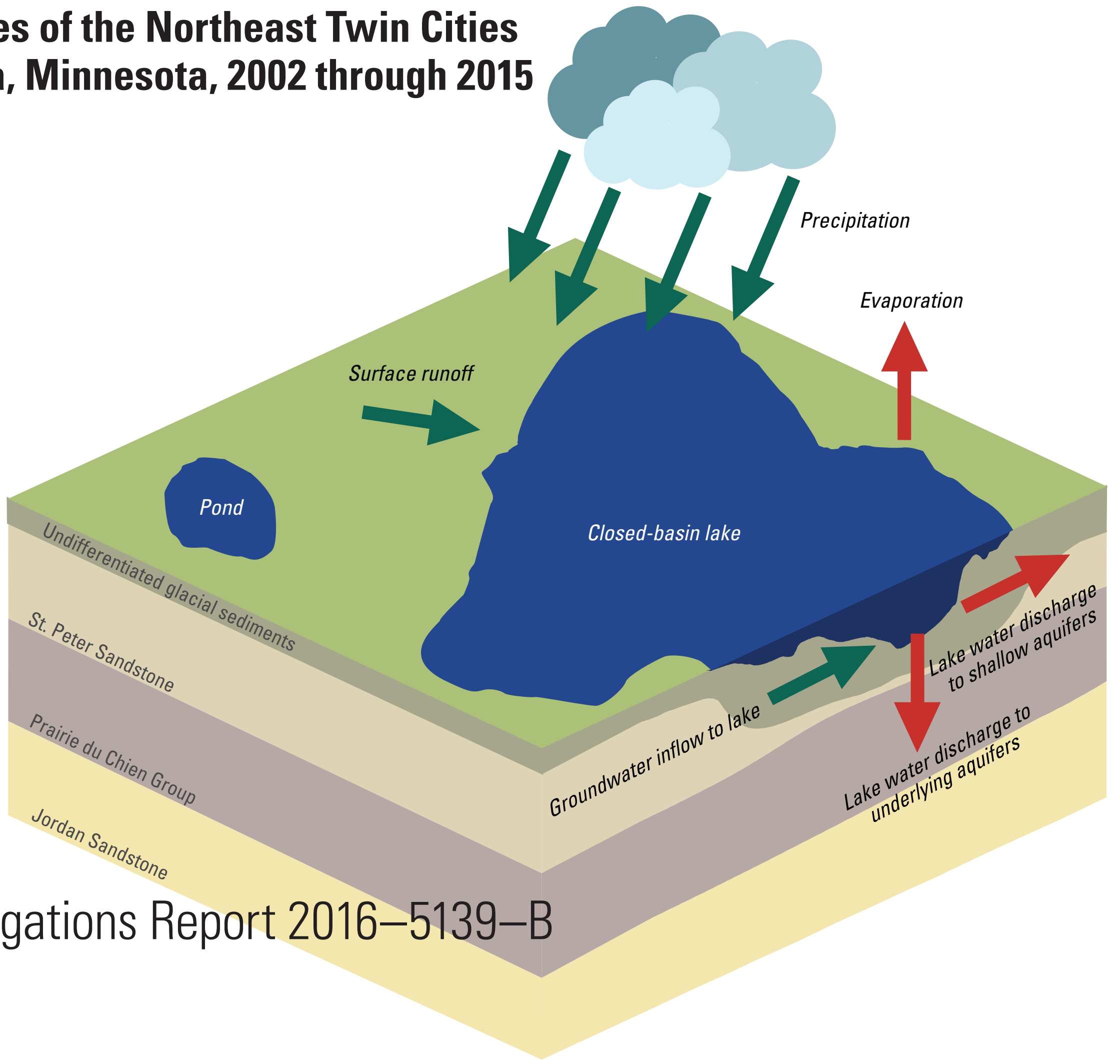
Prepared in cooperation with the Metropolitan Council and Minnesota Department of Health

Simulation and Assessment of Groundwater Flow and Groundwater and Surface-Water Exchanges in Lakes in the Northeast Twin Cities Metropolitan Area, Minnesota, 2003 through 2013

Chapter B 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–B

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

Cover. Schematic showing water-balance components for a closed-basin lake, northeast Twin Cities Metropolitan Area. General schematic of geology based on Swanson and Meyer (1990) and Meyer and Swanson (1992).

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U.S. Department of the Interior
RYAN K. ZINKE, Secretary

U.S. Geological Survey
William H. Werkheiser, Acting Director

U.S. Geological Survey, Reston, Virginia: 2017

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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)
square mile (mi^2)	2.590	square kilometer (km^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)
million cubic feet (Mft^3)	0.02832	million cubic meters (Mm^3)
	Flow rate	
acre-foot per day (acre-ft/d)	0.01427	cubic meter per second (m^3/s)
million cubic feet per day (Mft^3/d)	0.02832	million cubic meters per day (Mm^3/d)
foot per day (ft/d)	0.3048	meter per day (m/d)
cubic foot per day (ft^3/d)	0.02832	cubic meter per day (m^3/d)
gallon per minute (gal/min)	0.06309	liter per second (L/s)
million gallons per day (Mgal/d)	0.003785	million cubic meter per day (Mm^3/d)
million gallons per year (Mgal/yr)	0.003785	million cubic meters per year (Mm^3/yr)
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-water levels and surface-water outlets varied between the lakes. The datums for each lake can be found at Minnesota Department of Natural Resources, 2015.

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

The lake-water levels for the six lakes simulated with the U.S. Geological Survey's modular finite-difference groundwater-flow model (MODFLOW) Lake (LAK) package were converted from their native datum in the Minnesota Department of Natural Resources' Lake Finder web page (Minnesota Department of Natural Resources, 2015c) to the North American Vertical Datum of 1988 (NAVD 88) before their use as calibration targets for the NMLG model. Water levels for Turtle Lake, Snail Lake, and White Bear Lake were available in reference to the Mean Sea Level 1912 (MSL 1912) datum (Minnesota Department of Natural Resources, 2015c). Water levels for Big Marine Lake, Lake Elmo, and Pine Tree Lake were available in reference to the National Geodetic Vertical Datum of 1929 (NGVD 29).

Abbreviations

3D	three dimensional
DEM	digital elevation model
gSSURGO	gridded Soil Survey Geographic database (SSURGO) [dataset]
HSG	hydrologic soil group
HYSEP	hydrograph separation [program]
LAK	Lake [MODFLOW package]
MM3	Metro Model 3
MNDNR	Minnesota Department of Natural Resources
MNW	Multi-Node Well [MODFLOW package]
MODFLOW	modular finite-difference groundwater-flow model
MODFLOW–NWT	Newton formulation of the U.S. Geological Survey's modular finite-difference groundwater-flow model
MWI	Minnesota Well Index
NLCD	National Land Cover Database
NMLG	Northeast Metro Lakes Groundwater-Flow [model]
NSE	Nash-Suttcliffe efficiency
NWIS	National Water Information System
PEST	Parameter Estimation [software]
PHI	total difference between the simulated and calibrate target value [PEST]
PHIMLIM	target measurement objective function [PEST]
RIV	River [MODFLOW package]
RMSE	root mean square error
SWB	Soil-Water-Balance [model]
TauDEM	Terrain Analysis Using Digital Elevation Models
UPW	Upstream Weighting [MODFLOW package]
USGS	U.S. Geological Survey
UZF	Unsaturated-Zone Flow [MODFLOW package]

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The authors also would like to thank the following individuals for their assistance in providing and compiling hydrologic data for the model: Matthew Kocian, Rice Creek Watershed District; Erik Anderson, Washington Conservation District; James Shaver, Carnelian-Marine-St. Croix Watershed District; Alan Rupnow, Ramsey County Public Works; John Manske, Ramsey County Public Works; Peter Boulay and Greg Spoden, Minnesota State Climatological Office; Britta Suppes and Elizabeth Hosch, Capitol Region Watershed District; Brandon J. Barnes, Barr Engineering; Jeremy Walgrave, Limno-Tech; Dave Bucheck and John Hanson, Valley Branch Watershed District; Camilla Correll, Ryan Fleming, and Karen Kill, Brown’s Creek Watershed District; Jamie Schurbon and Andrew Dotseth, Anoka County Conservation District; Tom Wesolowski, City of Shoreview; and Lyn Bergquist, Sandy Fecht, James Solstad, Jenifer Sorenson, and Andrew Williquett, Minnesota Department of Natural Resources.

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Simulation and Assessment of Groundwater Flow and Groundwater and Surface-Water Exchanges in Lakes in the Northeast Twin Cities Metropolitan Area, Minnesota, 2003 through 2013

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This report is chapter B of a two-part report addressing groundwater and surface-water exchanges in the northeast Twin Cities Metropolitan Area. Chapter A of this report (Jones and others, 2016) described a field and statistical study of groundwater and surface-water exchanges and water levels of 96 lakes in the northeast Twin Cities Metropolitan Area, Minnesota, from 2002 through 2015. This report describes the development and application of a three-dimensional, numerical, groundwater-flow model, termed the "Northeast Metro Lakes Groundwater-Flow (NMLG) model," that was used to assess groundwater and surface-water exchanges and the effects of groundwater withdrawals and precipitation on water levels of lakes in the northeast Twin Cities Metropolitan Area.

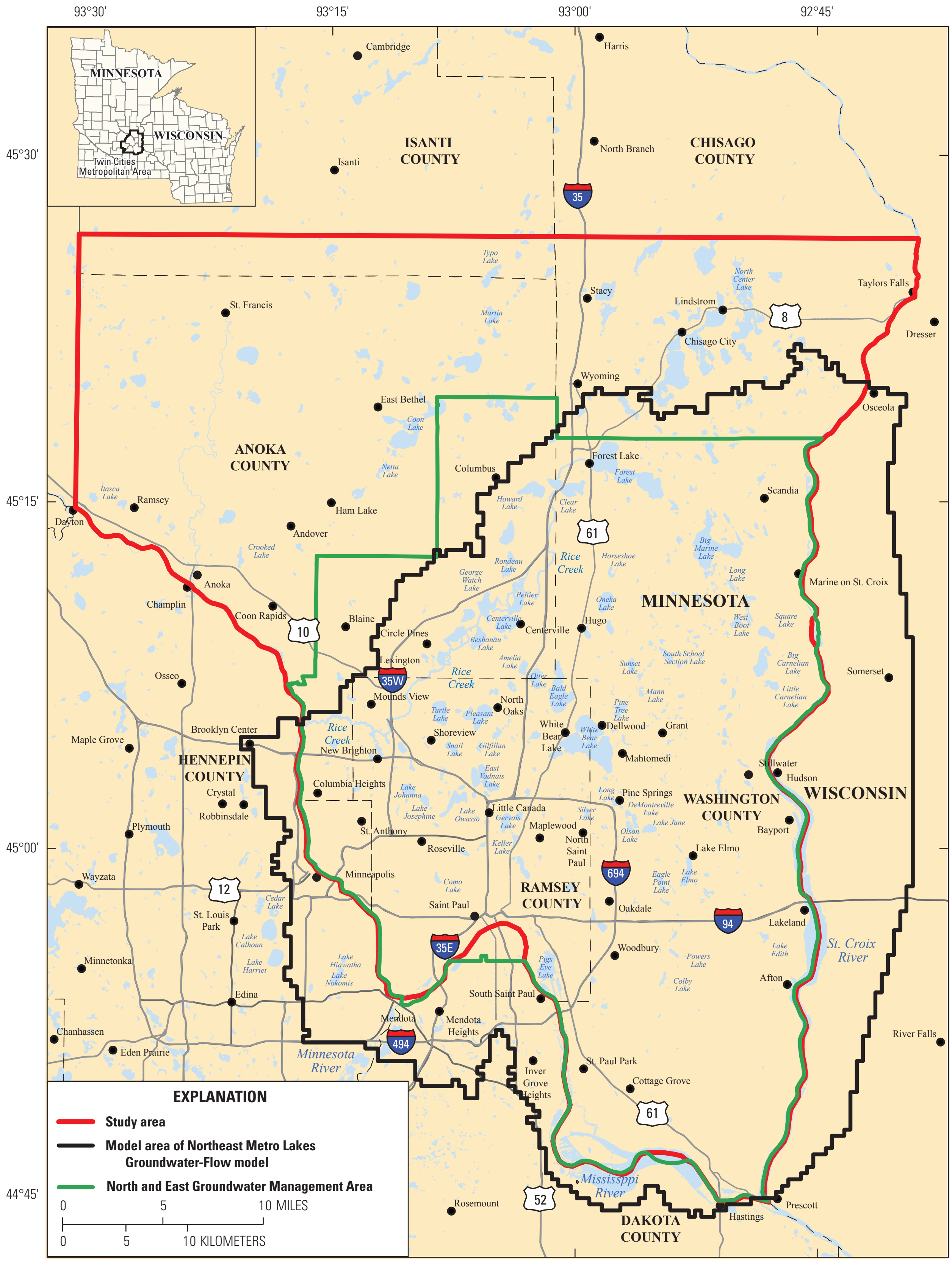
Abstract

Water levels during 2003 through 2013 were less than mean water levels for the period 1925–2013 for several lakes in the northeast Twin Cities Metropolitan Area in Minnesota. Previous periods of low lake-water levels generally were correlated with periods with less than mean precipitation. Increases in groundwater withdrawals and land-use changes have brought into question whether or not recent (2003–13) lake-water-level declines are solely caused by decreases in precipitation. A thorough understanding of groundwater and surface-water exchanges was needed to assess the effect of water-management decisions on lake-water levels. To address this need, the U.S. Geological Survey, in cooperation with the Metropolitan Council and the Minnesota Department of Health, developed and calibrated a three-dimensional, steady-state groundwater-flow model representing 2003–13 mean hydrologic conditions to assess groundwater and lake-water exchanges, and the effects of groundwater withdrawals and precipitation on water levels of 96 lakes in the northeast Twin Cities Metropolitan Area.

Lake-water budgets for the calibrated groundwater-flow model indicated that groundwater is flowing into lakes in the northeast Twin Cities Metropolitan Area and lakes are providing water to underlying aquifers. Lake-water outflow to the simulated groundwater system was a major outflow component for Big Marine Lake, Lake Elmo, Snail Lake, and White Bear Lake, accounting for 45 to 64 percent of the total outflows from the lakes. Evaporation and transpiration from the lake surface ranged from 19 to 52 percent of the total outflow from the four lakes. Groundwater withdrawals and precipitation were varied from the 2003–13 mean values used in the calibrated model (30-percent changes in groundwater withdrawals and 5-percent changes in precipitation) for hypothetical scenarios to assess the effects of groundwater withdrawals and precipitation on water budgets and levels in Big Marine Lake, Snail Lake, and White Bear Lake. Simulated lake-water levels and budgets for Snail Lake and White Bear Lake were affected by 30-percent changes in groundwater withdrawals and 5-percent changes in precipitation in the area, whereas the water level in Big Marine Lake was mainly affected by 5-percent precipita-tion changes. The effects of groundwater withdrawals on the lake-water levels depend on the number of wells and amount of withdrawals from wells near the lakes. Although lake-water levels are sensitive to precipitation changes, increases in groundwater withdrawals during dry periods exacerbate lake-water level declines. The calibrated, groundwater-flow model is a tool that water-resources managers can use to address future water management issues in the northeast Twin Cities Metropolitan Area.

Introduction

Recently (2003–13), water levels were low for sev-eral lakes in the northeast Twin Cities Metropolitan Area, Ramsey and Washington Counties, Minnesota. These lakes include White Bear and Turtle Lakes (fig. 1), which are being



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 the study area, groundwater-flow model area of the Northeast Metro Lakes Groundwater-Flow model, and North and East Groundwater Management Area, northeast Twin Cities Metropolitan Area, Minnesota. Modified from Jones and others (2016).

considered for lake-water-level augmentation (Minnesota Department of Natural Resources [MNDNR], 2016a; Short Elliott Hendrickson Inc., 2015). These low lake-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 (that is, Turtle and Snail Lakes) have recovered to near their ordinary high water levels, whereas others have not (that is, White Bear and South School Section Lakes) (Jones and others, 2016). The ordinary high water level for a lake is defined in Minnesota State Statutes as an elevation representing 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, 2016). Periods of historically lower water levels in White Bear Lake correlate with periods of below-normal precipitation (MNDNR, 1998).

Many hydrologic and physical characteristics affect water levels in lakes. Seasonal and long-term (1925–2014) changes in precipitation and evaporation have resulted in lower lake-water 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 generally have less of an effect on seasonal or annual changes in water levels and storage than on smaller lakes; water-level changes in large lakes generally are driven by snowmelt, precipitation, and evaporation (Wilcox and others, 2007; Watras and others, 2013). Nonetheless, groundwater levels in aquifers can decline, which can reduce water levels in lakes hydrologically connected to those aquifers (Zektser and others, 2005).

Urban expansion and associated land-use and groundwater-withdrawal modifications in Ramsey and Washington Counties brought into question whether or not recent (2003–13) water-level declines in White Bear Lake (fig. 1) and other northeast Twin Cities Metropolitan Area lakes are solely caused by decreases in precipitation during the same period. A U.S. Geological Survey (USGS) cooperative study of White Bear Lake determined that the 2003–11 water-level decline could be explained by changes in several variables and could not be explained based solely on decreases in precipitation (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 and Jordan aquifers could explain the measured change in lake-water level (Jones and others, 2013). The Prairie du Chien and Jordan aquifers are major bedrock aquifers used for water supply in the northeast Twin Cities Metropolitan Area. Annual and summer groundwater withdrawals from these aquifers in the northeast Twin Cities Metropolitan Area more than doubled from 1980 through 2010, primarily because of groundwater withdrawals by municipalities (Jones and others, 2013). Results from Jones and others (2013) indicated that increases in groundwater withdrawals from the Prairie du Chien and Jordan aquifers, coupled with hydraulic connection between the lake and aquifers, may result in increased water outflow from White Bear Lake to the aquifers and decreased groundwater discharge to the lake, resulting in lower water levels in the lake. One limitation of the Jones and others (2013) lake-water-level statistical analysis was that it did not account for the spatial effects of groundwater withdrawals on lake-water levels, which can only be accounted for through use of a groundwater-flow model.

Groundwater and surface-water exchange can be an important factor in lake-water-level and ecosystem management and must be understood to assess the potential effects of management decisions on lake-water 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). Two other lakes (White Bear and Turtle Lakes; fig. 1) are also being considered for lake-water-level augmentation (MNDNR, 2016a; Short Elliott Hendrickson Inc., 2015). Quantification of the water-balance components, including groundwater and surface-water exchanges of these lakes, is needed to assess the effects of any augmentation. An understanding of the effects of groundwater withdrawals on lake-water levels and balances also has been limited by the lack of tools needed to assess these exchanges. To address this need, the USGS, in cooperation with the Twin Cities Metropolitan Council and Minnesota Department of Health, developed a groundwater-flow model to simulated and assess groundwater-flow and groundwater and surface-water exchanges, and the effects of groundwater withdrawals and precipitation changes on lake-water levels in northeast Twin Cities Metropolitan Area lakes during 2003 through 2013.

Purpose and Scope

This report describes the simulation and assessment of groundwater and surface-water exchanges, and the effects of changes in groundwater withdrawals and precipitation on water levels of northeast Twin Cities Metropolitan Area lakes (fig. 1). The report documents the design, construction, calibration, sensitivity, and limitations of a steady-state groundwater-flow model representing mean conditions during 2003–13, a period when lake-water levels were declining in many lakes, such as White Bear Lake (fig. 1). The report presents simulated mean groundwater levels and flows, lake-water levels and budgets for the 2003–13 simulation period, and simulated levels for four lakes determined in eight hypothetical scenarios of different groundwater withdrawals and precipitation. This report further describes the steady-state response of lake and groundwater levels to (1) hypothetical increases and decreases in groundwater withdrawals in the northeast Twin Cities Metropolitan Area, and (2) the effects of hypothetical precipitation differences with the potential future increases and decreases in municipal groundwater withdrawals.

This report is chapter B of a two-part report addressing groundwater and surface-water exchanges in the northeast Twin Cities Metropolitan Area. Chapter A of this report (Jones and others, 2016) described a field and statistical study of groundwater and surface-water exchanges, and water levels of 96 lakes in the northeast Twin Cities Metropolitan Area, Minnesota, from 2002 through 2015. This report describes the development and application of a three-dimensional, numerical groundwater-flow model, termed the “Northeast Metro Lakes Groundwater-Flow (NMLG) model” in this report.

Description of Study Area, Hydrology, Geology, and Hydrogeology

The hydrology, geology, and hydrogeology of the study area (fig. 1) are described in Jones and others (2013, 2016) and briefly in the following sections. The northeast Twin Cities Metropolitan Area, which is the study area in this report, is a gently rolling to flat glaciated landscape in Ramsey, Washington, south Chisago, northeast Hennepin, and southeast Anoka Counties (fig. 1). Land use in Ramsey, northeast Hennepin, and south Anoka Counties is mainly developed land, and in the northern, eastern, and southeastern parts of the study area land use consists of a mixture of developed land, crop/pasture, forest, and wetlands (fig. 2).

Hydrology

The surface-water hydrology of the study area consists of rivers, streams, lakes, and wetlands; and much of the surface water eventually discharges to the Mississippi and St. Croix Rivers (fig. 1). About 300 lakes, and many more attached and isolated wetlands, streams, and rivers, are present in the study area. Some of the larger lakes include Bald Eagle, White Bear, Forest, and Big Marine Lakes (fig. 1). These larger lakes and rivers are used extensively for recreation, including fishing, boating, and swimming.

A strong correlation exists between temporal variations in water levels of closed-basin lakes, such as White Bear Lake, and local aquifers (MNDNR, 1998; Jones and others, 2016), indicating that groundwater contribution to these lakes is an important factor affecting lake-water levels. Groundwater and lake-water exchanges are common at shallow depths where more permeable sediments commonly are present and organic sediment thicknesses are less than thicknesses at deeper depths (Winter and others, 1998); however, stable isotope analyses of water samples, lake-sediment coring, continuous seismic-reflection profiling, and water-level and flow monitoring done by Jones and others (2016) indicated that lake water is outflowing from deep-water sites in White Bear Lake (fig. 1).

Geology and Hydrogeology

The geology and hydrogeology of the study area consists of Precambrian, Cambrian, and Ordovician bedrock (figs. 3, 4; table 1) that underlie unconsolidated Pleistocene glacial sediments of pre-Wisconsin and Wisconsin age (fig. 5), forming the northern part of the Twin Cities Basin (fig. 4). Swanson and Meyer (1990), Meyer and Swanson (1992), Setterholm (2010, 2013), Jones and others (2013), Bauer (2016), and MNDNR (2016b) also provide detailed descriptions of the geology and hydrogeology of the study area. Ordovician- and Cambrian-age bedrock underlies glacial deposits, or is exposed at the land surface, in southern Washington County (figs. 4, 5). The Cambrian and Ordovician-aged sedimentary bedrock formations in the study area contain several major bedrock aquifers and confining units, including the St. Peter aquifer, Prairie du Chien aquifer, and Jordan aquifer. These aquifers are used extensively as sources of water to wells in southeast Minnesota (Runkel and others, 2003a). Groundwater in Pleistocene glacial sediments is used mostly for domestic or commercial sources of water in the northeast Twin Cities Metropolitan Area. The following descriptions of the geologic formations and corresponding hydrogeologic units in the study area are described from oldest to youngest in terms of the geologic formations.

Paleozoic bedrock formations overlie Middle Proterozoic-age volcanics and sedimentary rocks throughout most of the study area. The volcanic sequences of Keweenawan Supergroup (Chengwatana volcanics, Clam Falls volcanics, North Branch volcanic sequence, and Powder Mill volcanic sequence) mainly consist of basalts with some conglomerate beds (Bauer, 2016; Morey, 1972; Setterholm, 2010) and are part of the Midcontinent Rift that extends from Kansas northeast to Lake Superior. Clam Falls Volcanics and Chengwatana Volcanics of Keweenawan Supergroup of Middle Proterozoic age (Wirth and others, 1997) underlie glacial deposits and are exposed at the land surface along a series of faults associated with horsts and grabens near the St. Croix River in Chisago County (figs. 1, 4). Middle Proterozoic-age red to reddish brown sedimentary rocks of the Solor Church Formation of Keweenawan Supergroup, Fond du Lac Formation of Keweenawan Supergroup, and Hinckley Sandstone of Keweenawan Supergroup overlie the volcanics (Bauer, 2016; Morey, 1972; Setterholm, 2010). The Solor Church Formation of Keweenawan Supergroup consists of reddish-brown shale interbedded with reddish-brown feldspathic sandstone, whereas the Fond du Lac Formation of Keweenawan Supergroup consists of poorly sorted, red sandstone with some shale (Morey, 1972). The Hinckley Sandstone of Keweenawan Supergroup is a pale-red to light-pink, well-sorted sandstone (Morey, 1972) that is hydraulically connected to the overlying Mount Simon Sandstone of Dresbach Group, forming the Mount Simon-Hinckley aquifer (Runkel and others, 2003a).

The Mount Simon Sandstone of Dresbach Group of Late Cambrian age is the lowest bedrock unit of the Paleozoic bedrock formations of southeastern Minnesota and the study area (fig. 3; table 1) (Swanson and Meyer, 1990; Meyer and Swanson, 1992). The lower part of the Mount Simon Sandstone of Dresbach Group is principally a fine- to coarse-grained sandstone that is moderately to poorly cemented. The upper part

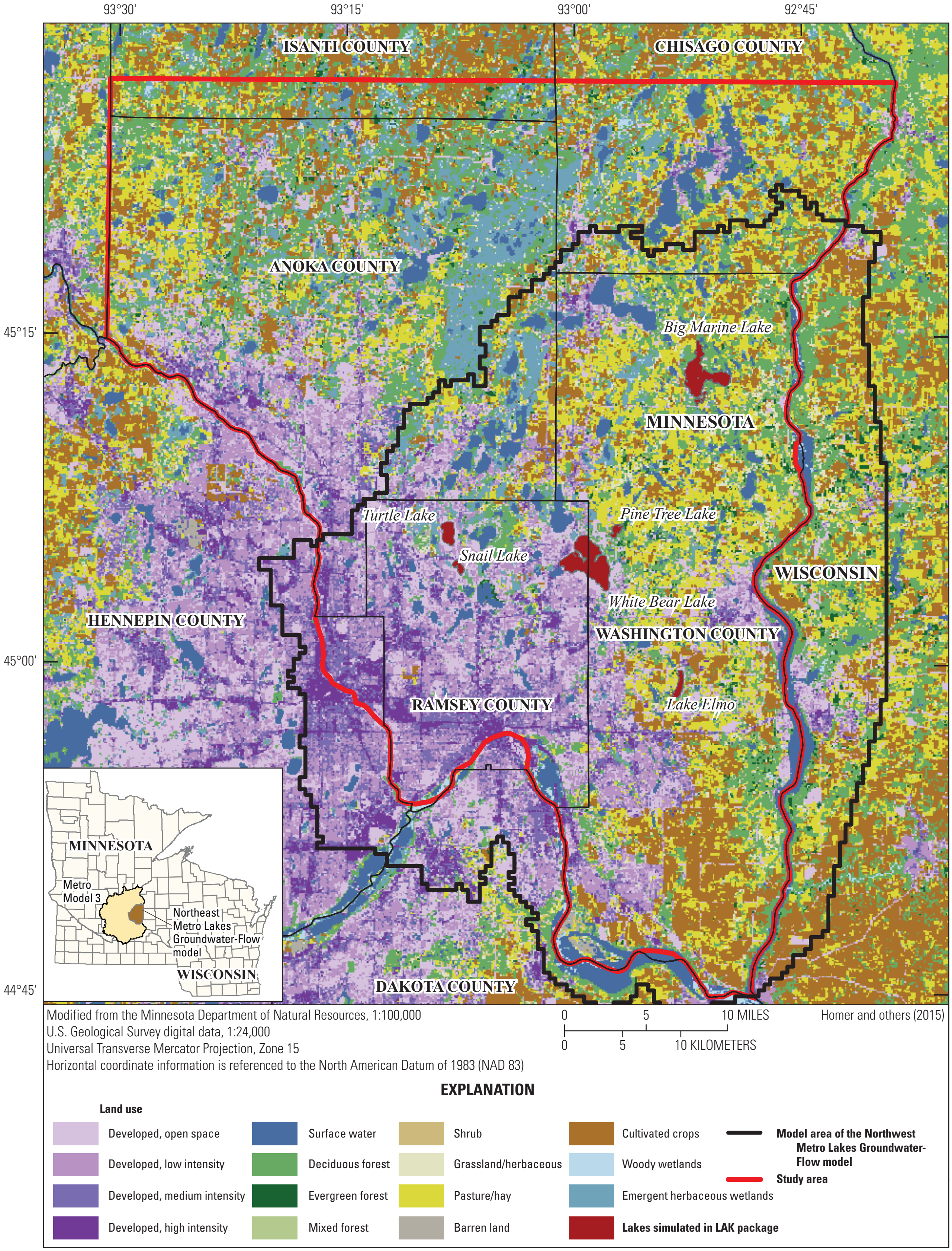


Figure 2. Land cover and lakes simulated in the U.S. Geological Survey's modular finite-difference groundwater-flow model (MODFLOW) Lake (LAK) package of the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota. Modified from Jones and others (2016).

Erathem	System	Formation or Group	General lithology	Thickness, in feet	Hydrogeologic unit and hydraulic properties	Primary groundwater model layer numbers, and model layer numbers where present as the secondary unit (range in parentheses)[1,2]
Cenozoic	Quaternary	Undifferentiated glacial sediments		0–100	**Glacial aquifers**—Horizontal hydraulic conductivities for unconsolidated sediments range from 323 to 240 feet per day (ft/d), mean of 79 ft/d; vertical hydraulic conductivities for unconsolidated sediments range from 321 to 88 ft/d, mean of 47 ft/d; horizontal hydraulic conductivities for till range from 33.3×10^{-5} to 26 ft/d.	1–4 (5–11)
Paleozoic	Ordovician	Decorah Shale		40	**Decorah-Platteville-Glenwood confining unit**—The vertical hydraulic conductivity ranges from about 1×10^{-5} to 1×10^{-4} ft/d; horizontal hydraulic conductivites for the Decorah Shale range from less than 1×10^{-6} to 98 feet/day; the Platteville Formation range from $1x10^{-2}$ to 98 ft/d, and the Glenwood Formation are approximately $1x10^{-2}$ ft/d.	1–4
		Platteville Formation		25		
		Glenwood Formation		5		
		St. Peter Sandstone		100	**St. Peter aquifer**—The aquifer is a major aquifer in southeastern Minnesota; horizontal hydraulic conductivities range from 1×10^{-3} to greater than 49 ft/d,vertical hydraulic conductivities range from 2×10^{-3} to 92 ft/d; effective porosity ranges from 0.28 to 0.3; storativity ranges from 9×10^{-5} to 9.8×10^{-3}.	5 (1–4)
		Prairie du Chien Group: Shakopee Formation		130	**Prairie du Chien aquifer, Shakopee Formation**—The aquifer is a part of a major aquifer in southeastern Minnesota; has well-developed secondary porosity; horizontal hydraulic conductivities range from 1.0 to 160 ft/d; vertical hydraulic conductivities range from 0.03 to 35 feet/day; has a well-developed secondary porosity; effective porosity of 0.06, but highly variable; storativity ranges from 1.1×10^{-5} to 3.4×10^{-5}.	6 (1–5)
		Prairie du Chien Group: Oneota Dolomite		170	**Prairie du Chien aquifer, Oneota Dolomite**—The aquifer is part of major aquifer in southeastern Minnesota; horizontal hydraulic conductivities range from 1.5×10^{4} to 740 ft/d, with the lower conductivities in the middle and lower parts of the unit; vertical hydraulic conductivities range from 1.5×10^{4} to 10^{-3} ft/d; the uppermost part of the unit has a well-developed secondary porosity; effective porosity of 0.06, but highly variable; storativity ranges from 1.1×10^{5} to 3.4×10^{4}.	
	Cambrian	Jordan Sandstone		100	**Jordan aquifer**—The aquifer is a major aquifer in southeastern Minnesota; horizontal hydraulic conductivities range from 1×10^{-2} to greater than 490 ft/d; vertical hydraulic conductivities range from 1×10^{-4} to 47 ft/d; effective porosity of 0.32; storativity ranges from 4.9×10^{-5} to 1.2×10^{-4}.	7 (1–6)
		St. Lawrence Formation		75	**St. Lawrence confininfg unit**—Parts of the formation are confining, other parts are used for water supplies; horizontal hydraulic conductivities range from less than 1×10^{-2} to 46 ft/d; vertical hydraulic conductivities range from 1×10^{-5} to 1.8 ft/d; effective porosity ranges from 0.15 to 0.20.	8 (1–7)
		Tunnel City Group		155 – 160	**Tunnel City aquifer**—The upper part of the group, the Mazomanie Formation, is an aquifer and the lower part is considered a confining unit; the Mazomanie Formation consists of a fine- to medium-grained sandstone that generally forms more than 20 percent of the Tunnel City Group in the northeast Twin Cities Metropolitan Area; hydraulic conductivities range from less than 1 to 340 ft/d, with conductivities varying with fracturing and depth, hydraulic conductivities are greater at shallower depths.	9 (1–8)
		Wonewoc Formation		40 – 45	**Wonewoc aquifer**—A single aqufer with parts of the upper part of the formation considered confining; hydraulic conductivities range from 1 to 230 ft/d, varying with fracturing.	10 (5–9)
		Eau Claire Formation		200	**Eau Clarie confining unit**—Vertical and horizontal hydraulic conductivities in deep bedrock settings range from 1×10^{-6} to 1×10^{2} ft/d, hydraulic conductivities in shallow bedrock settings range from less than 1 to 400 ft/d, sandstones in the unit are used as local aquifers.	11 (6–10)
		Mount Simon Sandstone of Dresbach Group		160 – 340	**Mount Simon-Hinckley aquifer**—The aquifer is used by some large high-capacity wells in northeast Twin Cities Metropolitan area. The fine clastic material in upper part of the Mount Simon Sandstone of Dresbach Group forms a confining unit in deep bedrock settings, the lower part of the formation consists of coarse clastic material that serves as an aquifer with the Hinckley Sandstone. The Hinckley Sandstone consists of reddish-brown mudstones and siltstones, interbedded with reddish-brown feldspathic sandstone. Hydraulic conductivities for the aquifer range from less than 1 to 200 ft/d, with the higher conductivities generally present in shallow bedrock settings, hydraulic conductivities in deep bedrock settings (greater than 200 feet below the land surface) generally range from less than 1 to 50 ft/d.	12 (10–11)
		Hinckley Sandstone of Keweenawan Supergroup				

[1]The primary and secondary layer numbers for each hydrogeologic unit will vary with the presence and absence of other hydrogeologic units over the area. Units commonly are absent in areas where faulting and buried bedrock valleys are present.
[2]Values in parentheses represent the range of model layers where the unit is the secondary unit in the layer.

