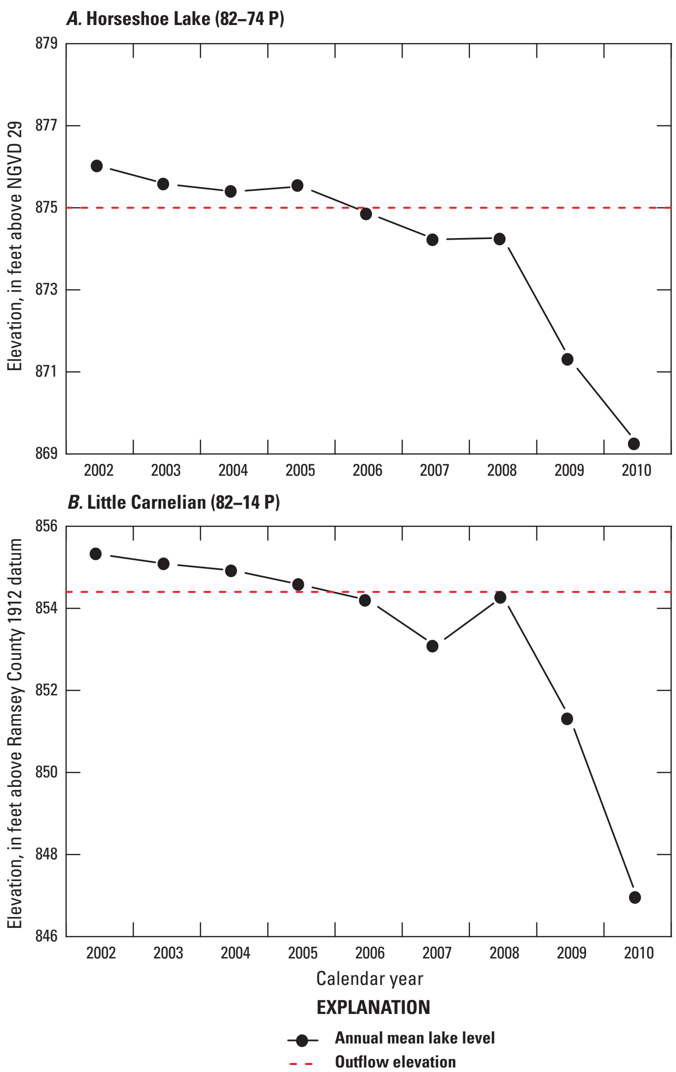



**Figure 15.** Annual mean lake levels and outflow elevations for 2002–10, northeast Twin Cities Metropolitan Area, Minnesota. *A*, Horseshoe Lake (82–74); *B*, Little Carnelian Lake.

Groundwater levels substantially declined in the groundwater system around Little Carnelian Lake despite potentiometric surfaces that were low compared to the rest of the study area. Two observation wells (MNDNR observation wells 227032 and 227033; fig. 5) 235 yards from Little Carnelian Lake were outliers in the relation between mean groundwater-level elevation and groundwater-level change from 2002 to 2010 (figs. 11*B*, *C*). Both wells are open to buried Quaternary aquifers. Little Carnelian Lake declined by 8.4 ft from 2002 to 2010 (fig. 15) and had a mean water level of 853.9 ft (NAVD 88). Well 227033 declined by 5.4 ft and had a mean water level of 817.5 ft (NAVD 88) (fig. 11*B*). The well is screened 126 to 130 ft below land surface. Well 227032 declined by 6.8 ft and had a mean water level of 787.3 ft (NAVD 88) (fig. 11*B*). The well is screened from 249 to 252 ft below land surface. These mean water levels indicate a substantial downward gradient between Little Carnelian Lake and buried Quaternary aquifers.

Generally, water levels in Little Carnelian Lake are not well-correlated to the groundwater levels in these two wells, though all three experienced near-record or record lows during 2010–14. The poor correlation between groundwater levels in the two wells and lake levels in Little Carnelian Lake are, in part, due to different response times to climatic events. The well hydrographs demonstrate a substantial lag in response to major climatic events; for example, there was a substantial drought in the northeast Twin Cities Metropolitan Area during 1988 and 1989 (Minnesota Department of Natural Resources, 2016d). The groundwater levels in both wells reached their lowest point in response to this event in 1991 (Minnesota Department of Natural Resources, 2016d). The year 2002 was wet, and Little Carnelian Lake water levels responded that year and then declined in subsequent years. The wells did not peak from this wet year until late 2003 and early 2004.

## Summary of Short-Term Statistical Analysis

The years 2002–10 were a period of declining water storage in the northeast Twin Cities Metropolitan Area (fig. 1), as indicated by declining water levels in most lakes (fig. 9) and observation wells (fig. 11) that met record completeness criteria. Relations between lake-level changes from 2002 to 2010 and many climatic, landscape, and geologic characteristics of lakes and their watersheds were examined to determine if any of these characteristics could explain the variability in lake-level changes. Two statistical approaches were used to explore these relations: nonparametric correlations (Spearman's ρ) and ANOVA.

A general conceptual model of lake-level changes across the northeast Twin Cities Metropolitan Area is evident from these statistical comparisons and is shown in figure 16. Annual mean lake levels were generally stable for lakes with large, well-connected watersheds that supported consistent lake outflow (fig. 16), such as Bald Eagle Lake. Closed-basin lake levels were less stable than flow-through lake levels (fig. 16), with lake levels changing more at high elevations (for example, South School Section Lake) compared to low elevations (for example, Crooked Lake). Similarly, groundwater levels declined more at high elevations compared to low elevations in the Quaternary, Prairie du Chien, and Jordan aquifers. Lakes at high elevations were typically in rural settings on Superior Lobe deposits (fig. 16). Closed-basin lake levels in watersheds with low clay and silt content and high infiltration capacity were more stable than lake levels in watersheds with high clay and silt contents. Flow-through and closed-basin lake levels were more stable in developed compared to rural settings (fig. 16). More flow-through lakes and outlet-control structures were present in developed compared to rural settings. The distribution of lake-level changes in the northeast Twin Cities Metropolitan Area was more strongly correlated with landscape variables than with local mean annual precipitation. Closed-basin lakes and groundwater levels in the region extending from White Bear Lake northeast to Goose Lake had the greatest annual mean water-level declines from 2002 to 2010 (figs. 5, 14).

## Long-Term Statistical Analysis—Relations Between Lake-Level Changes and Precipitation for 1925–2014

When reviewing results from the long-term statistical analysis, it should be noted that lake-level management practices for each of the 14 lakes (fig. 5) have changed through time, but these changes are not evaluated in detail in this report. Every lake has its story, and each of these lakes has a unique water-level management history; for example, several northeast Twin Cities Metropolitan Area lakes (fig. 1) were augmented from other surface-water and groundwater sources. White Bear Lake was augmented with groundwater from the 1920s through 1978 (Minnesota Department of Natural Resources, 1998). Snail Lake was augmented with groundwater until the late 1980s and augmented with Mississippi River water from 1993 to 2007 and again from 2009 through the present (2016) (City of Shoreview, 2016). Turtle Lake was augmented from 1923 through 1989 from local groundwater pumping and augmentation from the St. Paul Water Utility (Maloney, 2014). For this discussion, closed-basin or flow-through status of each lake assigned for 2002–10 was assumed to apply to each lake for 1925–2014. White Bear Lake had the most readily available historical outlet and water-level information, which confirms that it was a closed-basin lake with no outflow for 80 percent of the years from 1925 to 2014. Lake levels in White Bear Lake were above the outlet elevation for about 5 years in the 1940s, 2 years in the 1950s, 3 years in the 1980s, 5 years in the 1990s, and 2 years in the 2000s (fig. 2*A* of Jones and others, 2013).

Despite similar precipitation, the lakes have exhibited very different water-level responses to precipitation from 1925

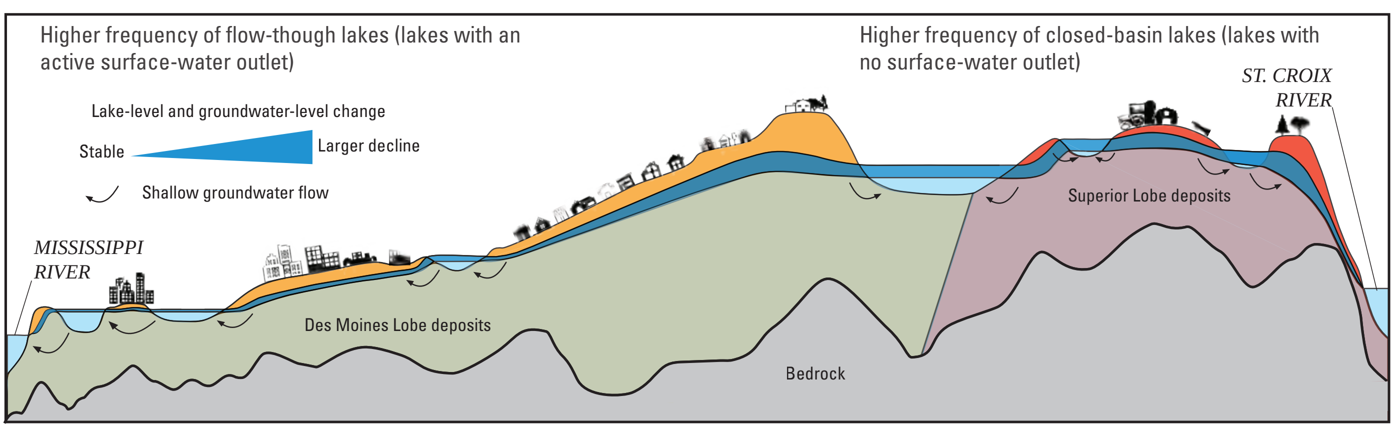



**Figure 16.** Conceptual diagram of land cover, geology, and water-level changes in the northeast Twin Cities Metropolitan Area, Minnesota.

to 2014; for example, Valentine Lake and White Bear Lake represent two very different lake-level change histories from 1925 to 2014 (fig. 17). Lake levels in Valentine Lake (fig. 5) have had a strong upward trend since 1925, similar to the long-term trend in the precipitation (fig. 17). Lake levels in White Bear Lake, a closed-basin lake, do not have the same trend as Valentine Lake (fig. 17). Lake levels in White Bear Lake were much more variable (commonly setting the upper and lower range of annual lake-level anomalies) compared to levels in the other 13 lakes in this analysis (fig. 17). Since 2000, lake levels in Valentine Lake generally have been above the long-term mean, whereas lake levels in White Bear Lake have oscillated from about 1.5 ft above to nearly 4 ft below the long-term mean of the lake (fig. 17). From 2005 to 2014, White Bear Lake had the most negative anomaly from its long-term mean compared to the rest of the lakes (fig 17).

Precipitation and most flow-through lake levels were more variable (higher CV) during 1943–78 as compared to 1979–2014 (table 14). For the first 35 years in the analysis (1943–78), the CV of annual total precipitation was 21.1, and the CV was substantially less (15.2) for the second 35 years in the analysis (1979–2014) (table 14). This decrease in the precipitation variability corresponded to decreases in the annual mean lake-level variability for flow-through lakes. Of the 14 lakes in the long-term analysis, 10 were classified as flow-through lakes. Of these lakes, eight had corresponding decreases in their CV, meaning the mean annual lake levels were less variable when annual total precipitation was less variable. Two of the flow-through lakes (Lake Josephine and Valentine Lake) had only very minor increases in CV, but their CVs were among the lowest of all the lakes.

The four closed-basin lakes (Silver [East], Snail, Turtle, and White Bear) had substantial changes in CV in the opposite direction; their water levels were more variable when precipitation was less variable. Detrended precipitation and lake-level data showed the same pattern between lake types (data not shown). Closed-basin lake levels increased in variability and flow-through lake levels decreased in variability when precipitation variability decreased. Temporal patterns in precipitation, lake augmentation with other water sources, or both are possible explanations for the contrasting behaviors of flow-through and closed-basin lakes.

The variability represented in the CV of annual precipitation totals does not capture the evenness of precipitation through time. The same precipitation CV would result if the 15 years with lowest precipitation were either all clustered together or dispersed equally across the 35-year period; however, the 5-year moving average would have very different patterns under these two scenarios. Despite more variable precipitation during 1943–78, the dry years and wet years seem to be more evenly dispersed because the 5-year moving average does not show multiyear periods of precipitation increases or decreases, with the exception of 1943–50. After 1950, the 5-year moving average generally oscillates up and down annually (fig. 17). In contrast, the period 1979–2014 has two distinct multiyear oscillations: 1988–98 and 2004–14. In these two periods, the 5-year moving average declines for several years and then increases for several years (fig. 17). Closed-basin lakes are primarily dependent on precipitation and groundwater inflow for water sources, and these two periods of multiyear oscillations are apparent in groundwater and White Bear Lake levels (Jones and others, 2013).

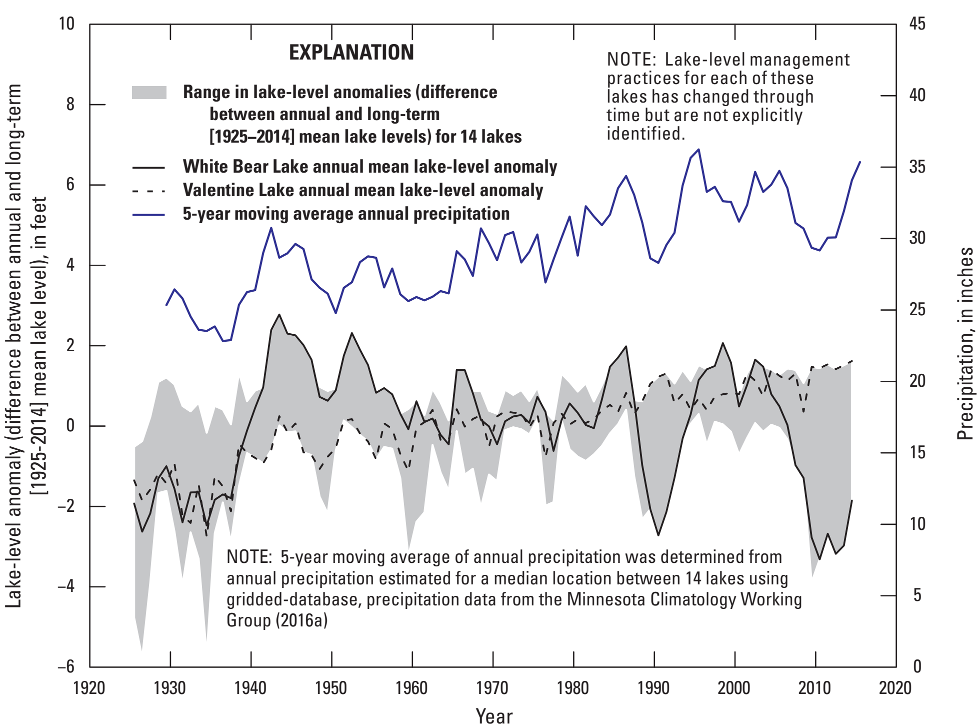



**Figure 17.** The 5-year moving average of annual precipitation and annual lake-level anomalies for 14 lakes, 1925–2014, northeast Twin Cities Metropolitan Area, Minnesota.

**Table 14.** Coefficient of variation in annual mean lake levels and precipitation during two 35-year periods, 1943–78 and 1979–2014, northeast Twin Cities Metropolitan Area, Minnesota.

[ID, identification; NA, not applicable; -, negative change; +, positive change]

| Precipitation or lake name[1] | Lake ID[1] | Lake type[2] | Period 1 (1943–78) | Period 2 (1979–2014) | Direction of change from period 1 to period 2 |
|---|---|---|---|---|---|
| | | | Coefficent of variation, in percent | | |
| Precipitation | NA | NA | 21.1 | 15.2 | - |
| Bald Eagle (62–2 P) | 62000200 | Flow through | 0.058 | 0.040 | - |
| Gervais (62–7 P) | 62000700 | Flow through | 0.066 | 0.044 | - |
| Johanna (62–78 P) | 62007800 | Flow through | 0.085 | 0.065 | - |
| Josephine (62–57 P) | 62005700 | Flow through | 0.033 | 0.041 | + |
| Long (62–67 P) | 62006700 | Flow through | 0.053 | 0.042 | - |
| McCarrons (62–54 P) | 62005400 | Flow through | 0.039 | 0.023 | - |
| Otter (2–3 P) | 02000300 | Flow through | 0.118 | 0.103 | - |
| Owasso (62–56 P) | 62005600 | Flow through | 0.051 | 0.048 | - |
| Phalen (62–13 P) | 62001300 | Flow through | 0.149 | 0.078 | - |
| Silver (East) (62–1 P) | 62000100 | Closed basin | 0.040 | 0.061 | + |
| Snail (62–73 P) | 62007300 | Closed basin | 0.056 | 0.122 | + |
| Turtle (62–61 P) | 62006100 | Closed basin | 0.044 | 0.091 | + |
| Valentine (62–71 P) | 62007100 | Flow through | 0.051 | 0.054 | + |
| White Bear (82–167 P) | 82016700 | Closed basin | 0.096 | 0.181 | + |

[1]Minnesota Department of Natural Resources (2013b).

[2]Flow through is synonymous with the variable LK_OUTLET = 1, closed basin is synonymous with LK_OUTLET = 0 in appendix table 1–1.

The recent increase in variability in closed-basin lakes also may result from less-frequent augmentation of closed-basin lakes after 1978. The variability in closed-basin lake levels may have been artificially reduced from 1943 to 1978 by augmentation and perhaps less reflective of precipitation patterns as compared to 1979–2014. The increased variability in lake levels from 1979 to 2014 could have resulted from lakes returning to a more natural and less regulated response to precipitation; however, many other lake-level management practices and hydrologic alterations on the landscape in 1943–2014 may have affected lake levels. It is difficult, therefore, to single out a specific augmentation effect on the lake levels.

The explanation for the recent increase in closed-basin lake-level variability likely is a combination of climatic and anthropogenic factors. The 5-year moving average of precipitation had more multiyear increases and decreases after 1979 than from 1943 to 1978 (fig. 17), and these patterns were reflected in closed-basin lake levels. These patterns indicate that closed-basin lake levels are very sensitive to changes in local climate under the present hydrologic conditions of water use and water routing in the northeast Twin Cities Metropolitan Area.

Considering the entire 1925–2014 period for all 14 lakes included in this analysis, lake levels responded to precipitation at different time scales. Annual changes in mean lake levels were positively correlated with within-year precipitation (no lag) for all the lakes; correlation coefficients ranged from 0.33 to 0.58 (table 15). This correlation indicates that more precipitation results in a more positive lake-level change; for example, the total precipitation that fell in 1981 is correlated to the change in mean annual lake level from 1980 to 1981. The positive correlation indicates that if more precipitation had fallen in 1981, then the lake-level change would have been more positive. For all lakes, regardless of flow-through or closed-basin status, this is the case for the within-year precipitation (no lag).

Lake levels in some lakes correlated with precipitation from the previous year (1-year lagged precipitation). Lake levels in some lakes had positive correlations (values greater than 0.20), and levels in some lakes had negative correlations (values less than -0.20) with 1-year lagged precipitation (table 15). White Bear Lake is a unique lake in this regard because it had the most positive correlation with precipitation from the previous year (1-year lagged precipitation; table 15). This correlation indicates that after a wet year, the lake-level change was positive (lake levels rose) for the year of the precipitation and the following year. This correlation also means that a dry year could lead to 2 consecutive years of reduced lake-level change (lake-level declines).

**Table 15.** Correlations between change in annual mean lake levels for 14 lakes and total annual precipitation at different time lags for 1925–2014, northeast Twin Cities Metropolitan Area, Minnesota.

[ID, identification]

| Lake name[1] | Lake ID[1] | Lake type[2] | Lag 0 years | Lag 1 year | Lag 2 years | Lag 3 years | Lag 4 years | Lag 5 years |
|---|---|---|---|---|---|---|---|---|
| | | | Correlation coefficient | | | | | |
| Bald Eagle (62–2 P) | 62000200 | Flow through | 0.44 | 0.02 | -0.26 | -0.01 | -0.10 | -0.07 |
| Gervais (62–7 P) | 62000700 | Flow through | 0.41 | -0.22 | -0.15 | 0.06 | -0.14 | -0.03 |
| Johanna (62–78 P) | 62007800 | Flow through | 0.43 | -0.18 | -0.22 | 0.12 | -0.01 | -0.03 |
| Josephine (62–57 P) | 62005700 | Flow through | 0.52 | -0.19 | -0.31 | 0.09 | -0.06 | -0.00 |
| Long (62–67 P) | 62006700 | Flow through | 0.58 | -0.33 | -0.20 | 0.12 | -0.09 | -0.00 |
| McCarrons (62–54 P) | 62005400 | Flow through | 0.48 | -0.22 | -0.06 | 0.03 | -0.10 | 0.07 |
| Otter (2–3 P) | 02000300 | Flow through | 0.48 | 0.11 | -0.28 | -0.06 | -0.11 | -0.08 |
| Owasso (62–56 P) | 62005600 | Flow through | 0.43 | -0.09 | -0.31 | 0.07 | -0.09 | -0.05 |
| Phalen (62–13 P) | 62001300 | Flow through | 0.38 | -0.12 | -0.27 | 0.08 | -0.00 | -0.07 |
| Silver (East) (62–1 P) | 62000100 | Closed basin | 0.33 | -0.14 | -0.26 | 0.00 | -0.09 | -0.03 |
| Snail (62–73 P) | 62007300 | Closed basin | 0.34 | 0.03 | -0.10 | 0.05 | -0.21 | -0.00 |
| Turtle (62–61 P) | 62006100 | Closed basin | 0.40 | 0.05 | -0.14 | -0.07 | -0.10 | -0.14 |
| Valentine (62–71 P) | 62007100 | Flow through | 0.38 | -0.34 | -0.02 | 0.07 | -0.14 | 0.19 |
| White Bear (82–167 P) | 82016700 | Closed basin | 0.43 | 0.25 | -0.22 | -0.06 | -0.13 | -0.19 |

[1]Minnesota Department of Natural Resources (2013b).

[2]Flow through is synonymous with the variable LK_OUTLET = 1, closed basin is synonymous with LK_OUTLET = 0 in appendix table 1–1.

Nine lakes (Bald Eagle, Johanna, Josephine, Long, Otter, Owasso, Phalen, Silver, White Bear) also had negative correlations of values less than or equal to -0.2 with 2-year lagged precipitation (table 15). Lake-level and lagged precipitation data (not shown) indicated that the negative correlations were highly influenced by extremely dry years (1930s and 1988–1989), years with less than about 24 in. of annual precipitation. These correlations indicate that typically 2 years after an extremely dry year, lake-levels increased. In almost all cases, correlation values were not greater than 0.20 or less than -0.20 after the 2-year lag.

In summary, lake levels respond to annual precipitation, and those responses differ among lakes and through time. A comparison of two 35-year periods during 1925–2014 revealed that variability of annual mean lake levels in flow-through lakes increased when annual precipitation totals were more variable, whereas variability of annual mean lake levels in closed-basin lakes had the opposite pattern, being more variable when annual precipitation totals were less variable.In contrast, closed-basin lakes were substantially less variable with increased precipitation variability. The closed-basin response could reflect the temporal patterns in precipitation, changes in lake-level augmentation practices, and (or) other hydrologic alterations from urban development that affect lake levels. Multiyear dry and wet cycles affected closed-basin lake levels much more than flow-through lake levels. Lagged correlations between mean annual lake levels and annual precipitation revealed that lake levels respond to precipitation at different time scales. All lake levels responded positively to within-year precipitation, and White Bear Lake was unique in having a strong positive correlation with the 1-year lagged precipitation.

# Field Study of Groundwater and Surface-Water Exchanges

Water-quality data, water-level differences, and seepage-flux rates indicated that surface water is flowing to lower aquifers, particularly south of White Bear Lake. Groundwater and surface-water exchanges for White Bear Lake occurred in shallow and deep waters, with groundwater entering the lake nearshore and lake water leaving the lake at the water-sediment interface in deep waters.

## Water Quality of Precipitation, Lakes, and Groundwater

Stable isotope analyses of water samples from precipitation, lakes, piezometers, and wells were used to determine groundwater and lake-water exchanges and surface-water contributions to wells in the northeast Twin Cities Metropolitan Area (fig. 1). Monthly water samples were collected from four northeast Twin Cities Metropolitan Area lakes to determine variations in oxygen-18/oxygen-16 and deuterium/

protium ratios in the lakes. Field water-quality properties were collected with water-quality samples at various depths in White Bear Lake to estimate variations in stable isotope ratios with depth and across the thermocline in the lake. Ages of recharge were estimated for 9 of the 40 well-water samples using CFC–11, CFC–12, and CFC–113 concentrations and the dispersion LPM.

## Precipitation

Similar to bulk precipitation samples collected at White Bear Lake (fig. 1) by Jones and others (2013), all the oxygen-18/oxygen-16 and deuterium/protium ratios for the bulk precipitation samples plot close to a meteoric waterline determined by Landon and others (2000) for precipitation in Princeton, Minn. (fig. 18*A*). Because most of the ratios for the precipitation samples plot close to this meteoric waterline, the meteoric waterline was used to assess the oxygen and hydrogen isotopes for the lake, piezometer, and well-water samples. Oxygen-18/oxygen-16 ratios for bulk precipitation samples collected at the precipitation station along the east shore of White Bear Lake ranged from -13.74 to -3.72 per mil (fig. 18*A*; table 6), and the deuterium/protium ratios ranged from -97.3 to -17.5 per mil. Both ratios were highest in early summer (June) and lowest in the fall (October) (table 6).

## Lakes

The oxygen-18/oxygen-16 and deuterium/protium ratios for lake-water samples collected from Bald Eagle, Big Marine, Turtle, and White Bear Lakes plotted along a linear regression trend line, referred to as the evaporation trend line for the four lakes for 2014 (fig. 18*A*). The slope of this trend line (5.2) was slightly higher than the slope determined by Jones and others (2013) for White Bear Lake (slope=4.6) but similar to slopes for evaporation trend lines determined for lakes in Wisconsin (slope=6) (Krabbenhoft and others, 1990) and Canada (slope=5) (Gibson and Edwards, 2002). The slope reflects the effects of local hydrologic conditions influencing evaporation integrated during the evaporation season (Gibson and others 1993). Oxygen-18/oxygen-16 ratios for lake-water samples from the four lakes ranged from - 4.61 to -2.41 per mil (fig. 18; table 6), and deuterium/protium ratios ranged from -40.7 to -28.6 per mil (table 6). The intersection point between the evaporation trend line and the Landon and others (2000) meteoric water line (fig. 18*A*) (-9.52 [oxygen-18/oxygen-16 ratio], -65.8 [deuterium/protium ratio]) is the approximate weighted mean isotopic composition of local precipitation (Gibson and Edwards, 2002). This point is slightly lower than the approximate weighted mean isotopic composition of local precipitation determined by Jones and others (2013) for White Bear Lake (-8.2 [oxygen-18/oxygen-16 ratio], -55.4 [deuterium/protium ratio]).

The lowest ratios and largest range of ratios were in Bald Eagle Lake, whereas the other three lakes (Big Marine, Turtle, and White Bear Lakes) had similar values and ranges (fig. 18*A*). Displacement along the evaporation line for lakes represents changes in the water balance for a lake (Gibson and others, 1993). This difference between the isotopic composition of Bald Eagle Lake and the other three lakes likely reflects differences in the water balances between the lakes. Bald Eagle Lake has a larger watershed than the other three lakes (variable WS_LkshedArea.ac, appendix table 1–1); direct surface-water inflow and outflow through ditches are important components to water balance of the lake. In 2014, the other three lakes were closed-basin lakes.

Oxygen-18/oxygen-16 and deuterium/protium ratios for lake-water samples collected from the four lakes varied seasonally with variations in evaporation rates. The lowest ratios were in the early summer samples after snowmelt; the highest ratios were in late summer and fall (table 6). Snow samples tend to have much lower oxygen-18/oxygen-16 and deuterium/protium ratios than the surface-water samples (Jones and others, 2013). As the snow melted in early spring, snowmelt waters with low ratios flowed into the lakes, which decreased the ratios for the surface water in May. As temperatures increased in the late spring and summer, evaporation rates on the lake surfaces gradually increased, preferentially evaporating oxygen-16 over oxygen-18. As a result, oxygen-18/oxygen-16 and deuterium/protium ratios in the lake waters gradually increased in late spring and summer to the highest ratios in late summer and fall (table 6). Oxygen-18/oxygen-16 and deuterium/protium ratios for surface-water samples collected from lakes in the northeast Twin Cities Metropolitan Area generally were lower than ratios measured in surface waters from White Bear Lake (table 6). The ratios from these lakes generally plotted below the evaporation trend line for White Bear Lake (fig. 18*A*).

Oxygen-18/oxygen-16 and deuterium/protium ratios for water samples collected at various depths in White Bear Lake indicated that ratios vary little with depth and likely do not vary with time below the thermocline. Variations in water temperature, dissolved oxygen concentration, pH, and specific conductance with depth indicated that the thermocline for White Bear Lake on September 12, 2014, was between 36 and 46 ft below the water surface (figs. 19*A*, *B*, *C*, *D*). Slight variations across the thermocline were determined in oxygen-18/oxygen-16 and deuterium/protium ratios (-0.2 and -1.4, respectively; figs. 19*E*, *F*). These variations were similar to or less than the estimated expanded uncertainty of replicate measurements for oxygen-18/oxygen-16 and deuterium/protium ratios determined by the USGS Reston Stable Isotope Laboratory (±0.2 and ±2.0 per mil, respectively; U.S. Geological Survey, 2016b). Ranges in oxygen-18/oxygen-16 and deuterium/protium ratios with depth are similar to seasonal variations in ratios determined for water samples collected at the water surface from White Bear Lake in 2014. Because ratios varied

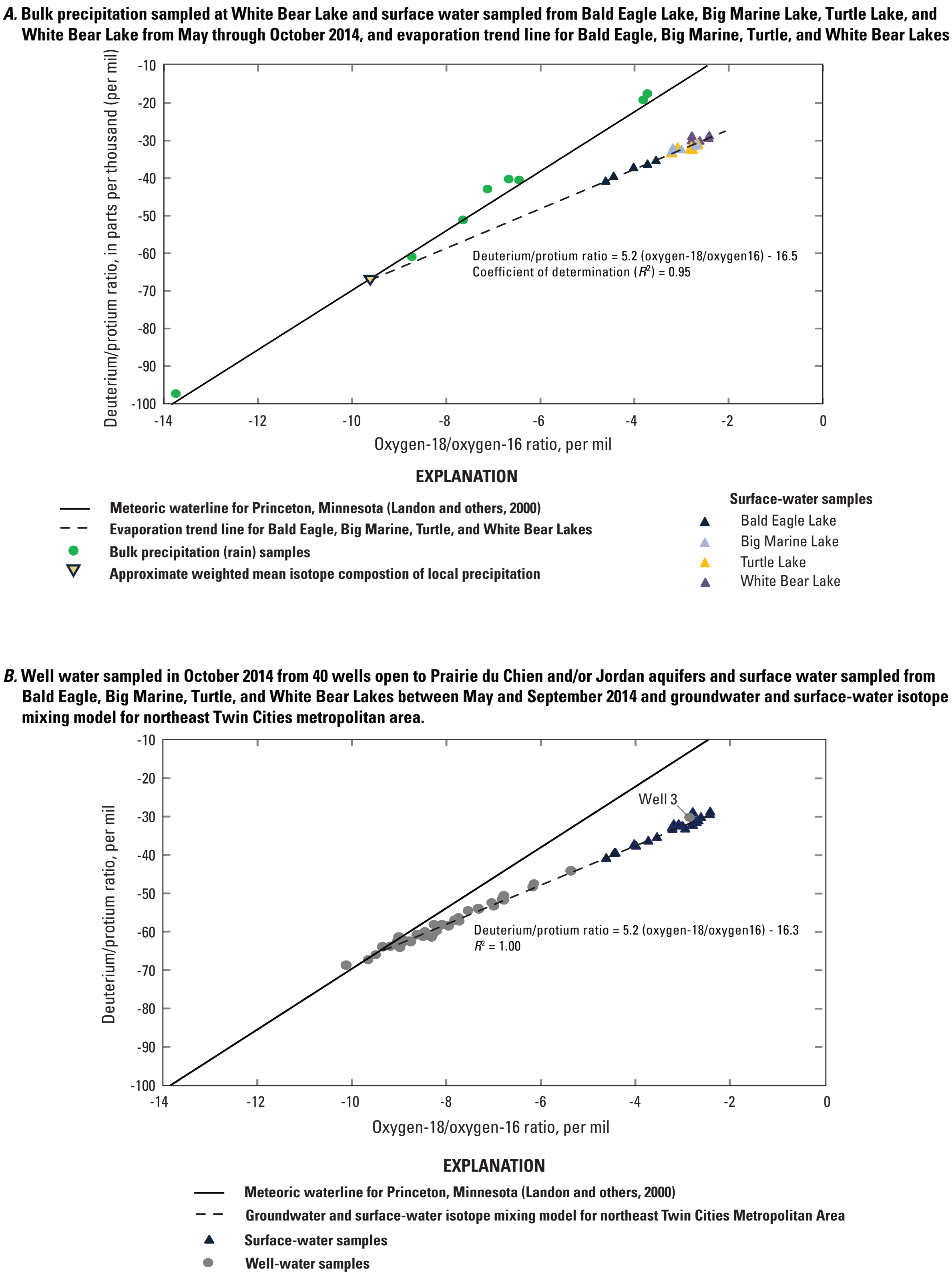



**Figure 18.** Comparison of oxygen-18/oxygen-16 ratios and deuterium/protium ratios with meteoric waterline for bulk precipitation, surface water from lakes, and well water, and groundwater and surface-water isotope mixing model for northeast Twin Cities Metropolitan Area, May–October 2014.