EXPLANATION

Till, sand, and gravel | Limestone/dolomite | Shale | Sandstone | Siltstone | Aquifer

Figure 3. Generalized hydrogeologic column of regional aquifers and confining units and their use in the groundwater-flow model, northeast Twin Cities Metropolitan Area, Minnesota. [Geology modified from Balaban (1988) and Lindgren (2001); hydrogeologic properties from Delin (1991), Metropolitan Council (2016), Meyer and Swanson (1992), Mossler (2008), Runkel and others (2003a), and Swanson and Meyer (1990)]

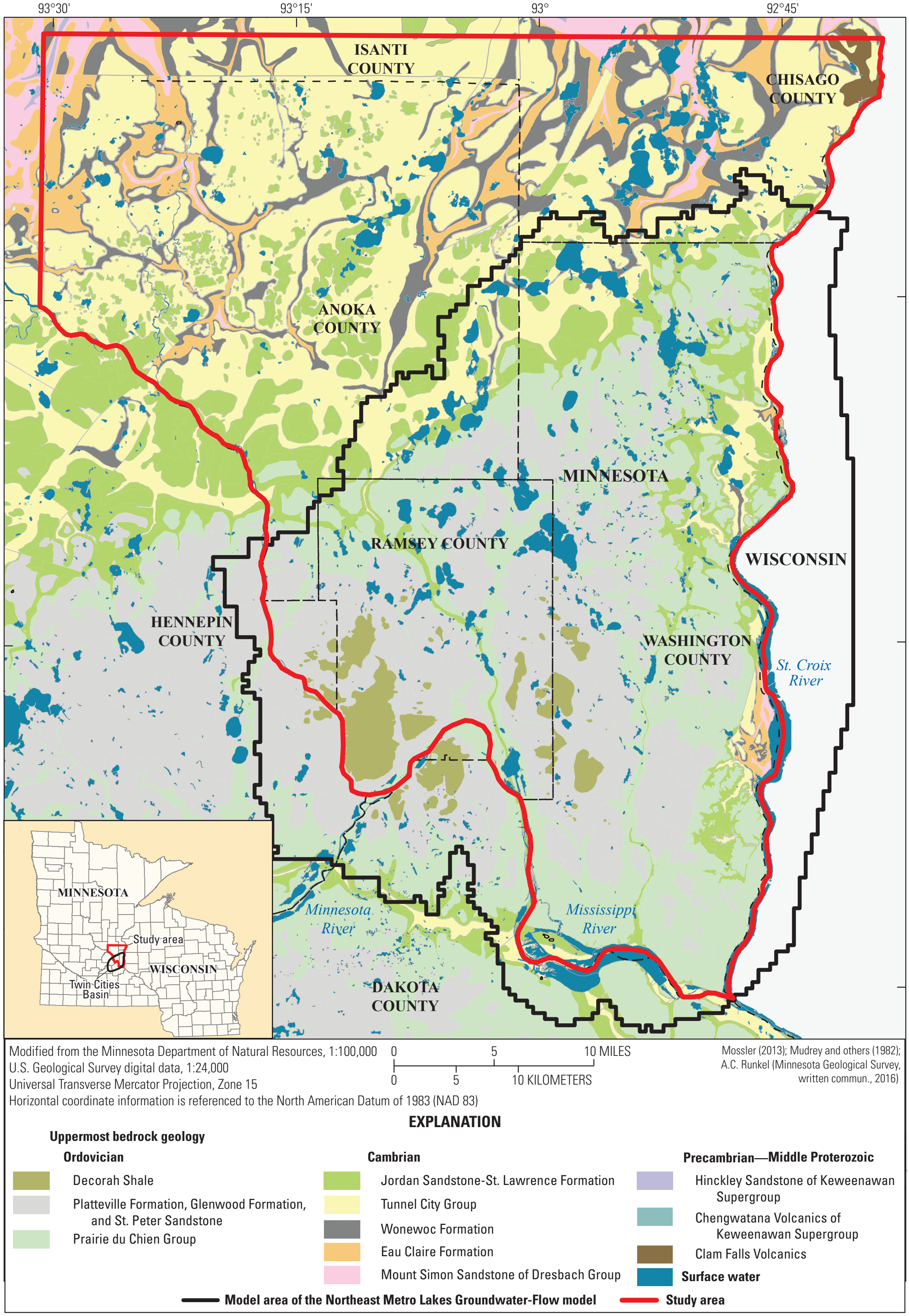


Figure 4. Uppermost bedrock geology of the study area, northeast Twin Cities Metropolitan Area, Minnesota. Modified from Jones and others (2016).

Table 1. Geologic units and aquifers in the northeast Twin Cities Metropolitan Area, Minnesota.

[Modified from Jones and others (2016). Aquifer nomenclature follows the geologic nomenclature of the U.S. Geological Survey. ft, foot; MM3, Metro Model 3; ft/d, foot per day]

System	Series	Geologic unit (aquifer)	Lithology	General thickness (ft)	Water-bearing characteristics
Quaternary	Pleistocene	Glacial lacustrine sediments (New Ulm and New Brighton Formations, Grantsburg sublobe of the Des Moines Lobe deposits)	Fine to medium sand, silt, and clay[1,2,3]	[1,2]Less than 50 to 250	Groundwater extraction mainly by commercial and domestic wells; horizontal hydraulic conductivities for unconsolidated sediments in MM3 range from [4]23 to 240 ft/d, mean of 79 ft/d; vertical hydraulic conductivities for unconsolidated sediments in MM3 range from [4]21 to 88 ft/d, mean of 47 ft/d; horizontal hydraulic conductivities for till range from [4]3.3×10^{-5} to 26 ft/d.
	Pleistocene	Ice-contact stratified deposits (New Ulm and New Brighton Formations, Grantsburg sublobe of the Des Moines Lobe deposits)	Sand, loamy sand, and gravel, interbedded with silt and glacial till[1,2,3]		
	Pleistocene	Glacial till (New Ulm and New Brighton Formations, Grantsburg sublobe of the 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[1,2,3]		
	Pleistocene	Glacial outwash (Cromwell Formation, Superior Lobe deposits)	Sand, loamy sand and gravel, commonly overlain by loess[2,3]		
	Pleistocene	Glacial lacustrine sediments (Cromwell Formation, Superior Lobe deposits)	Silt to medium-grained sand, interbedded with reddish-brown to reddish-gray silty clay and gravelly sand[2,3]		
	Pleistocene	Glacial till (Cromwell Formation, 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[2,3]		
	Pleistocene	Glacial outwash and ice-contact deposits (pre- to late Wisconsin, Keewatin)	Sand, loamy sand, and gravel, some lacustrine silt and clay[2,3]		
	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[2,3]		
	Pleistocene	Glacial till (Illinoian, River Falls Formation)	Complex mixture of sandy clay loams and sandy loams with gravels, sands, and cobbles[3]		
	Early Pleistocene	Glacial outwash (Pierce Formation)	Sand and gravels, well-compacted loams with gravel[3]		
Paleozoic	Ordovician (Middle)	Decorah Shale	Green calcareous shale interbedded with thin limestone[5]	[1,2]0–95	Produces little water; horizontal hydraulic conductivities range from [5]less than 1×10^{-6} ft/d (deep depths) to 98 ft/d (shallow depths).
	Ordovician (Middle)	Platteville Formation	Fine-grained dolostone and limestone[5]	[1,2]25–33	Produces little water; water flows mainly through bedding planes and vertical fractures. Horizontal hydraulic conductivities range from [5]less than 1×10^{-2} ft/d (deep depths) to 98 ft/d (shallow depths).
	Ordovician (Middle)	Glenwood Formation	Thin, green sandy shale[5]	[1,2]3–6	Produces little water; horizontal hydraulic conductivity of [5]1×10^{-2} ft/d (shallow and deep depths).
	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[5]	[1,2]146–166	Major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from 1×10^{-3} ft/d to greater than 49 ft/d; vertical hydraulic conductivities range from 2×10^{-3} to 92 ft/d; effective porosity ranges from [4]0.28 to 0.3; storativity ranges from [4]9×10^{-5} to 9.8×10^{-3}.

Table 1. Geologic units and aquifers in the northeast Twin Cities Metropolitan Area, Minnesota.—Continued

[Modified from Jones and others (2016). Aquifer nomenclature follows the geologic nomenclature of the U.S. Geological Survey. ft, foot; MM3, Metro Model 3; ft/d, foot per day]

System	Series	Geologic unit (aquifer)	Lithology	General thickness (ft)	Water-bearing characteristics
Paleozoic	Ordovician (Lower)	Prairie du Chien Group—Shakopee Formation (Prairie du Chien aquifer)	Thin to medium beds of dolostone, shale, and some siliciclastic sandstone[5]	[1,2]119–203 (Prairie du Chien Group)	Major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from [4,5]1.0 to 160 ft/d; vertical hydraulic conductivities range from [4,5]0.03 to 35 ft/d; effective porosity of [4]0.06; storativity ranges from [4]1.1×10^{-5} to 3.4×10^{-4}.
	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[5]		Part of major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from [5]1.5×10^{-4} to 740 ft/d; vertical hydraulic conductivities range from [5]1.5×10^{-4} to 10^{-3} ft/d; effective porosity of [4]0.06; storativity ranges from [4]1.1×10^{-5} to 3.4×10^{-4}.
	Cambrian (Upper)	Jordan Sandstone (Jordan aquifer)	Coarse and fine clastic sandstone[5]	[1,2]66–101	Major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from 1×10^{-2} to greater than [4,5]490 ft/d; vertical hydraulic conductivities range from [4,5]10^{-4} to 47 ft/d; effective porosity of [4]0.32; storativity ranges from [4]4.9×10^{-5} to 1.2×10^{-4}.
	Cambrian (Upper)	St. Lawrence Formation	Interbedded fine clastic (sandstone, siltstone, shale) and carbonate (dolostone) rock[5]	[1,2]30–60	Horizontal hydraulic conductivities range from less than [4,5]1×10^{-2} to 46 ft/d; vertical hydraulic conductivities range from [4,5]1×10^{-5} to 1.8 ft/d; effective porosity ranges from [4]0.15 to 0.20.
	Cambrian (Upper)	Tunnel City Group (formerly Franconia Formation[6])	Shale, siltstone, and fine-grained sandstone with beds of limestone and dolostone[5]	[1,2]116–166	Aquifer in southeast Minnesota; horizontal hydraulic conductivities range from less than [4,5]1×10^{-3} to 340 ft/d; vertical hydraulic conductivities range from [4,5]1×10^{-4} to 9.8 ft/d.
	Cambrian (Upper)	Wonewoc Formation (formerly the Ironton and Galesville Sandstones[6])	Silty, fine- to coarse-grained poorly sorted sandstones in the upper part, fine- to medium-grained sandstone in the lower part[5]	[1,2]42–67	Aquifer in southeast Minnesota; horizontal hydraulic conductivities range from [4,5]0.2 to 230 ft/d; vertical hydraulic conductivities range from [4,5]1×10^{-3} to 8 ft/d; effective porosity of [4]0.25; storativity ranges from [4]2.7×10^{-5} to 5.9×10^{-5}.
	Cambrian (Upper)	Eau Claire Formation	Siltstone, fine- to medium-grained glauconitic sandstones, and shales[5]	[1,2]63–114	Horizontal hydraulic conductivities range from less than [4,5]1×10^{-3} to 400 ft/d; vertical hydraulic conductivities range from [4,5]1×10^{-6} to 3×10^{-3} ft/d; effective porosity ranges from [4]0.28 to 0.35.
	Cambrian (Upper)	Mount Simon Sandstone of Dresbach Group	Fine- to coarse-grained quartzose sandstone, thin beds of siltstone, shale, and very fine-grained sandstone[5]	[1,2]160–336	Major aquifer in southeast Minnesota; horizontal hydraulic conductivities range from [4,5]1×10^{-2} to 200 ft/d; vertical hydraulic conductivities range from [4,5]1.10^{-4} to 14 ft/d; effective porosity of [4]0.23.
Precambrian	Proterozoic (Middle)	Hinckley Sandstone of Keweenawan Supergroup	Reddish-brown mudstones and siltstones, interbedded with pale-red to light pink feldspathic sandstone[7]	[8]More than 0.6 mile	Hydraulically connected to Mount Simon Sandstone forming a major aquifer; hydraulic conductivities and other hydrologic parameters unknown.
	Proterozoic (Middle)	Fond du Lac Formation of Keweenawan Supergroup	Poorly sorted, red sandstone with some shale[7]	Unknown	Not used as a source of water in the northeast Twin Cities Metropolitan Area; hydraulic conductivities and other hydrologic parameters unknown.
	Proterozoic (Middle)	Solor Church Formation of Keweenawan Supergroup	Reddish-brown shale interbedded with reddish-brown feldspathic sandstone[7]	Unknown	Not used as a source of water in the northeast Twin Cities Metropolitan Area; hydraulic conductivities and other hydrologic parameters unknown.
	Proterozoic (Middle)	Chengwatana Volcanics of Keweenawan Supergroup	Basalt, porphyritic, interlayered with conglomeratic sedimentary rock[8,9]	Unknown	Generally not used as a source of water in the northeast Twin Cities Metropolitan Area; hydraulic conductivities and other hydrologic parameters unknown.
	Proterozoic (Middle)	Clam Falls Volcanics, North Branch volcanic sequence, and Powder Mill volcanic sequence of Keweenawan Supergroup	Coarse-grained, ophitic basalt, with fine-grained, intergranular basalt and porphyritic basalt[8,9]	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 Bauer (2016).

[4]From Metropolitan Council (2016).

[5]From Runkel and others (2003a).

[6]From Mossler (2008).

[7]From Morey (1972).

[8]From Setterholm (2010).

[9]From Boerboom (2001).

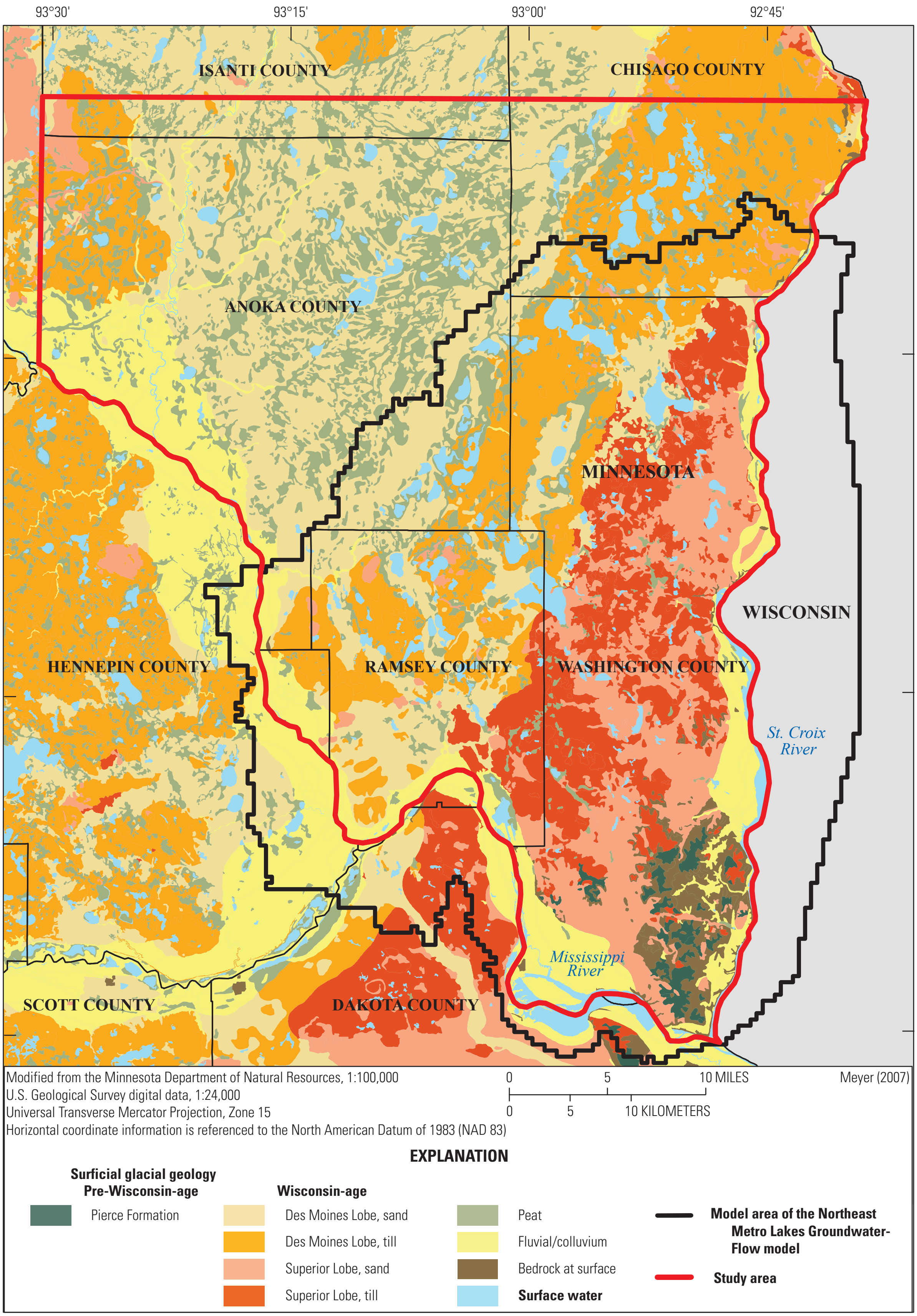


Figure 5. Surficial glacial geology of the study area, northeast Twin Cities Metropolitan Area, Minnesota. Modified from Jones and others (2016).

of the Mount Simon Sandstone of Dresbach Group consists of thin beds of siltstone and shale and very fine-grained sandstone (Runkel and others, 2003a). The corresponding hydrogeologic unit is referred to as the Mount Simon-Hinckley aquifer and is used by some large high-capacity wells in the northeast Twin Cities Metropolitan area (fig. 3).

Siltstones, fine- to medium-grained glauconitic sandstones, and shales of the Eau Claire Formation, of Late Cambrian age, overlie the Mount Simon Sandstone of Dresbach Group (fig. 3; table 1) (Swanson and Meyer, 1990; Meyer and Swanson, 1992). The upper part of the Eau Claire Formation mainly consists of shales and siltstones, and the lower part mainly consists of sandstones and siltstones (Mossler, 1992). The Eau Claire Formation commonly is considered a confining unit, but sandstones within the formation have been used for water supplies where it is shallow and fractured (Runkel and others, 2003b); the corresponding hydrogeologic unit is referred to as the Eau Claire confining unit (fig. 3).

The Wonewoc Formation, of Late Cambrian age, overlies the Eau Claire Formation and is divided into two parts: (1) an upper silty, fine- to coarse-grained, poorly sorted sandstone; and (2) a lower, more sorted, fine- to medium-grained sandstone (fig. 3; table 1) (Swanson and Meyer, 1990; Meyer and Swanson, 1992). The upper part of the formation is considered to be a single aquifer, and the lower part is considered to be a confining unit. The corresponding hydrogeologic unit is referred to as the Wonewoc aquifer (fig. 3).

The Tunnel City Group of Late Cambrian age overlies the Wonewoc Formation, and consists of shales, siltstones, and fine-grained sandstones with beds of carbonate strata (fig. 3) (Runkel and others, 2003a). The upper member of the group, the Mazomanie Formation, is a fine- to medium-grained sandstone that generally forms more than 20 percent of the Tunnel City Group in the northeast Twin Cities Metropolitan Area (Runkel and others, 2006) and is widely used as an aquifer in the Twin Cities Metropolitan Area. This upper member is a minor component of the group outside of the Twin Cities Metropolitan Area. The corresponding hydrogeologic unit is referred to as the Tunnel City aquifer (fig. 3) in this report.

The St. Lawrence Formation, of Late Cambrian age, lies between two formations that are important aquifers in the Twin Cities Metropolitan Area: the Jordan Sandstone and the Tunnel City Group (fig. 3; table 1). Parts of the St. Lawrence Formation are considered confining, and other parts are used for water supply. The formation consists of interbeds of the fine clastic and carbonate rocks, mostly sandstones, siltstones, dolostones, and shales (Mossler, 2008). Hydrologic studies of the St. Lawrence Formation determined that groundwater can travel at high flow rates through bedding planes and fractures in the formation (Green and others, 2008, 2010; Runkel and others, 2006); however, the corresponding hydrogeologic unit is referred to as the St. Lawrence confining unit in this report (fig. 3).

The Jordan Sandstone, of Late Cambrian age, consists of coarse to fine clastic sediments, with fractures at various depths below the land surface (Runkel and others, 2003a). The Jordan Sandstone contains the Jordan aquifer, which is an important source of water to wells in the Twin Cities Metropolitan Area. Values of hydraulic conductivity for the Jordan aquifer generally are greater and more variable at shallow depths, where groundwater flow through fractures is more prevalent (Runkel and others, 2003a).

The Prairie du Chien Group, St. Peter Sandstone, Glenwood Formation, Platteville Formation, and Decorah Shale of Ordovician age are the youngest bedrock formations in the study area (fig. 3; table 1). The Prairie du Chien Group, of Early Ordovician age, overlies the Jordan Sandstone and contains the Prairie du Chien aquifer. The Prairie du Chien Group consists of two formations: the Oneota Dolomite and the Shakopee Formation. The Oneota Dolomite is primarily thick beds of very fine-grained dolostone, and fine and coarse clastic interbeds are common in the lower part of the formation (Runkel and others, 2003a). The overlying Shakopee Formation consists of thin to medium beds of dolostone, shale, and minor amounts of siliciclastic sandstone (Runkel and others, 2003a). Solution-enhanced cavities along bedding planes and fractures are pronounced in the Shakopee Formation and along its contact with the Oneota Dolomite (Runkel and others, 2003a). These karst features have been identified and mapped throughout southeast Minnesota (Gao and others, 2002, 2005); most of this work was done south of the northeast Twin Cities Metropolitan Area. The Prairie du Chien aquifer is an important source of water to wells in the study area. Where karst features are present, the Prairie du Chien aquifer is sensitive to contamination (Minnesota Pollution Control Agency, 1999; Yingling, 2015). The farthest northern extent of the Prairie du Chien Group in the study area is in northern Washington and southern Anoka Counties (figs. 1, 4).

The St. Peter Sandstone of Middle Ordovician age unconformably overlies the Prairie du Chien Group (fig. 3; table 1). The upper one-half to two-thirds of the St. Peter Sandstone consists of fine- and medium-grained, quartz sandstone that is massive to thick bedded; the lower part of the formation consists of multicolored mudstones, siltstones, and shales interbedded with very coarse sandstone (Swanson and Meyer, 1990; Meyer and Swanson, 1992). The St. Peter Sandstone contains the St. Peter aquifer, which is an important source of water for southeastern Minnesota. The St. Peter aquifer can be hydraulically connected to the underlying Prairie du Chien and Jordan aquifers (Delin, 1991; Minnesota Pollution Control Agency, 1999); however, water levels in the St. Peter aquifer and the Prairie du Chien and Jordan aquifers indicate that the St. Peter aquifer is hydraulically separated from the underlying Prairie du Chien and Jordan aquifers (Runkel and others, 2003a).

The Glenwood Formation, Platteville Formation, and Decorah Shale of Middle Ordovician age overlie the St. Peter Sandstone (fig. 3; table 1). These geologic units generally are not used for sources of water to wells and are considered, collectively, as the Decorah-Platteville-Glenwood confining unit. The Glenwood Formation consists of thin, green, sandy shales ranging from 3 to 6 feet (ft) thick, and the overlying Platteville

Formation consists of fine-grained dolostone and limestone about 25 ft thick (Swanson and Meyer, 1990; Meyer and Swanson, 1992). The Decorah Shale overlies the Platteville Formation as thin caps in the southern parts of the study area and consists of green, calcareous shale interbedded with thin limestone (Swanson and Meyer, 1990; Meyer and Swanson, 1992).

The glacial and postglacial geology of the study area (fig. 5) is complex because numerous Pleistocene- and Holocene-age depositions and erosions formed the surficial landscape (Bauer, 2016). The oldest glacial deposits known in the study area are early Pleistocene-age (pre-Wisconsin age) deposits of the Pierce Formation (Bauer, 2016) (table 1). The Pierce Formation consists of sand and gravels deposited by glacial melt waters, and well-compacted loams with gravel deposited directly by glacial ice of the Winnipeg provenance earlier than about 781,000 years before the present (Bauer, 2016). Glacial sediments of the Pierce Formation are present at the land surface in southern Washington County (Bauer, 2016) (fig. 5). Advances and retreats of late Pliestocene glaciers eroded or buried much of the earlier glacial sediments. Late Pleistocene-age glacial sediments deposited during the Illinoian glaciation (about 191,000 to 130,000 years before present) and Wisconsin glaciation (about 33,000 to 13,300 years before present) cover bedrock and the Pierce Formation. Late Pleistocene-age glacial sediments of the River Falls Formation were deposited during the Illinoian glaciation (table 1). The formation consists of a complex mixture of sandy clay loams and sandy loams with gravels, sands, and cobbles (Bauer, 2016). The formation is deeply weathered and eroded, and thus is discontinuous in the study area.

The surficial geology mostly consists of glacial tills, sands, and gravels deposited by Wisconsin-age Superior Lobe and glacial sandy lake sediments, sands, gravels, and tills associated with the Grantsburg sublobe of the Wisconsin-age Des Moines Lobe (fig. 5; table 1) (Swanson and Meyer, 1990; Meyer and Swanson, 1992; Bauer, 2016). The Superior Lobe advanced, stagnated, and retreated over the study area in several Wisconsin-age phases (Wright and others, 1973). Glacial sediments of the Cromwell Formation were deposited by the Superior Lobe (table 1) during the St. Croix phase (about 20,000 to 18,000 years before present), forming the St. Croix moraine and other features that control the topography (Bauer, 2016). The formation mainly consists of (1) reddish-brown to reddish-gray gravelly loamy sand and sandy loam (glacial tills), forming hummocky topography in parts of the study area; (2) bedded outwash sand and gravels forming low hills; and (3) fine- to medium-grained and gravelly sand, silt, and clay formed in glacial lakes filled with melt waters (ice-walled lake plains) (Bauer, 2016). The Grantsburg sublobe, a branch of the Des Moines Lobe that flowed northeast-southwest over the study area, overrode the St. Croix moraine and Cromwell Formation about 19,000 to 14,450 years before present (Bauer, 2016), incorporating older sediments from the Superior Lobe deposits. Glacial sediments of the New Ulm and New Brighton Formations were deposited by the Grantsburg sublobe during this period (table 1). The New Ulm Formation mainly consists of (1) yellowish-brown to gray fine- to medium-grained sand, silt, and clay deposited in glacial lakes; (2) outwash sands and gravels; and (3) brownish-gray to gray, gravelly sandy loam to loam till (Bauer, 2016; Setterholm, 2013). The New Brighton Formation mainly consists of yellowish-brown to gray, fine- to medium-grained sand with some gravel deposits laid down as the lake bed of glacial Lake Anoka (Setterholm, 2013). Late-Wisconsin silt, clay, and sands were deposited as wind-blown loess, ice-contact deposits, outwash, lake deposits, and terrace deposits, respectively, above earlier glacial deposits (Bauer, 2016; Setterholm, 2013). Organic materials (peat) associated with wetlands and lakes and fluvial deposits commonly are on the land surface above the glacial deposits (fig. 5).

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 surfaces, and these melt waters deposited glacial sands and gravels and reworked till in the valleys (Bauer, 2016; Winter and Pfannkuch, 1976). Many northeast Twin Cities Metropolitan Area lakes, including Big Marine Lake, Lake Elmo, Turtle Lake, Snail Lake, and White Bear Lake (fig. 1), overlie buried bedrock valleys. These lakes may have resulted from the melting of remnant, buried ice blocks that settled and filled in the valleys (Bauer, 2016). Where the buried bedrock valleys are filled with permeable sands and gravels, the valleys can be areas of focused groundwater discharge to major rivers (Bauer, 2016). The buried bedrock valleys also can be sources of recharge to deep aquifers where the valleys incise confining units (Bauer, 2016).

Pleistocene- and Holocene-age depositions and erosions have resulted in discontinuous, mixed, and interfingered layers of pre-Wisconsin age, Wisconsin-age, and post-Wisconsin sediments. Sediment layers for the different glacial depositional events have been developed mainly based on sediment compositions (Swanson and Meyer, 1990; Meyer and Swanson, 1992; Setterholm, 2013; Bauer, 2016). Groundwater flow in these sediments is complex as sediments of different compositions and ages form glacial water-table and buried aquifers (Tipping, 2011; Minnesota Department of Natural Resources, 2016b). Various groundwater-flow modeling (Metropolitan Council, 2016) and water-quality techniques (Tipping, 2011) have been applied to determine the hydraulic properties (fig. 3; table 1) and the movement of groundwater flow in these complex aquifers.

Groundwater in the glacial aquifers generally moves through pores and fractures in interfingered sediment layers, and groundwater in bedrock aquifers moves through pores, fractures, and solution channels. Groundwater in glacial and bedrock aquifers in the study area generally flows south or west to the Mississippi River or to the east to the St. Croix River (fig. 1). Potentiometric maps for aquifers contained in glacial sediments and Ordovician-age formations for part of the study area indicated relatively high groundwater levels in an area between White Bear Lake and Big Marine Lake, with groundwater flowing toward the Mississippi or St. Croix Rivers (Jones and others, 2013). Groundwater flows are particularly high in solution channels present in the Shakopee Formation and along its contact with the Oneota Dolomite that compose the Prairie du Chien aquifer (Blum, 2015; Groten and Alexander, 2015; Yingling, 2015).

Hydrologic Properties of Aquifers and Confining Units

The hydraulic properties of aquifers and confining units, including horizontal and vertical hydraulic conductivities of the aquifers and confining units in the Twin Cities area, have been estimated by others (fig. 3; table 1) through aquifer testing in wells and boreholes, specific-capacity tests in wells, and groundwater-flow modeling. Horizontal hydraulic conductivities of the sand and gravel aquifer generally range from 21 to 240 feet per day (ft/d), and horizontal hydraulic conductivities for glacial till generally range from 3.0×10^{-6} to 26 ft/d (table 1; Tipping, 2011). Vertical hydraulic conductivity values for glacial sediments generally are one to three orders of magnitude less than horizontal hydraulic conductivity values for the same glacial sediments (Metropolitan Council, 2016).

Hydraulic conductivity values for each of the bedrock aquifers and, in particular, confining units tend to vary with depth and amount of fracturing (Runkel and others, 2003a). Horizontal hydraulic conductivities in the bedrock aquifers generally ranged from 1×10^{-3} to 740 ft/d (table 1); vertical hydraulic conductivities in the bedrock aquifers generally ranged from 1×10^{-4} to 92 ft/d. Hydraulic conductivities in the confining units generally were much lower than values from the bedrock aquifers at deeper depths but similar at shallow depths where fractures were commonly abundant (Runkel and others, 2003a). Horizontal hydraulic conductivities in the confining units generally ranged from less than 1×10^{-6} to 98 ft/d (table 1); vertical hydraulic conductivities in the confining units generally ranged from 1×10^{-7} to 1.8 ft/d (Metropolitan Council, 2016; table 1). The hydraulic properties of and groundwater-flow conditions in Middle Proterozoic sedimentary, volcanic, and mafic intrusive bedrock formations below the Hinckley Sandstone are poorly known (Bauer, 2016; Metropolitan Council, 2016).

Previous Investigations

Many hydrologic studies have assessed groundwater and surface-water exchanges in northeast Twin Cities Metropolitan Area lakes, some of which are described in Jones and others (2013, 2016). These studies have mainly focused on assessing the amount of groundwater inflow and lake-water outflow in nearshore environments and assessing the contribution of groundwater inflow to lake water budgets (Jones and others, 2016).

Groundwater movement in glacial and bedrock aquifers in the study area has been assessed through the delineation of potentiometric surfaces and application of groundwater-flow models. Norvitch and others (1973) delineated potentiometric surfaces for the winter of 1970–71, groundwater-level changes from December 1970 to August 1971, and long-term groundwater-level changes from winter 1965 to winter 1970 for the Prairie du Chien and Jordan aquifers to assess natural flow conditions and the effects of groundwater withdrawals on the aquifer. Larson-Higdem and others (1975) described vertical groundwater leakage rates from overlying aquifers to the Prairie du Chien and Jordan aquifers in the Twin Cities Metropolitan Area using potentiometric surfaces developed by Norvitch and others (1973). The estimate by Larson-Higdem and others (1975) of vertical leakage accounts for 10–20 percent of the increases in summer groundwater withdrawals in 1971 in the Twin Cities Metropolitan Area. Guswa and others (1982) and Schoenberg (1984) described groundwater flow for 1971–80 and changes in groundwater levels for the 1980s in the Prairie du Chien and Jordan aquifers. Horn (1983) investigated annual and seasonal groundwater-withdrawal rates from the Prairie du Chien and Jordan aquifers between 1880 and 1980 in the Twin Cities Metropolitan Area. Delin and Woodward (1984) delineated a potentiometric surface for the Prairie du Chien and Jordan aquifers for southeastern Minnesota, including the Twin Cities Basin. Schoenberg (1990) assessed the effect of groundwater-withdrawal rates on groundwater levels in the Twin Cities Metropolitan Area in the 1970s. Andrews and others (1995) investigated groundwater-level declines and associated groundwater withdrawals from the Prairie du Chien and Jordan aquifers from 1980 to 1990. Sanocki and others (2008) determined potentiometric surfaces and groundwater-level changes in the Prairie du Chien and Jordan aquifers, the Franconia-Ironton-Galesville aquifer, and the Mount Simon-Hinckley aquifer in the Twin Cities Metropolitan Area for March and August 2008; note that the formations containing the Franconia-Ironton-Galesville aquifer in Sanocki and others (2008) are referred to as the Tunnel City Group (Franconia Formation) and Wonewoc Formation (Ironton and Galesville Sandstones) (table 1, fig. 3). Jones and others (2013) developed potentiometric surfaces for the study area to assess groundwater flow and groundwater and surface-water interactions between glacial and bedrock aquifers near White Bear Lake (fig. 1). The MNDNR has used groundwater-level data to construct potentiometric surfaces for a series of glacial aquifers and major bedrock aquifers in Anoka County (MNDNR, 2016b) and the glacial aquifer in the Anoka Sand Plain (MNDNR, 1993)

Jones and others (2016) described a field and statistical study of groundwater and surface-water exchanges and water levels of 96 lakes in the northeast Twin Cities Metropolitan Area, Minnesota, for 2002 through 2015. Continuous seismic-reflection profiling done in this study 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. Results from the continuous seismic-reflection profiling in six lakes (Big Marine Lake, Lake Elmo, Pleasant Lake, South School Section Lake, Turtle Lake, and White Bear Lake, fig. 1) and slug tests completed in piezometers installed in White Bear Lake were used to establish zones of different lake-bed conductances for the six lakes. These zones were used in the NMLG model.

Several regional groundwater-flow models have been constructed to assess groundwater-flow conditions and the effects of groundwater withdrawals on groundwater levels in glacial and bedrock aquifers in the study area. The Metropolitan Council's Metro Model 3 (MM3) (Metropolitan Council, 2016) was developed to assess the effects of potential regional management scenarios on projected groundwater levels in the 11 counties of the Twin Cities Metropolitan Area, to identify areas possibly facing future water supply limitations, based on land-use changes, population growth, and water demand changes. A surface-water and groundwater-flow model was developed for southern Washington County (fig. 1) to evaluate aquifer sustainability of the Prairie du Chien and Jordan aquifers and the effects of groundwater withdrawals on surface waters (Barr Engineering Company and Washington County, 2005). Earlier regional groundwater-flow models in the study area designed to assess the effects of groundwater withdrawals on groundwater levels in bedrock aquifers include the Metropolitan Area Groundwater Model (Minnesota Pollution Control Agency, 2014) and the Twin Cities Metropolitan Area numerical groundwater-flow model (Schoenberg, 1990; Lindgren, 1996).

Methods of Model Development

A numerical, three-dimensional, steady-state, numerical groundwater-flow model, the NMLG model, was developed using the USGS's modular finite-difference groundwater-flow model (MODFLOW), Newton formulation (MODFLOW–NWT) program (Niswonger and others, 2011). MODFLOW–NWT is a Newton formulation variant of MODFLOW–2005 (Harbaugh, 2005), which is a numerical, three-dimensional, finite-difference groundwater modeling program. The Newton formulation described in this report extends the applications of MODFLOW–2005, especially to those problems representing unconfined aquifers and groundwater and surface-water exchanges (Niswonger and others, 2011). MODFLOW–NWT uses many of the packages distributed with MODFLOW–2005; however, additional packages used by MODFLOW–NWT are either modified MODFLOW–2005 packages or new packages. A steady-state, groundwater-flow model simulates stable groundwater conditions during a period where there is no change in groundwater levels or flow over time. Steady-state flow was simulated to represent 2003–13 mean groundwater-flow conditions.

The NMLG model incorporates an area of about 1,000 square miles (mi^2) in Ramsey, Washington, south Chisago, northeast Hennepin, and southeast Anoka Counties in Minnesota, and parts of western Wisconsin (fig. 1). The NMLG model was developed based in part on the parameter data and model designs of the Metropolitan Council's MM3 groundwater-flow model (Metropolitan Council, 2016). The MM3 was commissioned by the Metropolitan Council to provide understanding of regional groundwater processes and is used as a starting point for subsequent groundwater analyses in the Twin Cities Metropolitan Area (fig. 1). The MM3 covers an area of about 8,350 mi^2 and uses the MODFLOW–2005 model code (Harbaugh, 2005) to simulate groundwater-flow conditions throughout the 11 counties in and around the Twin Cities Metropolitan Area for 2003–13.

The NMLG model was developed and calibrated to represent mean hydrologic conditions in the study area during 2003–13. Groundwater-flow conditions in the NMLG model were simulated using a seven-step approach: (1) extract model layering and hydraulic property data from the MM3 model; (2) compile additional existing hydrologic and geologic data needed to construct the groundwater-flow model to provide more detailed assessment of groundwater and surface-water exchanges in glacial sediments; (3) compile precipitation and other climatic data needed for the Soil-Water-Balance (SWB) model (Westenbroek and others, 2010); (4) calibrate and run the SWB model to determine recharge and surface-runoff estimates used to determine recharge and runoff to lakes, respectively, for the groundwater-flow model during 2003–13; (5) discretize the compiled data using a finer grid than the MM3 model; (6) calibrate the NMLG model to mean 2003–13 conditions by comparing simulated and measured groundwater levels, lake-water levels, and river base-flow rates using the Parameter Estimation (PEST) calibration software (Doherty, 2010; Doherty and Hunt, 2010); (7) analyze how sensitive simulated groundwater levels, lake-water levels, and river base-flow rates were to changes in model parameters; and (8) run eight hypothetical groundwater-flow scenarios using different groundwater withdrawals and recharge, which results from different amounts of precipitation. The eight hypothetical scenarios include (1) a 30-percent increase and (2) a 30-percent decrease in mean 2000–13 groundwater withdrawals using 2000–13 mean precipitation; (3) 2000–13 mean groundwater withdrawals, (4) a 30-percent increase, and (5) a 30-percent decrease in 2000–13 mean groundwater withdrawals using a 5-percent increase in 2000–13 mean precipitation; and (6) 2000–13 mean groundwater withdrawals, (7) a 30-percent increase, and (8) a 30-percent decrease in 2000–13 mean groundwater withdrawals using a 5-percent decrease in 2000–13 mean precipitation. The SWB model is a hydrologic model that produces spatially distributed estimates of surface runoff and deep percolation estimates used in the NMLG model to simulated specified runoff for lakes and recharge, respectively, as a function of topography, land use, soil type, evapotranspiration, and precipitation (Westenbroek and others, 2010). PEST is a public-domain software that allows model-independent parameter optimization and parameter/predictive-uncertainty analysis through the fitting of simulated streamflows and water levels to measured streamflows and water levels (Doherty and Hunt, 2010).

Conceptual Groundwater-Flow Model

A conceptual model of a groundwater-flow system is a simple, descriptive representation that identifies important hydrologic features and physical processes that affect groundwater flow into and through the system (Anderson and others, 2015). Conceptual models are used to determine the design of and hydrologic features represented in numerical, groundwater-flow models (Peterson and others, 2015; Anderson and others, 2015). The development of a conceptual model involves identifying and characterizing the important components of a groundwater-flow system, including (1) aquifers and confining units, (2) water sources and sinks, and (3) hydrologic boundaries. Simplifying model assumptions are necessarily applied to the groundwater-flow system in the process of designing the conceptual model because it is not feasible to include all complexities of a groundwater-flow system into a model (Juckem, 2009); however, enough complexity must be retained in the model such that it represents the general behavior of the natural groundwater-flow system, including groundwater and surface-water exchanges. The conceptual model is based on and informed by existing data and represents how groundwater is thought to move and the relative importance of water sources and sinks. The resulting numerical model is a test of whether or not the hydrologic features and processes in the conceptual model accurately reproduce measured hydrologic data from and represent the natural behavior of the groundwater-flow system (Hill and Tiedeman, 2007). Numerical models are refined during model construction and calibration so they adequately reproduce measured hydrologic data; therefore, the conceptual understanding of the system may change. In this report, the MODFLOW–NWT version 1.08 program (Niswonger and others, 2011) was selected for construction of the numerical groundwater-flow model. MODFLOW, a finite difference model, was selected because components of lake-water budgets could be incorporated into the model using the MODLFOW Lake (LAK) package (Merritt and Konikow, 2000) to allow for detailed assessments of groundwater and surface-water exchanges at lakes. Other MODLFOW packages (Unsaturated Zone Flow [UZF], River [RIV], and Multi-Node Well [MNW] packages) were used to simulate areal recharge, rivers, streams, and groundwater withdrawals from the simulated aquifers. The application of the LAK and other MODFLOW packages are described in the “Water Sources and Sinks” section.

Aquifers and Confining Units

The physical descriptions of the aquifers and confining units used in the conceptual model of the northeast Twin Cities Metropolitan Area (fig. 1) are outlined in the “Description of Study Area, Hydrology, Geology, and Hydrogeology” section. The classification of aquifers and confining units in the conceptual model of the northeast Twin Cities Metropolitan Area consists of seven aquifers and three confining units. The aquifers are the glacial, St. Peter, Prairie du Chien, Jordan, Tunnel City, Wonewoc, and Mount Simon-Hinckley aquifers; the confining units are the Decorah Shale, Platteville Formation, Glenwood Formation, St. Lawrence Formation, and Eau Claire Formation confining units (fig. 3). Previous hydrologic studies of some of the bedrock aquifers have grouped aquifers as regional aquifers (for example, the Prairie du Chien and Jordan aquifers [Delin and Woodward, 1984]); however, these aquifers were simulated separately for the NMLG model and in the MM3 model (Metropolitan Council, 2016) because low hydraulic conductivities are present near the contacts between some of the aquifers in parts of the northeast Twin Cities Metropolitan Area (Runkel and others, 2003a), potentially hydraulically separating the aquifers. Although high groundwater flow rates have been measured in parts of the confining units where they are shallow and fractured (for example, the St. Lawrence Formation [Green and others, 2008, 2010; Runkel and others, 2006]), most of the confining units are deep and much less conductive than aquifers in the northeast Twin Cities Metropolitan Area (Runkel and others, 2003a).

Water-table and buried glacial aquifers are present throughout the model area, except for the southeast part of the model area where bedrock is at the land surface (fig. 5). These glacial aquifers incorporate unconsolidated glacial, fluvial, and colluvium sediments, and peat (fig. 5). Thicknesses of these sediments in the model area range from 0 to more than 500 ft, and are largest in buried bedrock valleys (Bauer, 2016; Meyer and Swanson, 1992). The glacial sediments are a complex mixture of gravels, sands, silts, and clays primarily originating from the Riding Mountain and Superior Provenances (Johnson and others, 2016). Specific aquifers and confining units within the glacial sediments were not delineated for this model because the interfingering of sediments of various compositions and origins is too complex to distinguish individual aquifers and confining units in the model area.

The bedrock aquifers and confining units underlie the glacial aquifer in the model area (fig. 3). Bedrock aquifers and confining units were absent in areas where faulting and erosion commonly associated with the development of buried bedrock valleys are present (Bauer, 2016; Meyer and Swanson, 1992). The lower parts of three bedrock hydrogeologic units (St. Peter, Prairie du Chien, and Tunnel City aquifers) generally have much lower hydraulic conductivities than the upper parts of the hydrogeologic units (Runkel and others, 2003a). Examples of these lower hydraulic conductivity parts of the hydrogeologic units include the basal part of the St. Peter Sandstone and the Oneota Dolomite of the Prairie du Chien Group (Runkel and others, 2003a).

Water Sources and Sinks

Areal recharge to the glacial aquifer and upper bedrock aquifer where glacial sediments are absent, particularly in upland areas, is the primary source of water to the northeast Twin Cities Metropolitan Area groundwater system. The spatial distribution of recharge varies according to land cover, evapotranspiration demands, and surficial (soil or bedrock) hydrologic characteristics. Rivers, streams, lakes, and wetlands can be sources or sinks to surrounding aquifers, depending on the difference between groundwater and surface-water levels, and on hydrologic, geologic and physiographic factors that affect groundwater and surface-water exchanges (see the following "Surface-Water and Groundwater Exchanges" section). The Mississippi and St. Croix Rivers are the main groundwater discharge features of the study area; less groundwater discharges at small perennial streams.

Groundwater withdrawal for municipal, domestic, and commercial water use in the northeast Twin Cities Metropolitan Area is another important sink from the groundwater system. Wells in the northeast Twin Cities Metropolitan Area extract groundwater from glacial aquifers and bedrock aquifers in hydrogeologic units of Ordovician and Cambrian age that are separated from lower aquifers by confining units. Within Minnesota, only wells used to extract more than 10,000 gallons per day (gal/d) or 1 million gallons per year (Mgal/yr) from bedrock and glacial aquifers were included in the NMLG model because annual withdrawal records were available for these wells. Water users withdrawing groundwater from these high-capacity wells are required by the State of Minnesota to obtain a water-use (appropriation) permit and report monthly water-use amounts to the MNDNR (MNDNR, 2015a). In Wisconsin, wells used to withdraw 100,000 gal/d were included in the model because annual withdrawal records were available for these wells. Water users withdrawing groundwater from these high-capacity wells are required by the State of Wisconsin to register and report annual water-use amounts to the Wisconsin Department of Natural Resources (Wisconsin Department of Natural Resources, 2017). Groundwater withdrawals from smaller capacity wells (less than 10,000 gal/d in Minnesota and 100,000 gal/d in Wisconsin) were assumed to account for a small fraction of the total groundwater withdrawals in the northeast Twin Cities Metropolitan Area and therefore were not included in the NMLG model.

Groundwater and Surface-Water Exchanges

Groundwater and surface-water exchanges are complex in the northeast Twin Cities Metropolitan Area, involving surface-water bodies such as rivers, streams, wetlands, and lakes. Most surface-water bodies receive groundwater from and discharge water to aquifers, and parts of the surface-water body are sources and other parts are sinks to the groundwater system; for example, Square Lake (fig. 1) receives substantial groundwater discharge on its western shore and provides water to the groundwater system on the eastern shore (Alexander and others, 2001). Surface-water bodies will be described with respect to their net contributions to the groundwater system in this report. The net contribution of a surface-water body to the groundwater system equals the difference between the amount of groundwater flowing into the surface-water body and the amount of water flowing from the surface-water body to the groundwater system. Surface-water bodies that provide less water to the groundwater system than they receive were identified as discharge features (sinks), and surface-water bodies that provide more water to the groundwater system than they receive were identified as recharge features (sources) in this report. A surface-water body identified as either a discharge or recharge feature still can receive water from and discharge water to the groundwater system, but the net water balance will differ.

For this discussion of groundwater and surface-water exchanges in lakes in this report and Jones and others (2016), lakes were classified as closed-basin and flow-through lakes. Closed-basin lakes are lakes that lack active surface outflow for most of the study period (2003–13), whereas flow-through lakes are lakes that had active surface outflow for most of the study period.

Groundwater and surface-water exchanges happen through bed materials underlying surface-water bodies and are limited by hydraulic conductivity and thickness of bed materials (bed conductances). Bed sediments in lakes and wetlands in the northeast Twin Cities Metropolitan Area generally are in direct contact with glacial sediments, and rivers and streams are in contact with groundwater in fluvial sediments, colluvium, and glacial sediments (fig. 5). Bed sediments in lakes and wetlands underlying surface-water bodies commonly are less hydraulically conductive than the glacial sediments in which they lie above because of the accumulation of organic matter and other fine particles (Jones and others, 2016); however, the bed of the deeper parts of some lakes have little organic material and consist of more permeable, glacial sediments (Jones and others, 2016). Direct exchanges across bed sediments have been measured with field studies in several lakes in the study area, including Big Marine Lake, Big Carnelian Lake, Little Carnelian Lake, Square Lake (Brown, 1985; Alexander and others, 2001), and White Bear Lake (Jones and others, 2013, 2016). Many lakes in the northeast Twin Cities Metropolitan Area overlie buried bedrock valleys (for example George Watch Lake (fig. 1) [Winter, 1999] and White Bear Lake [Jones and others, 2013]), which commonly contain highly permeable beds of coarse sands and gravels and low permeable beds of glacial till and clay.

Surface-water and groundwater elevations in previous studies of the northeast Twin Cities Metropolitan Area (MNDNR, 1998; Barr Engineering Company, 2010; Jones and others, 2013, 2016; Juckem and Robertson, 2013) provide insight into general groundwater flow directions and groundwater and surface-water exchanges. The highest surface-water elevations are in closed-basin lakes and wetlands generally extending between White Bear Lake northeast to Big Marine Lake (Jones and others, 2016). Previous studies indicate shallow and deep groundwater to the east of this zone generally flows east toward the St. Croix River, and groundwater west and south of this zone generally flows west and south toward Mississippi River or toward smaller streams that eventually discharge to the Mississippi River (Metropolitan Council, 2016; MNDNR, 1998; Jones and others, 2013). The high-elevation, closed-basin lakes, such as Big Marine Lake, and wetlands primarily are present on surficial glacial deposits from the Superior Provenance (Johnson and others, 2016; Jones and others, 2016) and are recharge features to the groundwater system.

Most of the closed-basin lakes in the study area are shallow and only are sources of water to local glacial aquifers; however, water from deeper closed-basin lakes can reach deeper bedrock aquifers. Variations in closed-basin lake-water levels in the northeast Twin Cities Metropolitan Area closely reflect variations in groundwater levels, indicating a strong connection between surface-water and groundwater (Jones and others, 2016). The potentiometric surfaces of the Prairie du Chien and Jordan aquifers in this region are typically 5 to 15 feet lower than lake surfaces and the water-table of aquifers in glacial sediments (Jones and others, 2013). These lower groundwater levels in the deeper aquifers indicate that closed-basin lakes are likely net sources of recharge to the groundwater system. Two studies done on White Bear Lake (MNDNR, 1998; Jones and others, 2013), a deep, closed-basin lake, indicate that the lake is a recharge feature to aquifers, with a net loss of lake water to surrounding glacial and bedrock aquifers. Seepage measurements made on White Bear Lake and water isotope ratios indicate groundwater is discharging to White Bear Lake in the nearshore littoral areas, and lake water is discharging to groundwater in deep (between 30 and 80 ft) parts of the lake (Jones and others, 2013, 2016). Water isotope ratios of lake waters and groundwater in the Prairie du Chien aquifer upgradient and downgradient from White Bear Lake (fig. 20 in Jones and others, 2016) indicate a hydrologic connection between the glacial aquifers underlying White Bear Lake and the Prairie du Chien aquifer (Jones and others, 2013). The Prairie du Chien aquifer is thought to be the uppermost bedrock unit under parts of White Bear Lake (Jones and others, 2013). A study done on Big Marine Lake by Brown (1985) indicated that the closed-basin lake also was a recharge feature, though a detailed lake-water budget was not calculated.

Lakes classified as flow-through systems often had more stable water levels during 2002–10 compared to closed-basin lakes (Jones and others, 2016). When flow-through lakes have stable surface outflow during long periods of low precipitation, the lakes likely are discharge features because most of this outflow results from groundwater discharge to the lake. Flow-through lakes are common at low elevations on Riding Mountain Provenance glacial deposits along the northwestern boundary of the study area, such as the lakes connected to Rice Creek (for example, George Watch and Centerville Lakes, fig. 1), but also are present at higher elevations in Superior Lobe deposits, such as Square Lake and Lake Elmo (figs. 1, 5).

Results from previous studies (Winter, 1999; Alexander and others, 2001; Jones and others, 2013) have demonstrated that flow-through lakes often are discharge features or sinks of the groundwater systems; for example, a groundwater-flow modeling study done by Winter (1999) on George Watch and Centerville Lakes (fig. 1) indicates that the lakes receive mostly groundwater inflow from local and more regional groundwater-flow systems. Potentiometric surfaces for the glacial, and Prairie du Chien and Jordan aquifers determined by Jones and others (2013) further indicate that these lakes and other lakes connected to Rice Creek (fig. 1) likely are sinks to local and more regional aquifers (Winter, 1999). A water budget done by Alexander and others (2001) for Square Lake indicated nearly 75 percent of water inflow to the lake was from groundwater discharge to the lake. Stable lake-water levels and continuous outflow measured in Lake Elmo during the drought conditions in 1987–88 indicated that groundwater discharge is an important component of the lake's water budget (Barr Engineering Company, 2015).

Urbanization substantially changes hydrology and the classification of flow-through lakes as groundwater sources or sinks in the model area. Many flow-through lakes and wetlands are in the highly urbanized southwestern part of the model area (fig. 2; Jones and others, 2016) and had relatively stable lake-water levels during the 2003–13 study period. The development of impervious surfaces in suburban and urbanized areas often reduces areal groundwater recharge and increases surface-water runoff to streams, rivers, lakes, and wetlands (for example, Spinello and Simmons, 1992). Surface-water bodies receiving increased surface-water runoff could serve as areas of focused groundwater recharge, or sources of water, to the groundwater-flow system, as hydraulic gradients between these surface-water bodies and the local aquifers increase with the development of the impervious surfaces.

Groundwater can discharge at the land surface in areas adjacent to, but beyond, the footprint of surface-water bodies. In this report, this groundwater discharge is referred to as surface leakage, which is a sink from the groundwater-flow system. Examples of this discharge include bluff-side springs that are commonly from bedrock bedding planes and fractures in outcrops along the St. Croix and Mississippi Rivers (Runkel and others, 2003a) and seeps identified by authors of this report along the east shore of White Bear Lake and along Big Marine Lake (Alexander and others, 2001). This surface leakage also commonly happens in wetlands adjacent to lakes, streams, and rivers.

Soil-Water-Balance Model

The SWB model was calibrated and used to determine a mean 2002–13 daily distribution of recharge and mean daily surface runoff values for the NMLG model. A different SWB model was used to determine recharge estimates for the MM3 model (Barr Engineering Company, 2012; Metropolitan Council, 2016). The SWB model calculates daily recharge and runoff from daily precipitation and air temperature data as a function of land cover (use) (fig. 2), hydrologic soil type, and antecedent moisture conditions (Westenbroek and others, 2010). In addition to these required input data and parameters, the model requires a flow-direction grid to determine runoff to surface-water bodies, including rivers and lakes, in the model area. The SWB model calculates runoff using the U.S. Department of Agriculture, Natural Resources Conservation Service (NRCS) runoff curve number method (NRCS, variously dated).

Model Inputs

Land-use data for the SWB model was acquired from the National Land Cover Database (NLCD) (USGS, 2015a) at a 98-ft (30-meter [m]) resolution (defined as a horizontally square cell of 98 ft [30 m] on a cell side) and resampled to the 410-ft (125-m) NMLG model grid. For the SWB model, the NLCD coverage for 2001 (Homer and others, 2007) was used for simulation periods from 2002 through 2004, the NLCD coverage from 2006 (Fry and others, 2011) was used for simulation periods from 2005 through 2009, and the NCLD coverage from 2011 (Homer and others, 2015) was used for simulation periods from 2010 through 2013. When using the routing functionality of the SWB model, the model calculates all runoff that flows to a surface-water body, as identified by land-use type, as runoff. Many lakes in the study area have fill and spill dynamics when water levels exceed outlet elevations (that is, not all runoff entering surface water bodies contributes to streamflow). Also, surface-water bodies may discharge water to the groundwater system; therefore, the land-use code used in the SWB model for nonflowing surface-water bodies was changed to a land-use code used in an adjacent model cell, thus leaving the aforementioned streamlines as the only cells at which runoff is calculated.

Soil properties of the hydrologic soil group (HSG) and the available water capacity were derived from the gridded Soil Survey Geographic database (SSURGO) dataset (gSSURGO) (Soil Survey Staff, U.S. Department of Agriculture, and NRCS, 2015). The HSGs were determined using the National Engineering Handbook (NRCS, variously dated) using properties derived from the gSSURGO dataset (Soil Survey Staff, U.S. Department of Agriculture, and NRCS, 2015). The HSGs identified as either drained or undrained were classified as drained because most soils in the model area would fall into the drained condition (Smith and Westenbroek, 2015). The HSG was originally calculated at the native 295-ft (90-m) resolution of the gSSURGO dataset and then resampled to 410 ft (125 m). Available water capacities, native to the gSSURGO dataset, were converted to the applicable units (inches of water per foot of soil) and bilinearly interpolated to the 410-ft (125-m) grid resolution.

A flow-direction grid accurately representing the flow of runoff from each cell within the SWB model area was supplied as model input. This grid was needed to ensure that the representative runoff in the SWB model was routed to the appropriate river or lake in the model. The flow-direction grid was a product of interpretation of the 410-ft (125-m) digital elevation model (DEM) generated for the NMLG model (Minnesota Geospatial Commons, 2016). The flow-direction grid was produced by making sure that the flow directions in the grid matched hypothesized flow directions in the model area. The elevations of the 410-ft (125-m) resampled DEM were raised 9.8 ft (3 m) along the perimeters of hydrologic unit code 8 watersheds available from the National Hydrography Dataset (USGS, 2015b) to ensure that the hydrologic unit code 8 watershed boundaries were preserved, which correspond to level 8 watershed boundaries used for NMLG model boundaries (MNDNR, 2013). Sinks for areas less than four model cells as identified from the Minnesota Department of Natural Resources (MNDNR) lakes polygon dataset (MNDNR, 2012) were incorporated into the flow-direction grid using the Terrain Analysis Using DEMs (TauDEM) software (Tarboton, 2015). To determine the appropriate flow lines to incorporate into the 410-ft (125-m) flow-corrected DEM, flow accumulation lines having a value of greater than 10,000 cells were selected from a 33-ft (10-m) resolution, flow-corrected DEM available through the USGS StreamStats program (USGS, 2015c). The “Agree” method (Hellweger, 1997) was used in geographic information system software to incorporate the flow lines into the 410-ft (125-m) DEM. Parameters in the “Agree” method (buffer distance, smoothing distance, and sharp drop distance; Hellweger, 1997) were increased incrementally until the flow trajectories of the 410-ft (125-m) flow-accumulation grid, derived from the corrected DEM, resembled those represented by the streamlines used to incorporate the flow routing. Runoff from wetlands connected to river segments simulated in the SWB model were included in the runoff values for the rivers used in the runoff calibration in the SWB model.

The SWB model requires two lookup tables and a control file (Westenbroek and others, 2010; Smith and Westenbroek, 2015). The lookup tables consist of one table containing soil and land-cover information needed for describing recharge, runoff, and storage processes; and another table containing the extended Thornthwaite-Mather soil-water retention curve (Thornthwaite and Mather, 1957) needed to estimate evapotranspiration rates from climatic data. The soil and land-cover lookup table consisted of runoff curve numbers and root-zone depths for each land-use and soil-type combination and an initial growing season interception value for each land use. The SWB model control file contains parameters describing evapotranspiration, continuous frozen ground index, and growing season parameters. Default parameters for the soil and land-cover lookup table and control file parameters were obtained from previous SWB models completed in and around the study area (Metropolitan Council, 2016; Smith and Westenbroek, 2015); however, many of these parameters were adjusted during the model calibration process.

Several published sets of suggested values for runoff curve numbers corresponding to specific land cover exist (Cronshey and others, 1986; Smith and Westenbroek, 2015; Westenbroek and others, 2010), whereas other previous studies have documented the locational and seasonal variability of these values associated with antecedent moisture conditions

(Hjelmfelt, 1991; NRCS, variously dated). Literature numbers commonly refer to watershed-scale characteristics, whereas with the SWB model, every grid cell is assigned a runoff curve number based on its associated land cover and soil type.

Model Calibration

Model calibration is the process of adjusting model parameters to improve the match between simulated and measured (observed) data. The SWB model was calibrated using annual base flows and monthly measured runoff as calibration targets in PEST (table 2). Annual base flows at one streamgage on Rice Creek (Rice Creek below Old Highway 8 in Mounds View, Minnesota [USGS streamgage 05288580]) and monthly measured runoff totals during 2003–13 at three streamgages on Rice Creek (USGS streamgage 05288580, Rice Creek Watershed streamgage R1, and Rice Creek Watershed streamgage R5), one streamgage on Valley Creek (Valley Branch Watershed District streamgage VA–1), and one streamgage on Brown's Creek (Brown's Creek Watershed District streamgage BA–0.3) were used as the calibration targets (fig. 6; table 2). Monthly simulated recharge and runoff values were determined from the calibration process for each month from January 2002 and December 2013 and then used to determine mean 2002–13 values. These monthly values were determined to ensure that simulated mean 2002–13 values represented a general value for the steady-state period and were not skewed toward data from periods when data were collected at a higher frequency. The stream base-flow components for Valley and Brown's Creeks were not used to inform the calibration because boundaries for the groundwater watershed and the surface-water watershed for each creek are not the same (Almendinger, 2014; Brown's Creek Watershed District, 2012).

Model parameters for the SWB model were calibrated using automated PEST (Doherty 2010). These model parameters were (1) groups of runoff curve numbers based on land covers, (2) root-zone depths for different land covers and soil types, and (3) soil parameters and conditions. The outputs from the calibrated SWB model were used to generate the spatial distribution of runoff, specifically to selected lakes (Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake), and recharge (water percolation below the root zone) throughout the study area. Runoff curve numbers, root-zone depths, and interception parameters from the lookup table and select control file variables describing runoff from frozen ground and growing season length in the SWB model were calibrated to reduce the difference between simulated values and measured data by using the Nash-Sutcliffe efficiency (NSE) coefficient (Nash and Sutcliffe, 1970) as the key parameter to assess the quality of the fit between simulated and measured data. An NSE value of 1 indicates that simulated values match the measure data exactly, and an NSE value of 0 indicates that the simulated values are no better than the mean of the measured data (Ellison and others, 2016). A negative NSE value indicates that the mean of the measured data is better than the simulated values at approximating individual measured values.

Groups of runoff curve numbers for different land covers were optimized in the calibration process to improve the accuracy of the simulated runoff and recharge processes compared to measured values throughout the study area. The group of runoff curve numbers contains all runoff curve numbers for a particular land cover over a range of soil types spanning from sandy to clayey soils. Runoff curve numbers for a particular land cover tended to increase as underlying soils transitioned from sandy to clayey. To ensure that groups of calibrated runoff curve numbers retained this trend and to reduce the number of calibrated parameters in the SWB model, a relation similar to a relation developed by Mockus (1964) was developed where a single runoff curve number (for example, the runoff curve number corresponding to the HSG D soil) and the coefficient for a land cover was calibrated. Runoff curve numbers for HSG A, B, and C soils were derived from the HSG D runoff curve number for each land cover according to the following inverse quadratic equation (for an example):

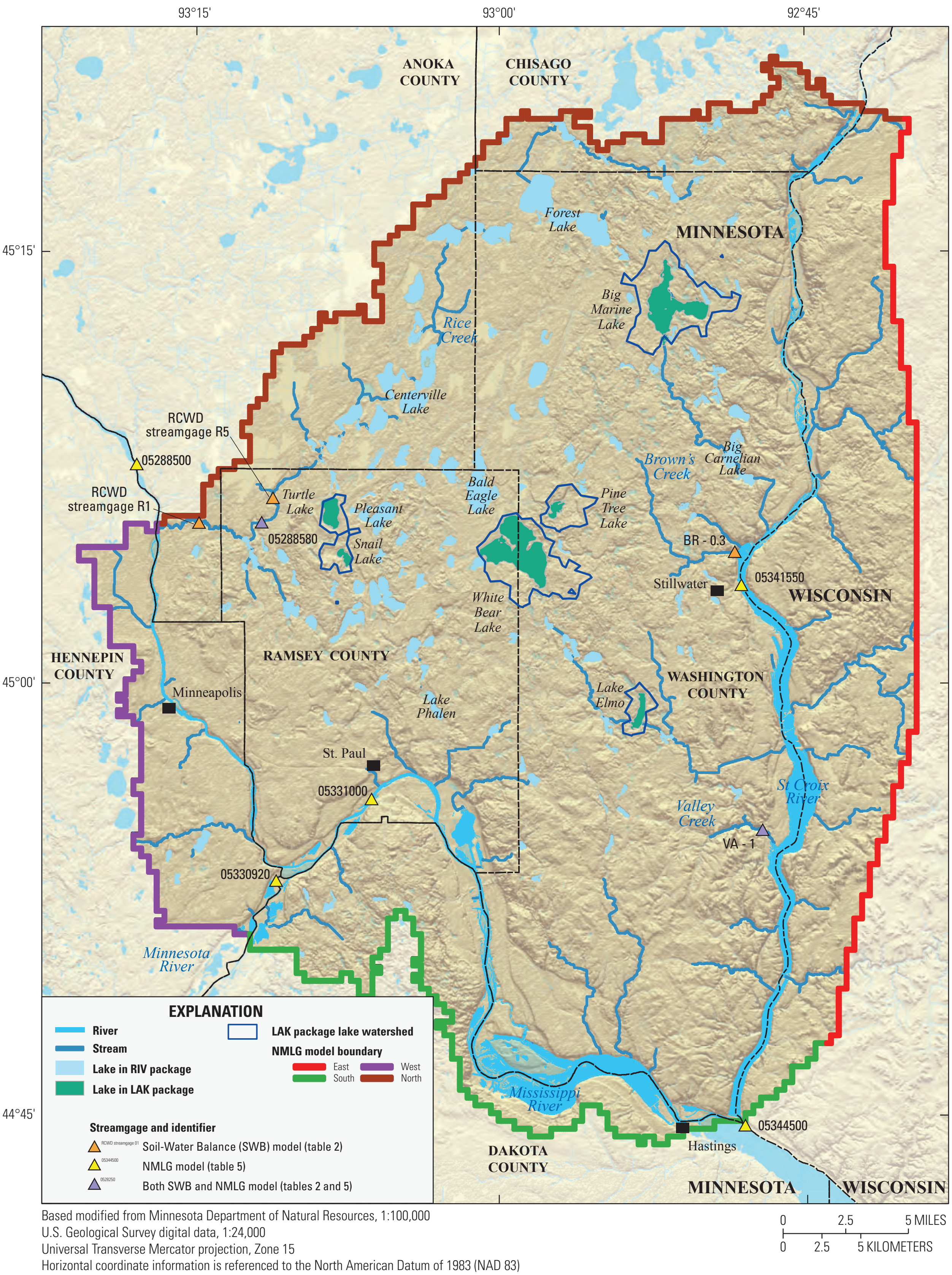
$$CN_{L_i} = CN_{L_D} * \left(-a_g * \left(100 - CN_{L_D}\right) * b_i^2 + 1\right) \tag{1}$$

where

- CN_{L_i} is the runoff curve number for a soil type (HSG A, B, or C) for each land cover L;
- CN_{L_D} is the runoff curve number for HSG D soil for land cover L;
- a_g is the coefficient for land-cover group g (land-cover groups are urban, forest, grass/cropland, and wetland); and
- b_i is the offset for soil type as follows: 3 for HSG A soil, 2 for HSG B soil, and 1 for HSG C soil.

Land covers with similar hydrologic characteristics were grouped into the following land-cover groups: urban (various developed lands), forest (shrub, deciduous, evergreen, and mixed forest), grass/cropland (barren land, grassland/herbaceous, pasture/hay, and cultivated crops), and wetlands (woody and emergent herbaceous) (fig. 2). Coefficients were specified based on these land-cover groups. Values for the curve number for HSG D soil for each land cover and coefficients for each land-cover group initially were determined so that they mostly agreed with the relation proposed by Mockus (1964). Upper and lower limits of these parameter values used during the calibration process were flexible enough so that the relations could accommodate the variety of reported surface runoff curve numbers determined in previous SWB modeling studies (Smith and Westenbroek, 2015).

Root-zone depths for each land-cover and soil-type permutation were determined as a model parameter calibrated for each land-cover group within the study area (fig. 2). Root-zone depths affect the storage capacity of soil that must be filled before groundwater recharge and the amount of water available for evapotranspiration determined using the Thornthwaite-Mather method (Thornthwaite and Mather, 1957). Root-zone



Based modified from Minnesota Department of Natural Resources, 1:100,000
U.S. Geological Survey digital data, 1:24,000
Universal Transverse Mercator projection, Zone 15
Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83)

Figure 6. Location of rivers and lakes simulated in the Northeast Metro Lakes Groundwater-Flow (NMLG) model using the U.S. Geological Survey's modular finite-difference groundwater-flow model (MODFLOW) River (RIV) or Lake (LAK) packages, and streamgages for which streamflow data were used in model calibration, northeast Twin Cities Metropolitan Area, Minnesota.

Table 2. Summary of measured and simulated runoff and base flow in select streams used to calibrate the Soil-Water-Balance model, northeast Twin Cities Metropolitan Area, Minnesota.