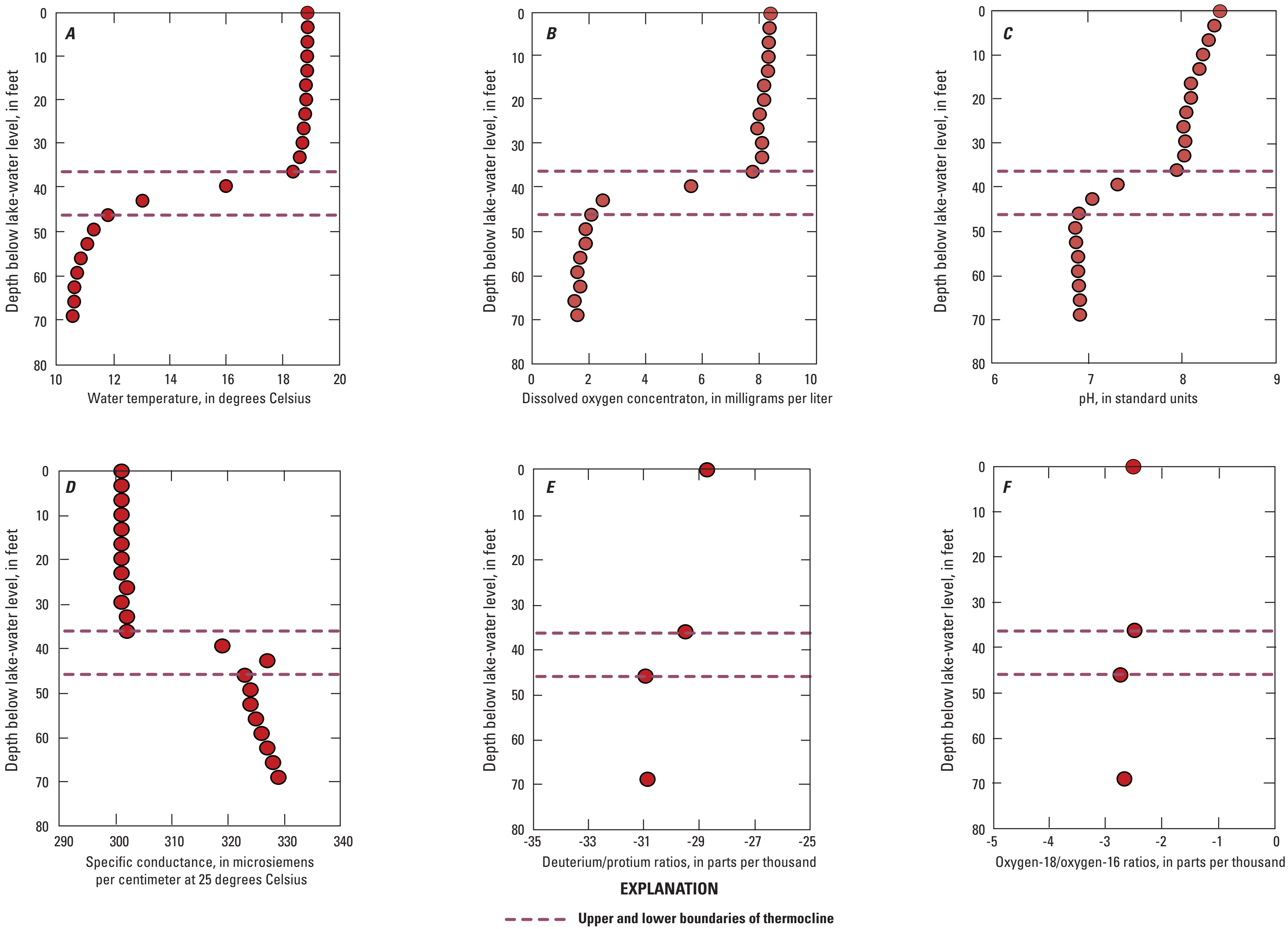



**Figure 19.** Surface-water-depth comparison of field water-quality parameters and stable isotope ratios near piezometer P4–6, White Bear Lake (U.S. Geological Survey station 450401092581302), September 12, 2014.

little with depth in White Bear Lake, a mean of the lake-water ratios from the four sampled lakes was compared to ratios for water samples from wells to estimate surface-water contributions to wells.

## Groundwater

Field water-quality properties and analyses of water samples collected from piezometers in White Bear Lake and wells near the lake indicated that lake water is flowing out of White Bear Lake and likely reaching the downgradient Prairie du Chien-Jordan aquifer. Stable isotope and other water-quality analyses of water samples collected from wells in the northeast Twin Cities Metropolitan Area indicated groundwater and surface-water exchanges with the Prairie du Chien-Jordan aquifer were occurring in other parts of the area.

### Piezometers

The oxygen-18/oxygen-16 and deuterium/protium ratios for water samples collected from piezometers were similar to ratios determined for lake-water samples collected at the water surface and in deeper parts of White Bear Lake (fig. 2; table 6), which indicated that lake water potentially could be reaching the screened intervals of the piezometers. Oxygen-18/oxygen-16 ratios for the six piezometer water samples ranged from -2.67 to -2.28 per mil, and deuterium/protium ratios ranged from -30.9 to -26.5 per mil (table 6). These ratios varied little with piezometer depth or location.

### Wells

Stable isotope concentrations in water samples from 40 wells provided valuable insight into groundwater and surface-water exchanges to the Prairie du Chien-Jordan aquifer. Dissolved gas concentrations indicated potentially large percentages of surface-water contribution to 3 of the 40 wells. The use of CFC concentrations to date groundwater from wells was limited to 9 of the 40 well-water samples.

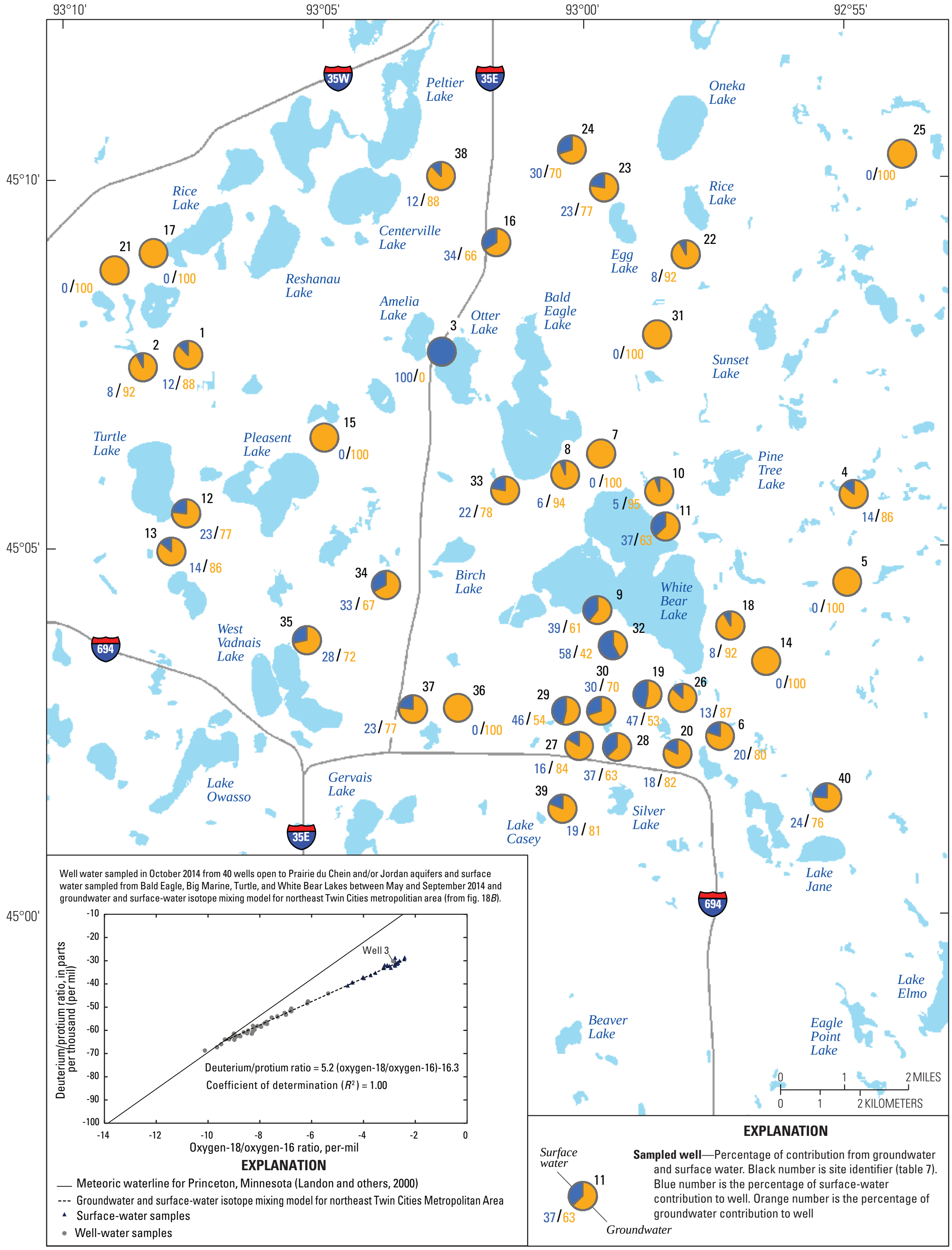
#### Stable Isotopes

Oxygen-18/oxygen-16 and deuterium/protium ratios for the water samples collected from wells open to Prairie du Chien-Jordan aquifer (including wells only open to either the Prairie du Chien or Jordan aquifer) in the northeast Twin Cities Metropolitan Area varied widely, from on the meteoric water line to next to ratios for the surface-water samples for Bald Eagle, Big Marine, Turtle, and White Bear Lakes (fig. 18*B*). These ratios plotted linearly between the meteoric water line and surface-water samples for the four lakes (fig. 18*B*).

The wide range in oxygen-18/oxygen-16 and deuterium/protium ratios for water sampled from wells completed in the Prairie du Chien-Jordan aquifer indicates different sources are supplying these wells with water. Well water with oxygen-18/oxygen-16 and deuterium/protium ratios that are on or near the meteoric water line consist mostly of groundwater that was recharged quickly after rainfall and had little time to be affected by evaporation. Of the 40 water samples, 9 (sites 5, 7, 14, 15, 17, 21, 25, 31, and 36; fig. 18*B*; table 7) plotted on or close to the meteoric water line near the approximate weighted mean isotope composition of local precipitation (fig. 18*B*), which indicates that water from these wells receive no surface-water contribution. These wells were located throughout the northeast Twin Cities Metropolitan Area (fig. 20), and well depth ranged from 135 to 490 ft below land surface (table 7).

Oxygen-18/oxygen-16 and deuterium/protium ratios varied little among water sampled from 29 wells (sampled two to four times between 2011 and 2014), which potentially indicates little to no changes in the surface-water contribution to the wells during 2011–14 (Jones and others, 2013; James Walsh, Minnesota Department of Health, written commun., November 2, 2015; table 7). For each well, the range of oxygen-18/oxygen-16 ratios was less than 0.63 per mil, and the range of deuterium/protium ratios was less than 3.0 per mil during the 2011–14 sampling period (Jones and others, 2013). These results indicate little to no changes in groundwater-flow conditions upgradient from these wells, and surface water likely contributed to these wells between 2011 and 2014.

The linear relation between the oxygen-18/oxygen-16 and deuterium/protium ratios for the well-water samples was used to develop a groundwater and lake-water isotope mixing model to estimate the percentage of surface-water contribution to the well water (fig. 18*B*, 20). This model is similar to the model developed for wells around White Bear Lake (Jones and others, 2013). Two end points were established for this relation: one point to represent 100 percent groundwater contribution and another point to represent 100 percent surface-water contribution (fig. 20). The mean oxygen-18/oxygen-16 and deuterium/protium ratios for the well-water samples near the meteoric water line were used for the 100 percent groundwater contribution end point. The mean oxygen-18/oxygen-16 and deuterium/protium ratios for the surface-water samples collected from Bald Eagle, Big Marine, Turtle, and White Bear Lakes were used for the 100 percent surface-water contribution end point. Using these end points, the percentage of surface-water contribution to the well water was estimated by comparing the linear distance of the oxygen-18/oxygen-16 and deuterium/protium ratios for the well-water sample from the two end points. Well-water samples with oxygen-18/oxygen-16 and deuterium/protium ratios closer to the mean ratios for the surface-water samples had a larger estimated percentage of surface-water contribution; well-water samples with oxygen-18/oxygen-16 and deuterium/protium ratios closer to the mean ratios for the well-water samples near the meteoric water line had a smaller percentage of surface-water contribution. Wells that had ratios higher than the mean oxygen-18/oxygen-16 and deuterium/protium ratios for the surface-water samples represented 100 percent surface-water contribution.




Base modified from Minnesota Department of Natural Resources digital data, 1:100,000
Minnesota Department of Transportation digital data, 1:100,000
ESRI digital data, 1:500,000, and U.S. Geological Survey digital data, 1:100,000
Universal Transverse Mercator projection, Zone 15
Horizontal coordinate information is referenced to
the North American Datum of 1983 (NAD 83)

**Figure 20.** Oxygen-18/oxygen-16 ratios and deuterium/protium ratios and percentage contribution from groundwater and surface water for well-water samples collected in October 2014, northeast Twin Cities Metropolitan Area, Minnesota.

The mean percentage of surface-water contributions for the 40 sampled wells was 20 percent. Wells with surface-water contributions greater than or equal to 30 percent were south of White Bear Lake (wells 9, 19, 28, 29, 30, and 32; fig. 20), northeast of West Vadnais Lake (well 34), north of Bald Eagle Lake (wells 16 and 24), and between Amelia and Otter Lakes (well 3). The depth of these wells ranged from 154 to 513 ft below land surface (table 7).

Water from 31 of the 40 wells that were sampled in October 2014 had oxygen-18/oxygen-16 and deuterium/protium ratios that indicated the well water was a mixture of surface water and groundwater (fig. 20; table 7). Of these 31 wells, 11 were south of White Bear Lake, and surface-water contributions ranged from 13 to 58 percent (fig. 20). White Bear Lake is the likely source of the surface water to these 11 wells with a mixture of surface water and groundwater because (1) these wells are near and downgradient from White Bear Lake, (2) these wells obtain their water from relatively deep depths, and (3) White Bear Lake is the only large, deep lake near to these 11 wells (Jones and others, 2013). The wells with a mixture of surface water and groundwater south of White Bear Lake were at depths ranging from 166 to 513 ft below the land surface and less than 3 mi south of White Bear Lake. Oxygen-18/oxygen-16 and deuterium/protium ratios for water from wells that were upgradient from White Bear Lake indicated the water mainly was only from groundwater and not considered a mixture of groundwater and surface water (greater than or equal to 30 percent surface-water contribution; fig. 20). One exception to this pattern is well 11 (USGS site 450528092582601; fig. 20). Well 11 is close to White Bear Lake (less than 800 ft from the shoreline) and is relatively shallow (175 ft deep) compared to the other sampled wells. Pumping and the associated drawdown from this well may bring in water from White Bear Lake.