[mi^2, square mile; in/yr, inch per year; USGS, U.S. Geological Survey; NSE, Nash-Sutcliffe efficiency]

Stream name	Data source and streamgage identifier	Type of measurement	Surface drainage area (mi^2)	Number of years of observation data	Runoff or base flow (in/yr)			Percent error between simulated and measured	Observation weight	Initial model NSE	Calibrated model NSE
					Measured	Simulated	Residual				
Brown's Creek	Brown's Creek Watershed District streamgage [1]BA–0.3	Runoff	31	11	1.43	1.70	0.27	18.7	0.15	0.2	0.4
Valley Creek	Valley Branch Watershed District streamgage [2]VA–1	Runoff	12	11	1.79	1.44	–0.35	–19.6	0.09	0.0	0.0
Rice Creek Below Old Highway 8 in Mounds View, Minnesota	USGS streamgage [3]05288580	Base flow	156	5	10.77	11.77	1.00	9.2	0.16	–1.4	0.8
Rice Creek Below Old Highway 8 in Mounds View, Minnesota	USGS streamgage [3]05288580	Runoff	156	5	2.36	2.10	–0.26	–11.1	0.22	0.3	0.3
Rice Creek	Rice Creek Watershed (RCWD) streamgage [4]R1	Runoff	180	2 years 6 months	4.74	3.70	–1.04	–21.9	0.14	0.2	0.4
Rice Creek	Rice Creek Watershed (RCWD) streamgage [4]R5	Runoff	154	2 years 8 months	2.13	1.66	–0.47	–22.1	0.11	0.0	0.0

[1]Karen Kill, Brown's Creek Watershed District, written commun., August 17, 2015.

[2]Jennifer Kostrzewski, Metropolitan Council, written commun., June 29, 2015.

[3]U.S. Geological Survey, 2015e.

[4]Matthew Kocian, Rice Creek Watershed District, written commun., September 1, 2015.

depths for land covers from previous SWB models (Smith and Westenbroek, 2015) indicate a general decline in root-zone depth from more permeable soils (HSG A and B soils) to less permeable soils (HSG C and D soil). The root zones typically were deepest on HSG B soil across most land-cover types in previous SWB models (Smith and Westenbroek, 2015). In a manner similar to the previously mentioned runoff curve numbers, a relation among a group of root-zone depths pertaining to a specific land cover was constructed to reduce the number of parameters and ensure a logical progression among these depths over different soil types for a particular land cover. To enforce this relation during the parameter estimation process, root-zone depths for HSG A, C, and D soils were derived from the HSG B root-zone depth for each land cover according to the following inverse quadratic equation (for an example):

$$RZ_{L_i} = RZ_{L_B}\left(m_{ga} * b_i^2 - m_{gb} * b_i + 1\right) \qquad (2)$$

where

RZ_{L_i} is the root-zone depth for a soil type (HSG A, C, or D) for each land cover *L*, in feet;

RZ_{L_B} is the root-zone depth for HSG B soil for land cover *L*, in feet,

m_{ga} and m_{gb} are coefficients for land-cover group *g* (urban, forest, grass/cropland, and wetland); and

b_i is the offset for soil type as follows: 1 for HSG A soil, 2 for HSG C soil, and 1 for HSG D soil.

Different land cover grids were used to simulate (1) runoff to the lakes simulated with the MODFLOW Lake (LAK) package (described in the “Water Sources and Sinks” section) and (2) recharge across the model area. The SWB model allows users to define an open-water cover type in which groundwater cannot recharge. For simulating runoff to LAK package lakes, small stream centerlines within their watersheds and open-water cells were assigned to the no-recharge, open-water cover type to ensure proper routing of surface runoff to the lakes. For simulating recharge across the model area, small stream centerlines within the watersheds of the lakes simulated with the LAK package were assigned to their respective NLCD classes. Open-water cells representing lakes were reclassified to a new open-water cover type that allowed recharge, ensuring that recharge was possible on all model cells that are not MODFLOW River (RIV) or LAK cells in layer 1 of the NMLG model.

Soils with high organic content (greater than 20 percent by mass) are present throughout the study area (Soil Survey Staff, U.S. Department of Agriculture, and NRCS, 2015), and therefore a specific hydrologic soil type for these soils was incorporated into the soil-type grid. Organic soils often are thought to behave similarly to HSG B and C soils; however, the calibration process provided flexibility in the calibration of runoff parameters for the organic soil type by allowing PEST to determine the best offset value (b_i) in runoff curve number and root-zone depth relations for organic soils.

Select soil parameters and conditions in the SWB control file that control water movement in the soils also were determined during the calibration process. These parameters and conditions were (1) the number of growing degree days that initiated the growing season, (2) the minimum temperature that ended the growing season, and (3) the upper and lower limits of the continuous frozen ground index, a factor that enhances runoff as soils freeze.

All runoff and recharge within the delineated watersheds upstream from the streamgages was assumed to be contributed to the streamflow measured at the streamgages in the calibration of the model. PEST was used to calibrate the SWB model to NSE coefficients for each watershed for monthly runoffs and annual base flows when assumptions of the method were appropriate and data were available for watersheds. Base-flow separations of streamflow hydrographs were completed using the local minimum method in the hydrograph separation program (HYSEP) (Sloto and Crouse, 1996) to determine the runoff and base-flow components of each streamflow record. Upstream watershed areas were calculated from the location of the streamgages using the 410-ft (125-m), flow-corrected DEM.

Monthly runoff and annual base-flow targets were only included in the analysis when a complete record for a particular month or year existed for the streamgage. Months and years with incomplete streamflow records were omitted from the calculation of the NSE coefficient. In one case, two streamgages, USGS streamgage 05288580 and Rice Creek Watershed Gage R5 (fig. 6), had overlapping data during October 30, 2008, and November 19, 2009, for practically the same watershed. To prevent undo impedance on the calibration for that period, the streamflow record for the Rice Creek Watershed Gage R5 streamgage was omitted from the calibration because it was the less complete record.

Observation weights for each of the calibration targets (table 2) were determined based on (1) the number of observations recorded at each streamgage, (2) the quality of the measurements, (3) the watershed area, and (4) the degree to which the method assumptions applied to the watershed. To devise an appropriate weighting scheme for each of the NSE calibration targets, a mean of the weights of the individual measurements used to produce the NSE coefficient was calculated and then scaled by the watershed area normalized to the study area. Individual weights were placed on the NSE coefficient for each calibration target site (streamgage). Weights for each streamgage were set in proportion to area of the watershed compared to the total area of all watersheds used in the calibration and raised to the exponent of 0.25. This weighting process allowed for larger watersheds to be more represented in the calibration while mitigating the large discrepancies in size such that smaller watersheds, specifically those for Brown’s Creek and Valley Creek (fig. 6), maintained influence in the calibration process. A quality weight factor for each watershed represented the quality of streamflow data for each streamgage ranked from most reliable to least reliable on a scale of 0 (least reliable) to 0.3 (most reliable). Factors considered in determining the quality values primarily were continuity, length, and documentation of the record. A component weight factor was used to weight the runoff and base-flow estimates for the watersheds. A component weight factor used a relative multiplier of 0.7, compared to 1.0 for runoff, placed on the base-flow estimates for Rice Creek based on the modeler's confidence in the estimates. The following equation was used to determine the observation weights for each calibration target:

$$w_{i,q} = B_i C_q \left(\frac{a_i}{A_T} \right)^{0.25} \quad (3)$$

where

- $w_{i,q}$ is the observation weight of Nash-Sutcliffe efficiency coefficient for watershed i and flow component q;
- a_i is the area of watershed i, in square miles;
- A_T is the total area of all calibration watersheds, in square miles;
- B_i is the quality weight factor for watershed i; and
- C_q is the component weight factor for flow component q.

Calibration was terminated when improvement in reduction of the objective function error was reduced to less than 0.5 percent for three successive optimization iterations.

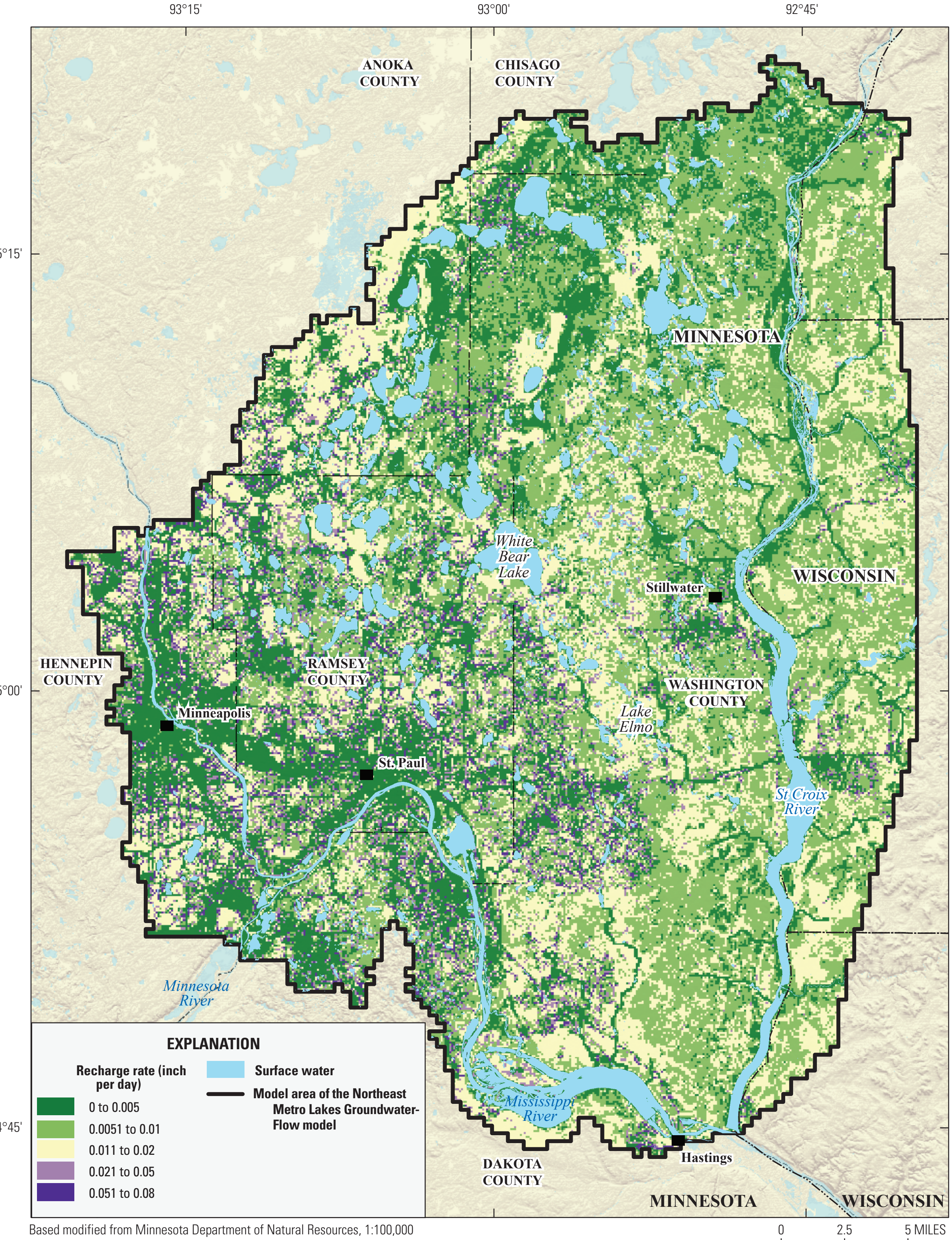
Model Results

The calibration of the SWB model improved the model fit to the measured surface-water runoffs at Brown's Creek streamgage BR-0.3 and Rice Creek streamgage R1 (fig. 6) and base flow at Rice Creek. The NSE values for these simulated runoffs improved by 0.2, and base flows improved by 2.2, moving closer to an ideal NSE value of 1 (table 2; Nash and Sutcliffe, 1970). The largest reduction in error was achieved for base flows at USGS streamgage 05288580 (fig. 6), which was the only base-flow target included in the calibration process. The NSE values for the base-flow values changed from a negative value, indicating that the linear model fit of the simulated to measured values was biased (McCuen and others, 2006) to a positive value, indicating that the linear model fit was unbiased (table 2). The percent error between simulated and measured base flow was 9.2 (table 2). Calibration of runoff for that same site resulted in the best fit between simulated and measured runoff values for all calibration sites, resulted in the lowest percent error for all calibration sites (table 2). The percent error between simulated and measured monthly runoff for the five streamgages ranged from –22.1 to 18.7 (table 2).

After calibration, the SWB model was run with the calibrated parameter set to generate 2002–13 mean recharge grids over the model area and 2003–13 mean daily surface-water runoff values for six lakes (Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake), which were used in the NMLG model. Unrealistically high and low recharge values were simulated by the SWB model for a subset of model cells. These unrealistic values were constrained by (1) an upper limit recharge value of the sum of the overall mean recharge value for all model cells and three times the standard deviation of the recharge values in all model cells, and (2) a lower limit recharge value of 5 percent of the upper limit recharge rate. All model cells with an SWB-simulated mean recharge value higher than the upper limit, as defined in the previous sentence, were assigned the upper limit value. All model cells with an SWB-simulated mean recharge rate below the lower limit (5 percent of the upper limit recharge rate) were assigned the lower limit value. Recharge rates were unrealistically high in low-lying areas where runoff accumulated during the SWB simulations. Recharge rates were extremely low, with some 0 ft/d, in areas with developed land cover (fig. 2). It was necessary to adjust 0 ft/d recharge rates to a nonzero value to make these areas responsive to the recharge multiplier used for NMLG model calibration (see "Model Calibration" in the "Steady-State Groundwater Flow Model" section). In the model area, 1 percent of the SWB model cells had recharge values above the upper limit, and 20 percent of the SWB model cells had recharge values below the lower limit.

For the calibrated SWB model, the mean recharge rate on nonopen water cells for the NMLG model grid was 4.6 inches per year (in/yr) before any upper or lower limit constraints placed on the SWB simulated recharge values and 4.4 in/yr (0.012 inches per day; fig. 7) after applying the upper and lower limits to the SWB simulated recharge values. Within the NMLG model grid, after applying the upper and lower limit constraints to the SWB simulated recharge values, 23 percent of the total recharge volume was present in low intensity developed land cover, 17 percent was present in pasture/hay land cover, 16 percent was present in deciduous forest land cover, and 12 percent was present in each of developed open space (for example, city parks) and cultivated crop land cover. The remaining land-cover types each had 6 percent or less of the total recharge volume. Using the runoff functionality of SWB resulted in higher recharge rates of cells in depressions and valleys where runoff accumulated. This was particularly true in low-density urban areas where runoff from high-density urban areas accumulated.

Recharge values produced from the SWB model generally were lower than recharge estimates determined in other hydrologic studies in the Twin Cities Metropolitan Area. Applying a regional regression method, Lorenz and Delin (2007) estimated a mean annual recharge rate range between 3 and 9 in/yr to surficial materials in the Twin Cities Metropolitan Area. A SWB model developed and run by Metropolitan Council (2016) for 1988–2011 produced recharge values ranging from 2.7 to 13 in/yr (mean=8.2 in/yr) for the Twin Cities Metropolitan Area. Ruhl and others (2002) used a variety of methods to determine a wide range of recharge values (1.2 to 13.6 in/yr) for the Twin Cities Metropolitan Area, Minnesota. Barr Engineering Company and Washington County (2005)



Based modified from Minnesota Department of Natural Resources, 1:100,000
U.S. Geological Survey digital data, 1:24,000
Universal Transverse Mercator projection, Zone 15
Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83)

Figure 7. Mean 2003–13 recharge rates determined by Soil-Water-Balance model and used in the Northeast Metro Lakes Groundwater-Flow (NMLG) model, northeast Twin Cities Metropolitan Area, Minnesota.

determined a mean recharge rate of 8.7 in/yr for Washington County using the watershed model component of MIKE SHE program (Refsgaard and Storm, 1995). The recharge values determined with the SWB model for the 2003–13 data in this report were lower than in other studies possibly because the base-flow values used to calibrate the model were determined in large watersheds and in rivers where the boundaries of the groundwater watershed and the surface watershed were different, or the baseflow estimates used to calibrate the SWB model were closer to actual baseflows and the multiplier value applied to the base-flow values (0.7) may not have been needed.

Steady-State Groundwater-Flow Model

The MODFLOW–NWT version 1.08 program (Niswonger and others, 2011) was selected for construction of the NMLG model. MODFLOW, a finite difference model, was selected because components of lake-water budgets could be incorporated into the model to allow for detailed assessments of groundwater and surface-water exchanges at lakes. The MODFLOW–NWT model code was selected primarily for its ability to simulate groundwater flow under unconfined and confined conditions and address issues associated with cell drying and rewetting when solving models representing unconfined systems. The NMLG model was calibrated and used to simulate mean groundwater-flow conditions and groundwater and surface-water exchanges in the model area during 2003–13.

The NMLG model domain and discretization was constructed to best simulate groundwater and surface-water interactions and the effects of groundwater withdrawals on lake-water levels under steady-state conditions while maintaining the general framework of the MM3 model (Metropolitan Council, 2016). Initial model parameters, boundary conditions along the eastern part of the model, and bedrock layering in the MM3 model were used in the NMLG model. Specifics as to MM3 model data use in the NMLG model are indicated throughout this report.

Model Domain

The domain of the active NMLG model was sized such that the boundaries were appropriately distanced (at least 3.1 miles [mi]) from the lakes of primary interest in the central part of the study area and included the MNDNR’s North and East Metro Groundwater Management Area in the model (fig. 1). The model boundaries to the east, south, and west (fig. 6) were placed on the basis of the following: (1) groundwater levels generated from the MM3 model and other existing groundwater-flow models (Juckem, 2009) and (2) the relative location of major rivers and lakes that are large groundwater sinks (fig. 6). To the east, the model boundary is in Wisconsin several miles east of the St. Croix River (figs. 1, 6), which is a major groundwater discharge feature, to allow streamflow data from streamgages on the St. Croix River (USGS streamgages 05344500 and 05341550, fig. 6) to be included in the model calibration. This boundary coincides with the MM3 model boundary. To the south and west, the model boundary (fig. 6) was defined by (1) the shortest distance of either the nearest MNDNR level 8 watershed boundary (MNDNR, 2013) to the far side of the model area of the Mississippi River or (2) 3.1 mi (5 kilometers [km]) outwardly perpendicular from the Mississippi River centerline (fig. 1). To the north, the model boundary (fig. 6) was defined by level 8 watershed boundaries produced by the MNDNR (MNDNR, 2013). The watersheds included in the NMLG model were selected by their inclusion of Forest and Clear Lakes, two large lakes of interest (fig. 1; MNDNR, 2016b).

Boundary Conditions

The lateral and vertical hydrogeologic boundaries of the model were based on boundaries of the MM3 model (Metropolitan Council, 2016), other existing steady-state groundwater-flow models (Juckem, 2009), and surface-watershed boundaries. The base of the Mount Simon-Hinckley aquifer, which consists of the Mount Simon Sandstone of Dresbach Group and Hinckley Sandstone of Keweenawan Supergroup, was the base of the conceptual and numerical model (fig. 3). It was assumed that there was no flow below this base because the Middle Proterozoic sedimentary, volcanic, and mafic intrusive bedrocks underlying the Hinckley Sandstone of Keweenawan Supergroup (table 1) are considered to be impermeable and a regionally confining unit (Delin and Woodward, 1984); however, hydrologic properties of and groundwater-flow conditions in Middle Proterozoic sedimentary, volcanic, and mafic intrusive bedrocks underlying the Hinckley Sandstone are poorly known (Bauer, 2016; Metropolitan Council, 2016). The base of the Mount Simon-Hinckley aquifer also was chosen as the base of the conceptual and numerical model (1) because no groundwater is withdrawn from high-capacity wells from Precambrian bedrock formations below the Mount Simon Sandstone of Dresbach Group and Hinckley Sandstone of Keweenawan Supergroup in the study area (MNDNR, 2015a), (2) because groundwater withdrawals from the Mount Simon-Hinckley aquifer likely have little effect on groundwater and surface-water exchanges in lakes in the model area, and (3) to be consistent with the base of the MM3 model.

The lateral boundary of the model is based on the location of major rivers and watersheds, some of which were used in the MM3 model (Metropolitan Council, 2016). These boundaries were chosen to be more than 3 mi away from the six lakes of interest (Big Marine Lake, Lake Elmo, Pine Tree Lake, Square Lake, Turtle Lake, and White Bear Lake) and to include most of the MNDNR’s North and East Metro Groundwater Management Area (fig. 1). Major rivers in the model area are the Mississippi and St. Croix Rivers (fig. 1). Model boundaries were extended several miles beyond these major rivers to include them in the model, allow direct simulation of potential underflow beneath the St. Croix River (Schoenberg, 1984, 1990), and allow streamflow data from streamgages on these rivers to be included in the model calibration. Level 8 watersheds produced by the MNDNR (MNDNR, 2013) were selected to represent northern parts of the model boundary because lakes with measured water levels were present near the boundary and they were more than 3 mi away from the six lakes of interest.

Groundwater levels from previous groundwater-flow models (Metropolitan Council, 2016; Juckem, 2009) were represented as constant-head cells at the lateral boundaries of the NMLG model in each model layer. Constant-head cells are model cells in which the water levels are held constant during the simulation period (Harbaugh, 2005). Constant-head cells were used at these boundaries because steady-state water levels from the MM3 model (Metropolitan Council, 2016) and other existing groundwater-flow models (Juckem, 2009) were previously used along the eastern domain of the MM3 model. These steady-state water levels were thought to be a reasonable representation of 2003–13 mean steady-state water levels along the lateral boundaries. The use of constant-head cells to represent hydrologic conditions along the lateral boundaries of the model allows for unlimited flow in the groundwater-flow system, which can result in simulated groundwater and lake-water levels being higher than measured levels.

Model Discretization

Uniform horizontal grid cell dimensions of 410 by 410 ft (125 by 125 m) were selected for the NMLG model. For comparison, the MM3 model had a uniform grid of 1,640 by 1,640 ft (500 by 500 m). The finer grid for the NMLG model was needed to adequately represent the complexity of surface-water features important to groundwater and surface-water exchanges within the region. Cell dimensions of 328 ft (100 m), 164 ft (50 m), and 1,640 ft (500 m) also were considered. Based on the 410-ft (125-m) grid, the NMLG model consists of 560 rows, 432 columns, and 12 layers, with a total of 1,992,000 active and 911,040 inactive model cells.

Layering the NMLG model was developed with the simultaneous objectives of accurately simulating groundwater and surface-water exchanges and maintaining some consistency with the layering of the MM3 model. The geologic deposits and formations represented in the NMLG model are the same deposits and formations represented in the MM3 model, but the upper layers of the two models differ. The NMLG model consists of 12 layers (fig. 3), whereas the MM3 model consists of 9 layers (Metropolitan Council, 2016). The top four layers of the NMLG model primarily represent glacial aquifers (undifferentiated glacial sediments consisting of glacial tills, glacial outwash, buried sand and gravel, colluvium, alluvium, and terrace deposits), and the Decorah-Platteville-Glenwood confining unit. Glacial aquifers and the Decorah-Platteville-Glenwood confining unit are represented in the top layer of the MM3 model (Metropolitan Council, 2016). This more detailed layering of the glacial aquifers was needed to complete a more detailed simulation of the water exchanges between the glacial aquifers and the lakes and rivers in the model area. The top layer in the MM3 model had a mean thickness of 98 ft (30 m) and consisted of composite properties of all sediments present from the land surface to the bottom of the layer (Metropolitan Council, 2016), which is appropriate for such a large-scale regional model; however, this single layer is not adequate for simulating shallow groundwater and surface-water exchanges affected by interbedding of glacial sediments of different compositions and sizes. Explicitly simulated lakes and rivers were represented in the top three layers of the model to simulate the groundwater and surface-water exchanges in more detail. Glacial aquifers are thought to be present below all the lakes in the model area (Swanson and Meyer, 1990; Meyer and Swanson, 1992; Bauer, 2016; Jones and others, 2016), and therefore groundwater and surface-water exchanges are present between the lakes and these aquifers. The glacial aquifers also were simulated in parts of layers 5 through 11 in the model where buried bedrock valleys existed.

The top layer of the NMLG model was constructed to provide a more detailed representation of (1) surface-water elevations, (2) groundwater and surface-water exchanges at shallow depths, and (3) the three-dimensional arrangement of glacial sediments. The top elevation of the model was created by applying a Gaussian filter to the 98-ft (30-m) DEM (USGS, 2015d) to remove topographic extremes (for example, chopped-off peaks and filled-in valleys) over the entire model area and resampling the new DEM using bilinear interpolation to the 410-ft (125-m) model resolution. The 98-ft (30-m) DEM was used to apply more recent elevation data for determining elevations for the top layer of the model compared to the MM3 model (Metropolitan Council, 2016). The elevations of model cells within enclosed surface-water features having historical stage data (USGS, 2015e) were set to the maximum measured stage in the stage record to make it likely that the simulated water levels would be lower than the model-cell elevations. After refinement of the top layer in the NMLG model, elevations for the top four layers of the NMLG model were determined by dividing the top layer of the MM3 model at depths of 9.8, 20, and 39 ft (3, 6, and 12 m, respectively) below the 410-ft-gridded (125-m-gridded) model top.

The lower eight layers of the NMLG and MM3 models primarily consist of bedrock aquifers in hydrogeologic units of Ordovician- and Cambrian-age and confining units vertically from the St. Peter aquifer to the Mount Simon-Hinckley aquifer (fig. 3). The layer elevations and hydrogeologic representation (aquifers and confining units) of these bottom eight layers were obtained from the bottom eight layers of the MM3 model through bilinear interpolation of the 1,640-ft by 1,640-ft (500-m by 500-m) grid of the MM3 model to the 410-ft by 410-ft (125-m by 125-m) grid used in the NMLG model. A minimum cell thickness of 3.3 ft (1 m) was enforced when the interpolation yielded a thickness of less than 3.3 ft (1 m) so that the subsequent layer elevation was adjusted downward slightly to accommodate this tolerance.

As in the MM3 model, simulated aquifers and confining units in the NMLG model were present in different layers over the model area (fig. 3, range of model layers), although each simulated aquifer and confining unit mostly were present in a primary layer (fig. 3, primary model layer). Aquifers and confining units are not continuous throughout the study area because either geologic units containing the aquifers and confining units were not deposited or parts of the aquifers and confining units have been eroded away during glacial advances and retreats. As a result, the simulated aquifers and confining units had to be simulated in multiple layers.

In addition to the 12 layers of the model, the NMLG model uses quasi-three-dimensional (3D) confining layers from the MM3 model to simulate the vertical flow between low-permeability conditions in the base of three simulated aquifers (St. Peter aquifer, Prairie du Chien aquifer, and Tunnel City aquifer) that exist in these aquifers (Runkel and others, 2003a). Quasi-3D confining layers are not layers explicitly simulated in the model but are used in the calibration process to determine vertical hydraulic conductance between two model cells or layers (Harbaugh, 2005). Groundwater levels are not simulated in the quasi-3D confining layers because they are not explicitly simulated in the model. In addition to the primary layers for the three simulated aquifers with quasi-3D-confining units, quasi-3D confining layers also were applied to the other layers of the model because the three simulated aquifers were present in other layers of the model. Vertical hydraulic conductivity values for the quasi-3D confining layers in the calibrated MM3 model were used as initial vertical hydraulic conductivity values for the quasi-3D confining layers in the NMLG model. Where the low-permeability basal parts of these three simulated aquifers are not present, the vertical hydraulic conductivity was set at 32,808 ft/d (10,000 m/d) in the NMLG model, as it was in the MM3 model, so that the layer has no substantial effect on the vertical hydraulic conductance between the model layers. The thicknesses of each quasi-3D confining layer were 0.33 ft (0.1 m) in the NMLG and MM3 models.

Hydraulic Properties and Zonation

Groundwater-flow models using MODFLOW–NWT require that the Upstream Weighting (UPW) package be used as the groundwater-flow package (Niswonger and others, 2011). The UPW package uses a continuous groundwater-level function, rather than the discrete approach of drying and rewetting used by other MODFLOW packages (Harbaugh, 2005) to treat nonlinearities of cell drying and rewetting (Niswonger and others, 2011). For steady-state simulations, the UPW package requires (1) horizontal hydraulic conductivity values for aquifers and confining units, (2) anisotropy of horizontal hydraulic conductivity or vertical hydraulic conductivity values for aquifers and confining units, and (3) vertical hydraulic conductivity values of any existing quasi-3D-confining layers.

The distribution of hydraulic properties for the simulated glacial aquifers was determined primarily from the gridded dataset of 11 sediment textural classifications characterized from geologic descriptions in geologic cores and well-drilling logs (Tipping, 2011), and supplemented with geologic information using surficial glacial information from Meyer (2007). The gridded dataset (Tipping, 2011) also was used to determine the hydraulic properties for the simulated glacial aquifer in the MM3 model (Metropolitan Council, 2016). This dataset consists of a 3D point dataset with points spaced at 820 ft (250 m) in the horizontal plane and at about 20 ft (6.1 m) in the vertical plane beginning at about 10 ft (3.1 m) below land surface and proceeding to the top of a bedrock surface. At each horizontal point, a maximum of eight vertical glacial sediment data points were described in the Tipping (2011) dataset. The number of the vertical glacial sediment data points at each horizontal point depended on the depth of the geologic core or well at or near the horizontal point. Each of the vertical glacial sediment data points were assigned a given point (*x*-*y*) location and a sediment class value ranging from 1 to 11, representing various glacial sediment classes varying from highly conductive sand and gravel to deep impermeable clays and tills (Tipping, 2011). These 11 sediment class values were assigned to the 11 parameters in the horizontal hydraulic conductivity parameter group (qhk) based on the geologic descriptions (parameter names quat_hk1 through quat_hk11, table 3 [available for download in Excel format at https://doi.org/10.3133/sir20165139B]) and to 11 parameters in the vertical hydraulic conductivity parameter groups (qvk and qvkd) based on the geologic descriptions (parameter names quat_vk1 through quat_vk11, table 3).

To ensure complete coverage of all NMLG model cells, the glacial sediment point dataset derived from Tipping (2011) was augmented horizontally and vertically. The Tipping (2011) dataset is the most complete and uniform classification of the glacial sediments that covers most of the study area; however, glacial sediments in some of the model area were not classified by Tipping (2011) because data were lacking.

Vertical augmentation of the Tipping (2011) dataset was required where (1) sediment class data were not present at all depths above the upper bedrock surface, and (2) the upper bedrock surface elevations in the MM3 model did not match any bedrock surface elevations in the dataset. To provide glacial sediment information over the model area in Minnesota, the Tipping (2011) dataset was augmented vertically at all eight potential vertical depths at each point in the dataset missing data at a 820-ft (250-m) grid spacing. The process of augmenting the dataset began by identifying points lacking elevation data and assigning elevation values to them at all depths. Points missing elevations in layer 1 were set to 10 ft (3.1 m) below the land-surface elevation at that location on the 98-ft (30-m) DEM. Points missing elevations in glacial sediment layers beneath layer 1 were set to 20 ft (6.1 m) below the elevation for the preceding (above) layer because this spacing was the vertical offset for most attributed points in the Tipping (2011) dataset. After filling all the missing point elevations, a series of eight sediment elevation grids were produced containing the subsequent point elevations at 820-ft (250-m) spacing and subsequently sampled horizontally to the cell spacing of the NMLG model grid (410-ft [125-m] spacing). Points without sediment class data were then assigned sediment class values at the eight sediment depths based on the most frequently present sediment classes associated with the surficial glacial units defined by Meyer (2007).

Sediment class data also were augmented horizontally and vertically at areas where no horizontal or vertical glacial sediment information was present in the Tipping (2011) dataset. Points were created in the eight augmented Tipping (2011) datasets in these areas at the 820-ft (250-m) grid spacing. Sediment class data were assigned sediment class values to these points at the eight sediment depths based on the most frequently present sediment classes associated with the surficial glacial units defined by Meyer (2007). For cases where all points within a surficial glacial unit at a specific depth were missing geologic information, the most frequently present sediment class from the same surficial glacial unit within the above layer was assigned to the missing points in the current layer. Points within lakes were assigned sediment class values down to the bottom of the lakes to complete the dataset in these areas. After filling in all missing values of sediment class, the point dataset was used to produce eight 410-ft (125-m) grids, with each cell in the eight sediment class grids containing a value representing a sediment class.

Initial values of the hydraulic properties of eight sediment class grids were calculated in a similar manner as was completed for the MM3 model. Composite values of horizontal and vertical conductivities were calculated by assigning hydraulic property values from Tipping (2011) for the 11 sediment classes (table 3) to cells in the 8 sediment class grids. These hydraulic property values represent values for the sediment classes in the glacial sediment class datasets. The resulting grids were then intersected with the model grid, and composite horizontal and vertical conductivity properties were calculated for each model cell. This intersection of the model and sediment class grids was necessary because elevations for the two grids were misaligned. This process for the NMLG model resulted in a more detailed representation in comparison to the MM3 model for the hydraulic properties for the complex, interlayered glacial sediments throughout the model area.

Composite horizontal hydraulic conductivities for each NMLG model cell represented within glacial sediments between layers 1 and 8 were calculated using the following equation:

$$K_{x_{i,j,k}} = \frac{\sum_{l=1}^{n} KCL_{x_{l,j,k}} THK_{l,j,k}}{E_{top_{i,j,k}} - E_{bot_{i,j,k}}} \tag{4}$$

where

$K_{x_{i,j,k}}$ is the composite horizontal hydraulic conductivity for model cell *i*,*j*,*k*, in feet per day;

$KCL_{x_{l,j,k}}$ is the horizontal hydraulic conductivity of glacial sediment classes for zone *l* between the top and bottom of model cell *i*,*j*,*k*, in feet per day;

$THK_{l,j,k}$ is the thickness of the glacial sediment classes for zone *l* between the top and bottom of model cell *i*,*j*,*k*, in feet;

$E_{top_{i,j,k}}$ is the elevation of the top of model cell *i*,*j*,*k*, in feet; and

$E_{bot_{i,j,k}}$ is the elevation of the bottom of model cell i,j,k, in feet.

Composite vertical conductivities were calculated for each cell using the following equation:

$$K_{z_{i,j,k}} = \frac{E_{top_{i,j,k}} - E_{bot_{i,j,k}}}{\sum_{l=1}^{n} \frac{THK_{l,j,k}}{KCL_{z_{l,j,k}}}} \tag{5}$$

where

$K_{z_{i,j,k}}$ is the composite vertical hydraulic conductivity for model cell *i*,*j*,*k*, in feet per day;

$E_{top_{i,j,k}}$ is the elevation of the top of model cell *i*,*j*,*k*, in feet;

$E_{bot_{i,j,k}}$ is the elevation of the bottom of model cell *i,j,k,* in feet;

$THK_{l,j,k}$ is the thickness of the glacial sediment classes for zone *l* between the top and bottom of model cell *i*,*j*,*k*, in feet; and

$KCL_{z_{l,j,k}}$ is the vertical hydraulic conductivity of glacial sediment classes for zone *l* between the top and bottom of model cell *i*,*j*,*k*, in feet per day.