Other water-quality data, besides stable isotope data, indicated that surface water was contributing water to northeast Twin Cities Metropolitan Area wells; for example, the oxygen-18/oxygen-16 ratio, deuterium/protium ratio, and methane concentration determined for water sampled from one well (well 3, USGS site 450741093024301) indicated that all the water from the well was from a surface-water source (figs. 18*B*, 20). The well was between two lakes (Otter and Amelia Lakes) and withdraws water from the Prairie du Chien aquifer between 149 and 154 ft below the land surface. The oxygen-18/oxygen-16 and deuterium/protium ratios for the water sample were much higher than ratios from the other sampled wells, and were similar to ratios for Big Marine, Turtle, and White Bear Lakes (figs. 18*B*, 20; tables 6, 7). The methane concentration in the water (4.0 milligrams per liter [mg/L]) was more than one order of magnitude greater than methane concentrations in water samples collected from the other 39 wells (table 7). The dinitrogen concentration in water from well 3 was the lowest concentration (16.7 mg/L) measured in water from the 40 wells (table 7). Methane gas commonly is generated in organic-rich lake or wetland sediments as a result of microbe-mediated decomposition of the organic materials (Beckwith and Baird, 2001; Chen and others, 1972; Lojen and others, 1999; Williams and Crawford, 1984; Vreča, 2003). Lake waters discharging to aquifers can accumulate methane as the waters flow through the lake sediments before reaching the aquifers and nearby wells. Water in wells that take in a larger amount of this discharged lake water can have high methane and low dinitrogen concentrations. Similarly, relatively low dinitrogen concentrations were measured in pore waters of organic materials in peatlands in the northeast Twin Cities Metropolitan Area with high methane concentrations (Williams and Crawford, 1984). Ammonia production and binding to deep-water lake sediments can result in low dinitrogen concentrations in pore waters of lake sediments (Jones and others, 1982; Kuivila and Murray, 1984), which can be transported in groundwater-flow systems to nearby wells.

Oxygen-18/oxygen-16 and deuterium/protium ratios for waters sampled from wells indicate that surface water is reaching the Prairie du Chien aquifer; percentages of surface-water contribution generally were greater downgradient or near some of the larger northeast Twin Cities Metropolitan Area lakes, such as White Bear and Bald Eagle Lakes (fig. 20). These percentages tend to decrease in wells farther away from the lakes because water in the aquifer consists more of water recharged directly from the land surface.

### Dissolved Gases and Age Dating

Dissolved gas concentrations in water sampled from wells used to estimate ages of well waters indicated reduced conditions in all the sampled wells and potentially large percentages of surface-water contribution to some wells. Dissolved oxygen concentrations measured on site and in water samples were less than 2.5 mg/L (table 7). High methane concentrations, low dinitrogen concentrations, and high oxygen-18/oxygen-16 and deuterium/protium ratios in water from well 3 (fig. 20; table 7) indicated that water in this well was from a surface-water source (as described previously in the “Stable Isotopes” section). High methane concentrations and high oxygen-18/oxygen-16 and deuterium/protium ratios in wells 19 and 37 (table 7) relative to concentrations and ratios measured in water samples from other northeast Twin Cities Metropolitan Area wells indicate a large percentage of surface-water contribution to the well.

Carbon dioxide concentrations measured in water samples from the 40 wells ranged from 2.2 to 57 mg/L (table 7), and the highest concentrations present in water were from wells 25 and 40 (fig. 20; table 7). These two domestic wells were constructed to relatively shallow depths (135 and 165 ft below land surface) compared to other wells and were open to the Prairie du Chien aquifer, which consists mainly of dolostones and limestones (Runkel and others, 2003). Carbon dioxide gas commonly builds up in waters from limestone aquifers with the dissolution of calcite and carbonate minerals in soils and limestones (Atkinson, 1977; Pearson and

Hanshaw, 1970), and can be present in high concentrations in water from wells open to the aquifer, particularly in low-production wells, such as domestic wells with fluctuating water levels. Groundwater withdrawal rates from domestic wells typically are low compared to municipal and commercial wells, potentially allowing for the accumulation of gases in well water. Elevated carbon dioxide concentrations are often associated with lower pH values (Carroll and others, 2009). The lowest pH values were measured in water from wells 25 and 40 (table 7).

Ages of recharge were estimated for 9 of the 40 well-water samples using CFC–11, CFC–12, and CFC–113 concentrations using the dispersion LPM (table 7). The dispersion model uses two parameters (mean age and the dispersion parameter) in the following equation (Jurgens and others, 2012):

$$DM_{g(t-t')} = \frac{1}{\tau_s} \frac{1}{\sqrt{4\pi DP \frac{t-t'}{\tau_s}}} e^{\left(-\frac{\left(1-\frac{t-t'}{\tau_s}\right)^2}{4DP\frac{t-t'}{\tau_s}}\right)} \qquad (4)$$

where

- $DM$ is the dispersion model;
- $g$ is the transit-time or exit-age distribution function (Maloszewski and Zuber, 1982);
- $t$ is the sample date;
- $t'$ is the date at which a water parcel entered the system;
- $\tau_s$ is the mean age of the sample;
- $e$ is a mathematical constant that is the base of the natural logarithm; and
- $DP$ is the dispersion parameter, which equals the dispersion coefficient ($D$) divided by the velocity ($v$) and outlet position ($x$).

The dispersion parameter ($DP$) is the inverse of the Peclet number or the ratio of the dispersion coefficient ($D$) to the velocity ($v$) and outlet position ($x$). In practice, the dispersion parameter describes the relative width and height of the age distribution, and is mainly a measure of the relative importance of dispersion (mixing) to advection (Zuber and Maloszewski, 2001).

Simulated CFC concentrations fit the observed CFC data the best for the dispersion model for all nine well-water samples. Of the five LPMs used in the analysis, the dispersion LPM likely best represents the flow conditions at the nine wells in the Prairie du Chien and Jordan aquifers because (1) the water sampled from the long open-hole intervals of the wells likely result in a mixture of water of various ages and (2) groundwater flow in the karstic Prairie du Chien and Jordan aquifers is complex and heterogeneous. The dispersion model provides an approximate description of age distributions in samples from a variety of aquifer configurations (Jurgens and others, 2012). The chi-square (weighted sum of squares) for observed and simulated CFC concentrations were less than 3.0 for each of the model runs used to estimate ages of recharge. Estimated ages of recharge for the nine well-water samples ranged widely from the early 1940s to the mid-1970s (table 7). These dates were estimated using a combination of concentration data for CFC–11 and CFC–12; CFC–12 and CFC–113; CFC–11 and CFC–113; or CFC–11, CFC–12, and CFC–113 determined for the water samples (table 7). Wells with longer open intervals are more likely to extract waters varying widely in sources and ages compared to wells that have shorter open intervals. The wide range in estimated ages of recharge may have resulted from the wide range in the lengths of open intervals and depths for the sampled wells. The open-interval lengths and depths for these nine wells ranged from 16 to 224 ft and from 140 to 513 ft below land surface, respectively (table 7). In porous media aquifers, wells that are deeper are more likely to have older water compared to shallower wells because it would likely take longer for recharged waters to reach the open intervals; however, the presence of karstic features in the Prairie du Chien and other aquifers in the northeast Twin Cities Metropolitan Area (Runkel and others, 2003) may result in recharged waters reaching deeper wells in shorter-than-anticipated arrival times because of conduit flow.

Estimated age of recharge is generally correlated with percentages of surface-water contributions to wells, with the younger ages associated with higher percentages of surface-water contributions. Waters with the youngest estimated ages of recharge (the early to mid-1970s) were sampled from three wells (wells 6, 27, and 28) south of White Bear Lake (fig. 20; table 7). These wells received estimated surface-water contributions ranging from 16 to 37 percent (fig. 20). Water from three wells (wells 2, 13, and 16) about 5 mi west and north of White Bear Lake had estimated age of recharge ranging from early 1940s to mid-1950s, with estimated surface-water contributions ranging from 8 to 34 percent (fig. 20). Water collected from three wells (wells 10, 11, and 14) north or east of White Bear Lake was dated between the late 1940s to late 1960s, with estimated surface-water contributions ranging from 0 to 37 percent (fig. 20; table 7).

## Lake-Sediment Characteristics, Water-Level Differences, Hydraulic Conductivity, and Seepage-Flux Rates

Results from lake-sediment coring, CSPs, and water-level and flow monitoring indicated that groundwater inflow and lake-water outflow in White Bear Lake can be happening in shallow and deep water settings. The CSPs indicated that deep sections of White Bear, Pleasant, Turtle, and Big Marine Lakes (fig. 1) have few trapped gases and little organic material, indicating areas where groundwater and surface-water exchanges may happen. Water-level differences between White Bear Lake and piezometers and seepage measurements in deep-water parts of the lake indicate that groundwater and lake-water exchange may be happening in deep waters, which is not typical in most Minnesota lakes.

## Lake-Sediment Coring in White Bear Lake

The lithology of lake-sediment cores collected in the deep waters of White Bear Lake (fig. 2) indicated that organic-rich sediment thicknesses generally are small (range from 1.8 to 13.6 ft). The sediment cores collected at the four sites consisted of lacustrine and glacial sediments. Cumulative core depths ranged from 7.0 to 20.2 ft below the water-sediment interface. Lacustrine sediment extended down to 1.8 to 13.6 ft below the water-sediment interface in the four cores. The lacustrine sediments in the cores consisted of organic-rich clayey sapropel, with silt, diatoms, macrofossils, and shells present in some intervals (table 16). The glacial sediments are marked by transitions to clays, silts, sands, and gravels (table 16). A detailed description of the core lithologies is in Heck and others (2014).

The coring devices at site P1 (fig. 2) were driven a total of 10.1 ft below the water-sediment interface (table 16). Total sediment recovery was 6.5 ft, and the upper 6.2 ft of the core consisted of organic-rich lacustrine sediments with diatomaceous silty sapropel transitioning to sandy silt and clayey silt with sands (table 16). Coarse to very coarse lithic sands, gravels, silts, and clays were present below 6.2 ft in the core, which likely are glacial deposits that may have slumped into a basin (see the “Continuous Seismic-Reflection Profiling in Six Northeast Twin Cities Metropolitan Area Lakes” section).

The coring devices at site P2 (fig. 2) were driven a total of 20.2 ft below the water-sediment interface (table 16). The entire 13.6 ft of the core consisted of organic-rich lacustrine sediments with layers of clayey and silty sapropel and silty fine sands (fig. 2; table 16). Clays transitioning to sands and gravels are inferred below 13.6 ft in the core (fig. 2; table 16). This transition likely represents the contact between lacustrine to glacial sediments.

The coring devices at site P3 (fig. 2) were driven a total of 10.4 ft below the water-sediment interface (table 16). Total core recovery was 9.7 ft. The upper 6.5 ft of the core consisted of lacustrine layers of organic-rich clays and clayey silt with shells and sands. Reddish-gray sands, sandy silts, and clays were present in the core between 6.5 and 6.7 ft below the top of the core, and 1.6 ft of olive-brown, clayey silt were present below the reddish-gray sediments (fig. 21; table 16). The reddish-gray sediments may be glacial sediments slumped on top of olive brown, lacustrine sediments (see the “Continuous Seismic-Reflection Profiling in Six Northeast Twin Cities Metropolitan Area Lakes” section). Layers of reddish-brown silts and sands were present below the 1.6 ft of olive-brown, clayey silt, and are likely glacial in origin. The reddish-brown sands that were at the bottom of the core likely were glacial outwash of the Cromwell Formation, which fills the bedrock valley present in the north and east bays of White Bear Lake (Bauer, 2016).

The coring devices at site P4 (site 2) were driven a total of 9.0 ft below the water-sediment interface (table 16). Total core recovery was 5.8 ft. Organic-rich material, including sapropel, only was present in the upper 1.8 ft of the core (table 16). Thin clay and silt layers present below the organic-rich material between 1.8 to 5.8 ft may be glacial or lacustrine in origin. Compact sand at a depth 5.8 ft below the water-sediment interface fell out of the corer during sampling. These sediments likely are glacial outwash of the Cromwell Formation (Bauer, 2016). The lithology of this lake-sediment core is similar to the lithology of a core collected at a site 430 ft west-southwest of site P4 (Jones and others, 2013); organic-rich material was present in the upper 2.0 ft of the core, and layers of clays, silts, sands and gravels were present below the organic-rich material.

A lack of thick lacustrine organic deposits and the presence of more permeable, slumped sands and gravels in White Bear Lake cores indicate the lakebed may be permeable in some areas of deep waters, and groundwater and surface-water exchanges may be possible. Minnesota lakes that have persisted for the Holocene typically will have about 23 to 46 ft of lake sediments (Webb and Webb, 1988; Dean and Gorham, 1998). The thicknesses of the organic-rich, lacustrine deposits in the White Bear Lake sediment cores could range from 2.0 to more than 17 ft, depending on the classification of the silt and clay layers in the cores. Physical characteristics of White Bear Lake, such as the presence of slumped sands, small watershed size, and deep-water areas, may explain the lack of thick organic lake sediments at some deep-water sites (Jones and others, 2013).

## Continuous Seismic-Reflection Profiling in Six Northeast Twin Cities Metropolitan Area Lakes

The CSPs collected on six lakes (fig. 2) indicated potential areas where groundwater and surface-water exchange are more likely to happen in the lakes based on the presence of trapped gases and organic layer thicknesses. A total of 141 survey lines (about 68 linear mi) of CSPs were collected in the six lakes (table 17). Survey line lengths ranged from 59 to 15,000 ft, and maximum water depths along the profiles ranged from 13 ft in South School Section Lake to 135 ft in Lake Elmo. Depth of seismic signal penetration into the lake sediments ranged from less than 0.3 to about 20 ft below the lake bottom, and was controlled by (1) water-column thickness, (2) the compaction of the lake sediments, (3) presence of gas in subbottom sediments, and (4) geologic properties of the bottom sediments (figs. 22, 23). Trapped gas in the subbottom attenuated the signal to depths of less than 0.3 ft into the lake sediments in areas where the CSPs indicated the presence of thick sapropel deposits (fig. 22*B*). Characteristic hard bottom sediments in these lakes are tightly compacted sands and gravels, and clayey silt layers with shells.

The CSPs at the four lake sediment core sites in White Bear Lake indicated that organic-rich sediments (including sapropel) could be distinguished from layers of clays, silts, sands, and gravels in areas where gases were not abundant in the sediments (figs. 22, 23). Organic-rich materials, such as sapropel, in the upper parts of the lake-sediment cores

**Table 16.** Lithology from lake-bottom sediment cores at piezometer nest locations in White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.

[ft, foot; NAVD 88, North American Vertical Datum of 1988; --, no data]

| Unit number | Depth below lake bottom (ft) | | Depth below lake bottom (ft above NAVD 88) | | Color | Unit lithology |
|---|---|---|---|---|---|---|
| | Top of unit | Bottom of unit | Top of unit | Bottom of unit | | |
| Lithology of lake-sediment core WBL–P1 collected at piezometer nest P1 from the lake-bottom sediment of White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota | | | | | | |
| 1 | 0.0 | 4.4 | 891.45 | 887.05 | Dark and light brown | Diamtomacous silty sapropel with large diatoms and clay, gradual transition to more consolidated and more clastic sediment. |
| 2 | 4.4 | 5.1 | 887.05 | 886.40 | Very dark brown | Diatomacous sapropel with silt and large plan macrofossils, gradual transition ot clay. |
| 3 | 5.1 | 5.6 | 886.40 | 885.89 | Brown | Sandy silt with clay. |
| 4 | 5.6 | 5.9 | 885.89 | 885.51 | Light olive brown | Shelly fine sandy silt with shells and clay, lithic sand and black horizontal organic smears. |
| 5 | 5.9 | 6.2 | 885.51 | 885.22 | Very dark brown | Organic-rich clayey silty, some coarse-sand sized quartz. |
| 6 | 6.2 | 6.3 | 885.22 | 885.10 | Gray | Lithic gravelly coarse sand, very coarse quartz sand grains. |
| 7 | 6.3 | 6.5 | 885.10 | 884.99 | Gray | Lithic coarse sandy silt with clay. |
| 8 | 6.5 | 7.0 | 884.99 | 884.49 | -- | Sand—fell out of core barrel, some sandy water in barrel. |
| 9 | 7.0 | 10.1 | 884.49 | 881.35 | -- | Gravelly sand—fell out of core barrel, recovered 1.6 inches of material. |
| Lithology of lake-sediment core WBL–P2 collected at piezometer nest P2 from the lake-bottom sediment of White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota | | | | | | |
| 1 | 0.0 | 4.2 | 869.95 | 865.73 | Very dark brown | Organic-rich, massive clayey sapropel with silt. |
| 2 | 4.2 | 7.2 | 865.73 | 862.75 | Very dark brown to black | Organic-rich, massivly bedded silty sapropel with clay and diffuse transitions. |
| 3 | 7.2 | 10.5 | 862.75 | 859.45 | Very dark borwn and gray | Bedded clayey sapropel with silt, increasing clay. |
| 4 | 10.5 | 12.3 | 859.45 | 857.65 | Very dark gray | Organic-rich silty clay sapropel. |
| 5 | 12.3 | 12.4 | 857.65 | 857.55 | Very dark grayish brown | Silty fine sand. |
| 6 | 12.4 | 13.4 | 857.55 | 856.55 | Grayish brown | Organic-rich silty clay sapropel with plant macrofossils, large increase in fine grained quartz. |
| 7 | 13.4 | 13.6 | 856.55 | 856.35 | Very dark gray | Organic-rich clayey silt. |
| 8 | 13.6 | 16.9 | 856.35 | 853.05 | -- | Clay with early transition to sand. |
| 9 | 16.9 | 20.2 | 853.05 | 849.75 | -- | Sand and gravel. |
| Lithology of lake-sediment core WBL–P3 collected at piezometer nest P3 from the lake-bottom sediment of White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota | | | | | | |
| 1 | 0.0 | 2.3 | 873.55 | 871.25 | Very dark gray | Organic-rich silty clay, gradually increasing in fine sand. |
| 2 | 2.3 | 3.1 | 871.25 | 870.47 | Very dark grayish brown | Fine grained quartz, clayey, silt with shell fragments. |
| 3 | 3.1 | 4.6 | 870.47 | 868.99 | Dark olive brown | Clayey silt with white shells. |
| 4 | 4.6 | 5.9 | 868.99 | 867.66 | Dark olive brown | Clayey silt, no shells. |
| 5 | 5.9 | 6.5 | 867.66 | 867.09 | Very dark gray | Organic-rich massive clay. |
| 6 | 6.5 | 6.6 | 867.09 | 866.94 | Reddish gray | Poorly sorted lithic sand, some clay, mostly coarse-grained quartz. |
| 7 | 6.6 | 6.7 | 866.94 | 866.86 | Reddish gray | Fine sandy silt with mafics. |
| 8 | 6.7 | 8.3 | 866.86 | 865.25 | Olive brown | Massive clayey silt. |
| 9 | 8.3 | 9.7 | 865.25 | 863.84 | Reddish brown | Silt with mafics, alternating bands of fine, medium, and coarse-very coarse lithic sands. |
| 10 | 9.7 | 10.4 | 863.84 | 863.18 | -- | Sand—fell out of core barrel. |

**Table 16.** Lithology from lake-bottom sediment cores at piezometer nest locations in White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.—Continued

[ft, foot; NAVD 88, North American Vertical Datum of 1988; --, no data]

| Unit number | Depth below lake bottom (ft) | | Depth below lake bottom (ft above NAVD 88) | | Color | Unit lithology |
|---|---|---|---|---|---|---|
| | Top of unit | Bottom of unit | Top of unit | Bottom of unit | | |
| Lithology of lake-sediment core WBL–P4 collected at piezometer nest P4 from the lake-bottom sediment of White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota | | | | | | |
| 1 | 0.0 | 1.7 | 851.85 | 850.11 | Very dark grayish brown | Massive clayey sapropel with silt. |
| 2 | 1.7 | 1.8 | 850.11 | 850.08 | Very dark grayish brown | Massive organic-rich fine sandy clay with silt. |
| 3 | 1.8 | 2.6 | 850.08 | 849.24 | Grayish brown | Fine silty clay. |
| 4 | 2.6 | 2.7 | 849.24 | 849.18 | Reddish brown | Silty fine sand. |
| 5 | 2.7 | 2.9 | 849.18 | 849.00 | Grayish brown | Fine silty clay. |
| 6 | 2.9 | 3.0 | 849.00 | 848.86 | Reddish brown | Silty fine sand. |
| 7 | 3.0 | 5.8 | 848.86 | 846.05 | Reddish gray | Fine silty clay. |
| 8 | 5.8 | 9.0 | 846.05 | 842.85 | -- | Sand—fell out of core barrel. |

were identifiable in the CSPs as higher-frequency reflections, appearing as faint lines in the CSPs (figs. 22, 23). Layers of silts, sands, and gravels in the lake-sediment cores were observed in the CSPs as characteristic lower-frequency, higher-amplitude reflections and more pronounced lines; the darkest lines indicate hard, compact layers of sands and gravels (fig. 23*A*). These compacted layers commonly prevented further penetration of the seismic signal to lower depths. Structural features in subbottom sediments, such as slumped materials and sediment-filled basins, were identified in the CSPs (figs. 22*A*, 23*A*) by their characteristic set of high-amplitude reflections. These features commonly are present downslope of or in areas of White Bear Lake where large changes in lake-depth elevations happened (fig. 23*A*).

The presence of abundant gases could be identified by dark sections of the CSPs (fig. 22*B*). Typically, these dark sections were within 3 ft of the water-sediment interface (water-bottom). Gas was often present in clayey organic material, such as sapropels (fig. 22*B*). Often gases present in organic-rich lake sediments are hydrogen sulfide and methane gases generated as a result of microbe-mediated decomposition of the organic materials (Beckwith and Baird, 2001; Chen and others, 1972; Lojen and others, 1999; Vreča 2003). Areas where gases were abundant were considered to be areas where little water exchange happened between the lake and lakebed sediments because permeabilities of fine-grained, organic-rich materials (such as sapropel or gyttja) typically are low, and the gases filling the void spaces limit water exchange (Christiansen, 1944; Heilweil and Solomon, 2002). Under fully saturated conditions, the permeability of sapropels typically ranges from $5x10^{-4}$ to $1x10^{-7}$ foot per day (ft/d) (Karls, 1982; Tiedeman and others, 1997; Winter, 1983). Areas where few gases and little sapropel are present were considered areas where potential water exchange between the lake and lake sediments could happen.

The CSPs in the six lakes were classified into three groups based on geological and sediment gas-content interpretation of the CSPs (figs. 24, 25). The three groups were (1) areas where abundant trapped gases were present, typically with clayey sapropel (gyttja) deposits greater than 5.0 ft thick; (2) nearshore or shallower water areas where few gases were present; and (3) deep-water areas where few gases were present. The groups represent areas of different potentials for water exchange between the lake and lake sediments. Sections of the CSPs where dark blotched areas covered the seismic-reflection lines were placed in the first group, whereas other sections of the CSPs were placed in the other two groups based on the water depth. The first group (gas-filled sediments) represents areas where little to no water exchange between the lake and lower sediments was thought to happen, whereas the other two groups (low-gas sediments) represent areas where water exchange potentially may happen.

**Table 17.** Total number of miles and survey lines of continuous seismic-reflection profiling in selected lakes, November 2013, northeast Twin Cities Metropolitan Area, Minnesota.

| Lake name (fig. 2) | Number of miles | Number of survey lines |
|---|---|---|
| Big Marine Lake | 11 | 21 |
| Lake Elmo | 5.6 | 15 |
| Pleasant Lake | 9.3 | 20 |
| South School Section Lake | 3.4 | 15 |
| Turtle Lake | 6.1 | 20 |
| White Bear Lake | 32 | 50 |

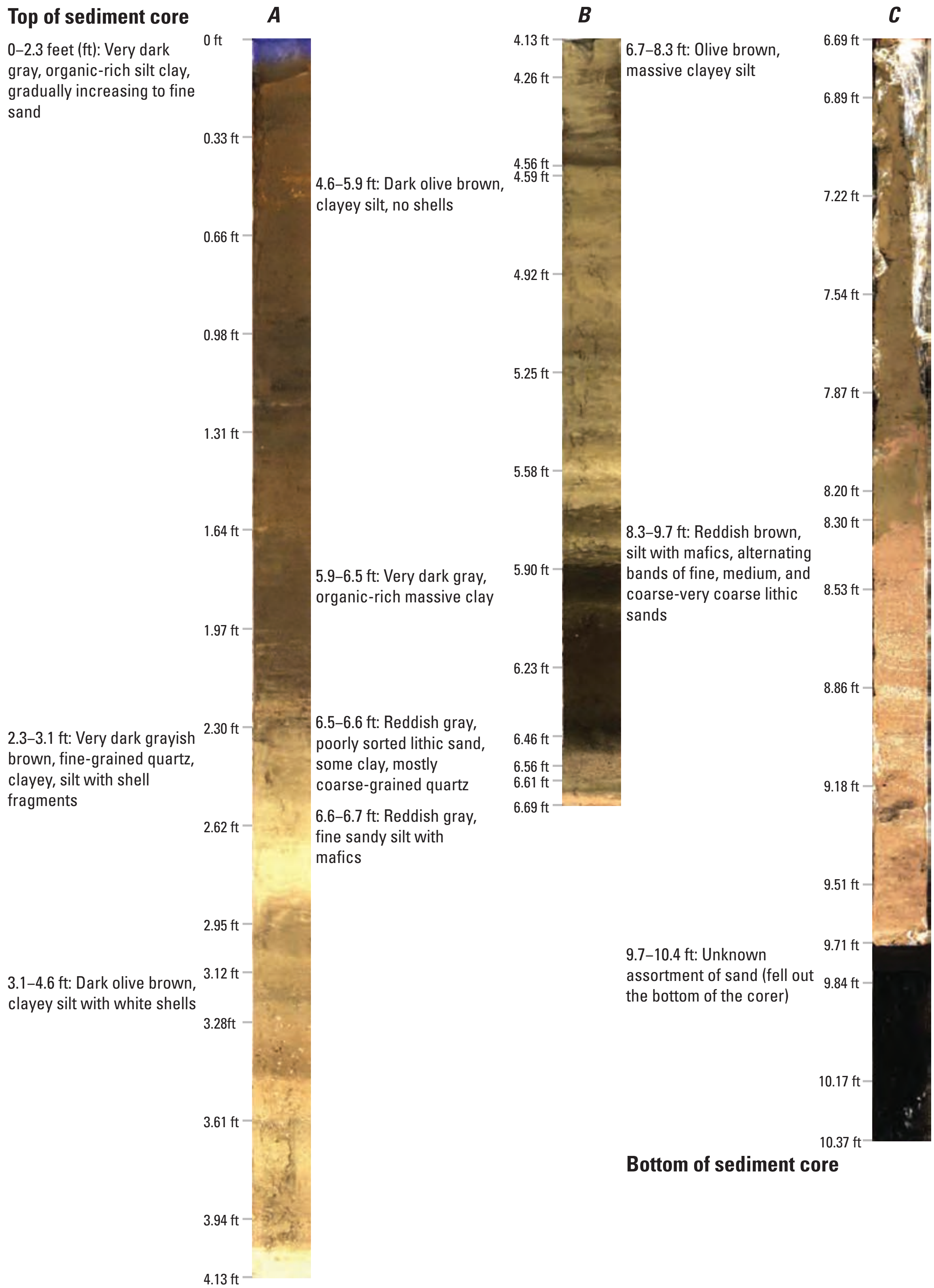
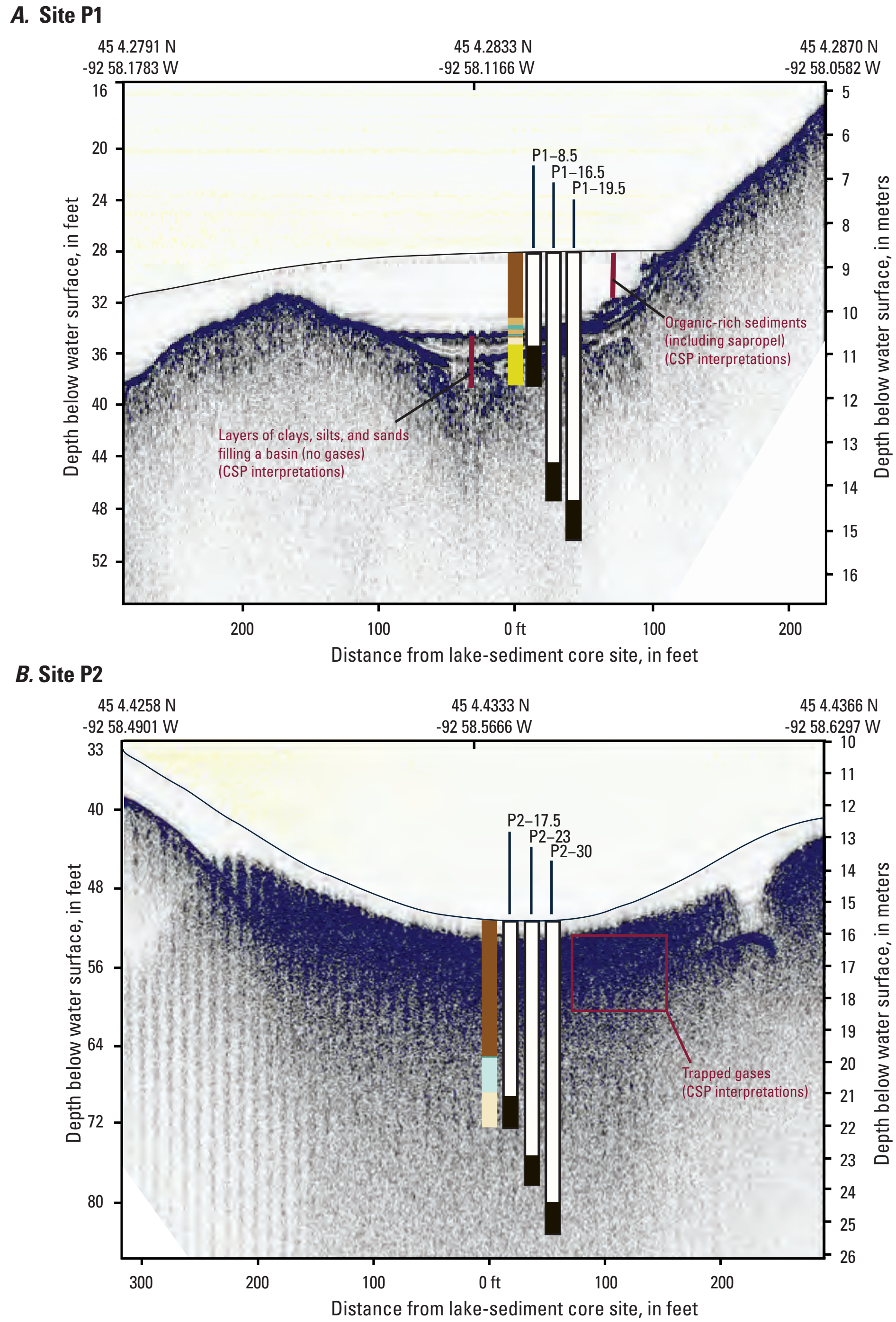



**Figure 21.** Lithology of lake-sediment cores collected at piezometer nest P3 from the bottom of White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.