Hydraulic properties were calculated for model cells in the glacial zone from layers 1 through 8 using parameter optimization and calibration by applying automated calibration methods within PEST software (Doherty, 2010). Hydraulic properties for cells in the glacial sediments in layers below layer 8 were given the hydraulic properties of spatially corresponding cells in the MM3 model.

The Decorah-Platteville-Glenwood confining layer was represented as a single, restricted (low hydraulic conductivity) zone within the lowest glacial layer of the NMLG and MM3 models in those areas where at least one of the three bedrock formations (Decorah Shale, Platteville Formation, and Glenwood Formation) was present (fig. 3). Bedrock elevations from the Minnesota Geological Survey (Robert Tipping, Minnesota Geological Survey, written commun., December 10, 2015) were used to define the location and the top and bottom of these three bedrock formations (Decorah Shale, Platteville Formation, and Glenwood Formation) within the NMLG model area. Where the Decorah-Platteville-Glenwood confining unit was present in the dataset, this unit was incorporated into the glacial sediment class grids as a new sediment class to cells in the glacial sediment dataset that were more than 50 percent occupied by the Decorah-Platteville-Glenwood confining unit.

Bedrock layering and zonation of hydraulic conductivities in the MM3 model were used as the initial bedrock hydraulic conductivities and their distribution in the NMLG model. Bedrock hydraulic properties zones in the MM3 model were resampled from the original 1,640-ft (500-m) grid-cell size to the current 410-ft (125-m) grid-cell size without interpolation. Initial values of hydraulic properties for horizontal and vertical hydraulic conductivity in each bedrock hydraulic conductivity zone were set to values contained in the MM3 model.

Water Sources and Sinks

A series of water sources and sinks were incorporated into the NMLG model to simulate the effects of these sources and sinks on groundwater flow and groundwater and surface-water exchanges. Simulated sources to groundwater were (1) areal recharge to the top layer of the model; and (2) recharge from rivers, lakes, and streams. Simulated sinks from groundwater in the model were (1) withdrawals from wells; and (2) discharge to rivers, lakes, and streams.

Recharge

Recharge to the upper model layers was incorporated into the NMLG model using the MODFLOW Unsaturated-Zone Flow (UZF) package with output from the SWB model (Niswonger and others, 2006). The UZF package was used in the model to simulate water flow and storage in the unsaturated zone, partition flow into evapotranspiration and recharge, and account for land-surface runoff to lakes and streams (Niswonger and others, 2006). A kinematic wave approximation to Richards' equation is used in the UZF package to simulate vertical unsaturated flow (Niswonger and others, 2006). Grids representing the spatial distribution of daily recharge for the respective simulation periods were generated using the SWB model (Westenbroek and others, 2010) and used to determine a mean 2002–13 daily distribution of recharge to the top layer of the model, except where lakes and rivers are present (fig. 7). A recharge multiplier (rchgm in table 3) was used with this mean 2003–13 distribution of recharge to determine the final recharge for the NMLG model. The SWB model for the study area was calibrated to measured base flows and streamflows for streams that had sufficient streamflow data within the model area as described in the "Soil-Water-Balance Model" section.

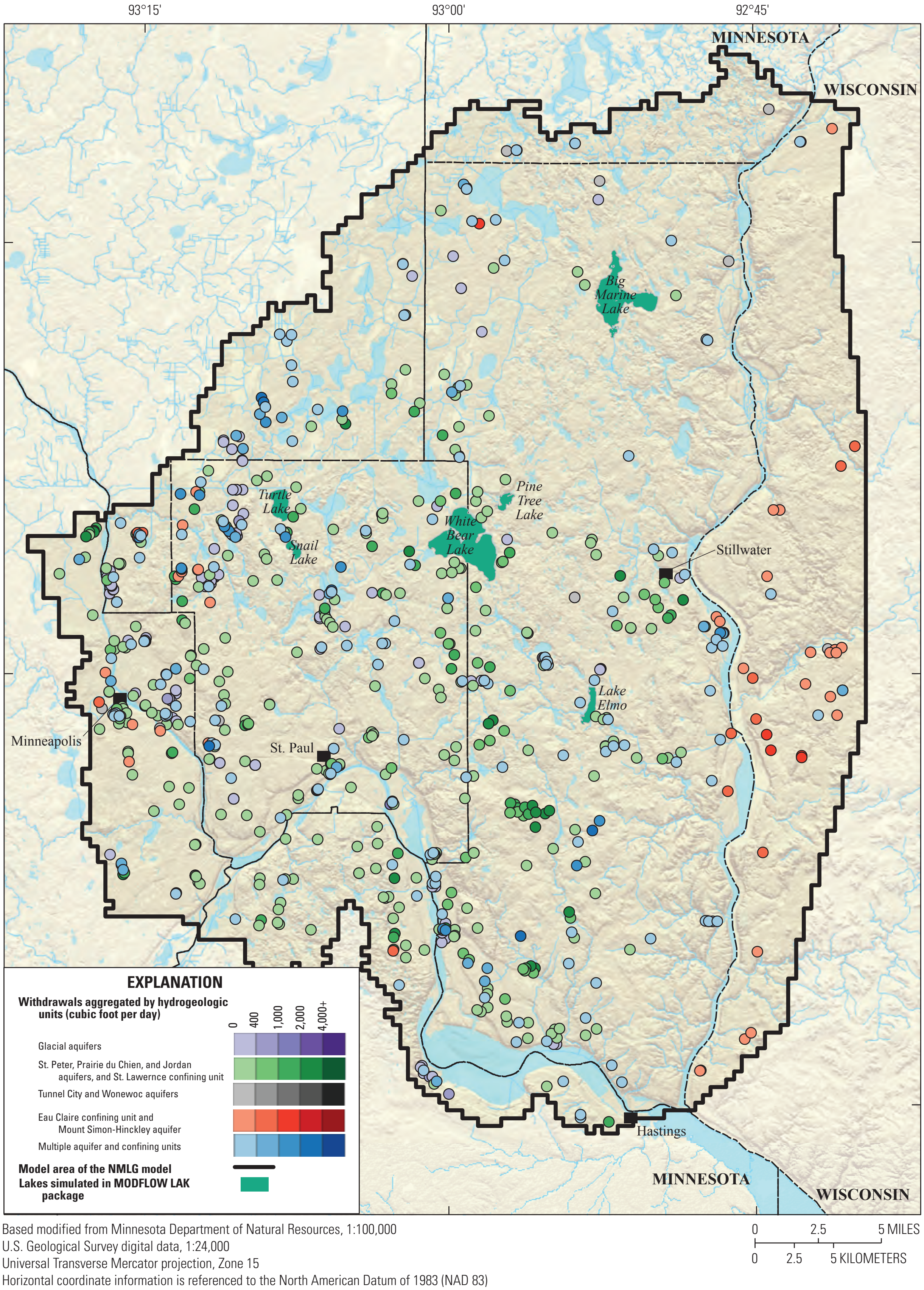
The UZF package was configured using the mean 2002–13 recharge values obtained from the SWB model (fig. 7), after applying the upper and lower limit constraints to the SWB simulated recharge values, as recharge to the NMLG model. The vertical conductivity used to calculate recharge rates was specified as the vertical conductivity of the first (uppermost) model layer. Surface leakage was activated, and simulated leakage within lake watersheds was routed accordingly. The undulation depth, which is the depth to water table at which the unsaturated zone begins to generate surface leakage, was set to 1.6 ft (0.5 m). Additional gridded properties affecting the performance of the UZF package also were specified. Spatially distributed saturated moisture content was calculated, and the Brooks-Corey function was determined using textural data obtained from the gSSURGO data (Soil Survey Staff, U.S. Department of Agriculture, and NRCS, 2015) and pedo-transfer functions developed by Saxton and Rawls (2006). The Brooks-Corey function, or epsilon (Brooks and Corey, 1964), was used to relate the saturated moisture content to unsaturated hydraulic conductivity. The 295-ft (90-m) grids generated using the gSSURGO data were then bilinearly interpolated to the 410-ft (125-m) grid. A uniform recharge multiplier that modified the SWB-generated recharge grid was adjusted during the model calibration process.

Groundwater Withdrawals from Wells

Groundwater withdrawals from 838 high-capacity wells were simulated in the model (fig. 8). Data for the high-capacity wells in the model were aggregated from the MNDNR (Minnesota Department of Natural Resources, 2015a), the Wisconsin Department of Natural Resources (Robert Small, Wisconsin Department of Natural Resources, written commun., December 16, 2015), or from the USGS (Cheryl Buckwald, U.S. Geological Survey, written commun., December 17, 2015). The MODFLOW Multi-Node Well (MNW) package (Konikow and others, 2009) was used to simulate groundwater withdrawals from high-capacity wells having open or screened intervals spanning one or more model layers. High-capacity wells represented in the MM3 model that were within a 1,640-ft (500-m) internal buffer of the model area for the NMLG model and had a nonzero value for groundwater withdrawal were included in the NMLG model.

Groundwater-withdrawal data through 2013 for all high-capacity wells in Minnesota were obtained from the Minnesota Water Use Data System (MNDNR, 2015a). Groundwater withdrawals for wells in the NMLG model in Minnesota were identified using a combination of the Minnesota Department of Health's Minnesota Well Index (MWI) unique identification number, the MNDNR permit number, the MNDNR installation identification number, and the MNDNR use code. Monthly groundwater withdrawals from high-capacity wells were aggregated from January 2003 through December 2013 to determine mean 2003–13 values. Wells that were not included in MM3 model records through 2010 were assigned to the MNW package depending on the open-hole or screened elevation intervals. If open-hole or screened elevation intervals were not available, elevations were assigned to them using the geologic stratigraphy defined for each well from MWI well records. If no stratigraphy, open-hole, or screened interval information were available, the wells were not included in the NMLG model.

Groundwater withdrawals for high-capacity wells in Wisconsin (fig. 8) in the east part of the NMLG model area were obtained from the Wisconsin Department of Natural Resources (Robert Small, Wisconsin Department of Natural Resources,



Based modified from Minnesota Department of Natural Resources, 1:100,000
U.S. Geological Survey digital data, 1:24,000
Universal Transverse Mercator projection, Zone 15
Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83)

Figure 8. Mean 2003–13 groundwater withdrawals from high-capacity wells used in the Northeast Metro Lakes Groundwater-Flow (NMLG) model, northeast Twin Cities Metropolitan Area, Minnesota. [MODFLOW, modular finite-difference groundwaterflow model; LAK, Lake]

written commun., December 16, 2015) and from the USGS (Cheryl Buckwald, U.S. Geological Survey, written commun., December 17, 2015). Well construction information for these wells were obtained from USGS (Cheryl Buckwald, U.S. Geological Survey, written commun., December 17, 2015).

Total reported groundwater withdrawals for the model area that could not be assigned to wells or model layers were not incorporated into the model. These withdrawals accounted for less than 2 percent of the total groundwater withdrawals reported to the MNDNR and Wisconsin Department of Natural Resources during 2003–13.

For many wells, construction information was not fully known. Assumptions were therefore made regarding well construction information to the extent necessary for inclusion of the groundwater withdrawals in the MNW package. In most cases, the elevations of open or screened intervals or the tops and bottoms of aquifers the wells were open to or screened in was not known. In all cases, however, the groundwater-withdrawal rates and elevation of the pump were known and used to make assumptions on the elevation of the open/screened interval or the aquifers the wells were open to or screened in.

The effects of groundwater withdrawals from domestic and other unpermitted wells were not simulated in the groundwater-flow model because no known records exist regarding the amounts of groundwater withdrawn from individual domestic wells during 2003–13. Permitted public water system wells withdrew an estimated 66 million gallons per day (Mgal/d), serving about 554,500 people throughout Anoka, Ramsey, and Washington Counties in 2010 (Maupin and others, 2014). About 115,000 people are served by domestic wells withdrawing an estimated total of 8 Mgal/d throughout Anoka, Ramsey, and Washington Counties in 2010 (Maupin and others, 2014).

Rivers, Lakes, and Streams

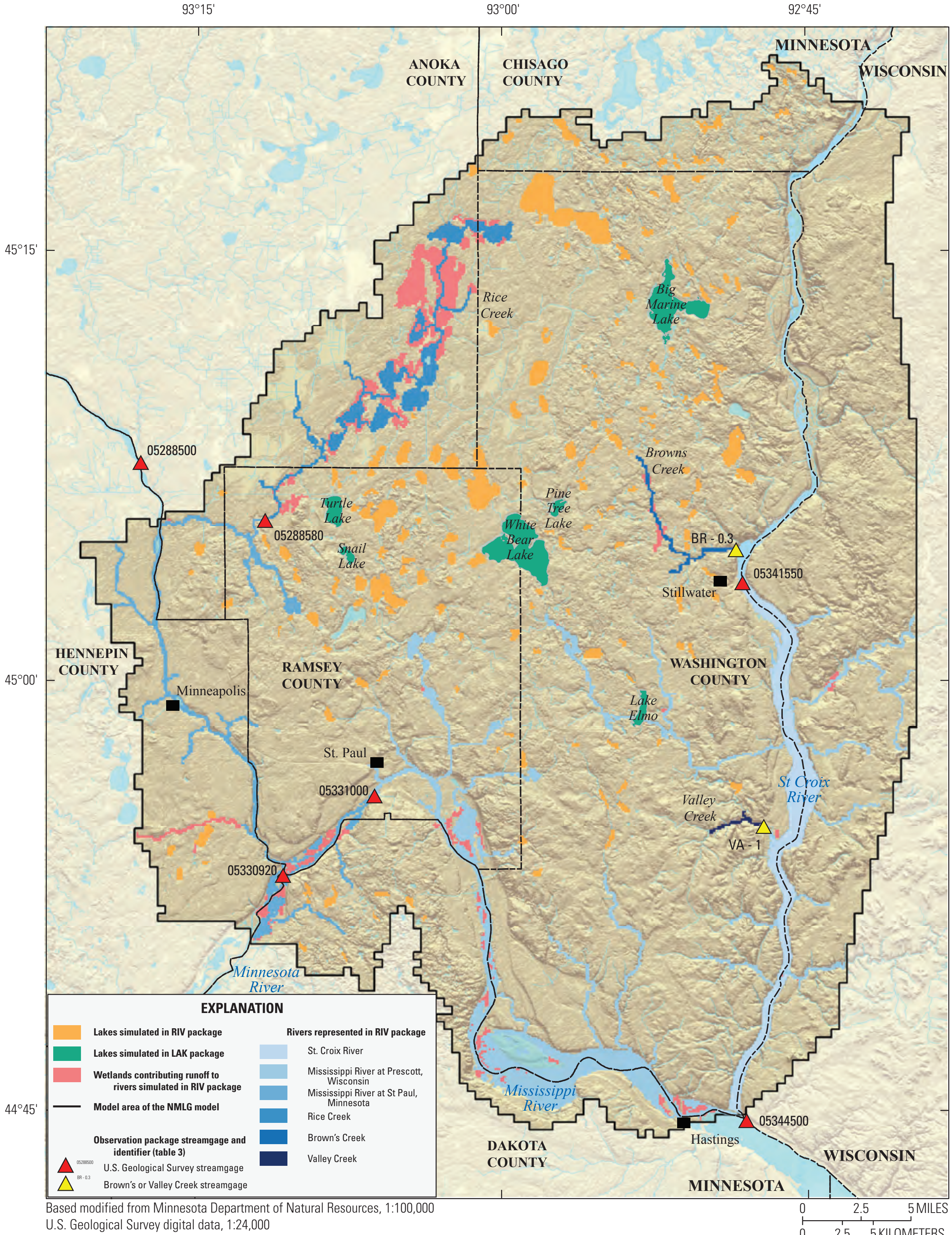
Rivers and streams were simulated using the MODFLOW RIV package (fig. 9) (Harbaugh, 2005). These rivers and streams included the Mississippi River, St. Croix River, Rice Creek, Brown's Creek, and Valley Creek (fig. 9). Lakes not selected to be simulated with the MODLFOW LAK package were simulated using the RIV package (figs. 6, 9). The RIV package is used to apply head boundary conditions, or head-dependent fluxes, to simulate the exchange of water between groundwater and model cells containing surface-water features (Harbaugh, 2005). Each cell containing a RIV package boundary condition requires information on the water level or stage, the cell bottom, and conductance of the feature. Values for the riverbed conductances were adjusted during the calibration of the model based on the surface-water feature type (river or lake, fig. 9) and the major surficial geologic unit at the feature location.

Major river features within the model area, such as the Mississippi, Minnesota, and St. Croix Rivers (fig. 9), were included in the model because they are strong sinks to groundwater flow. Depths for all major river cells were 9.8 ft (3 m) below the top (land surface) of the 98-ft (30-m) DEM (USGS, 2015d) at the center of the river feature. For the 2003–13 steady-state simulation and the eight hypothetical scenarios, the stage of a RIV cell was assumed to be the value of the 98-ft (30-m) DEM at the center of the surface-water feature.

The same set of streams and rivers used for calculating runoff in the SWB model (table 2) were used to simulated streams with the RIV package of the NMLG model. Streamlines were made into 98-ft-wide (30-m-wide) polygons, and the bottom elevations of RIV cells representing streams were set to a depth of 3.3 ft (1 m) below the elevation of the 98-ft (30-m) grid at the geometric center of the stream feature within each model cell the stream intersected. Values of stage for these streams and rivers were set to elevations equal to the bottom of the model cells representing the RIV feature assuming that smaller streams and rivers only behave as sinks with respect to the groundwater system. Initial riverbed conductance values were assigned as the product of the stream or river area and an initial estimated riverbed conductance value of the underlying surficial geology co-located with the RIV feature (table 3).

Lakes having an area greater than four model cells and that were not simulated with the LAK package were included as features in the RIV package (figs. 6, 9). These features were derived from the MNDNR Lakes and Open-Water water dataset (MNDNR, 2012) as classified by "Use Class 421." Gridded bathymetry data were used to determine the RIV cell bottom of these features where applicable. Bathymetry data were acquired from MNDNR (2015b) at a 16-ft (5-m) resolution and were resampled to the 410-ft (125-m) model grid. To accurately assign the depths of these cells, the elevations of the model top contained within these lakes were set to the ordinary high water levels (Minnesota Office of the Revisor of Statutes, 2016) obtained from the MNDNR (2015c). The bottom of the RIV cells representing the lakes were then calculated as the top of the model minus the bathymetry value for all cells having bathymetry data. The bottom of the RIV cells was set at a depth of 1.6 ft (0.5 m) in the case where no bathymetry data were available. Stage data from historical observations (USGS, 2015e) were used to assign stage when available; otherwise the elevation of the 98-ft (30-m) DEM at the center of the feature was used. The RIV cell conductivities were assigned based on the lake's waterbody type and the underlying geological unit.

Wetlands are common features throughout the study area and are likely important hydrologic components; however, wetland function with respect to the groundwater system is variable and complex. Because of these complexities and uncertainties, wetlands were not explicitly simulated with the RIV package. Instead the effect of wetlands on the groundwater system were included in the model by the addition of the SWB recharge coverage and the UZF package. The SWB-generated recharge rates provide initial recharge rates to the top layer of the model, and the UZF package is allowed to proportionally discharge groundwater, or surface leakage, in model cells where the groundwater elevations are within 1.6 ft



Based modified from Minnesota Department of Natural Resources, 1:100,000
U.S. Geological Survey digital data, 1:24,000
Universal Transverse Mercator projection, Zone 15
Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83)

Figure 9. Rivers and lakes represented in the U.S. Geological Survey's modular finite-difference groundwater-flow model (MODFLOW) River (RIV) package, lakes represented in the MODFLOW Lake (LAK) package, and streamgages used in calibration of the Northeast Metro Lakes Groundwater-Flow (NMLG) model, northeast Twin Cities Metropolitan Area, Minnesota.

(0.5 m) of the model top. This approach allows for inclusion of wetland behavior without constraining the model with numerous RIV cells.

Several adjustments were made to RIV cells to address multiple surface-water features, recharge, and overlapping of RIV cells with constant-head boundaries. Multiple surface-water features represented in the model by RIV cells were present within a single model cell. To address this issue, area-weighted means were computed for the bottom and stage of the cell, and a cumulative streambed conductance for the cell was calculated. The recharge rates in UZF model cells where a RIV feature occupied more than 75 percent of the cell area also were set to 0 ft/d to reduce overlap in boundary conditions and allow the RIV package to control model fluxes in these cells. The RIV cells also were deactivated along model boundaries in cells where they would be present with constant-head lateral boundary conditions.

Measured water elevation (stage) data for lakes and major rivers, simulated using the RIV package, were compiled for 2003–13 from the MNDNR (2015c), USGS National Water Information System (NWIS) database (USGS, 2015e), Valley Branch Watershed District (2015a), Brown's Creek Watershed District (Karen Kill, Administrator, Brown's Creek Watershed District, written commun., August 17, 2015), Rice Creek Watershed District (Matthew Kocian, Lake and Stream Specialist, Rice Creek Watershed District, written commun., September 1, 2015), and the Metropolitan Council (Jennifer Kostrzewski, Principal Environmental Scientist, Metropolitan Council, written commun., June 24, 2015). To ensure that simulated RIV cell elevations represented a general elevation for the steady-state period and were not skewed toward elevations from periods when data were collected at a higher frequency, a monthly value was first assigned to each water body for each month from January 2003 through December 2013. The monthly value was assigned by calculating the mean value of elevation data within the month; if no elevation data existed during a given month, the next closest preceding (backward in time) value was assigned as the monthly value. This methodology was chosen because typically data were missing in the late fall and winter months, and assigning late fall data to winter months was more appropriate than assigning early spring values to winter months. No minimum data threshold was used because extracting any real measured value for a RIV cell is preferable to the alternative method of extracting the DEM elevation. Mean values were then determined from the monthly values for 2003–13 for the steady-state simulation.

Lake Simulation with Lake Package

Lake-water levels and budgets were simulated for six lakes in the NMLG model using the LAK package (figs. 1, 6, 9; Merritt and Konikow, 2000). These lakes were Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake (figs. 1, 2, 6). The LAK package in the MODFLOW–NWT simulates lake-groundwater exchanges and the response of lake-water levels to hydraulic stresses, such as groundwater withdrawals, applied to the aquifer. Variations in lake-water levels are determined by individual water budgets computed by the model for each lake simulated in the model. To determine the lake-water budget, input hydrologic parameter values representing the rate of atmospheric recharge and evaporation, overland runoff, and the rate of any direct withdrawal from, or augmentation of, the lake volume are required (Merritt and Konikow, 2000).

The six lakes simulated with the LAK package were selected based on meeting the following criteria: (1) maximum lake depth of greater than 25 ft, (2) water surface area of greater than 75 acres, (3) location greater than 3 mi away from the peripheral boundaries of the NMLG model, and (4) lake water-level data available for at least 50 percent of the months during the open-water periods (April–November) of 2003–13. Other lakes met this criteria but were not selected because they lack surface-water outflow or inflow data, or other data required for the LAK package. The selected lakes consist of closed-basin and flow-through lakes (table 4) (Jones and others, 2016) and some of the larger lakes (such as Big Marine and White Bear Lake) in the northeast Twin Cities Metropolitan Area (fig. 1). Mean water levels during 2003–13 for each of the six lakes were calculated and used to calibrate the 2003–13 steady-state model (table 4).

The LAK package was used to simulate mean lake-water levels and budgets for 2003–13 based on (1) simulated groundwater inflow and lake-water outflows from the lakes, (2) relations between lake volume and depth for the lakes, and (3) hydraulic conductivity zones. The areal extent of lakes simulated with the LAK package were determined by creating a 410-ft (125-m) raster grid from the vector representation of each of the six lakes obtained from the MNDNR Open-Water dataset (MNDNR, 2012). This grid was compared to bathymetry datasets with 16-ft (5-m) resolution (MNDNR, 2015b) to determine the model cells and layers representing the lakes. To simulate the interface and exchanges between lakes and the groundwater system, lakes were activated in all model cells where the elevation of the top of a model cell underlying the areal extent of the lake was greater than the bathymetric elevation. Lake cells occupied the top three layers of the model. Where the lakes simulated with the LAK package exceeded the depth of layer three, elevations in the third layer in the NMLG model were brought down by 3.3 ft (1 m) below the bottom of the lake bathymetry. This was done to ensure that glacial sediments were represented in model cells below lakes represented in the LAK package.

Relations between lake volume and depth were specified for each lake to simulate the water budgets of the six lakes. The relations between lake volume and depth for Big Marine Lake, Lake Elmo, Lake Owasso, Square Lake, Turtle Lake, and White Bear Lake (fig. 1) were derived from bathymetry datasets with 16-ft (5-m) resolution (MNDNR, 2015b). The relations between lake volume and depth for Snail Lake and Pine Tree Lake were not included in the bathymetry dataset but were provided by the MNDNR (Andrew Williquette, MNDNR, written commun., November 19, 2015). The 16-ft

Table 4. Summary statistics of lake-water-level data for the period 2003–13 for the six lakes simulated with the U.S. Geological Survey's modular finite-difference groundwater-flow model (MODFLOW) Lake (LAK) package in the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[MNDNR, Minnesota Department of Natural Resources; ID, identification number; ft, foot; NAVD 88, North America Vertical Datum of 1988]

MNDNR lake ID	Lake name	Lake type[1]	Reference datum	Approximate mean lake surface area (acre)	Lake watershed area (acre)	Lake watershed area to lake surface area ratio	Measured lake-water levels[2] (feet)								Simulated mean lake-water level (ft)	Simulated mean minus measured mean lake-water level (ft)
							Number of measurements	Minimum	25th percentile	Median	75th percentile	Maximum	Range	Mean		
62006100	Turtle (62–61 P)	Closed basin	NAVD 88	447	837	1.9	308	889.5	890.4	891.0	891.8	893.4	3.8	891.1	895.2	4.1
62007300	Snail (62–73 P)	Closed basin	NAVD 88	155	1,050	6.8	308	877.6	881.1	882.0	882.6	884.1	6.5	881.6	880.8	–0.8
82005200	Big Marine (82–52 P)	Closed basin	NAVD 88	2,049	7,818	3.8	500	939.2	940.5	940.7	941.1	941.7	2.5	940.8	940.7	0.1
82010600	Elmo (82–106 P)	Flow through	NAVD 88	297	1,686	5.7	112	882.9	884.3	884.8	884.9	885.5	2.5	884.6	884.1	–0.5
82012200	Pine Tree (82–122 P)	Closed basin	NAVD 88	181	4,424	24	146	942.5	943.6	944.2	944.8	945.9	3.5	944.2	938.6	–5.6
82016700	White Bear (82–167 P)	Closed basin	NAVD 88	2,424	7,099	2.9	489	919.3	920.5	921.0	922.4	925.9	6.6	921.5	923.0	1.5

MNDNR lake ID	Lake name	Lake type[1]	Reference datum	Approximate mean lake surface area (acre)	Lake watershed area (acre)	Lake watershed area to lake surface area ratio	Calibration weight of lake-water-level measurement (eq. 9)			
							Factor based on priority of lake-level target (L_i)	Cummulative error (c_i)	Estimate error (n_i)	Measurement weight (w_{l_i})
62006100	Turtle (62–61 P)	Closed basin	NAVD 88	447	837	1.9	1.08	0.05	0.125	15.7
62007300	Snail (62–73 P)	Closed basin	NAVD 88	155	1,050	6.8	1.08	0.05	0.125	15.7
82005200	Big Marine (82–52 P)	Closed basin	NAVD 88	2,049	7,818	3.8	1.32	0.05	0.125	19.2
82010600	Elmo (82–106 P)	Flow through	NAVD 88	297	1,686	5.7	0.98	0.05	0.125	14.3
82012200	Pine Tree (82–122 P)	Closed basin	NAVD 88	181	4,424	24	0.98	0.05	0.125	14.3
82016700	White Bear (82–167 P)	Closed basin	NAVD 88	2,424	7,099	2.9	1.65	0.05	0.125	24

[1]As defined in Jones and others, 2016.

[2]Minnesota Department of Natural Resources, 2015.

(5-m) bathymetric data were aggregated to a 410-ft (125-m) resolution by calculating the mean depth for blocks of 25 cells on a side using the R raster package (Hijmans, 2015). Aggregation started in the upper-left corner of a raster that covered the extent of all lakes to be simulated with the LAK package. Each lake was divided into 150 equal-depth intervals to meet the data requirements for the LAK package (Merritt and Konikow, 2000). The volume of the lake at each of the 150 depth intervals was calculated according to the following equation:

$$V = \mathrm{sum}[L^2 * (x_i - d)] \tag{6}$$

where

V is the volume, in cubic feet;
L is the length of a raster cell side (410 ft [125 m]), in feet;
x_i is the lake depth in raster cell i, in feet; and
d is the depth interval, in feet.

The elevations of lake surfaces (at 0 ft depth) were determined by calculating the mean elevation of all light detection and ranging data (MNDNR, 2015d) within the extent of the bathymetric data for each lake.

Zones representing vertical lakebed leakance values, which are equal to the vertical hydraulic conductivity of a lakebed divided by its thickness, were determined for each lake based on interpretations of continuous seismic-reflection profiles, lake-sediment cores, slug tests completed in lakebed and glacial sediments beneath White Bear Lake, and near-shore wetland evaluations. Model cells representing the lakebed leakance zones for the six lakes were assigned to one of three hydraulic conductivity zones: (1) shallow-water permeable sediments, (2) low-permeability sediments, and (3) deep-water permeable sediments (for lake-water depth of greater than 30 ft). The hydraulic conductivity values for these three zones in the six lakes were used to estimate ranges of lakebed leakance values for each zone (table 3). Lakebed leakance values for each zone in each of the six lakes were assigned as calibration parameters in PEST (Doherty, 2010; table 3). Low-permeability sediments and deep-water permeable sediments were identified in four of the six lakes (Big Marine Lake, Lake Elmo, Turtle Lake, and White Bear Lake, fig. 9) based on interpretation of continuous seismic-reflection profiles (Jones and others, 2016). Areas with deep-water permeable sediments identified in the continuous seismic-reflection profiles were large enough to be included in the model only in White Bear Lake. Locations of low-permeablility lake sediments were identified based on the presence of trapped gases, whereas areas with no trapped gases were identified as areas of more permeable sediments in the seismic-reflection profiles (Jones and others, 2016). Trapped gases often are present in organic-rich lake sediments. Under fully saturated conditions, the permeability of sapropels typically ranges from 5×10^{-4} to 1×10^{-7} ft/d (Karls, 1982; Tiedeman and others, 1997; Winter, 1983). Sapropels are fine-grained, organic-rich sediment (Calvert and Fontugne, 2001) commonly present in lake bottoms.

Values for the water-budget components that were entered into the model input files for the LAK package were (1) precipitation directly on the lakes, (2) evaporation directly from the lake, and (3) surface-water outflows and runoff. Mean annual precipitation values were determined for each lake for 2003–13 using daily total precipitation from the Daymet dataset (Thornton and others, 2014) at the centroid of each lake. As a verification check for the Daymet precipitation dataset, the annual total precipitation from 2003 through 2011 over White Bear Lake was compared to the precipitation record from a single, long-term observation site about 4 mi southwest of White Bear Lake (Peter Boulay, MNDNR, written commun., January 13, 2012; Jones and others, 2013). Daymet annual precipitation totals over White Bear Lake were all within 10 percent of the long-term measured annual precipitation values. Evaporation was calculated for the open-water season for each lake using the Hargreaves-Samani method with daily mean temperature data from the Daymet daily temperature dataset (Thornton and others, 2014) at each lake as described in Jones and others (2016). 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 Lake and Turtle Lake (fig. 9). 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 a monthly basis, and mean 2003–13 values were determined and used in the water-budget calculations for the LAK package. Surface-water runoff into each of the six lakes was estimated using mean runoff determined by the SWB model.

Although five of the six lakes simulated with the LAK package were classified as closed-basin lakes (table 4), surface water outflowed in four of the five closed-basin lakes (Big Marine Lake, Snail Lake, Turtle Lake, and White Bear Lake) for a few months during 2003–13. These outflows generally happened in the spring and early summer of 2003 after increased precipitation in spring 2003 (fig. 2 in Jones and others, 2013), and were included in the mean 2003–13 estimate of surface-water outflow for each lake in the NMLG model. Snail Lake was the only lake not to have any surface-water outflows during 2003–13 (Alan Rupnow, Ramsey County Public Works, oral commun., March 3, 2016).

Mean 2003–13 daily surface-water outflows were determined on the basis of monthly estimates during 2003–13 for the five lakes (Big Marine Lake, Lake Elmo, Snail Lake, Turtle Lake, and White Bear Lake) having surface-water outflows. The mean 2003–13 daily surface-water outflows were used in the LAK package to simulate surface-water runoff to the five lakes in the NMLG model. Total monthly outflows for the lakes for 2003–13 were estimated based on lake-water levels and outflow rating curves for each of the five lakes (Big Marine Lake, Lake Elmo, Snail Lake, Turtle Lake, and White

Bear Lake) with surface-water outflows. The lake-water-level data were obtained from the MNDNR Lakefinder database (MNDNR, 2015c). Existing surface-water outflow rating curves obtained from hydrologic studies, watershed organizations, and engineering firms (MNDNR, 1998; Erik Anderson, Washington Conservation District, written commun., March 1, 2016; John P. Hansen, Valley Creek Watershed District, written commun., March 4, 2016) were used to estimate surface-water outflow for three lakes (Big Marine Lake, Lake Elmo, and White Bear Lake) (fig. 9). Surface water for Pine Tree Lake was determined to outflow for only 2 months (May 2003 and April 2005) during 2003–13. This outflow happened in culverts that often were clogged with sediments, so outflows were estimated based on rating curves for the culverts and observations by watershed organization staff (Matt Kocian, Rice Creek Watershed District, oral commun., March 7, 2016). The outflow for Turtle Lake was estimated based on an existing rating curve for culvert structure (Hulsing, 1967) on the lake and water-level data.

Mean daily runoff volumes contributing to each of the six lakes in the MODFLOW LAK package of the NMLG model were simulated with the calibrated SWB model. A mean daily runoff volume was extracted from the 2003–13 SWB model output within each lake's watershed that accompanies the SWB model (Westenbroek and others, 2010).

Model Solver

The generalized minimum residual solver included with MODFLOW–NWT (Niswonger and others, 2011) was selected as the solution method. The default parameters for the generalized minimum residual solver under the "COMPLEX" option were used (Niswonger and others, 2011). The criteria for model convergence were set at 0.33 ft (0.1 m) for the water-level change tolerance and at 35,300 cubic feet per day (ft^3/d) (1,000 cubic meters per day [m^3/d]) for the flux tolerance.

Model Calibration

The NMLG model was calibrated using existing groundwater-level, lake-water-level, and streamflow data in the model area as targets. The model parameters of hydraulic conductivities, riverbed conductance, lakebed leakance, and recharge rates for the NMLG model were calibrated through manual and automated methods with PEST (Doherty, 2010). The model fit of the simulated groundwater-level, lake-water-level, and streamflow data to similar measured data were evaluated using graphical analysis and error evaluation statistics.