**EXPLANATION**

[Continuous seismic-reflection profiling (CSP) surveys were completed on November 13, 2013. The water level of White Bear Lake on that date was about 919.76 feet above the Ramsey County 1912 datum]

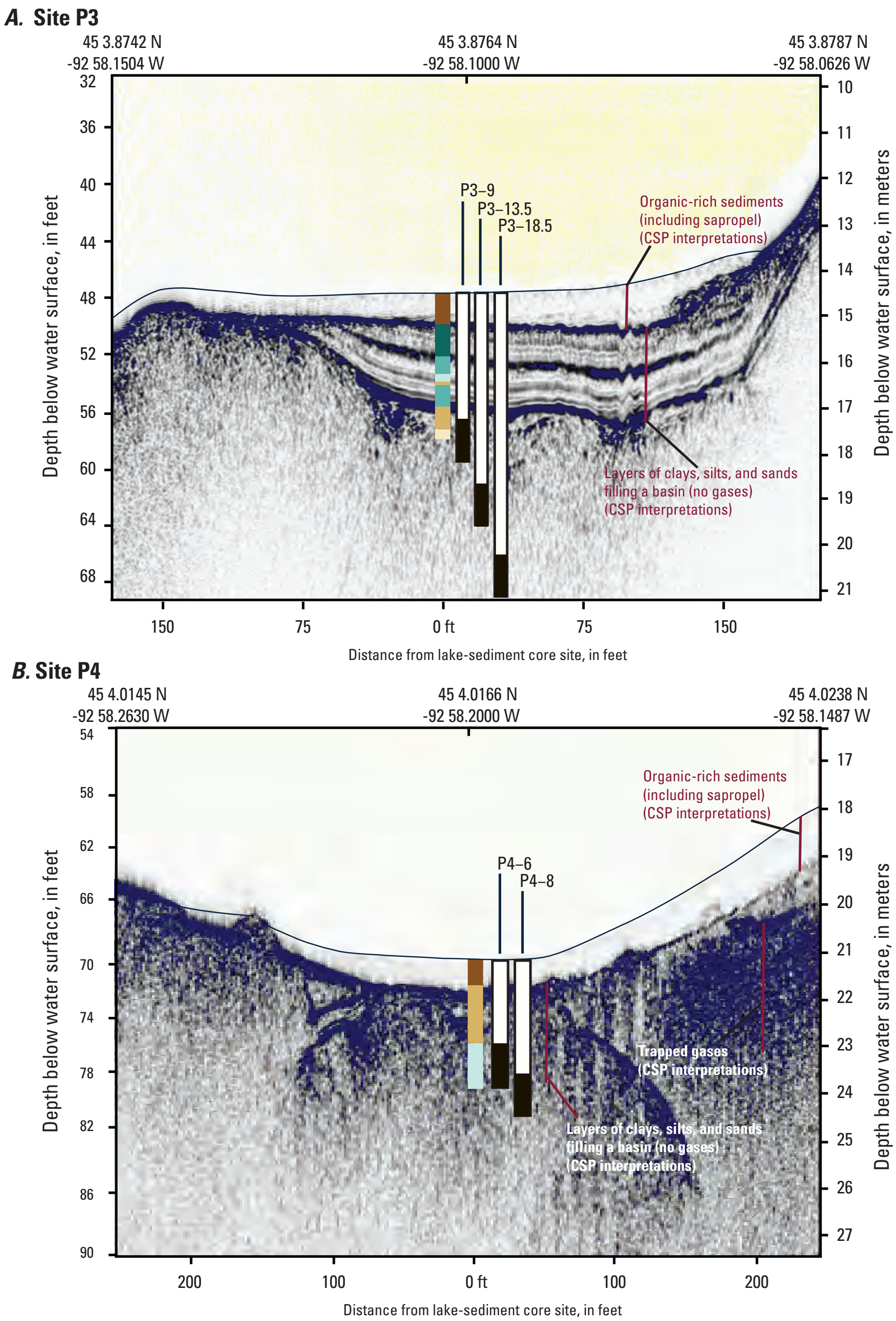
**Lake-sediment core lithology**

- Sapropel and clay
- Silt, clay, and sand
- Sand
- Gravel and sand
- Clay
- Clayey silt, no shells

**Piezometer construction**

- Casing in lake sediment
- Screen
- P2–30 Piezometers about 6 feet from core and site number
- Lake bottom

**Figure 22.** Continuous seismic-reflection profiling, lake-sediment core lithology, and piezometer construction for sites P1 and P2, White Bear Lake, Minnesota.




**EXPLANATION**

[Continuous seismic-reflection profiling (CSP) surveys were completed on November 13, 2013. The water level of White Bear Lake on that date was about 919.76 feet above the Ramsey County 1912 datum]

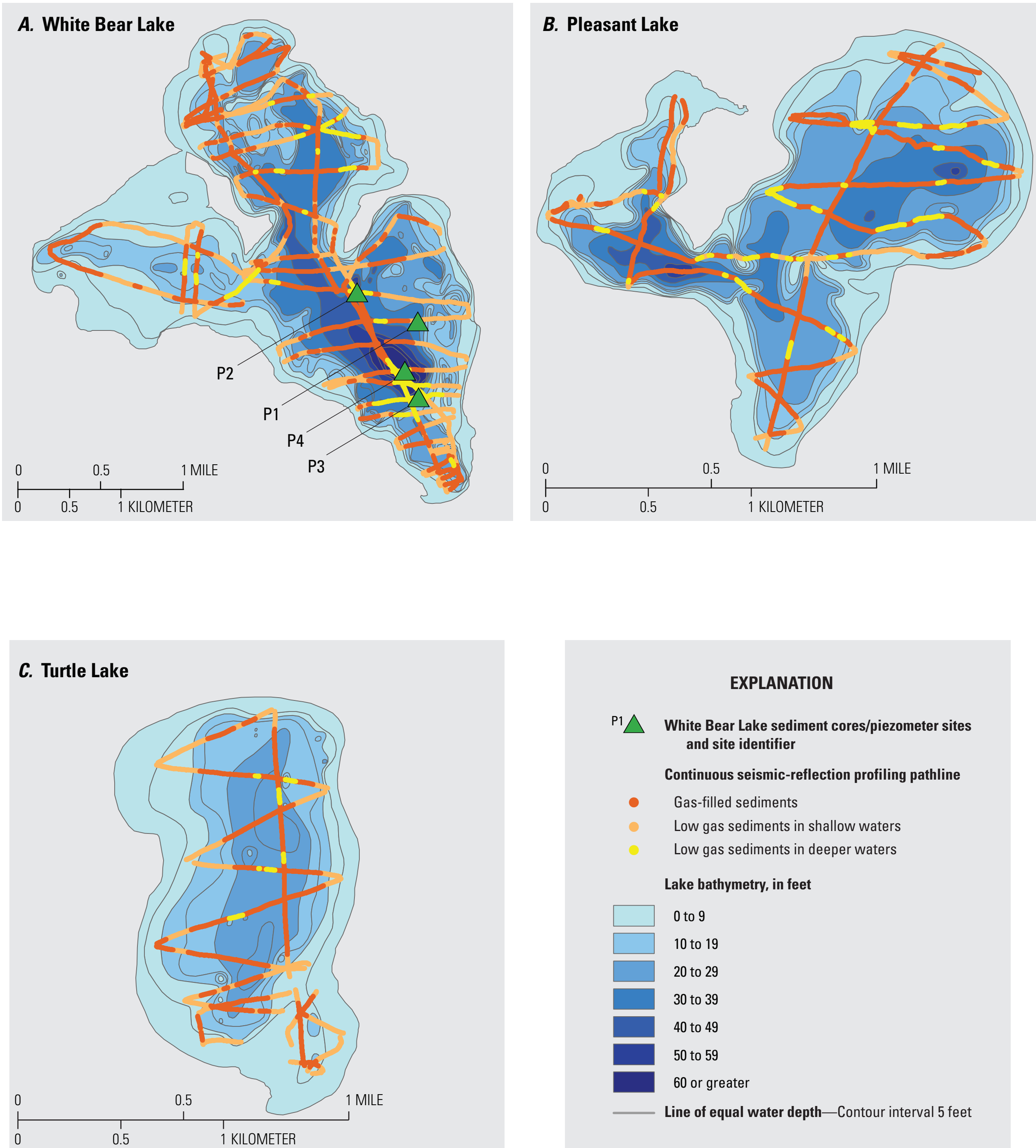
**Lake-sediment core lithology**

- Sapropel and clay
- Silt, clay, and sand
- Sand
- Clay
- Clayey silt, no shells
- Clayey silt, shells

**Piezometer construction**

- Casing in lake sediment
- Screen
- P3–9 Piezometers about 6 feet from core and site number
- Lake bottom

**Figure 23.** Continuous seismic-reflection profiling, lake-sediment core lithology, and piezometer construction for sites P3 and P4, White Bear Lake, Minnesota.




Lake bathymetry from Minnesota Department of Natural Resouces (2006)

**Figure 24.** Location of identified gas-filled and low-gas sediments along continuous seismic-reflection profile pathlines and lake bathymetry, northeast Twin Cities Metropolitan Area, Minnesota. *A*, White Bear Lake. *B*, Pleasant Lake. *C*, Turtle Lake.

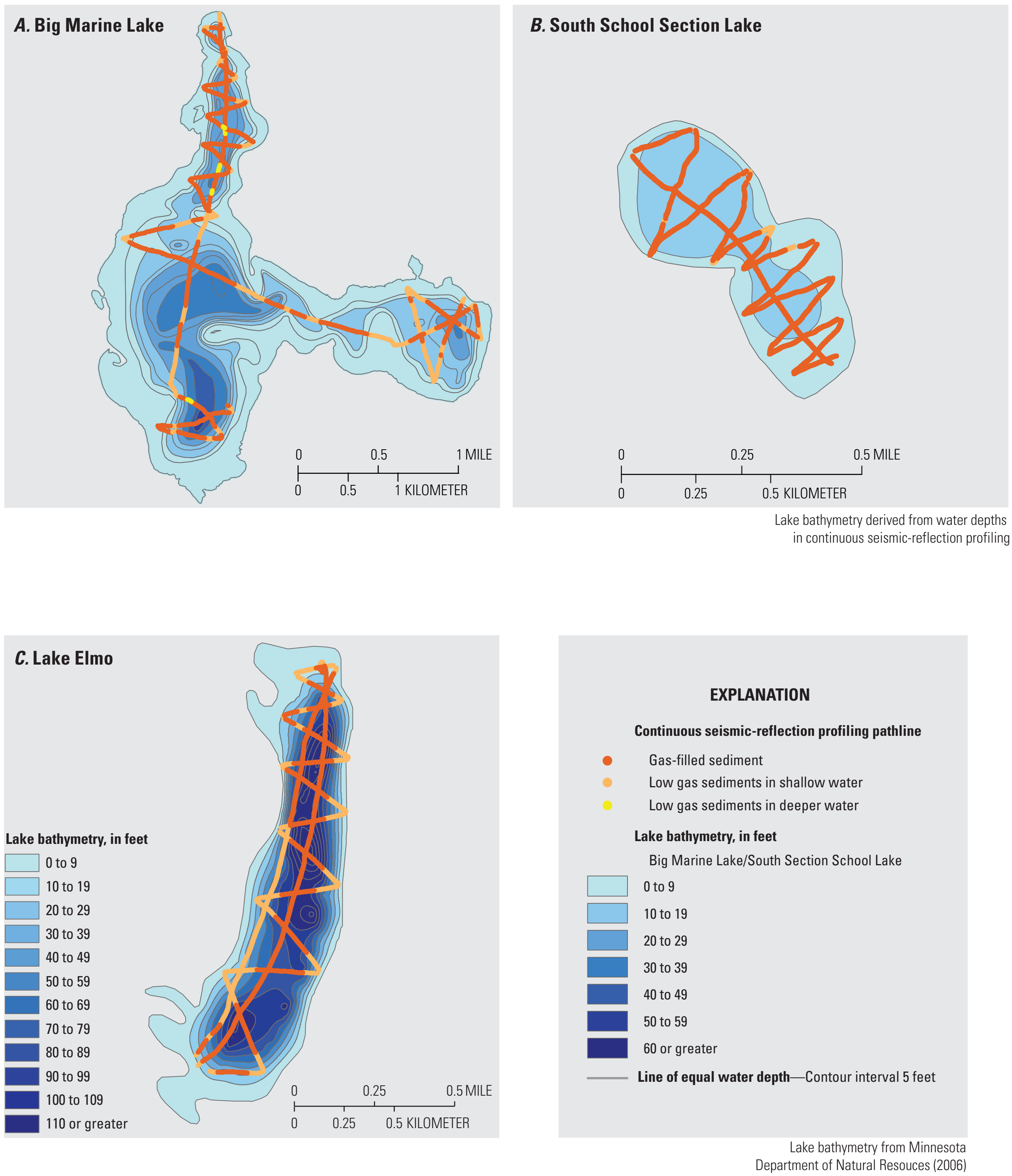



**Figure 25.** Location of identified gas-filled and low-gas sediments along continuous seismic-reflection profile pathlines and lake bathymetry, northeast Twin Cities Metropolitan Area, Minnesota. *A*, Big Marine Lake. *B*, South School Section Lake. *C*, Lake Elmo.

Lake sediments with few gases, slumping, and in sediment-filled basins were identified in deeper-water areas in White Bear, Pleasant, Turtle, and Big Marine Lakes (figs. 24, 25). A particularly large area of these low-gas sediments was identified in six CSPs in the southeast section of White Bear Lake (fig. 24*A*). Two of the four piezometer nests installed in White Bear Lake were placed in this area based on the lack of trapped gases and potential for water exchange. Although areas of little to no gases were present in these lakes, lake sediments in most of the deeper-water areas of these four lakes had trapped gases (figs. 24, 25).

The CSP penetration depths in the seismic profiles for South School Section Lake generally were less than 2 ft below the water-sediment interface. The lake-bottom sediments in most of the lake consisted of fine clastic sediments with gases, allowing little penetration of the CSP signal (fig. 25*B*).

The CSPs in Lake Elmo indicated the presence of trapped gases in lake sediments at water depths below 82 ft, and in some shallow areas, indicating little water exchange in these areas (fig. 25*C*). Water exchange potentially could happen at shallow depths where trapped gases were not indicated in the CSPs.

## Deep-Water Piezometer and Lake Water-Level Differences at White Bear Lake

The geology in the screened intervals of the piezometers was interpreted from (1) the lithology of lake-sediment cores, (2) CSPs, (3) driller's notes and insights during the driving of the galvanized steel pipe, and (4) existing geologic reports and logs of wells near the shoreline of White Bear Lake (tables 9, 16; fig. 2) (Bauer, 2016). The lake-sediment cores at each of the four sites only were collected to depths that included a part or the entire screened intervals in the shallowest piezometers (figs. 22, 23; table 16). Sediment refusal in the lake-sediment corers prevented collection of sediments to farther depths. At each site, the lake-sediment cores indicated that glacial sands and gravels were present at the screened intervals of the shallowest piezometers (figs. 22, 23; table 16). The depth of penetration of the CSP seismic signals into the lake sediments ranged from 1.0 ft at site P2 to 8.9 ft at site P3. These penetration depths were shallower than the top of the screened intervals of the piezometers at all the sites, except for P1–8.5, P4–6, and P4–8 (table 9). The CSPs at P1 and P4 indicated hard sediments, likely the sands and gravels indicated in the lake-sediment cores at the sites, at the upper parts of the screened intervals of P1–8.5, P4–6, and P4–8 (figs. 22*A*, 23*B*).

Errors in the water-level measurements in the piezometers ranged from ±0.04 to 0.12 ft, and generally were less than the tape-down, water-level measurements below the lake surface in each piezometer (table 10). These errors could have resulted from bending of the casing in the lake and wave action preventing an accurate reading. The measuring point errors associated with the tape-down measurements in the piezometers were calculated based on the mean difference between the measured and calculated stickup values of each tape-down, water-level measurement.

Water-level differences between piezometers and the lake varied seasonally and with depth at the four sites in 2014. Water levels in the lake and piezometers rose from May 30 to early July, gradually decreasing from July to November (fig. 26). The rise between May and July was a result of above-normal precipitation in the spring and early summer of 2014 in the Twin Cities Metropolitan Area (Minnesota Department of Natural Resources, 2016e). Between May 30 and November 7, 2014, water levels in White Bear Lake fluctuated within a 1.1-ft interval between a maximum elevation of 922.2 ft (NAVD 88) on June 28, 2014, and a minimum elevation of 921.1 ft (NAVD 88) on November 7, 2014 (fig. 26). The water level at all piezometer sites ranged from 0.11 ft above the lake surface to 0.37 ft below the lake surface (table 10). Daily minor water-level fluctuations were measured in the continuous piezometer and lake elevations resulting from evaporation from the lake surface and changes in the barometric pressures. Continuous piezometer and lake elevations rose sharply in the summer and fall after large amounts of precipitation.

Water levels in the deepest piezometers at three of the four sites (P1–19.5, P2–30, and P3–18.5) were lower than the lake level throughout the May 30–November 7, 2014, monitoring period, indicating downward vertical hydraulic gradients (fig. 26). The only exception to downward gradients was at site P4, where the screen for the deepest piezometer (P4–8) only was 8 to 11 ft below the water-sediment interface (table 9). The lower water levels in the three deep piezometers relative to the lake indicated a potential for downward flow of lake water into the lake sediments to the locations of the piezometer screens for these piezometers. Water-level differences between the lake and the three deep piezometers ranged from 0.15 to 0.37 ft below the lake surface during the monitoring period (table 10). The depths of the screens for the three deep wells ranged from 18.5 to 33 ft below land surface (lake bottom) (table 9). Water levels in piezometer P4–8 generally were lower than the lake levels between May 30 and August 10, 2014, similar to lake levels between August 10 and September 21, 2014, and higher than lake levels between September 21 and November 7, 2014 (fig. 26*D*).