Targets and Target Weights

A total of 3,403 calibration targets were used during the NMLG model calibration process. Data used for these calibration targets were (1) groundwater levels measured in 3,392 wells open to or screened in glacial aquifers or bedrock hydrogeologic units (fig. 10), (2) base flows at 5 streamgages in the model area (fig. 9; table 5), and (3) lake-water levels measured in the 6 lakes simulated in the model using the LAK package (fig. 9; table 4). Many of the groundwater-level calibration targets used to calibrate the NMLG model were used to calibrate the MM3 model. Base flows for Brown's Creek and Valley Creek were used to calibrate the NMLG model but were not used to calibrate the SWB model because the groundwater contributing area and the surface watershed for each creek are not aligned. Mean 2003–13 values were determined and used as calibration targets for calibration at sites with multiple values during 2003–13. Groundwater-level calibration targets were compiled from the USGS NWIS database (USGS, 2015e), MNDNR Cooperative Groundwater program (MNDNR, 2015e), Valley Branch Watershed District (Valley Branch Watershed District, 2015b), and the Minnesota Department of Health MWI (Minnesota Department of Health and Minnesota Geological Survey, 2016). The number of groundwater-level measurements in individual wells varied from 1 to 900 measurements during 2003–13. Wells with groundwater levels used in the model calibration were not closer than 410 ft (125 m) from MNW-package nodes for high-capacity wells simulated in the model using the MNW package. Lake-water-level calibration targets during 2003–13 were compiled from the MNDNR Lake Finder database (MNDNR, 2015c). Streamflow calibration targets were compiled from the USGS NWIS database (USGS, 2015e) and the Metropolitan Council (Jennifer Kostrzewski, Principal Environmental Scientist, Metropolitan Council, written commun., June 24, 2015).

Calibration weights were assigned to each of the individual or mean groundwater levels, lake-water levels, and base flows to ensure that the calibration process was prioritized to fit the higher integrity data points throughout the model area. The calibration weights for groundwater levels were based on (1) the number of measurements at or near the site during 2003–13, (2) the source of the data, and (3) the type of well measured. The MNDNR and USGS groundwater-level measurements were presumed to be more accurate than groundwater-level measurements in MWI wells because (1) MNDNR and USGS measurements were often taken from observation wells that were dedicated for groundwater-level measurements, whereas MWI measurements were typically taken from municipal, domestic, commercial, and irrigation wells immediately following well construction and development (2) the

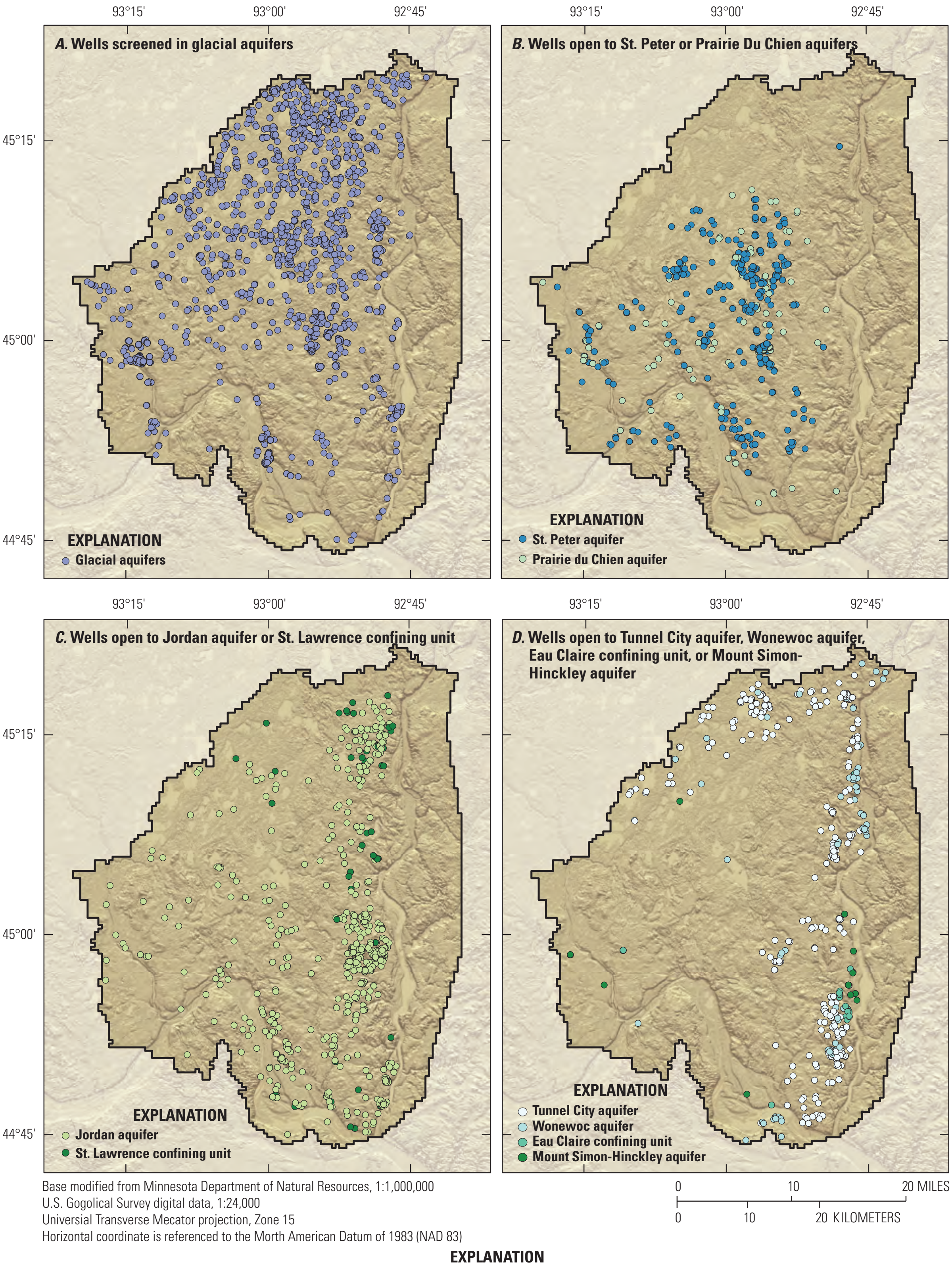


Figure 10. Observation wells with groundwater levels used in calibration of the groundwater-flow model by simulated hydrogeologic units, northeast Twin Cities Metropolitan Area, Minnesota. *A*, wells screened in glacial aquifers; *B*, wells open to the St. Peter or Prairie du Chien aquifers; *C*, wells open to the Jordan aquifer or St. Lawrence confining unit; *D*, wells open to the Tunnel city aquifer, Wonewoc aquifer, Eau Claire confining unit, or Mount Simon-Hinckley aquifer.

Table 5. Summary statistics of 2003–13 stream base flows used to calibrate the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[Negative minimum observed and 25th-percentile values were the result of subtracting base flow at one streamgage from base flow at another streamgage. Values in parenthesis are percentages of measured mean base flows. mi^2, square mile; Mft^3/s, million cubic foot per second]

River name	Data source and streamgage identifier	Surface drainge area (mi^2)	Number of months with complete base-flow data	Measured net base flows (Mft^3/s)							Simulated mean base flow (Mft^3/s)	Mean simulated minus measured base flow (Mft^3/s)
				Minimum	25th percentile	Median	75th percentile	Maximum	Range	Mean		
Brown's Creek	Brown's Creek Watershed District [1]BR-0.3	31	132	0.187	0.39	0.509	0.663	1.61	1.42	0.54	0.54 (100)	0.00 (0)
Mississippi River at Prescott, Wisconsin[2]	USGS streamgage [3]05344500	44,800	108	–548	–3.2	45.9	96.3	900	1,448	53	57 (108)	4 (8)
Rice Creek below Old Highway 8 in Mounds View, Minnesota	USGS streamgage [3]05288580	156	60	0.289	3.00	4.80	6.59	8.44	8.15	4.7	5 (106)	0.3 (6)
Mississippi River at St. Paul, Minnesota[4]	USGS streamgage [3]05331000	36,800	108	–518	–7.7	0.75	48.9	3,229	3,747	53	28 (53)	–25 (47)
Valley Creek	Valley Branch Watershed District [5]VA-1	12	132	0.881	1.10	1.25	1.49	2.04	1.16	1.3	1.3 (100)	0.00 (0)

River name	Data source and streamgage identifier	Surface drainge area (mi^2)	Number of months with complete base-flow data	Calibration weight of base-flow measurement			
				Factor of part of flow (F)	Relative error based on accuracy (c)	Relative error based on record and number of measurements (n)	Initial weight (w, in day per millions of cubic feet)
Brown's Creek	Brown's Creek Watershed District [1]BR-0.3	31	132	1.0	0.16	0.21	109
Mississippi River at Prescott, Wisconsin[2]	USGS streamgage [3]05344500	44,800	108	0.5	0.20	0.21	0.44
Rice Creek below Old Highway 8 in Mounds View, Minnesota	USGS streamgage [3]05288580	156	60	1.0	0.10	0.42	10.0
Mississippi River at St. Paul, Minnesota[4]	USGS streamgage [3]05331000	36,800	108	0.5	0.23	0.21	0.38
Valley Creek	Valley Branch Watershed District [5]VA-1	12	132	1.0	0.16	0.21	45

[1]Karen Kill, Brown's Creek Watershed District, written commun., August 17, 2015.

[2]Flows measured on the St. Croix River at Stillwater, Minnesota (streamgage 05341550), Mississippi River at St. Paul, Minnesota (streamgage 05331000), and Valley Creek were substracted from the flows measured at this streamgage to obtain net base flow. Negative values indicate the base flow at streamgage 05344500 was lower than the combined total of the other gages used in this calculation. It is an artifact of the method and not necessarily a representation of water loss to groundwater.

[3]U.S. Geological Survey, 2015e.

[4]Flows measured on the Mississippi River at Highway 610 in Brooklyn Park, Minnesota (streamgage 05288500), Mississippi River at Fort Snelling State Park, Minnesota (streamgage 05330920) and Rice Creek below Old Highway 8 in Mounds View, Minnesota (streamgage 05288580) were substracted from the flows measured at this streamgage to obtain net base flow. Negative values indicate the base flow at streamgage 05331000 was lower than the combined total of the other gages used in this calculation. It is an artifact of the method and not necessarily a representation of water loss to groundwater.

[5]Jennifer Kostrzewski, Metropolitan Council, written commun., June 29, 2015.

accuracy of water-level tapes used to take the measurements often were checked by the measurer on a semiannual basis, and (3) the reference points, or measuring points, on the wells often were surveyed using local benchmarks.

Groundwater-level targets were initially weighted using the following equation:

$$w_{h_i} = \frac{P_i G_i}{D_i \sqrt{\left(c_i / 1.96\right)^2 + \left(n_i / 1.96\right)^2}} \tag{7}$$

where

w_{h_i} is the initial weight of the well;

G_i is the group weighting factor (table 6) based on the hydrogeologic unit the well is open to or screened in;

P_i is the proximity factor for wells within 10 model cells of a lake cell;

D_i is the proximity factor based on the density of groundwater-level targets within a 410-ft (125-m) grid cell;

c_i is the expected cumulative measurement error (table 6) for a 95-percent confidence interval of the groundwater-level measurement; and

n_i is the estimated error based on the expected confidence interval of the mean value (table 6) and the number of measurements at the well.

Group weighting factors for wells in hydrogeologic units (table 6) were assigned based on determining the exchanges between groundwater and surface-water features within the model area. Wells open to or screened in stratigraphically higher hydrogeologic units (glacial and upper bedrock aquifers), which generally exchanged more groundwater with surface-water bodies, were given higher prioritization, or higher weighting factors (table 6). Weighting factors also were adjusted to accommodate tightly clustered wells. Weights for wells completed in the same hydrogeologic unit and within 410 ft (125 m) of other wells of the same observation class or higher were adjusted by a proximity factor equal to *Di*, which is equal to the square root of the number of wells. Cumulative measurement errors in wells from each source of the groundwater-level data were assigned based on assumed errors with respect to measurement methods and positional accuracy of the well (table 6). The weighting factors for wells screened in glacial aquifers and within a radius of 10 model cells of 1 of the 6 lakes simulated by the LAK package were increased by 25 percent to ensure accurate groundwater and surface-water exchange near these lakes of interest.

Table 6. Summary of weighting factors, expected cumulative measurement errors, and expected confidence intervals used in calculating measured groundwater-level target weights for the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[m, meter]

Hydrogeologic unit well is open to or screened in	Group weighting factor (G_i in eq. 7)
Glacial aquifers, St. Peter aquifer, Prairie du Chien aquifer, Jordan aquifer	1.25
St. Lawrence confining unit, Tunnel City aquifer, Wonewoc aquifer, Eau Claire confining unit, Mount Simon-Hinckley aquifer	1
Data source	**Expected cumulative measurement error (m) (c_i in eq. 7)**
Minnesota Department of Natural Resources (2015e), Valley Branch Watershed District (2015b)	0.6
Minnesota Department of Health and Minnesota Geological Survey (2016), Minnesota Well Index	4.5
U.S. Geological Survey (2015e)	0.3
Observation classes (number of observations)	**Expected confidence interval of mean value (m) (n_i in eq. 7)**
Greater than 100	0.125
11–100	0.25
6–10	0.5
3–5	1
1–2	2
Hydrogeologic unit proximity weight factor	**Weight factor for proximity to lakes (m) (P_i in eq. 7)**
Glacial aquifers	1.5
St. Peter aquifer, Prairie du Chien aquifer	1.25
Jordan aquifer, St. Lawrence confining unit, Tunnel City aquifer, Wonewoc aquifer, Eau Claire confining unit, Mount Simon-Hinckley aquifer	1

Net base flow in designated river and stream reaches throughout the study area were used as calibration targets (fig. 9; table 5). These targets represent the mean contribution of groundwater discharge or base flow to streamflow within the designated reach areas. Base-flow separation was completed using the local minimum method in the HYSEP program (Sloto and Crouse, 1996) on all streamflow data for the streamgages for the period of record available during 2003–13 to produce the calibration targets of mean base flow per day. For streams originating within the model area, the mean daily base flow was the calibration target. For major rivers flowing through the model area (Mississippi and St. Croix Rivers), base flows from the upstream streamgages were subtracted from base flow at the downstream streamgage to get the mean net base flow contributed to the designated reaches within the model area (table 5). Reaches were designated to include all connected flowing and flow-through surface-water features between the downstream steamgage and either the headwaters of the river or an upstream streamgage.

Weighting of stream and river base-flow targets gave consideration to the number of measurements, the quality of the data, and an adjustment factor based on the amount of flow originating within the model area for a measurement by using the following equation:

$$w_{q_i} = \frac{1.96F_i}{c_i n_i Q_i} \tag{8}$$

where

w_{q_i} is the initial weight of the base-flow measurement,

F_i is the factor of part of the flow originating within model area based on watershed boundaries and the model boundary,

c_i is the relative error based on the measurer’s perceived accuracy of the measurement,

n_i is the relative error based on length of record and number of measurements, and

Q_i is the flow observation, in cubic feet per day.

Values for the factors of part of the flow originating within the model area (F) were 0.5 for the Mississippi River base-flow targets (USGS streamgages 05344500 and 05331000) where one-half of the watershed upgradient of the streamgages were within the model boundaries, and 1.0 for the Browns Creek (BR-0.3), Rice Creek (USGS streamgages 05288580), and Valley Creek (VA-1) base-flow targets (fig. 9) where the entire watersheds upgradient of the streamgages were within the model boundaries (table 5). The relative errors based on the measurer’s perceived accuracy of the measurement (c) ranged from 0.10 to 0.23 (table 5) with the highest values (0.20 and 0.23) for the largest river, the Mississippi River. Values for the relative errors based on length of record and number of measurements (n) were 0.21 for four of the base-flow targets, and 0.42 for the Rice Creek base-flow target (table 5), which had only 60 months of base-flow data compared to more than 100 months for each of the other base-flow targets used to calibrate the NMLG model.

Simulated groundwater discharge (base flow) to designated river reaches used in the calibration of the model were compiled from the leakage in RIV cells representing the reach and from UZF-package produced surface leakage in cells containing wetlands contiguously connected to RIV cells representing the reach (fig. 9). Wetland areas contributing to the river observations were determined from the 2006 NLCD 98-ft (30-m) land-cover data (Fry and others, 2011).

Mean 2003–13 lake-water levels in six lakes (fig. 2; table 4; Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake) were the primary calibration targets for the model. Lake-water-level calibration targets were weighted in a similar manner as groundwater-level targets. Individual weights of lake-water-level calibration targets were determined based on (1) the priority of the lake-water level target, (2) the accuracy of the measurements, and (3) the number of observations by using the following equation:

$$w_{l_i} = \frac{L_i}{\sqrt{\left(c_i / 1.96\right)^2 + \left(n_i / 1.96\right)^2}} \tag{9}$$

where

w_{l_i} is the weight of the lake-water-level measurement,

L_i is the factor based on priority of lake-water-level calibration target,

c_i is the cumulative error for a 95-percent confidence interval of measurement, and

n_i is the estimated error based on the number of measurements at the lake.

Values for the factor based on priority of lake-water-level calibration target (L) ranged from 0.98 to 1.65 (table 4), with the White Bear Lake target having the highest value because the lake had one of the highest number of lake-water-level measurements and was of high priority due to a 2003–2010 lake-level decline (Jones and others, 2013). Values of 0.05 and 0.125 (table 4) were used for the cumulative error for a 95-percent confidence interval of measurement (c) and the estimated error based on the number of measurements at the lake (n), respectively, for all the lake-water-level calibration targets because water levels for each of the lakes were measured using similar techniques.

Model Parameters

Horizontal and vertical hydraulic conductivity values were calibrated for glacial aquifers and zones within the bedrock hydrogeologic units in the NMLG model (table 3). Horizontal and vertical hydraulic conductivity values for glacial aquifers and the bedrock zones were calibrated with initial values for each sediment class and bedrock hydrogeologic unit derived from the MM3 model. Hydraulic conductivities for the model layers and multipliers representing bedrock hydrogeologic units were calibrated using a series of pilot points to adjust the hydraulic conductivities in a particular unit incrementally from the values in the MM3 model. Pilot points are surrogate-parameter, point locations in the model where parameter values are estimated and interpolated in the NMLG model area using PEST in such a way that heterogeneity can be represented in the model without estimating hydraulic property values in every cell of the model (Doherty and others, 2010). This series used to calibrate model parameters in the bedrock hydrogeologic units consisted of 40 pilot points (fig. 11). The 40 pilot points were used to calibrate multipliers for the horizontal and vertical hydraulic conductivity values for zones in the St. Peter aquifer (8 pilot points), Prairie du Chien aquifer (12 pilot points), Jordan aquifer (12 pilot points), and St. Lawrence confining unit (8 pilot points) (table 3). Pilot points also were used to calibrate multipliers for the vertical hydraulic conductivity values for the confining parts of the St. Peter aquifer (8 pilot points) and Prairie du Chien aquifer (12 pilot points). The pilot points were located horizontally at 40 locations distributed throughout the central part of the NMLG model area (fig. 11). Of all the bedrock units in the NMLG model, pilot points only were used to calibrate horizontal and vertical hydraulic conductivity values in the St. Peter aquifer, Prairie du Chien aquifer, Jordan aquifer, and St. Lawrence confining unit because (1) they were the shallowest bedrock units underneath the six lakes of interest (Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake) (Bauer, 2016; Meyer and Swanson, 1992) and therefore were likely to have the greatest effect on groundwater and surface-water exchanges, (2) most of the groundwater was withdrawn in aquifers above the St. Lawrence confining unit (Jones and others, 2013), and (3) the number of pilot points used in the model calibration were limited to keep model run times reasonable. Multipliers were used to calibrate for the horizontal and vertical hydraulic conductivity values over the entire model area for the Tunnel City, Wonewoc, and Mount Simon-Hinckley aquifers; and the Eau Claire confining unit (table 3).

Recharge in the model was calibrated by applying a recharge multiplier (table 3) to the mean 2003–13 daily distribution of recharge values produced from the SWB model (fig. 7). During model calibration, the recharge multiplier was allowed to vary between 1 and 2 because recharge values produced from the SWB model were lower than recharge values estimated in other hydrologic studies (Lorenz and Delin, 2007; Metropolitan Council, 2016).

Riverbed conductance values for each surface-water feature were calibrated in PEST based on zonation of the dominant underlying surficial geological unit and the type of surface-water feature (river, lake, or stream) that composed the river cell (table 3). Seven surficial geological zones were created parsimoniously from the aggregation of the surficial glacial geology defined by Meyer (2007) (fig. 5). Areas identified as “modern lacustrine” by Meyer (2007) were grouped with sediments identified as “peat,” and areas identified as “Labradorian till or sand” by Meyer (2007) were grouped with sediments identified as “Superior Lobe till or sand” groups, respectively (fig. 5). Lakes underlain by more than one surficial geological zone were assigned to the zone covering the largest area beneath the lake. Riverbed conductance values for surface-water features represented by the RIV package in the Wisconsin part of the model area (fig. 9) were grouped into a separate zone because an equivalent geologic analysis was lacking for that part of the model area, and the number of surface-water features in the Wisconsin part of the model was small compared to the Minnesota part of the model (fig. 9). Initial values for the 16 riverbed conductances were set to 0.03 ft/d (0.01 m/d) (table 3), but each conductance value was adjusted during calibration.

To improve the refinement of riverbed conductances for river cells representing lakes, an exponential decline function was created to reduce the lake-sediment conductivity with respect to lake-water depth. This general relation between the conductivity of lake sediments and lake depth was gleaned from results of continuous seismic-reflection profiling of four lakes (Big Marine Lake, Lake Elmo, Turtle Lake, and White Bear Lake) in the northeast Twin Cities Metropolitan Area (fig. 2 in Jones and others, 2016). A first-order exponential decline function was applied to all lakes simulated with the RIV package that had bathymetry data. The following equation describes the exponential decline function of lake-sediment conductivity from an initial hydraulic conductivity value for glacial sediments at the shoreline (lake-water depth=0 ft) to a minimum value at a specified maximum depth of 66 ft (20 m).

$$C_d = C_0 e^{kd} \tag{10}$$

where

C_d is the conductivity at depth d, in feet per day;

C_0 is the initial hydraulic conductivity value for glacial sediments at the shoreline (depth d=0 feet), in feet per day;

k the exponential decline coefficient; and

d the depth of water feature, in feet.

The minimum conductivity was determined as a factor of the initial conductivity, and the maximum depth was set to 66 ft (20 m) after review of the lake bathymetry data. The exponential decline coefficient (k) was solved for by using the following equation:

$$k = \frac{\ln\left(C_{min}\right)}{d} \tag{11}$$

where

C_{min} is the exponential decline factor by which C_0 is reduced at depth d.

The exponential decline factors for the lakes represented in the RIV package were adjusted during the model calibration.

Unlike the approach used to compute lakebed conductance values for lakes simulated with the RIV package, lakebed leakance values were calibrated for the three lakebed-conductance zones in the six lakes (Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake) simulated with the LAK package using PEST (table 3). A range in permeability of sapropels (see the “Lake

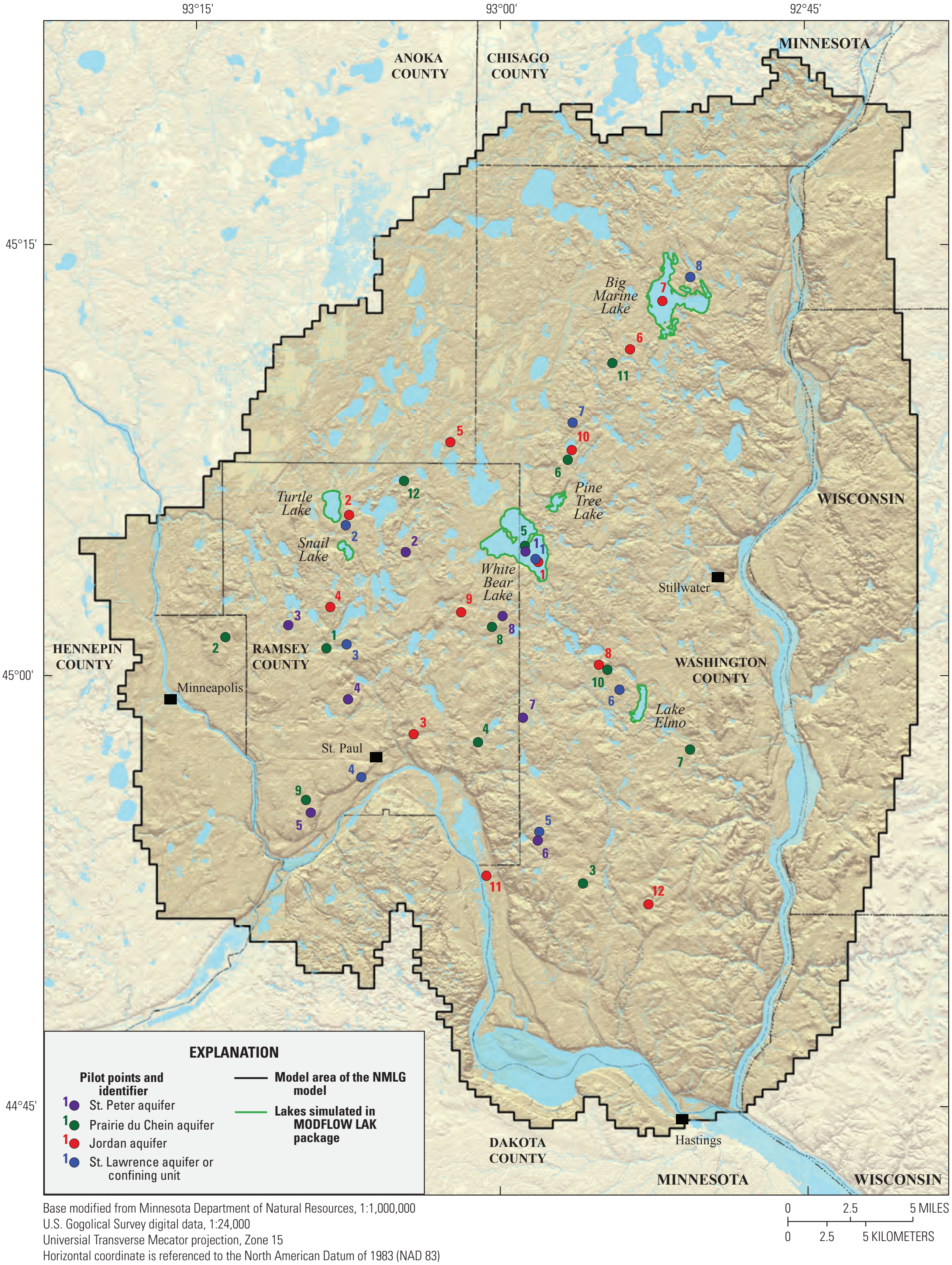


Figure 11. Pilot points used to calibrate the Northeast Metro Lakes Groundwater-Flow (NMLG) model, northeast Twin Cities Metropolitan Area, Minnesota. [MODFLOW, modular finite-difference groundwater-flow model; LAK, Lake]

Simulation with Lake Package" section) was used to estimate a range of initial lakebed leakance values (3.3×10^{-5} to 32 ft/d [1×10^{-5} to 10 m/d]) used as the calibration target range for the low-permeability sediments in PEST (Doherty, 2010). A range in hydraulic conductivity values (0.1 to 3.2 ft/d) determined from slug tests completed using deep-water piezometers in White Bear Lake (Jones and others, 2016) was used to estimate a range of lakebed leakance values (3.3×10^{-4} to 32 ft/d [1×10^{-4} to 10 m/d]) as a calibration target range for the deep-water permeable sediments in PEST (Doherty, 2010). A similar range of lakebed leakance values (3.3×10^{-4} to 49 ft/d [1×10^{-4} to 15 m/d]) was used as a calibration range for the shallow-water permeable sediments in PEST (Doherty, 2010) based on hydraulic conductivity values used in the MM3 model (Metropolitan Council, 2016) for glacial sediments along the six lakes.

Tikhonov regularization was used to limit the deviation of model parameters from their initial values, allowing the modeler's hydrogeologic knowledge and expertise to be incorporated into the calibration process (Doherty and others, 2010). This regularization was done to address the difficulty with using discrete layers to simulate heterogeneity in the glacial and bedrock aquifers and confining units and complex exchanges of groundwater and surface waters near lakes and rivers. Regularization was applied to (1) horizontal and vertical hydraulic conductivity values for glacial sediments (aquifers); (2) horizontal and vertical hydraulic conductivity values and their multipliers for bedrock aquifers and confining units; (3) riverbed conductances for rivers, streams, and lakes; (4) lakebed leakance values for lakes simulated with the LAK package; (5) the recharge multiplier; and (6) surface-water runoff multipliers for lakes simulated with the LAK package (table 3). Regularization was used with pilot points for the upper bedrock aquifers and confining units (St. Peter aquifer, Prairie du Chien aquifer, Jordan aquifer, and St. Lawrence confining unit) to estimate a reasonable field of horizontal and vertical hydraulic conductivity values that balance the need to match calibration targets with limiting the amount of heterogeneity introduced to the horizontal and vertical hydraulic conductivity fields. This approach allows for estimations of hydraulic conductivity values at specific, pilot-point locations and uses a kriging method to interpolate hydraulic conductivity values between the pilot points (Doherty and others, 2010). Regularization weights, which limit model parameter deviations from their preferred values, were calculated as the inverse of the expected parameter uncertainty (as indicated by the lower and upper parameter bounds) multiplied by a weight adjustment factor:

$$w_{r_p} = \frac{F_{pg}}{log\left(UB_p\right) - log\left(LB_p\right)} \qquad (12)$$

where

w_{r_p} is the regularization weight for parameter p,

F_{pg} is the regularization weight adjustment factor for parameter group pg,

UB_p is the upper parameter limit for parameter p, and

LB_p is the lower parameter limit for parameter p.

The regularization weight adjustment factor was used to adjust the regularization weights based on the parameter groups defined in table 3. Parameter groups are groups of hydrogeologic parameters required in PEST that control the model equations used to calculate groundwater levels and flows in the model (Doherty and others, 2010). The parameter groups were delineated based on parameters for hydrogeologic units of similar common characteristics or links, such as the pilot points for a single bedrock unit, for which regularization weights were adjusted as a group to represent common links. The regularization weight adjustment factors were used to weaken the regularization weights on parameters that are relatively estimable based on the current calibration dataset and strengthen the regularization weights on parameters that are relatively inestimable. The information content of the calibration dataset can therefore be transferred to parameters of which the dataset is informative, without having to relax the application of default conditions on parameters for which information in the calibration dataset is weak or absent.

Model parameter regularization was balanced during the calibration process to obtain the best match between simulated values with calibration targets while obtaining hydrologically reasonable parameters during calibration. This was completed by specifying a "lower limit" to the match between simulated and target values, or the target measurement objective function (PHIMLIM, Doherty and others, 2010), and comparing this value to the total difference between the simulated and calibration target value (PHI) for the model. PHIMLIM controls the strength of the model parameter regularization. Low values for specified PHIMLIM weaken the regularization constraint, potentially overfitting simulated and measured values, resulting in parameter "bulls eyes" near observations. High PHIMLIM values result in a strong smoothing of the values but can result in an underfitting of the measured values (Fienen and others, 2009). Achieving a balance between regularization and matching simulated and measured values can be obtained by selecting a PHIMLIM value that meets the goal of matching simulated and measured target values and prevents an unrealistic matching of simulated and measured values with sets of unrealistic model parameters (Juckem and Robertson, 2013).

The selection of PHIMLIM values for the calibration of the NMLG model was done using an iterative process where the PHIMLIM value was set to a relatively low value (500), allowing overfitting, and reviewing the resulting model parameters. The PHIMLIM value was gradually increased until the resulting model parameters showed no signs of overfitting and the model produced a reasonable set of model parameters. This approach is suggested by Doherty and others (2010).

Calibration Process

The MODFLOW groundwater model was calibrated in two phases. First, a base groundwater model was constructed without the LAK package. In the base groundwater model, horizontal and vertical hydraulic conductivities of glacial sediments and bedrock aquifers, riverbed conductances, and the recharge multiplier were calibrated to groundwater-level and streamflow calibration targets (table 3). The only exceptions to this were horizontal and vertical hydraulic conductivities of glacial loam to sandy clay loam sediments, which were kept fixed for both calibration phases (table 3). These parameters were held fixed to their initial values because the glacial loam to sandy clay loam sediments were present in only 3 percent of the model area; therefore, changing these parameters would have little effect on the calibration process. In the base model, lakebed leakance values and runoff multipliers for the six lakes simulated with the LAK package were entirely excluded from the model so that model parameters being calibrated near these lakes during the first calibration phase were calibrated without being affected by the LAK package.

This first calibration phase was done using the Tikhonov regularization scheme for all adjustable parameters and provided the best model onto which to impose the LAK package and the accompanying stresses. A PHIMLIM value of 4,000 was determined for use in the first calibration phase of the NMLG model based on the approach described in the “Model Parameters” section. Fienen and others (2009) indicated that a reasonable PHIMLIM value could be the total number of targets used in the model calibration. The PHIMLIM value determined for the NMLG model calibration was about 18 percent higher than the total number of targets (3,403) used in the model calibration.

Regularization weights and regularization weight adjustment factors generally were set to lower values, allowing more flexibility in exploring parameter values during the calibration process, for model parameters with little data to justify their initial values and were set to higher values for parameters with data to support the initial parameter values set in the model. Regularization weight adjustment factors were set to 1.0 for shallow glacial sediments and 0.5 for deep glacial sediments during the first calibration phase (table 3). The regularization was set to be more flexible for the deeper glacial sediments because sparse hydraulic conductivity data were available for deeper depths. Regularization weight adjustment factors also were set to low values (0.25 and 0.5) for horizontal hydraulic conductivity values for low-permeability, glacial sediments (loam, silt rich, silt and clay); and the Decorah-Platteville-Glenwood confining unit because few hydraulic conductivity values exist for these units (table 3). A similar approach was used for regularization of vertical hydraulic conductivity values for all glacial sediments because of a lack of data. Regularization weight adjustment factors were set low (0.25, table 3) for horizontal hydraulic conductivity multipliers at each of the pilot points for the upper bedrock aquifers and confining units (St. Peter aquifer, Prairie du Chien aquifer, Jordan aquifer, and St. Lawrence confining unit) and for the lower aquifers and confining units (Tunnel City aquifer, Wonewoc aquifer, Eau Claire confining unit, and Mount Simon-Hinckley aquifer) to allow flexibility for determining the conductivity values for the bedrock units across the model area. Regularization weight adjustment factors for the vertical hydraulic conductivity multipliers were set slightly higher (0.5) for the bedrock aquifers and confining units because multipliers varied less than the actual hydraulic conductivity values (table 3). For the confining parts of the St. Peter aquifer and Prairie du Chien aquifer, regularization weight adjustment factors for the vertical hydraulic conductivity multipliers were set to 1.0 (table 3) but were allowed to vary over a large range of values to provide some flexibility during calibration. A large range of values and regularization weight adjustment factors of 0.5 were set for calibration of the riverbed conductances and riverbed-conductance exponential decline factors to allow flexibility of calibration for these values because no data exist for these values in the model area (table 3). The regularization weight adjustment factor for the recharge multiplier was set to 1.25 to reduce the flexibility of changing the recharge values from the SWB-calibrated values (table 3). Base model results during the first phase of calibration provided insight into the initial and expected ranges of values for parameters that were not available from the MM3 model (such as riverbed conductances, exponential decline parameters, and the recharge multiplier).