Water-level differences between the lake and different piezometers in each site varied between the sites during the May 30–November 7 monitoring period. Tape-down water levels in all three piezometers at site P1 were lower than the lake levels (fig. 26*A*), which indicates a potential for lake water to flow downward into the lake sediments over the range of the screen intervals of the three piezometers (table 9). Tape-down water levels in the shallowest piezometer at site P2 (piezometer P2–17.5) were higher or similar to the lake level on the dates measured during the monitoring period (fig. 26*B*), which indicates a small vertical hydraulic gradient between the lake water and pore water at the piezometer screened interval (table 9). Tape-down water levels in the two deeper piezometers at site P2 were lower

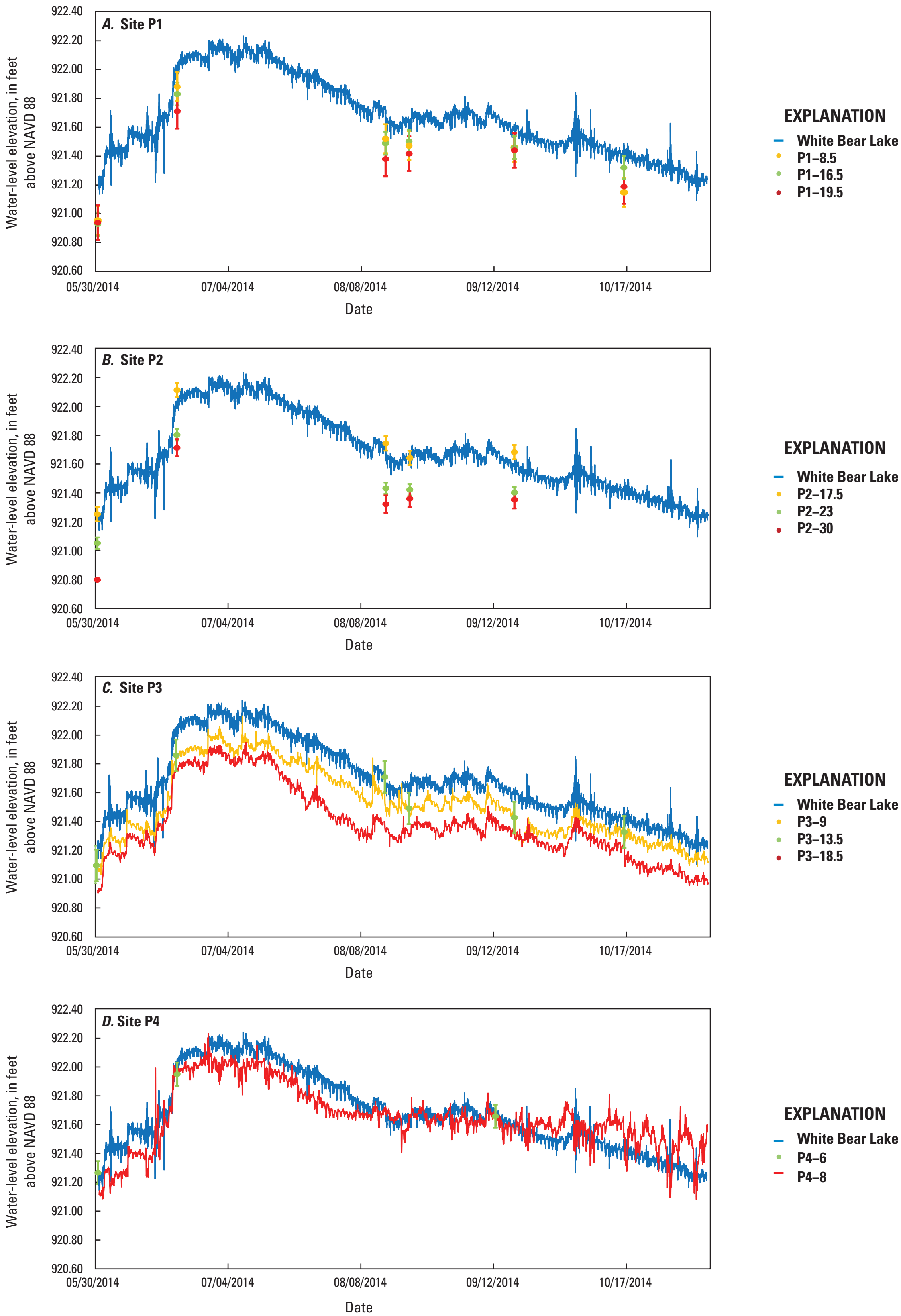



**Figure 26.** Water levels in deep-water piezometer nests installed in White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota, May–November 2014.

than lake levels on the dates measured during the monitoring period (fig. 26*B*), which indicates potential downward flow. The potential for little exchange between lake water and pore water in the lake sediments at site P2 likely is because of 16 ft of low-permeable sapropel, silty sand, and clay between the lake and the piezometer's screened interval at the site (fig. 22*B*; table 16). Water levels in the three piezometers at site P3 were lower than lake water levels during the monitoring period, with the exception of the August 14, 2014, tape-down measurement in piezometer P3–13.5, which was similar to the lake level measured on that day (fig. 26*C*). Water-level differences between the lake and the two piezometers at site P4 generally were small compared to piezometers at the other three sites, and little differences in water levels were measured in the piezometers on the measurement dates (fig. 26*D*). The water-level differences between the lake and piezometers likely were small because the 2 piezometers were the shallowest of the 11 piezometers (table 9); and only 4.2 ft of sapropel, silty clay, and fine sand hydraulically separated the lake from the piezometer screens.

## Hydraulic Conductivities of White Bear Lake Sediments

Hydraulic conductivity values determined from the slug tests completed in piezometers in White Bear Lake (fig. 2) were used to determine the potential for groundwater and surface-water exchanges in deep-water areas of the lake relative to nearshore sites. Hydraulic conductivity values ranged from 0.02 to 3.2 ft/d (table 18). Water-level displacement data with time during falling- and rising-head slug tests in piezometers P1–16.5, P2–30, P3–9, P3–18.5, and P4–8 fit well to the KGS solution method curves. The water-level data collected at each site produced similar water-level displacement data between falling- and rising-head slug tests, except for tests completed in piezometer P4–8.

Analysis of water-level displacement data from the falling-head slug tests completed in piezometer P4–8 produced lower hydraulic conductivity estimates (1.4–1.8 ft/d) than the rising-head slug test hydraulic conductivity estimates (2.6–3.0 ft/d). The differences between falling- and rising-head hydraulic conductivity estimates at piezometer P4–8 may be due to varying presence of finer particles in the screened interval during the tests. These finer particles may have stuck to the screen upon slug introduction, acting as a check-valve during the falling-head test and causing resistance to flow and a lower hydraulic conductivity estimate. During the rising-head tests, the finer particles may have been drawn away from the sides of the screen during slug removal, resulting in a decreased resistance to flow and a higher hydraulic conductivity estimate (Powers and others, 2007).

Slug tests completed in P2–30 produced similar hydraulic conductivity estimates between the rising- and falling-head slug tests (table 18). The falling and rising-head slug tests produced hydraulic conductivity values of 3.1–3.2 and 3.0–3.2 ft/d, respectively. The hydraulic conductivity estimates between falling- and rising-head slug tests were similar in P1–16.5, producing falling-head estimates of 0.6–0.8 ft/day and rising-head estimates of 0.8 ft/day.

**Table 18.** Estimated hydraulic conductivity in selected deep-water piezometers in White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.

[ft, foot; ft/d, foot per day]

| Piezometer number (figs. 22, 23) | Slug test number | Date of slug test | Volume of slug (gallons) | Falling-head measured instantaneous water-level rise (ft) | Rising-head measured instantaneous water-level rise (ft) | Theoretical instantaneous water-level rise (ft) | Hydraulic conductivity | |
|---|---|---|---|---|---|---|---|---|
| | | | | | | | Falling head (ft/d) | Rising head (ft/d) |
| P1–16.5 | 1 | 2/9/2015 | 0.047 | 0.95 | 0.93 | 0.96 | 0.8 | 0.8 |
| P1–16.5 | 2 | 2/9/2015 | 0.047 | 0.99 | 0.94 | 0.96 | 0.7 | 0.8 |
| P1–16.5 | 3 | 2/9/2015 | 0.047 | 0.94 | 0.91 | 0.96 | 0.6 | 0.8 |
| P2–30 | 1 | 2/12/2015 | 0.047 | 0.89 | 0.83 | 0.96 | 3.1 | 3.2 |
| P2–30 | 2 | 2/12/2015 | 0.047 | 0.92 | 0.81 | 0.96 | 3.2 | 3.0 |
| P2–30 | 3 | 2/12/2015 | 0.047 | 0.96 | 0.95 | 0.96 | 3.1 | 3.1 |
| P3–9 | 1 | 2/26/2015 | 0.047 | 0.88 | 0.97 | 0.96 | 0.1 | 0.1 |
| P3–9 | 2 | 2/26/2015 | 0.047 | 0.88 | 0.97 | 0.96 | 0.02 | 0.1 |
| P3–18.5 | 1 | 2/13/2015 | 0.047 | 0.96 | 0.99 | 0.96 | 0.1 | 0.1 |
| P3–18.5 | 2 | 2/13/2015 | 0.047 | 0.96 | 0.97 | 0.96 | 0.1 | 0.1 |
| P4–8 | 1 | 2/25/2015 | 0.066 | 1.46 | 1.52 | 1.44 | 1.8 | 3.0 |
| P4–8 | 2 | 2/25/2015 | 0.066 | 1.26 | 1.36 | 1.44 | 1.4 | 3.0 |
| P4–8 | 3 | 2/25/2015 | 0.066 | 1.82 | 1.31 | 1.44 | 1.4 | 2.6 |

Similar estimated hydraulic conductivities for piezometers P1–16.5, P2–30, P3–9, P3–18.5, and P4–8 may have resulted from the use of a solid slug in a small diameter piezometer (1-in.-inside diameter), which caused uncontrolled water-level bounce during the test initiation and may have resulted in an underdamped (oscillating) response or caused some frictional well loss (Butler, 2002). Variability could also have resulted because of the 1-year delay between the time the piezometers were developed and when the slug tests were completed. An instantaneous water-level change is required for any slug-test method, which results in more accurate early-time water-level data. Matching of early- and later-time water-level data is needed to accurately estimate hydraulic conductivity. Slug tests completed in P1–16.5, P2–30, P3–9, P3–18.5, and P4–8 indicated a maximum instantaneous water-level displacement similar to the theoretical instantaneous water-level displacement (table 18). Because the measured and theoretical instantaneous water-level displacements were similar, the error was small.

## Mini-Piezometer and Seepage-Meter Surveys in White Bear Lake

Hydraulic-head differences measured in August 2014 around White Bear Lake (fig. 2) indicated groundwater was flowing into the lake along the shoreline from shallow aquifers at seven of the nine locations where seepage fluxes and hydraulic-head differences were measured. Hydraulic-head differences (hydraulic head of lake-sediment pore water minus hydraulic head of the lake) measured along the shoreline of White Bear Lake ranged from 0.00 to 0.33 ft (fig. 27; table 11). Positive differences indicated the hydraulic head is higher in the lake sediments, with potential groundwater inflow to the lake, and negative differences indicated the lake level is higher, with potential lake water discharging into surrounding aquifers (Jones and others, 2013). Of the nine sites where hydraulic-head differences were measured, hydraulic-head differences were positive at seven sites and equal to zero at two sites (fig. 27; table 11). Positive hydraulic-head differences were measured at all sites along the east, southeast, and south shore of the lake (fig. 27). Hydraulic-head differences of zero were measured along the northwest shore at Ramsey County Beach and West Ramsey County Beach, and hydraulic-head differences of less than 0.02 ft were measured at Bellaire Beach (fig. 27), indicating minimal to no water movement in the lake sediments. The highest positive hydraulic-head differences (0.33 ft) were measured at Mahtomedi Beach (fig. 27). No negative hydraulic-head differences were measured at any of the nine sites.

Positive or zero median seepage fluxes were measured at all nine sites along the northeast, east, northwest, southeast, and southwest shore of White Bear Lake (fig. 27). Median seepage-meter fluxes at the nine sites along the shores of White Bear Lake ranged from 0.00 to 0.94 ft/d (table 11). Median fluxes greater than or equal to 0.5 ft/d were measured at the White Bear Yacht Club, Mahtomedi Beach, and south of Mahtomedi Beach (fig. 27). Positive hydraulic-head differences were measured at these three sites (table 11), indicating potential groundwater inflow to the lake at these sites. Low positive (less than 0.02 ft/d) to zero median fluxes and zero to minor hydraulic-head differences were measured at West Ramsey County Beach, Ramsey County Beach, and Bellaire Beach (fig. 27; table 11), likely indicating only minor to no groundwater inflow to the lake at the sites.

Vertical hydraulic conductivity values in August 2014 are smaller than values determined in August and September 2011 by Jones and others (2013) for White Bear Lake but are typical of values for sands (Domenico and Schwartz, 1990). Values of vertical hydraulic conductivity determined from the nonzero hydraulic-head differences and median seepage-flux measurements ranged from 0.14 to 36 ft/d (table 11), with a median and a mean of 1.9 and 7.4 ft/d, respectively. Higher lake levels in White Bear Lake in August 2014 (0.7 ft higher) than in August and September 2011 (Minnesota Department of Natural Resources, 2016f) may have resulted either in lower hydraulic-head differences and lower seepage flux at measured sites, or both.

Seepage fluxes measured in the four deep-water sites generally were lower than seepage fluxes measured at the nearshore sites (tables 11, 19), indicating groundwater-outflow rates at the deep-water sites were lower than groundwater-inflow at most of the nearshore sites. Negative seepage fluxes were measured at all four deep-water sites from March 10 to 21, 2014, indicating downward flow of lake water into the lake sediments. The mean of the laboratory-adjusted mean seepage fluxes was -0.5 in/d, and values ranged from -1.0 to 0.2 in/d, whereas the mean of the field-adjusted mean seepage fluxes was -0.3 in/d, and values ranged from -0.8 to 0.4 in/d (table 19). Positive laboratory-adjusted and field-adjusted seepage fluxes were measured during one of two deployments at site P3 on March 16–17, 2014 (table 19), indicating pore-water discharge to the lake from the lake sediments. The magnitude of the seepage fluxes at the deep-water sites are similar to magnitude of seepage fluxes measured along the shoreline at seepage meter sites 2, 5, 6, 7, 8, and 9 (fig. 27; tables 11, 19) but generally differ in the direction of flow.

A large amount of noise was recorded in the seepage flux data, which made data analysis difficult. This noise could have resulted from (1) the operation of the electromagnetic sensor near its minimum output threshold or (2) the vertical movement of the lake's ice surface in response to wind. Averaging the data with time resulted in a much more robust dataset that could be used to evaluate flux during a day.

This study applied multiple hydrologic field methods and statistical methods to gather multiple lines of evidence about the groundwater and surface-water exchanges. The findings described herein provide unprecedented spatial detail on groundwater and surface-water exchanges in White Bear Lake and other lakes in the northeast Twin Cities Metropolitan Area, which is important in assessing the effects of groundwater withdrawals and lake-level augmentation activities on lake levels.