The second phase of the calibration process implemented the LAK package for the six lakes (Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake) using the previously calibrated base model. Lake-water-level targets for the six lakes were added and assigned weights. Horizontal and vertical conductivities of glacial aquifers, riverbed conductance, lakebed leakance values, recharge multipliers, and lake-specific runoff multipliers were adjusted during this second phase of calibration. The hydraulic conductivity and multiplier values for bedrock aquifers and confining units determined from the first phase of the calibration were fixed during the second phase of the calibration (table 3). The PHIMLIM value was increased from 4,000 used during the first calibration phase to 5,000 for the second calibration phase to account for additional complexities associated with the addition of the six lakes.

Regularization weights and regularization weight adjustment factors were changed for several parameters based partially on results from the first phase of calibration. Regularization weight adjustment factors were doubled for some model parameters to limit the change in these parameters during the second phase of the calibration from values determined from the first phase of calibration. These parameters were the (1) horizontal and vertical hydraulic conductivity values for the glacial sediments, with the exception of the hydraulic conductivity values for the glacial loam to sandy clay loam sediments; (2) horizontal and vertical hydraulic conductivity values for the Decorah-Platteville-Glenwood confining unit; (3) riverbed conductance values for streams and major rivers; and (4) the recharge multiplier.

Regularization weight adjustment factors for these model parameters were doubled because (1) the features represented by these parameters covered large parts of the model area and therefore the values determined during the first phase of calibration likely are the best representation for the hydraulic conductivity values for the glacial sediments and the confining unit over the model area, and (2) the addition of the six lakes simulated with the LAK package cover only small parts of the model area and would likely not have a great effect on these parameters during the second phase of calibration. The regularization factors for the riverbed conductance exponential decline factors were not changed from the first phase of calibration to allow these factors to vary during the second phase as much as they were allowed to vary during the first phase (table 3). This consistency in the regularization factor was done because these factors did not deviate appreciably from their prior initial values during the first phase of calibration of the base model. The riverbed conductance exponential decline parameters were not fixed in the second phase of the calibration because (1) many of the lakes represented by river cells were close to lakes represented by the LAK package and (2) no data were available to confirm or deny the values determined during the first phase of calibration represented changes in riverbed conductance declines in the model area. The regularization factor applied to conductance values of lakebed-conductance zones for the six lakes was set to 1, with exception of the shallow-water permeable lakebed sediments in White Bear Lake, to allow flexibility of the values during the second phase of calibration. This flexibility was done because little information existed on the lakebed hydraulic conductivity for five of the six lakes (Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, and Turtle Lake). The regularization factor for the lakebed leakance values for the shallow-water permeable lakebed sediments in White Bear Lake was set to 2 to allow less flexibility during calibration because more hydraulic conductivity data were available for these sediments (Jones and others, 2013, 2016), providing more confidence in the initial values. The regularization factor for the lake-specific runoff multipliers was set to 2.0 (table 3) because preliminary model runs revealed high sensitivity for the multipliers and the runoff values were generated from calibrated SWB model output. These runoff values were thought to be more accurate than conductance values of lakebed-conductance zones calibrated in the NMLG model because the runoff values were generated from a model calibrated to actual hydrologic data. Conductance values of lakebed-conductance zones generally were unknown for most of the lakes simulated with the RIV or LAK packages.

Model Fit and Model Error

Two techniques were applied to evaluate the fit of the groundwater-flow model to measured lake-water-level, base-flow data, and groundwater level: (1) graphical analysis and (2) model error statistics. The graphical analyses provide a visual comparison of simulated and measured water levels and flows and an overview of model performance (American Society of Civil Engineers, 1993). Residuals (the difference between simulated and measured values) were calculated for lake-water levels, base flows, and groundwater levels. Summary statistics of model residuals used to evaluate the model fit and error include the mean, minimum, maximum, absolute mean, root-mean-square error (RMSE), and the ratio of RMSE to the range of measured values (table 7). Model fit and error

Table 7. Summary of residual statistics for observations in lakes, rivers, and hydrogeologic units in the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[Mft^3/d; million cubic foot per day]

Observation group	Number of observations	Number of observation groups used in model calibration	Units	Mean residual	Minimum residual	Maximum residual	Absolute mean error	Root-mean-square error (RMSE)	Range of observed values	Ratio of RMSE to the range of observed values
Lakes	6	[1]6	Foot	–0.2	–5.6	4.2	2.1	2.9	62.6	0.05
Rivers (base flow)	5	[2]5	Mft^3/s	–4.3	–25.1	3.3	5.8	11.3	52.8	0.21
Glacial aquifers	1,761	1	Foot	2.3	–79.5	84.2	13.9	19.4	370.5	0.05
St. Peter and Prairie du Chien aquifers	492	[3]2	Foot	12.3	–121.7	92.3	17.3	23.3	295.1	0.08
Jordan aquifer and St. Lawrence confining unit	692	[4]2	Foot	0.7	–67.2	105.9	17.1	22.6	361.0	0.06
Tunnel City and Wonewoc aquifers	408	[3]2	Foot	2.3	–70.7	108.0	24.7	31.4	269.0	0.12
Eau Claire confining unit and Mount Simon-Hinckley aquifer	39	[4]2	Foot	1.5	–52.9	44.5	26.5	28.8	146.0	0.20

[1]One group for each lake.

[2]One group for each river.

[3]One group for each aquifer.

[4]One group for the aquifer and one group for the confining unit.

were evaluated for seven sets of observations during 2003–13 (table 7). The mean of residuals indicates model bias depending on the magnitude and direction of the mean from zero. A mean value close to zero indicates a balance between positive and negative residuals, or less model bias. A large positive mean indicates that the model primarily overpredicts measured values (simulated values are greater than measured), and a negative mean indicates that the model primarily underpredicts measured values (simulated values are less than measured). Low RMSE indicates a better model fit to measured values (Anderson and others, 2015; Reilly and Harbaugh, 2004). If a model accurately represents the groundwater-flow system, the residuals are expected to be random, independent, and normally distributed (Hill, 1998). The RMSE is determined by using the following equation:

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n} \left(h_s - h_0\right)^2}{n}} \tag{13}$$

where

h_s is the simulated water-level or flow value, in feet;

h_0 is measured water-level or flow value, in feet; and

n is number of observations.

A calibration criteria (goal) having the RMSE within plus or minus 5 percent of the range of measured mean lake-water levels for the six lakes in the LAK package was applied in the calibration of lake-water levels in the NMLG model. The same calibration goal was applied in the calibration of the mean groundwater levels for five observation groups, or sets of hydrogeologic units, defined in table 7. A calibration goal having total simulated base flows within plus or minus 5 percent of total measured base flows was applied in the calibration of groundwater discharge in the NMLG model.

Model Sensitivity

A sensitivity analysis was completed to determine the response of the model to changes in model parameters. Parameter sensitivities indicate the degree to which additional information could improve model calibration and the relative dependence of simulations on certain model parameters (Delin, 1991). Parameter sensitivities were characterized using PEST (Doherty, 2010), which quantified the changes in simulated lake-water levels, base flows, or groundwater levels resulting from adjustments to model parameters. For the sensitivity analysis, parameters were categorized into 18 parameter groups ("Parameter Group" column in table 3) according to the scheme established for model calibration. Observations were split into 20 observation groups that represented separate hydrologic features belonging to 3 general classes: 6 mean lake-water levels (each lake is a separate observation group), 5 mean base flows, and groundwater level observations made in each of 9 hydrogeologic units (table 8 [available for download at https://doi.org/10.3133/sir20165139B]). The lake-water level, base-flow, and groundwater-level observa-tions that were used for the sensitivity analysis are identical to those used as calibration targets.

After the model was calibrated, a separate PEST run was completed in which all observation weights were set to 1, each parameter was adjusted by 10 percent of its calibrated value, and regularization was not enforced. This approach was done to evaluate the relative importance of model parameters on simulated lake-water levels, base flows, and groundwater levels free of the weighting constraints applied during model calibration. The resulting parameter sensitivities represent the normalized magnitude of the changes in model-simulated values, not scaled by any weighting factors, relative to the calibrated model (Doherty, 2010). PEST calculated the composite sensitivity of each parameter within each observation group (Doherty, 2010). The sensitivities of each parameter within each observation group were then summed for all observation groups in each of the 3 general classes. The values in the last column of table 8 are these summed composite sensitivities in each class. Assigning an equal weight of 1 to each observation allowed for direct comparisons of the composite parameter sensitivities among observation groups within each observation class; for example, the sensitivity of simulated groundwater levels in each of the nine aquifers to the horizontal hydraulic conductivity of the glacial sediments could be compared directly because all the groundwater-level observations were weighted equally.

Information about model parameters during the model calibration is summarized in table 3. The sensitivity of parameters can change as their values change; thus, the sensitivities in table 8 are quantitative only for the parameter values in the "Final calibrated value (Calibrated with Lake Package lakes)" column in table 3.

Hypothetical Scenarios

Eight hypothetical steady-state groundwater-flow scenarios were simulated using the calibrated mean 2003–13 steady-state model to (1) assess the effects of groundwater withdrawals on lake-water levels and budgets, groundwater levels, and groundwater and surface-water exchanges in the northeast Twin Cities Metropolitan Area; and (2) assess the combined effects of different precipitation and groundwater-withdrawal conditions within the 2003–13 simulation period. Only groundwater withdrawals and precipitation were changed in the calibrated mean 2003–13 model for these scenarios, with no changes made to surface-water outflows or other hydrologic components of the water budgets of the lakes simulated with the LAK package.

Changes in lake-water levels for Big Marine Lake, Snail Lake, and White Bear Lake were assessed in these hypothetical scenarios. Turtle Lake and Pine Tree Lake were not included in the hypothetical scenarios because their simulated mean 2003–13 lake-water levels did not fall between maximum and minimum measured lake-water levels recorded in the lakes during 2003–13 (table 4), and simulated water levels in Lake Elmo were above the surface-water outlet elevation for the lake in the hypothetical scenarios, as described in the “Targets” section of this report. Turtle Lake, Pine Tree Lake, and Lake Elmo were simulated with the LAK package in these scenarios, but the results are not presented in this report.

Groundwater-Withdrawal Scenarios

Two hypothetical scenarios were simulated using the calibrated groundwater-flow model to assess potential effects of only changes in groundwater withdrawals from high-capacity wells on lake-water levels for three lakes—Big Marine Lake, Snail Lake, and White Bear Lake. These scenarios were simulated by increasing groundwater withdrawals from high-capacity wells in the model. No new wells were added to the model in these hypothetical scenarios. These hypothetical steady-state scenarios were simulated to assess the sensitivity of lake-water levels to groundwater withdrawals under mean hydrologic conditions during 2003–13.

The two scenarios to assess hypothetical effects of groundwater withdrawals involved (1) increasing mean 2003–13 groundwater withdrawals in the high-capacity wells in the model by 30 percent and (2) decreasing mean 2003–13 groundwater withdrawals in the high-capacity wells in the model by 30 percent. The 30-percent groundwater-withdrawal increases and decreases were chosen based on potential projected population increases in the northeast Twin Cities Metropolitan Area for the year 2040 (Metropolitan Council, 2015). No other changes, including the addition of other wells, were made for these simulations. These groundwater-withdrawal simulations do not account for the potential temporal effects of groundwater withdrawals on lake water levels and water budgets over any specific period of time because the calibrated model simulates steady-state conditions, not transient conditions, over the 2003–13 period.

Changing the 2003–13 mean groundwater withdrawals by 30 percent changed the total amount of water withdrawn from the groundwater system by about 3.5 million cubic feet per day (Mft^3/d), or 3.5 percent of the total outflow from the calibrated model (see the “2003–13 Steady-State Simulation” section). Simulated lake-water levels for the three lakes for the two scenarios were compared to the lake-water levels simulated using the mean 2003–13 groundwater withdrawals for the calibrated steady-state model. The results from the groundwater-withdrawals scenarios are described in the “Simulation Results of Groundwater Flow and Groundwater and Surface-Water Exchanges” section.

Precipitation and Groundwater-Withdrawal Scenarios

To assess the combined effects of different precipitation amounts and groundwater withdrawals on lake-water levels, six hypothetical steady-state scenarios were simulated by altering the mean 2003–13 precipitation by 5 percent to represent dry and wet conditions while either simultaneously varying groundwater withdrawals by 30 percent or using the 2003–13 mean groundwater withdrawals used in the calibrated model. Recharge values, surface-water runoff to, and direct precipitation on lakes simulated with the LAK package in the NMLG model were changed to simulate the effects of changing the precipitation (table 9). Changing the 2003–13 mean precipitation by 5 percent changed the total amount of recharge in the model by 4.7 Mft^3/d, which is about 4.7 percent of the total outflow from the calibrated model. In comparison, changing groundwater withdrawals by 30 percent changed the total amount of water withdrawn from the groundwater system by about 3.5 percent of the total outflow from the calibrated model (see the “2003–13 Steady-State Simulation” section).

The six hypothetical precipitation and groundwater-withdrawal scenarios were constructed to depict drier and wetter precipitation conditions than precipitation measured within the base period of 2003–13 with different groundwater withdrawals. To simulate these scenarios, recharge, precipitation, and runoff to lakes were altered to reflect drier and wetter conditions than mean 2003–13 conditions. Recharge and surface-water runoff to lakes representative of dry and wet conditions were generated with SWB running simulations from 2003 to 2013 with daily precipitation values either decreased or increased by 5 percent, representing dry or wet conditions, respectively (table 9). No other SWB model parameters or variables were changed from the original values for these simulations. These simulations generated mean daily cell-by-cell recharge rates used as input to the UZF package and runoff volumes to lakes for 2003–13 reflective of the 5-percent decrease or increase in daily precipitation; furthermore, daily precipitation values onto the lake surface were decreased and increased by 5 percent for the drier and wetter simulations in the NMLG model, respectively.

The six hypothetical precipitation and groundwater-withdrawal scenarios were: (1) 2003–13 mean groundwater withdrawals, (2) a 30-percent increase, and (3) a 30-percent decrease in 2003–13 mean groundwater withdrawals using a 5-percent increase in 2003–13 mean precipitation; and (4) 2003–13 mean groundwater withdrawals, (5) a 30-percent increase, and (6) a 30-percent decrease in 2000–13 mean groundwater withdrawals using a 5-percent decrease in 2003–13 mean precipitation. By including results from the mean 2003–13 steady-state calibrated simulation and the two groundwater-withdrawal scenario simulations described in the “Groundwater-Withdrawal Scenarios” section, lake-water levels on Big Marine Lake, Snail Lake, and White Bear Lake were compared in nine unique simulations demonstrating the expected relative effects of changes in precipitation and groundwater withdrawals.

Differences among the hypothetical precipitation and groundwater-withdrawal steady-state scenarios do not represent actual precipitation and groundwater-withdrawal changes, but results represent the potential effects of these hypothetical changes on mean, steady-state changes in

Table 9. Mean annual mean surface-water-runoff and precipitation rates for four lakes simulated using the calibrated Soil-Water-Balance model during 2003–13, a 5-percent increase in mean annual total 2003–13 precipitation, and a 5-percent decrease in the mean annual total 2003–13 precipitation, northeast Twin Cities Metropolitan Area, Minnesota.

[MNDNR, Minnesota Department of Natural Resources; ID, identification number; ft^3/day, cubic foot per day; ft/d, foot per day; NAVD 88, North America Vertical Datum of 1988]

MNDNR lake ID	Lake name	Lake type[1]	Reference datum	Surface-water runoff (ft^3/d)		
				5-percent-precipitation decrease from 2003–13 mean	2003–2013 mean	5-percent-precipitation increase from 2003–13 mean
62007300	Snail (62–73 P)	Closed basin	NAVD 88	33,300	38,200	38,600
82005200	Big Marine (82–52 P)	Closed basin	NAVD 88	381,000	408,000	437,000
82010600	Elmo (82–106 P)	Flow through	NAVD 88	63,100	67,700	72,200
82016700	White Bear (82–167 P)	Closed basin	NAVD 88	514,000	552,000	590,000

MNDNR lake ID	Lake name	Lake type[1]	Reference datum	Surface-water runoff per watershed area (ft/d)		
				5-percent-precipitation decrease from 2003–13 mean	2003–2013 mean	5-percent-precipitation increase from 2003–13 mean
62007300	Snail (62–73 P)	Closed basin	NAVD 88	0.00086	0.00098	0.0010
82005200	Big Marine (82–52 P)	Closed basin	NAVD 88	0.0014	0.0015	0.0017
82010600	Elmo (82–106 P)	Flow through	NAVD 88	0.00025	0.00027	0.00029
82016700	White Bear (82–167 P)	Closed basin	NAVD 88	0.0020	0.0022	0.0023

MNDNR lake ID	Lake name	Lake type[1]	Reference datum	Precipitation rate (ft/d)		
				5-percent-precipitation decrease from 2003–13 mean	2003–2013 mean	5-percent-precipitation increase from 2003–13 mean
62007300	Snail (62–73 P)	Closed basin	NAVD 88	0.0071	0.0075	0.0078
82005200	Big Marine (82–52 P)	Closed basin	NAVD 88	0.0067	0.0071	0.0074
82010600	Elmo (82–106 P)	Flow through	NAVD 88	0.0069	0.0073	0.0076
82016700	White Bear (82–167 P)	Closed basin	NAVD 88	0.0071	0.0074	0.0078

[1]As defined in Jones and others, 2016.

lake-water levels. The results from these simulations varying precipitation and groundwater withdrawals are described in the "Precipitation and Groundwater-Withdrawal Scenarios" section.

Simulation and Assessment of Groundwater Flow and Groundwater and Surface-Water Exchanges

This section of the report describes results from the model calibration, sensitivity analysis, and simulations; and outlines some of the limitations of the model. Mean residuals for the simulated lake-water levels, base-flow data, and groundwater levels indicated that the NMLG groundwater-flow model tended to fit measured levels and flows over the model area. Simulated lake-water levels, base-flow data, and groundwater-levels generally were most sensitive to changes in the recharge multiplier, riverbed conductances, and the horizontal hydraulic conductivity values of glacial aquifers. Simulated increases in mean 2003–13 groundwater withdrawals by 30 percent resulted in changes in simulated lake-water levels ranging from –5.5 to 4.38 ft from the mean 2003–13 levels for Big Marine Lake, Snail Lake, and White Bear Lake.

Model Calibration Results

This section of the report describes the evaluation of the NMLG model calibration results. The evaluation is based on (1) the match of simulated values to the calibration targets and (2) the estimation of model parameters.

Targets

As an observation group, lake-water levels for the six lakes simulated with the LAK package (Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake) only were 0.2 ft lower than the measured mean 2003–13 lake-water levels (table 7). The residuals for the lake-water levels seem to be normally distributed (fig. 12*A*). The RMSE of the simulated lake-water levels compared to the measured mean lake-water levels was 2.9 ft, or about 5 percent of the range of measured mean lake-water levels (table 7).

Individually, simulated lake-water levels were within acceptable limits for four of the six lakes simulated with the LAK package. The simulated mean lake-water levels for Big Marine Lake, Lake Elmo, Snail Lake, and White Bear Lake (fig. 1) agreed well with their measured mean 2003–13 lake-water levels (figs. 12*A*, 13*A*), each being within 1.5 ft of their respective measured mean lake-water level (table 4). The simulated lake-water levels for Turtle Lake and Pine Tree Lake did not fall between their maximum and minimum measured lake-water levels during 2003–13 (table 4); therefore, Turtle Lake and Pine Tree Lake were excluded from additional analysis and discussion of hypothetical model simulations. Possible explanations for the poor match between simulated and measured lake-water levels for Turtle Lake and Pine Tree Lake were misrepresentation of the hydrogeology beneath the lakes or misrepresentation of the lake sediments of the lakes (described in the "Model Sensitivity Results" section).

The simulated groundwater discharge into rivers and streams (base flow) acceptably agreed with base-flow values calculated from measured streamflow data for reaches within the model area, with the exception of the calculated base flows for the upstream reach of the Mississippi River at the St. Paul, Minnesota, streamgage (fig. 12*B*; table 5). Simulated base flows for the five streamflow gages (table 5) were less than measured base flows by an average of 4.3 Mft^3/d (table 7), which is only 8 percent of the range of measured base flows. With the exception of the upstream reach of the Mississippi River at the St. Paul, Minnesota, (streamgage 05331000), the simulated base-flow values account for 100 to 108 percent of the measured base flow (table 5). Simulated base flow for the river reach of the Mississippi River between streamgage 05288500 in Brooklyn Park, Minnesota, and streamgage 05331000 in St. Paul, Minnesota (fig. 9), only accounts for 53 percent of the mesaured base flow over this river reach (table 5). This measured base-flow value may be high because the drainage area of the watershed that incorporates the flow for the streamgage likely is too large for use in estimating recharge using the measured streamflows with HYSEP software (table 5; Sloto and Crouse, 1996). The simulated minus measured base-flow residuals were negatively skewed (fig. 12*B*) mainly because of the underestimation of the base flow over the reach of the Mississippi River between streamgage 05288500 in Brooklyn Park, Minnesota, and streamgage 05331000 in St. Paul, Minnesota (fig. 9). The measured base flows at the Mississippi River streamgages spanned 2 orders of magnitude and were much larger than base flows calculated at the other streamgages (table 5). The RMSE of the five simulated base flows was 11.3 Mft^3/d, or 21 percent of the range of the measured base flows (ratio of RMSE to the range of measured base-flow values was 0.21) (table 7). Anderson and others (2015) indicated that if the ratio of RMSE to the range of measured values is small, then the model error for the simulated values are only a small part of the overall model response.

Overall, the simulated groundwater levels were higher than, but linear trends agreed well with, the measured groundwater levels for all hydrogeologic units (figs. 13*C*–13*G*, fig. 14; table 7). Simulated groundwater levels generally followed the 1:1 correlation line over the range of measured values (fig. 13*C*–13*G*). Simulated groundwater levels in the St. Peter and Prairie du Chien aquifers were higher than measured groundwater levels by an average of 12.3 ft (table 7). Very few simulated groundwater levels in these aquifers were more than 5 ft lower than the measured water levels (fig. 14*B*). Simulated groundwater levels in all other hydrogeologic units

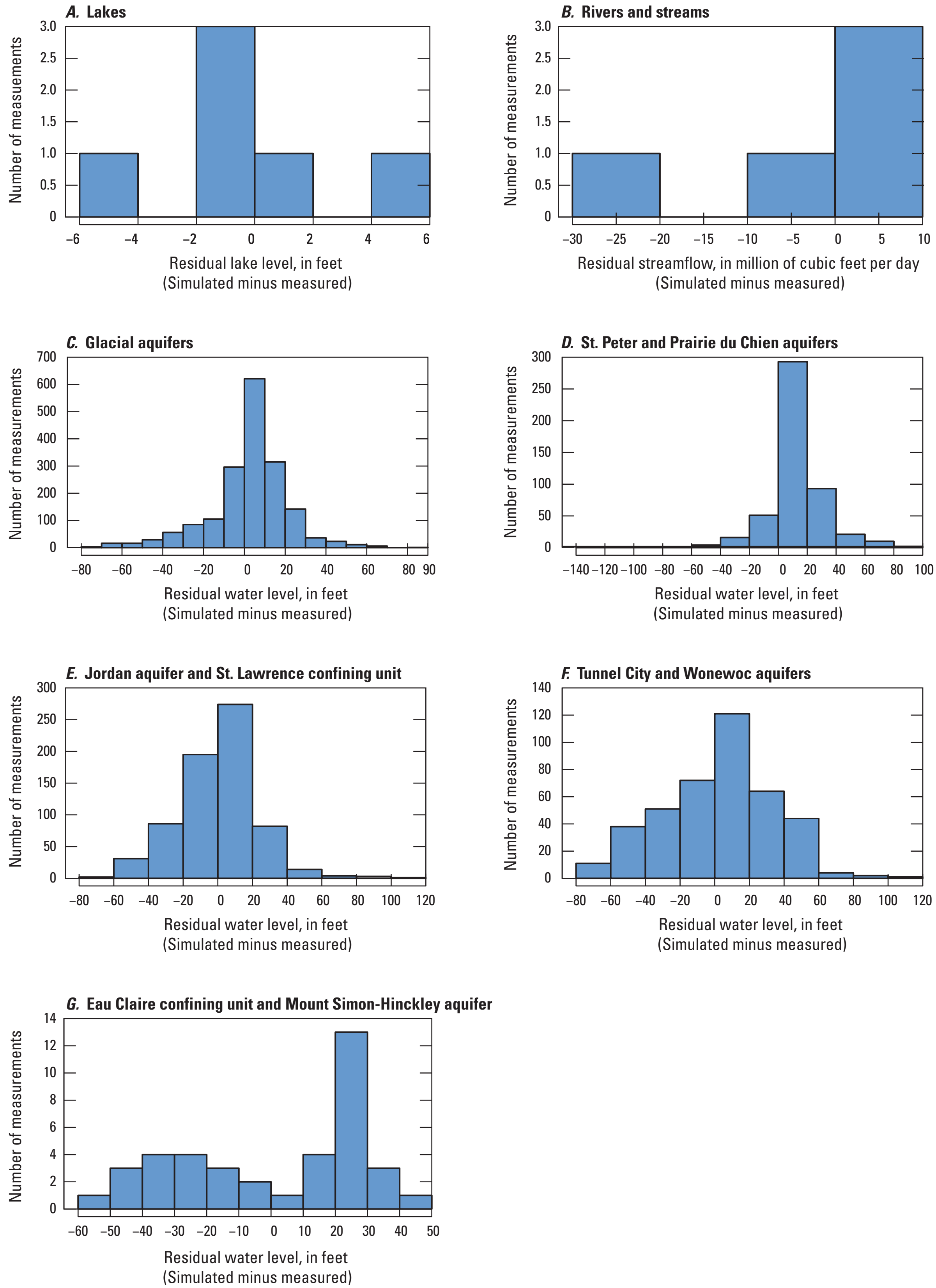


Figure 12. Histograms of residuals for lake-water levels, streamflows, and groundwater levels in observation wells for the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota, 2003–13. *A*, lakes; *B*, rivers and streams; *C*, glacial aquifers; *D*, St. Peter and Prairie du Chien aquifer; *E*, Jordan aquifer and St. Lawrence aquifer or confining unit; *F*, Tunnel City and Wonewoc aquifers; *G*, Eau Claire confining unit and Mount Simon-Hinckley aquifer.

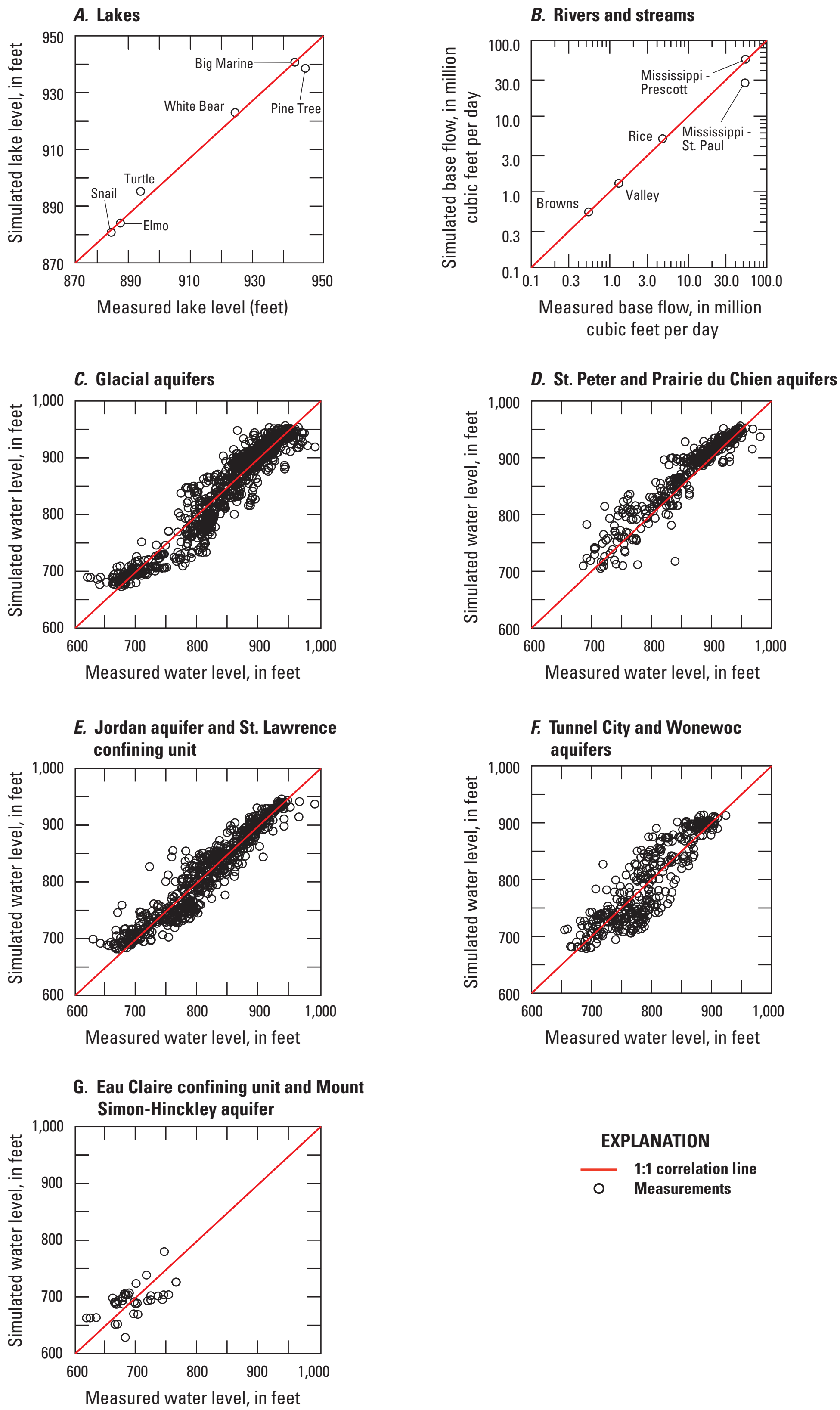


Figure 13. Relations between simulated and measured values for lake-water levels, streamflows, and groundwater levels in observation wells for the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota, 2003–13. *A*, lakes; *B*, rivers and streams; *C*, glacial aquifers; *D*, St. Peter and Prairie du Chien aquifer; *E*, Jordan aquifer and St. Lawrence confining unit; *F*, Tunnel City and Wonewoc aquifers; *G*, Eau Claire confining unit and Mount Simon-Hinckley aquifer.

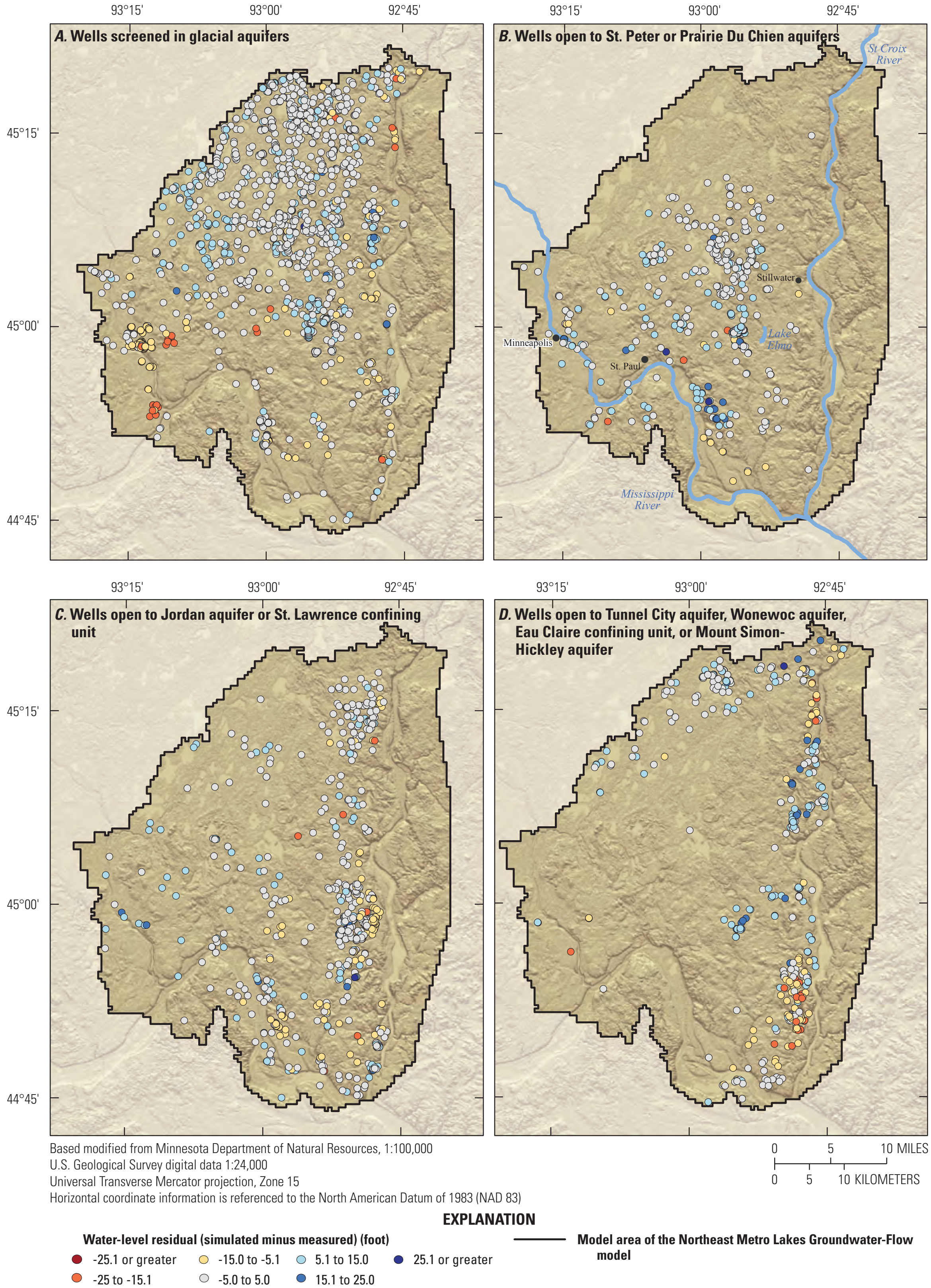


Figure 14. Residuals between simulated and measured groundwater levels by simulated hydrogeologic units in the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota. *A*, wells screened in glacial aquifers; *B*, wells open to the St. Peter or Prairie du Chien aquifers; C, wells open to the Jordan aquifer or St. Lawrence confining unit; *D*, wells open to the Tunnel city aquifer, Wonewoc aquifer, Eau Claire confining unit, or Mount Simon-Hinckley aquifer.

also were higher than measured water levels, but by substantially smaller amounts, with mean residuals for these other hydrogeologic units ranging from 0.7 to 2.3 ft (table 7). The RMSE for all of the hydrogeologic units ranged from 5 to 20 percent of the measured range of groundwater levels in wells open to the hydrogeologic units (table 7). The residuals were approximately normally distributed for hydrogeologic units stratigraphically above the Eau Claire confining unit and Mount Simon-Hinckley aquifer (fig. 12*C*–12*F*) but bimodally distributed for the Eau Claire confining unit and Mount Simon-Hinckley aquifer (fig. 12*G*).