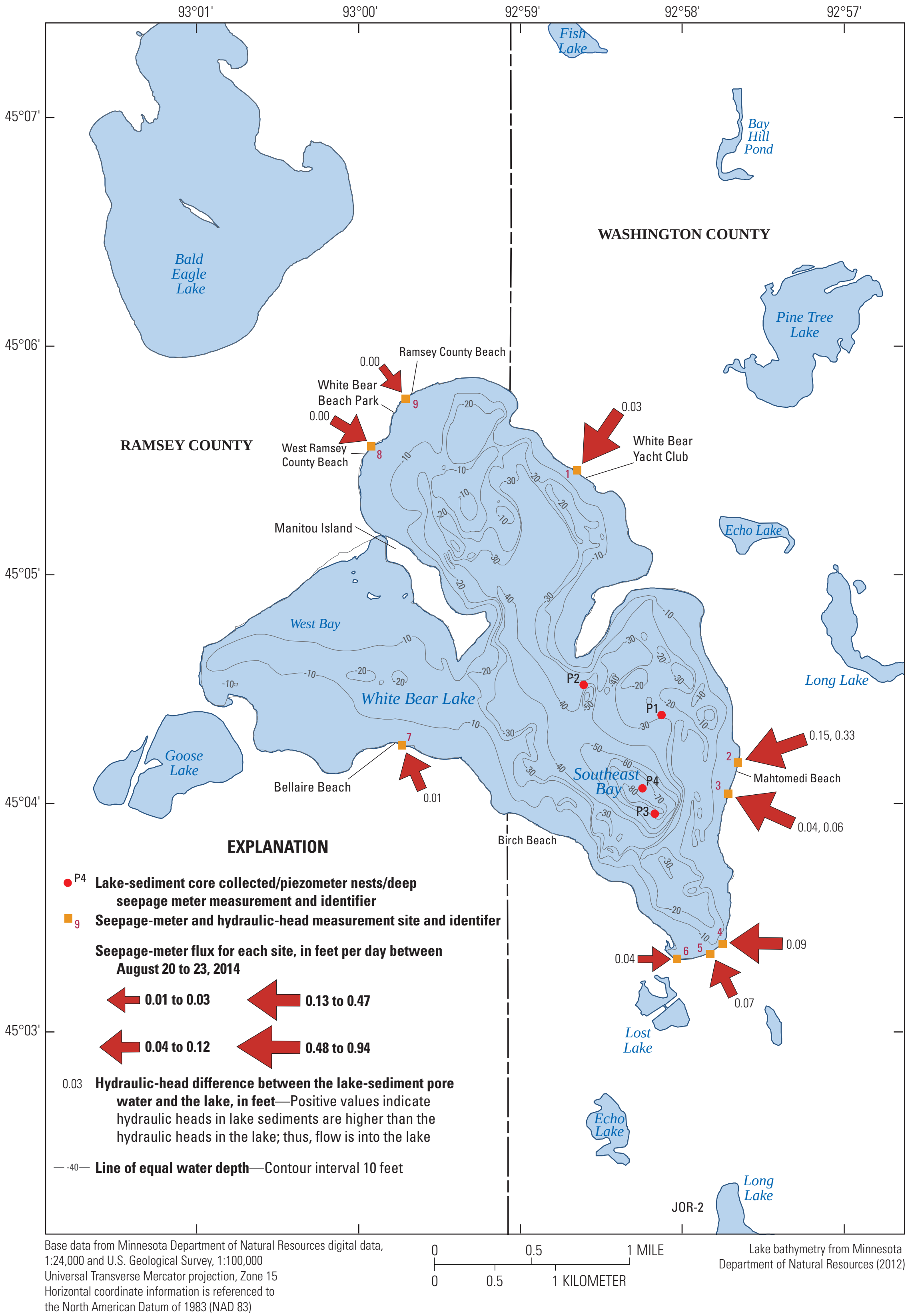



**Figure 27.** Locations for lake-sediment cores, piezometer nests, seepage-meter measurements, hydraulic-head measurements, and line of equal water-depth for White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.

**Table 19.** Seepage-flux measurements and hydraulic-head measurements at deep-water sites on White Bear Lake, northeast Twin Cities Metropolitan Area, Minnesota.

[in/d, inch per day]

| Piezometer site number (fig. 24*A*) | Start date of seepage meter measurement | End date of seepage meter measurement | Lab-adjusted mean seepage flux (in/d)[1] | Field-adjusted mean seepage flux (in/d)[1] | Number of measurements |
|---|---|---|---|---|---|
| P1 | 3/11/2014 | 3/12/2014 | -0.6 | -0.4 | 2,014 |
| P2 | 3/13/2014 | 3/14/2014 | -0.4 | -0.2 | 840 |
| P3 | 3/17/2014 | 3/18/2014 | -0.2 | -0.04 | 1,130 |
| P3 | 3/16/2014 | 3/17/2014 | 0.2 | 0.4 | 1,310 |
| P4 | 3/18/2014 | 3/19/2014 | -1.0 | -0.8 | 950 |
| P4 | 3/14/2014 | 3/15/2014 | -0.7 | -0.6 | 1,230 |

[1]Positive values indicate upward flow of water into the lake. Negative values indicate downward flow of water from the lake into the sediments.

# Implications

Closed-basin lakes, such as White Bear and South School Section Lakes (fig. 1), are more vulnerable to low and fluctuating lake levels than flow-through lakes because closed-basin lakes have more limited sources of water compared to flow-through lakes. These closed-basin lakes only rely on a balance between two main sources of water, direct precipitation and groundwater inflow to the lake, and two components of water loss, evapotranspiration and lake water outflow to aquifers, from small contribution areas to maintain their levels. Water levels in closed-basin lakes tend to reflect groundwater levels in nearby aquifers, which vary in response to multiyear climatic events (Watras and others, 2013; fig. 8 of Jones and others, 2013). Any factor that affects groundwater levels will be reflected in the water levels of closed-basin lakes sooner and more noticeably than in flow-through lakes. Any additional stresses, such as groundwater withdrawals, added to the watershed of the lake can make these lake-level changes more pronounced. Water levels in flow-through lakes are more stable (figs. 9, 10*C*, 11*A*, *B*, *D*) because they typically have larger, more well-connected watersheds than closed-basin lakes that can provide more consistent sources of water to the lakes; however, even flow-through lakes are not immune to level fluctuations, as was demonstrated by the sharp declines in Horseshoe and Little Carnelian Lakes (fig. 15) after water levels dropped below their outflow elevations.

Closed-basin lakes and observation wells in the region extending from White Bear Lake northeast to Goose Lake had the greatest water-level declines during 2002–10 (figs. 1, 9, 12). The combination of high elevation, geologic material, and isolated lake systems make the lakes in this area extremely vulnerable to slight shifts in water inputs and losses. Water levels in these high-elevation, closed-basin lakes likely will continue to be highly responsive to climatic events or other factors that cause consistent year-after-year increases or decreases in groundwater levels. These closed-basin lakes are well connected hydrologically to the groundwater system; therefore, water levels in these lakes will be more responsive to hydrologic factors that affect the groundwater system than water levels in flow-through lakes and low-elevation, closed-basin lakes. Water levels in the high-elevation, closed-basin lakes, for example, likely will be more responsive to nearby groundwater-level changes because of groundwater withdrawals than relatively low-elevation, flow-through lakes in urban areas above Des Moines Lobe glacial deposits. Inlets to and outlets from flow-through lakes were constructed to stabilize water levels in many northeast Twin Cities Metropolitan Area lakes where the percentage of development in the lake's watershed was high.

Groundwater and surface-water exchanges in shallow and deep waters can be an important component of the water balance of a lake and can be important in any lake-level management and lake-augmentation plan. Water balances determined for deep lakes often do not consider groundwater and surface-water exchanges at deep depths, with calculations of exchanges only between shallow groundwater systems and the lake, because it is difficult to quantify exchanges in deep waters. Areas of groundwater flow into and out of lakes commonly is thought to happen in shallow waters around lakes (less exchanges in deep parts of lakes) (Winter and others, 1998). In deep lakes, groundwater from shallow and more regional groundwater systems may be involved in deep-water exchanges.

Deep-water groundwater and surface-water exchanges happen in White Bear Lake and likely result in lake-water contribution to glacial deposits and bedrock aquifers. Stable isotope data collected from wells south of White Bear Lake indicated that surface water is being contributed to the wells downgradient from White Bear Lake, and lake water flowing out of the deeper sections of White Bear Lake likely are a surface-water source. The CSPs, water-level data, and seepage fluxes collected in this study support the conclusion that lake water is flowing out of deep-water parts of White Bear Lake.

Seepage fluxes and water levels measured in nearshore and deep-water sites of White Bear Lake indicated that much of the groundwater inflow is happening at the nearshore sites, and groundwater outflow is happening in deep waters. Seepage fluxes measured in the nearshore sites (table 11) generally were higher than seepage fluxes measured in the deep-water sites (table 19), which indicates that groundwater-inflow rates at most of the nearshore sites are higher than lake-water outflow from the deep-water sites. Groundwater inflow at the nearshore sites was anticipated because groundwater inflow was measured by Jones and others (2013) at the sites in 2001, and the sites were selected based on cool pore-water temperatures determined by Jones and others (2013), which can indicate locations of groundwater inflow.

The CSPs provided valuable real-time insight in identifying areas of potential deep-water exchange between lake water and pore water in the lake sediments of northeast Twin Cities Metropolitan Area lakes. To the authors' knowledge, CSP data have not been applied directly towards determining potential areas of groundwater and surface-water exchange. The CSPs have been used to delineate lake-sediment lithology (Banks and Johnson, 2011) to distinguish clastic sediments from organic-rich materials. Areas of thick organic material with trapped gases present were easily distinguishable from areas with thin layers of organic material and no trapped gases in the CSPs. Studies have determined that trapped gases will further reduce the permeability of low-permeable organic material (Christiansen, 1944; Heilweil and Solomon, 2002); therefore, little groundwater and surface-water exchanges likely happen in lake sediments with trapped gases. Further review of the CSPs with lake-sediment coring and water-level data would be needed to confirm the lack of groundwater and surface-water exchange.

Stable isotope and other water-quality data indicated that surface water is contributing water to wells in the northeast Twin Cities Metropolitan Area, particularly south of White Bear Lake. Oxygen-18/oxygen-16 and deuterium/protium ratios in lake water, precipitation, and well water indicate that there are variations in the contribution of surface water to the sampled wells in the northeast Twin Cities Metropolitan Area (varying from no contribution in 9 of the 40 wells to 100 percent contribution for well 3; fig. 20). The surface waters that are entering these wells may be from lakes or wetlands; however, physical and water-quality data indicate that lakes are the sources of surface-water contribution to some wells. White Bear Lake, for example, is the likely source of the surface water to wells south of the lake with a mixture of surface water and groundwater because (1) it is the only large, deep lake near these wells, (2) these wells are near and downgradient from White Bear Lake, (3) these wells obtain their water from relatively deep depths, and (4) oxygen-18/oxygen-16 and deuterium/protium ratios in water collected from White Bear Lake and the deep-water piezometers in the lake indicate the potential for downward flow of lake water into the lower lake sediments. Otter or Amelia Lakes are likely sources of surface-water contribution to well 3 because the well is close to both lakes and only 154 ft deep. Other wells with surface-water contributions of more than 20 percent are not near any large, deep lakes (for example, wells 16, 23, and 24; fig. 20). Large wetland complexes near these wells may be the sources of surface water to these wells.

Estimating the age of well waters using CFC and $SF_6$ concentrations in well-water samples was difficult in the northeast Twin Cities Metropolitan Area because CFC and $SF_6$ concentrations were high enough in many water samples to indicate potential nonatmospheric and terrigenic sources of contamination. These results demonstrate that these age-dating methods may not be reliable for dating groundwater in the northeast Twin Cities Metropolitan Area, and multiple methods for age dating waters should be used when dating waters if possible.

# Summary

Water levels declined from 2003 to 2011 in many lakes in Ramsey and Washington Counties in the northeastern part of the Twin Cities Metropolitan Area, Minnesota; however, water levels in other northeast Twin Cities Metropolitan Area lakes increased during the same period. Groundwater and surface-water exchanges can be important in determining lake levels where these exchanges are an important component of the water budget of a lake. An understanding of groundwater and surface-water exchanges in the northeast Twin Cities Metropolitan Area has been limited by the lack of hydrologic data. The U.S. Geological Survey, in cooperation with the Metropolitan Council and Minnesota Department of Health, completed a field and statistical study assessing lake-water levels and regional and local groundwater and surface-water exchanges near northeast Twin Cities Metropolitan Area lakes. This report documents the analysis of collected hydrologic, water-quality, and geophysical data; and existing hydrologic and geologic data to (1) assess the effect of physical setting and climate on lake-level fluctuations of selected lakes, (2) estimate potential percentages of surface-water contributions to well water across the northeast Twin Cities Metropolitan Area, (3) estimate general ages for waters extracted from the wells, and (4) assess groundwater inflow to lakes and lake-water outflow to aquifers downgradient from White Bear Lake.

Statistical analyses of lake levels during short-term (2002–10) and long-term (1925–2014) periods were completed to help understand changes in lake levels across the northeast Twin Cities Metropolitan Area. Comparison of 2002–10 lake levels to several landscape and geologic characteristics explained variability in lake-level changes for 96 northeast Twin Cities Metropolitan Area lakes. Application of several statistical methods determined that (1) closed-basin lakes (those lacking an active outlet) had larger lake-level declines than lakes with an outlet; (2) closed-basin lake-level changes reflected groundwater-level changes in the Quaternary, Prairie du Chien, and Jordan aquifers; (3) the installation of outlet-control structures, such as culverts and weirs, resulted in smaller multiyear lake-level changes than lakes without outlet-control structures; (4) water levels in lakes lying primarily above Superior Lobe deposits were significantly more variable than lakes lying primarily above Des Moines Lobe deposits; (5) lake-level declines were larger with increasing mean lake-level elevation; and (6) the frequency of some of these characteristics varies by landscape position. Closed-basin lakes and groundwater levels in the region extending from White Bear Lake northeast to Goose Lake had the greatest water-level declines from 2002 to 2010. Flow-through lakes and lakes with outlet-control structures were more common in watersheds with more than 50 percent urban development compared to watersheds with less than 50 percent urban development. A comparison of two 35-year periods during 1925–2014 revealed that variability of annual mean lake levels in flow-through lakes increased when annual precipitation totals were more variable, whereas variability of annual mean lake levels in closed-basin lakes had the opposite pattern, being more variable when annual precipitation totals were less variable.

Oxygen-18/oxygen-16 and hydrogen-2/hydrogen-1 ratios for water samples from 40 wells indicated the well water was a mixture of surface water and groundwater in 31 wells, whereas ratios from water sampled from 9 other wells indicated that water from these wells receive no surface-water contribution. Of the 31 wells with a mixture of surface water and groundwater, 11 were downgradient from White Bear Lake, likely receiving water from deeper parts of the lake; surface-water contributions ranged from 13 to 58 percent in the 11 wells. White Bear Lake is the likely source of the surface water to these 11 wells with a mixture of surface water and groundwater because (1) these wells are near and downgradient from White Bear Lake, (2) these wells obtain their water from relatively deep depths, and (3) White Bear Lake is the only large, deep lake near to these 11 wells.

Age dating of water samples from wells indicated that the age of water in the Prairie du Chien and Jordan aquifers can vary widely across the northeast Twin Cities Metropolitan Area. Estimated ages of recharge for 9 of the 40 wells sampled for chlorofluorocarbon concentrations ranged widely from the early 1940s to mid-1970s. The wide range in estimated ages of recharge may have resulted from the wide range in the open-interval lengths and depths for the wells. Contamination from nonatmospheric or terrigenic sources prevented the use of sulfur hexafluoride and dichlorodifluoromethane concentrations in water samples from 16 of the 40 wells for age dating.

Results from stable isotope analysis of water samples, lake-sediment coring, continuous seismic-reflection profiling, and water-level and flow monitoring indicated that there is groundwater inflow from nearshore sites and lake-water outflow happens in deep-water sites in White Bear Lake. Continuous seismic-reflection profiling indicated that deep sections of White Bear, Pleasant, Turtle, and Big Marine Lakes have few trapped gases and little organic material, which indicates where groundwater and lake-water exchanges are more likely. Water-level differences between White Bear Lake and piezometers and seepage measurements in deep waters of the lake indicate that groundwater and lake-water exchange is happening in deep waters, predominantly downgradient from the lake and into the lake sediment. Seepage fluxes measured in the nearshore sites of White Bear Lake generally were higher than seepage fluxes measured in the deep-water sites, which indicates that groundwater-inflow rates at most of the nearshore sites are higher than lake-water outflow from the deep-water sites.

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# Appendix

# Appendix 1. Additional Information for Lakes in the Northeast Twin Cities Metropolitan Area

Lake names, Minnesota Department of Natural Resources lake identification numbers, 2002-10 lake-level changes, and variables used for evaluating lake-level changes for 96 lakes of northeast Twin Cities Metropolitan Area lakes, Minnesota are provided in table 1–1. Information on the surface-water inlets and outlets for the 96 lakes assessed in the statistical analysis is provided in appendix table 1–2. Cross references of how surficial geology units were reclassified into sand or till for this study based on the classifications assigned by Meyer (2007) are provided in appendix table 1–3. The Microsoft® Excel file for appendix tables 1–1 through 1–3 is available for download at http://dx.doi.org/10.3133/sir20165139A. The Excel file also contains two worksheet tabs (table1–1_Metadata and table1–2_Metadata) for metadata associated with appendix tables 1–1 and 1–2, respectively.

**Table 1–1.** Lake names, Minnesota Department of Natural Resources lake identification numbers, 2002–10 lake-level changes, and variables used for evaluating lake-level changes for 96 lakes of northeast Twin Cities Metropolitan Area lakes, Minnesota.

**Table 1–2.** Surface-water inlets and outlets for northeast Twin Cities Metropolitan Area lakes, Minnesota.

**Table 1–3.** Table of the reclassification of surficial geology units from Meyer (2007).

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