The largest positive and negative residuals for groundwater levels in all of the simulated aquifers and confining units, particularly the Tunnel City, Wonewoc, and Mount Simon-Hinckley aquifers and the Eau Claire confining unit, tended to be near the St. Croix and Mississippi Rivers, although occasional large residuals were closer to the model interior (fig. 14). Steep hydraulic and topographical gradients are near the major rivers and are difficult to represent in a gridded model. Many of the wells open to the Tunnel City, Wonewoc, and Mount Simon-Hinckley aquifers and the Eau Claire confining unit with large positive and negative residuals are near the St. Croix River (fig. 14*D*). Model errors were introduced in these areas because of the simplifications required to represent steep landscapes adjacent to designated river cells in the model, which could have resulted in the large positive and negative residuals. These model errors could explain the bimodal distribution of the model residuals for wells open to the Eau Claire confining unit and Mount Simon-Hinckley aquifer (fig. 12*G*). Steep hydraulic and topographical gradients along the Mississippi River also may explain the higher-than-measured simulated groundwater levels in the St. Peter and Prairie du Chien aquifers near Pigs Eye Lake along the Mississippi River (figs. 1, 14*B*). Throughout the rest of the model area, positive and negative residuals in these two aquifers were relatively evenly distributed (fig. 14*B*).

Simulated groundwater levels in the glacial aquifer in the southwest part of the model tended to be lower than the measured water levels (fig. 14*A*). This area of the model is highly developed (urbanized) (fig. 2) with extremely low simulated recharge rates (0–0.005 inches per year [in/yr], fig. 7). The southwest part of the model contains the highest density of developed medium and high intensity land covers (fig. 2). The developed medium and high intensity land covers were not well represented in the Rice Creek watershed where SWB was calibrated to base flow, or in the other watersheds where SWB was calibrated to runoff (table 2); therefore, the calibrated SWB model may underpredict recharge in this part of the model because the developed land-cover parameters were not well represented and accounted for during the calibration process. The extremely low recharge rates in the southwest part of the model may result in low simulated groundwater levels in this part of the model, underpredicting the measured groundwater levels.

Residuals were generally of greater magnitude for the Tunnel City and Wonewoc aquifers and the Eau Claire confining unit and Mount Simon-Hinckley aquifer compared to the overlying hydrogeologic units (figs. 13*F*–13*G*, 14). The RMSE and absolute mean error for these units are of greater magnitude relative to the other aquifers and confining units (table 7). Simulated groundwater levels for the Tunnel City, Wonewoc, and Mount Simon-Hinckley aquifers and the Eau Claire confining unit typically were at least 5 ft higher than measured groundwater levels near the northern model boundary, along the central part of the St. Croix River, and near Lake Elmo (fig. 14*D*). Simulated groundwater levels for these aquifers and the confining unit typically were at least 5 ft less than measured groundwater levels along the St. Croix River in the northeastern and southeastern part of the model area (fig. 14*D*). The large residuals for these aquifers and the confining unit may have resulted in part from a lack of spatial calibration of the hydraulic conductivity values of theses aquifers and confining unit. During calibration of the NMLG model, a single multiplier parameter for each of these units was used to calibrate and uniformly adjust the initial hydraulic conductivity values for each of these units (Tunnel City aquifer, Wonewoc aquifer, Eau Claire confining unit, and the Mount Simon-Hinckley aquifer) wherever they were present within the model area. In contrast, the initial hydraulic conductivities of the other bedrock units were adjusted by multipliers that varied spatially between pilot points (table 3).

In summary, the NMLG model adequately reproduced measured data as indicated by the model fit and error distribution for lake-water levels, groundwater discharge to streams and rivers (base flow), and groundwater levels. With respect to groundwater levels, the model ran best in the interior parts far from dense urban development and the steep hydraulic and land surface elevation gradients near the St. Croix and Mississippi Rivers (fig. 14).

Model Parameters

Final calibrated parameter values for hydraulic conductivity of the glacial sediments (table 3) were related to the sediment size because the final hydraulic conductivity values for clays were lower than hydraulic conductivity values for sands. Hydraulic conductivity values ranged from 92 ft/d (28 m/d) for the horizontal hydraulic conductivity of shallow sand and gravel sediments (parameter quat hk5 in table 3) to 3.6×10^{-4} ft/d (1.1×10^{-4} m/d) for vertical hydraulic conductivity of deep clay loams (parameter quat_vk8 in table 3). The final calibrated hydraulic conductivity values for these glacial aquifers (table 3) generally were higher than the values suggested by Tipping (2011) but were similar to values in the MM3 model. The medians of horizontal hydraulic conductivity values for the glacial aquifers in the NMLG model and MM3 model were the same, and ranges in the values were similar (table 10). Medians and ranges of vertical hydraulic conductivity values for the glacial aquifers in the NMLG model were an order of magnitude higher than values in the MM3 model. This indicates that the more detailed designation of the glacial sediments in the NMLG model provided more conductivity vertically between layers of the glacial sediments.

Table 10. Horizontal and vertical hydraulic conductivity values for aquifers and confining units simulated in the Metropolitan Council Metro Model 3 and the calibrated Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[ft/d, foot per day]

Aquifer or confining unit	Metro Model 3[1]					
	Horizontal hydraulic conductivity (ft/d)			Vertical hydraulic conductivity (ft/d)		
	25th percentile	Median	75th percentile	25th percentile	Median	75th percentile
Glacial aquifer	5.8	38	71	0.049	0.1	1.1
Decorah Shale, Platteville Formation, and Glenwood Formation	33	45	77	0.013	0.022	0.036
St. Peter aquifer	41	67	117	13	22	38
Prairie du Chien aquifer	26	51	97	1.4	1.7	2.6
Jordan aquifer	23	32	36	1.7	2.4	2.8
St. Lawrence confining unit	0.8	0.8	0.9	0.010	0.011	0.012
Tunnel City aquifer	1.2	1.8	3.1	0.0024	0.0039	0.0086
Wonewoc aquifer	1.1	1.5	2.4	0.1	0.14	0.2
Eau Claire confining unit	0.000106	0.000108	0.000110	0.00000013	0.00000013	0.00000014
Mount Simon-Hinckley aquifer	2.3	3.0	4.4	0.6	0.8	1.1

Aquifer or confining unit	Calibrated Northeast Metro Lakes Groundwater-Flow model					
	Horizontal hydraulic conductivity (ft/d)			Vertical hydraulic conductivity (ft/d)		
	25th percentile	Median	75th percentile	25th percentile	Median	75th percentile
Glacial aquifer	7.7	38	92	0.08	1.3	23.1
Decorah Shale, Platteville Formation, and Glenwood Formation	2.3	16	46	0.00018	0.00026	0.00047
St. Peter aquifer	41	66	117	13.5	21.7	38.0
Prairie du Chien aquifer	23	49	96	1.4	1.7	2.6
Jordan aquifer	22	30	34	1.7	2.5	2.8
St. Lawrence confining unit	0.8	0.8	0.9	0.010	0.011	0.012
Tunnel City aquifer	1.2	1.8	3.1	0.0024	0.0039	0.0086
Wonewoc aquifer	1.1	1.5	2.4	0.1	0.14	0.2
Eau Claire confining unit	0.000106	0.000108	0.000110	0.000000128	0.000000132	0.000000138
Mount Simon-Hinckley aquifer	2.3	3.0	4.4	0.6	0.8	1.1

[1]Metropolitan Council, 2016.

Horizontal and vertical hydraulic conductivity values for the Decorah-Platteville-Glenwood confining unit in the NMLG model generally were lower than values for the MM3 model. Median values for the horizontal hydraulic conductivity values for this confining unit in the NMLG and MM3 models were 16 and 45 ft/d (4.9 to 14 m/d), respectively (table 10). Vertical hydraulic conductivity values for the Decorah-Platteville-Glenwood confining unit in the NMLG model generally were an order of magnitude lower than values in the MM3 model (table 10). During the calibration of the NMLG model, the regularization weight adjustment factor was set to a low value for the Decorah-Platteville-Glenwood confining unit, allowing for flexibility in the estimation of hydraulic conductivity values for the unit. This may have allowed the calibration process to match lower values for the horizontal and vertical hydraulic conductivities. The horizontal and vertical hydraulic conductivity values for the NMLG and MM3 models fall within the ranges of values reported by Runkel and others (2003a).

Horizontal and vertical hydraulic conductivity values for the aquifers and confining units below the Decorah-Platteville-Glenwood confining unit did not vary much from values in the MM3 model. This may be because many of the groundwater-level calibration targets used in calibrating the NMLG model were used in the calibration of the MM3 model. Horizontal and vertical hydraulic conductivity values for the St. Peter aquifer, Prairie du Chien aquifer, and Jordan aquifer were similar to values in the MM3 model even though flexibility was high in the calibration of these parameters using pilot points and multipliers. Median values for the horizontal and vertical hydraulic conductivity values for these aquifers for the NMLG model varied by less than 3 ft/d (0.9 m/d) from values for these aquifers in the MM3 model (table 10). The ranges and median horizontal and vertical hydraulic conductivity values for the Eau Claire confining unit and aquifers in hydrogeologic units beneath the St. Lawrence confining unit for the two models were the same (table 10) even though flexibility was high in the calibration of these parameters in the NMLG model using pilot points and multipliers. The ranges and median horizontal and vertical hydraulic conductivity values for aquifers and the Eau Claire confining unit below the St. Lawrence confining unit were the same as the values in the MM3 model (table 10). Horizontal and vertical hydraulic conductivity multipliers for bedrock hydrogeologic units generally were 1, ranging from 0.95 to 1.03, for most locations (table 3). Lower values for the mulitpliers for the horizontal hydraulic conductivities of the Prairie du Chien aquifer (parameters opdc_hk3, opdc_hk4 in table 3) and Jordan aquifer (parameters cjdn_hk11, cjdn_hk12 in table 3) in the southern part of the model area indicated that these units are less conductive within this model than the initial values derived from the MM3 model.

Calibrated riverbed conductances (table 3) indicate that the conductance of lake sediments within lakes above Superior Lobe sediments (fig. 5) had more conductance than those above Des Moines Lobe sediments (fig. 5). Riverbed conductance values for river cells representing lakes above Superior Lobe sediments were 1.1 ft^2/d (0.1 m^2/d) and 0.44 ft^2/d (0.041 m^2/d) for sand and till, respectively (parameters con3_11 and con4_11, respectively, in table 3), compared with 0.029 ft^2/d (0.0027 m^2/d) and 0.026 ft^2/d (0.0024 m^2/d) above Des Moines Lobe sand and till, respectively (parameters con1_11 and con2_11, respectively, in table 3). This indicates that seepage from lakes underlain by Superior Lobe sediments may be disproportionately important hydrologic features in the region with respect to groundwater flow. The riverbed conductance value of river cells representing lakes underlain by peat or modern lacustrine deposits (fig. 5, parameter con6_11 in table 3) was the lowest riverbed conductance value (0.0086 ft^2/d [0.00080 m^2/d]) used to represent surface-water features with the RIV package in the model. The calibrated values for riverbed conductances representing streams were most conductive on fluvial sediments (0.10 ft^2/d [0.010 m^2/d], parameter con5_12 in table 3), and least conductive on peat or modern lacustrine deposits (0.012 ft^2/d [0.0011 m^2/d], parameter con6_12 in table 3).

Calibrated lakebed leakance values for the six lakes represented by the LAK package generally were one to two orders of magnitude higher in shallow-water, permeable sediments than in the low-permeable sediments. Lakebed leakance values ranged from 0.098 per day (parameter ptr_1) to 1.3 per day (parameter bgm_1) for shallow-water, permeable sediments and ranged from 0.0076 per day (parameter bgm_2) to 0.014 per day (parameter wbl_2) for low-permeable sediments (table 3). These values for low-permeable lakebed sediments are similar to lakebed leakance values applied to the LAK package in groundwater-flow simulations in Wisconsin. Juckem (2009) determined a single lakebed leakance value of 0.019 per day for a simulation of groundwater and surface-water exchanges in Twin Lakes near Roberts, Wisconsin (not on figures), using the LAK package. Juckem and Robertson (2013) used a single lakebed leakance value of 0.0053 per day for a simulation of groundwater and surface-water exchanges in Shell Lake in Washburn County in northern Wisconsin (not on figures) using the LAK package. The lakebed leakance value for the deep-water, permeable sediments was 0.093 per day (parameter wbl_3 in table 3), which was between the ranges of leakance values for the low-permeable sediments and shallow-water, permeable sediments.

The calibrated recharge multiplier value (1.75, parameter rchgm in table 3) indicated that the recharge values for the SWB model may have underestimated the total recharge to the groundwater system by 70 percent. Recharge values produced from the SWB model generally were lower than recharge estimates determined in other hydrologic studies in the Twin Cities Metropolitan Area (as described in the “Soil-Water-Balance Model, Model Results” section). After multiplying the SWB output by the calibrated recharge multiplier of 1.75, the mean recharge rate was 7.7 in/yr. This new mean recharge rate (7.7 in/yr) is similar to the 1988–2011 mean recharge rate (8.2 in/yr) determined by Metropolitan Council (Metropolitan Council, 2016) from a SWB model for the Twin Cities Metropolitan Area.

Final calibration parameters for the two phases of model calibration indicated that regularization of the model parameters had a small effect on the calibration of the model. The PHI value for the first calibration phase was 4,699, which only was 18 percent higher than the PHIMLIM value (4,000) set for the model, indicating that the model likely was well calibrated by the final iteration of the calibration phase. Matching of simulated values to the target values accounted for 86 percent of the total PHI value (4,053), whereas regularization of the model only accounted for 14 percent of the PHI value (646), indicating that matching the calibration targets was a higher priority than regularization during the final iterations of the calibration phase. At the end of the second calibration phase, the PHI value was 6,571, which was 31 percent higher than the PHIMLIM value (5,000) set for the model, indicating that model parameters calibrated during the second calibration phase likely were more difficult to estimate from matching simulated values to the calibration targets. Matching of simulated values to the target values accounted for 89 percent of the total PHI value (5,816), whereas regularization of the model only accounted for 11 percent of the PHI value (755), indicating that matching simulated values to the calibration targets was a higher priority than regularization during the final iterations of the calibration phase.

The more difficult estimation of model parameters during the second calibration phase likely was because of the variability and the reduced regularization of the model parameters during the calibration. Model parameters calibrated during the second calibration phase include hydraulic conductivities for glacial aquifers and Decorah-Platteville-Glenwood confining unit, riverbed conductances, lakebed leakances, and runoff multipliers for the lakes simulated with the LAK package. Generally, values tend to be more variable for these model parameters than for the model parameters calibrated during the first calibration phase, resulting in more difficulty in estimating model parameters during the second phase from matching simulated values to the calibration targets. Also, little data exist for many of the model parameters calibrated during the second calibration phase; therefore, regularization weights were low for these parameters. These relatively low regularization weights allowed for more flexibility in the calibration of these parameters, potentially resulting in more deviation from the calibration target values.

Model Sensitivity Results

Simulated lake-water levels, base flows, and groundwater levels generally were most sensitive to changes in the recharge multiplier, riverbed conductances, and the horizontal hydraulic conductivity values of glacial aquifers (table 8). The three most sensitive parameters for each observation group are indicated with a "++" in table 8. No statistical basis exists for identifying the three most sensitive parameters in each observation group column but was done to provide a context for discussion. The recharge multiplier (parameter group rchgm in table 8) has a large effect on the model's ability to simulate lake-water levels, base flows, and groundwater levels, being 1 of the 3 most sensitive parameters in 16 of 20 observation groups. The recharge multiplier is an important parameter in the model sensitivity because recharge is simulated over most of the model area, and the final value for the multiplier (1.75, table 3) is relatively large compared to other multipliers used in the calibration of the model. Riverbed conductances (parameter group riv_con), which limit the amount of water that can enter or leave aquifers through river cells, were sensitive parameters in 10 observation groups: all 5 base flows and groundwater levels in wells screened in or open to 5 aquifers (glacial, St. Peter, Prairie du Chien, Wonewoc, and Mount Simon-Hinckley aquifers) (table 8). Base flows and groundwater levels for these five aquifers are sensitive to riverbed conductances because base flow values are determined from flows in the rivers, and many of the wells for the five aquifers with high residuals are near the Mississippi or St. Croix Rivers (fig. 14).

The horizontal hydraulic conductivity values for the glacial, Prairie du Chien, and Jordan aquifers are sensitive parameters to the calibration targets. The horizontal hydraulic conductivity values of glacial aquifers (parameter group qhk) have a large effect on the model's ability to simulate lake-water levels and base flows, being sensitive for five of the six lake-water levels simulated with the LAK package and base flows at all five streamgages used in the calibration of the model (table 8). Only the water level in Snail Lake (fig. 9) was not sensitive to the qhk parameter group. This high sensitivity of simulated lake-water levels and base flows to the horizontal hydraulic conductivity values of glacial aquifers supported the need for a detailed discretization of the glacial sediments for the model. The simulated glacial aquifer is in direct contact with the lakes and rivers, and therefore is a major factor in controlling groundwater and surface-water exchanges. The horizontal hydraulic conductivity of the Prairie du Chien aquifer (parameter group opdchk) was sensitive in 10 observation groups, including the lake-water level in 2 lakes (Lake Elmo and Snail Lake) and groundwater levels in wells screened in or open to 6 aquifers (glacial, St. Peter, Prairie du Chien, Jordan, Tunnel City, and Mount Simon and Hinckley aquifers) and 2 confining units (St. Lawrence and Eau Claire confining units) (table 8). The horizontal hydraulic conductivity of the Jordan aquifer (parameter group cjdnhk) was sensitive in seven observation groups, including lake-water levels in three lakes (Big Marine Lake, Turtle Lake, and White Bear Lake) and groundwater levels in wells open to three aquifers (Jordan, Tunnel City, and Woneoc aquifers) and the St. Lawrence confining unit (table 8). The model results indicate that horizontal hydraulic conductivities of the glacial, Prairie du Chien, and Jordan aquifers are major controlling factors in groundwater and surface-water exchanges, because the glacial aquifer is in direct contact with the lakes, the Prairie du Chien aquifer is the shallowest bedrock aquifer in a large part of the model area, and the

Jordan aquifer is the shallowest bedrock aquifer beneath Snail Lake and parts of Turtle Lake and Big Marine Lake (fig. 4). The horizontal hydraulic conductivities the Prairie du Chien aquifer (parameter group opdchk) and Jordan aquifer (parameter group cjdnhk) were sensitive for lake-water levels in five of six lakes simulated with the LAK package (table 8). Only the water level in Pine Tree Lake was not sensitive to these bedrock horizontal hydraulic conductivities.

Water levels in four lakes (Big Marine Lake, Pine Tree Lake, Turtle Lake, and White Bear Lake, [fig. 9]) were sensitive to the lakebed leakance (parameter group lak_k, table 8), which limits the flow of water at the lake sediment and aquifer interface. The three most sensitive parameter groups for Turtle Lake, White Bear Lake, and Big Marine Lake (fig. 9) were lakebed leakance values (parameter group lak_k), the horizontal hydraulic conductivity of glacial aquifers (parameter group qhk), and the horizontal hydraulic conductivity of the Jordan aquifer (parameter group cjdnhk) (table 8). Water levels in the four lakes likely are sensitive to lakebed leakance values and horizontal hydraulic conductivity of glacial aquifers because simulated lake water is in direct contact with the lakebed, and the glacial aquifers are the nearest aquifer to the lakes. The Jordan Sandstone is the lowest bedrock unit beneath parts of Turtle Lake and Big Marine Lake, and therefore changes in the hydraulic conductivity of the Jordan aquifer would likely have a stronger effect on water levels for the two lakes compared to changes in the hydraulic conductivity of the other bedrock hydrogeologic units.

Results from the sensitivity analysis of the model provided some insight into possible explanations for the poor match between simulated and measured lake-water levels for Turtle Lake and Pine Tree Lake. Misrepresentation of the hydrogeology below the lakes or misrepresentation of the lake sediments of the lakes in the NMLG model could explain the poor match. Simulated water levels in Turtle Lake and Pine Tree Lake were sensitive to lakebed leakance values (parameter lak_k, table 8) and the horizontal hydraulic conductivity of glacial aquifers (parameter qvkd, table 8). No geologic logging or hydraulic testing or data exist for lake sediments of and glacial sediments below Turtle Lake and Pine Tree Lake to confirm the geology and associated hydraulic properties below the lakes. Continuous seismic-reflection profiles are available only for Turtle Lake (Jones and others, 2016). An improved conceptual understanding of properties of the lake sediments and glacial sediments below Turtle Lake and Pine Tree Lake may be needed to successfully simulate the lakes in the NMLG model.

Model Simulation Results

Results from the 2003–13 steady-state simulation (calibrated model) and the eight hypothetical scenarios (table 11) are described in this section of the report. The final calibrated steady-state model, the eight hypothetical scenarios, and supporting data are available from Trost and others (2017).

Simulated lake-water outflows to the groundwater system were major outflow components in the simulated water budgets for Lake Elmo, Snail Lake, and White Bear Lake, indicating that the lakes were sources of recharge to the groundwater system. Using the calibrated steady-state NMLG model, simulated lake-water levels for Big Marine Lake, Snail Lake, and White Bear Lake changed with hypothetical changes in precipitation and groundwater withdrawals, indicating that both variables could result in lake-water-level changes in the lakes.

2003–13 Steady-State Simulation

Simulated total inflows and outflows are balanced in the NMLG model, with percent differences between total inflow and outflows being zero (model number 5, table 11). The largest inflow to the model was from areal groundwater recharge to the upper layer of the model, accounting for 41 percent of the total inflow. This recharge was simulated as recharge from the SWB model and calibrated with a recharge multiplier (table 3). Inflow from constant-head cells around the model boundary (37 percent) and inflow from rivers and lakes simulated with cells in the RIV package (17 percent) account for most of the remaining inflows. The largest outflows from the simulated groundwater system were groundwater discharges to the land surface and surface-water features (surface leakage), which accounted for 41 percent of the total outflows (table 11). Most of this surface leakage discharges as groundwater to springs along bluffs and adjacent wetlands of the major rivers (Mississippi and St. Croix Rivers) (61 percent) and to the land surface more than one model cell (410 ft) from a RIV cell and not within the watersheds of the lakes simulated with the LAK package (18 percent) (fig. 15; table 12). The rest of the surface leakage discharges as groundwater to streams, lakes, and wetlands throughout the model area (fig. 15; table 12). Groundwater discharge to river cells (river leakage) (28 percent), constant-head cells (18 percent), and wells (13 percent) accounted for the remaining outflows from the model (table 11). Most of the river leakage discharged to the Mississippi and St. Croix Rivers (88 percent, table 12). Groundwater discharge to the lakes (lake seepage) simulated with the LAK package accounted for less than 1 percent of the total outflow in the model (table 11).

The simulated potentiometric surfaces for the 2003–13 steady-state simulation indicate that general groundwater flow in aquifers is from areas with high potentiometric surfaces (and hence high groundwater levels) in the central part of the model area to areas with low potentiometric surfaces (and hence low groundwater levels), such as along major rivers in the northeast Twin Cities Metropolitan Area (fig. 1). For example, groundwater in the glacial aquifer and the Prairie du Chien aquifer flows eastward from locally high potentiometric surfaces in the north-central model area between White Bear Lake and Big Marine Lake (fig. 16) to the St. Croix River and northwestward towards Rice Creek and ultimately to the Mississippi River (fig. 16). General groundwater flow is from these high potentiometric surfaces to the Mississippi

Table 11. Simulated water budgets for the 2003–13 mean and different hypothetical groundwater-withdrawal and precipitation scenarios using the calibrated Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[Mft^3/d, million cubic foot per day; RIV, MODFLOW River package; LAK, MODFLOW Lake package; UZF, MODFLOW unsaturated-zone flow package; in/yr, inch per year]

Model number	Precipitation conditions	Groundwater-withdrawal conditions	Inflows to simulated groundwater system (Mft^3/d)[1]					
			Constant-head cells	River leakage (rivers, streams, and lakes in RIV package)	Lake seepage (LAK package)	Areal groundwater recharge (UZF package)	Wells (multi-node)	Total inflow
5	2003–13 mean (32.3 in/yr)	2003–13 mean groundwater withdrawals (calibrated model)	37.2 (37)	17.0 (17)	1.7 (2)	41.1 (41)	3.4 (3)	100.4
		Groundwater-withdrawal scenarios						
8		30-percent increase in mean 2003–13 groundwater withdrawals	37.8 (37)	17.5 (17)	1.7 (2)	41.1 (41)	3.5 (3)	101.6
2		30-percent decrease in mean 2003–13 groundwater withdrawals	36.7 (37)	16.4 (17)	2.0 (2)	41.1 (41)	3.3 (3)	99.5
		Precipitation and groundwater-withdrawal scenarios						
6	5-percent-precipitation increase from 2003–13 mean (33.9 in/yr)	2003–13 mean groundwater withdrawals	36.8 (35)	16.1 (15)	3.1 (3)	45.8 (44)	3.3 (3)	105.1
9		30-percent increase in mean 2003–13 groundwater withdrawals	37.3 (35)	16.7 (16)	2.8 (3)	45.8 (43)	3.4 (3)	106.0
3		30-percent decrease in mean 2003–13 groundwater withdrawals	36.2 (34)	15.6 (15)	3.9 (4)	45.8 (44)	3.2 (3)	104.7
4	5-percent-precipitation decrease from 2003–13 mean (30.7 in/yr)	2003–13 mean groundwater withdrawals	38.2 (41)	18.4 (19)	1.5 (1)	32.6 (35)	3.5 (4)	94.2
7		30-percent increase in mean 2003–13 groundwater withdrawals	38.7 (41)	19.1 (20)	1.4 (1)	32.6 (34)	3.6 (4)	95.4
1		30-percent decrease in mean 2003–13 groundwater withdrawals	37.6 (41)	17.8 (19)	1.5 (1)	32.6 (35)	3.4 (4)	92.9

Table 11. Simulated water budgets for the 2003–13 mean and different hypothetical groundwater-withdrawal and precipitation scenarios using the calibrated Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.—Continued

[Mft^3/d, million cubic foot per day; RIV, MODFLOW River package; LAK, MODFLOW Lake package; UZF, MODFLOW unsaturated-zone flow package; in/yr, inch per year]

Model number	Precipitation conditions	Groundwater-withdrawal conditions	Outflows from simulated groundwater system (Mft^3/d)[2] Constant-head cells	River leakage (rivers, streams, and lakes)	Lake seepage (LAK package)	Surface leakage	Wells	Total outflow	Difference between total inflows and outflows (Mft^3/d)[3]
5	2003–13 mean (32.3 in/yr)	2003–13 mean groundwater withdrawals (calibrated model)	18.2 (18)	28.0 (28)	0.5 (0)	40.9 (41)	13.0 (13)	100.6	-0.2 (0)
		Groundwater-withdrawal scenarios							
8		30-percent increase in mean 2003–13 groundwater withdrawals	18.0 (18)	27.4 (27)	0.6 (0)	39.2 (39)	16.5 (16)	101.7	-0.1 (0)
2		30-percent decrease in mean 2003–13 groundwater withdrawals	18.4 (18)	28.6 (29)	0.3 (0)	42.8 (43)	9.4 (10)	99.5	0 (0)
		Precipitation and groundwater-withdrawal scenarios							
6	5-percent-precipitation increase from 2003–13 mean (33.9 in/yr)	2003–13 mean groundwater withdrawals	18.5 (18)	29.0 (28)	0.2 (0)	44.5 (42)	13.0 (12)	105.2	-0.1 (0)
9		30-percent increase in mean 2003–13 groundwater withdrawals	18.3 (17)	28.4 (27)	0.3 (0)	42.6 (40)	16.5 (16)	106.1	-0.1 (0)
3		30-percent decrease in mean 2003–13 groundwater withdrawals	18.7 (18)	29.6 (28)	0.1 (0)	46.9 (45)	9.4 (9)	104.7	0 (0)
4	5-percent-precipitation decrease from 2003–13 mean (30.7 in/yr)	2003–13 mean groundwater withdrawals	17.6 (19)	26.0 (27)	0.7 (1)	37.0 (39)	13.0 (14)	94.3	-0.1 (0)
7		30-percent increase in mean 2003–13 groundwater withdrawals	17.4 (18)	25.4 (27)	0.7 (1)	35.5 (37)	16.5 (17)	95.5	-0.1 (0)
1		30-percent decrease in mean 2003–13 groundwater withdrawals	17.8 (19)	26.6 (29)	0.7 (1)	38.5 (41)	9.4 (10)	93.0	-0.1 (0)

[1]Values in parenthesis are percentages of total inflow.

[2] Values in parenthesis are percentages of total outflow.

[3]Values in parenthesis are percent differences.

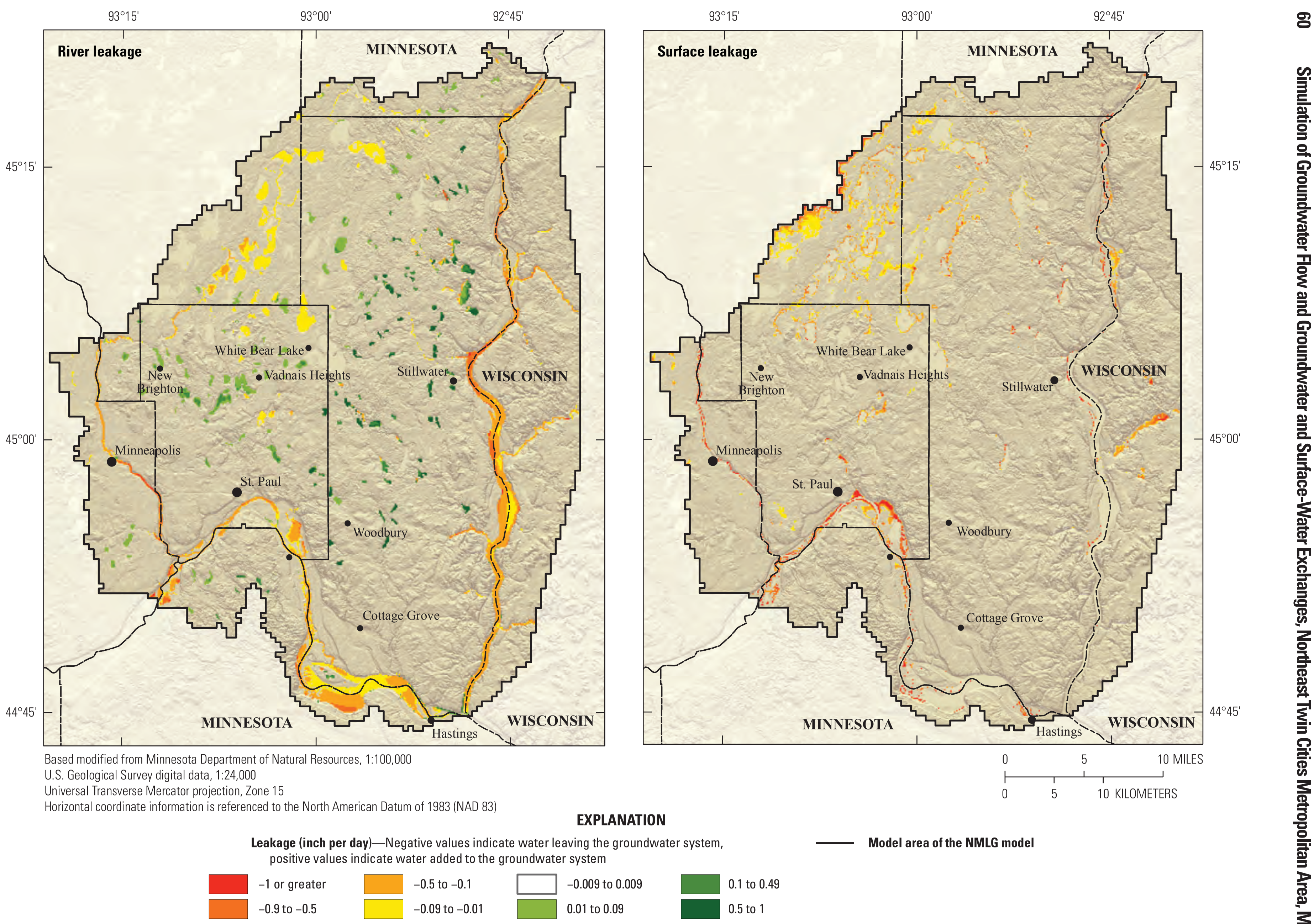


Figure 15. Simulated mean 2003–13 river and surface leakage in the calibrated Northeast Metro Lakes Groundwater-Flow (NMLG) model, northeast Twin Cities Metropolitan Area, Minnesota.

Table 12. Simulated surface and river leakage in the 2003–13 mean and different hypothetical groundwater-withdrawal scenarios using the calibrated Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[UZF, MODFLOW unsaturated-zone flow package; RIV, MODFLOW River package; Mft^3/d, million cubic foot per day; LAK, MODFLOW Lake package; ft, foot; in/yr, inch per year]

Model number	Precipitation conditions	Groundwater-withdrawal conditions	Simulated with the UZF package: Simulated surface leakage out of groundwater system (Mft^3/d)[1]						
			Leakage to major rivers	Leakage to streams	Leakage to lakes in RIV package	Leakage to lakes in LAK package	Leakage to wetlands	Leakage to land surface more than 410 ft from lakes and rivers	Total surface leakage
5	2003–13 mean (32.3 in/yr)	2003–13 mean groundwater withdrawals (calibrated model)	25.1 (61)	3.3 (8)	2.4 (6)	0.4 (1)	2.3 (6)	7.4 (18)	40.9
		Groundwater-withdrawal scenarios							
8		30-percent increase in mean 2003–13 groundwater withdrawals	24.2 (62)	3.2 (8)	2.2 (6)	0.2 (0)	2.3 (6)	7.1 (18)	39.2
2		30-percent decrease in mean 2003–13 groundwater withdrawals	25.9 (60)	3.4 (8)	2.6 (6)	0.8 (2)	2.4 (6)	7.7 (18)	42.8
		Precipitation and groundwater-withdrawal scenarios							
6	5-percent-precipitation increase from 2003–13 mean (33.9 in/yr)	2003–13 mean groundwater withdrawals	25.9 (58)	3.5 (8)	2.8 (6)	1.8 (4)	2.5 (6)	8.0 (18)	44.5
9		30-percent increase in mean 2003–13 groundwater withdrawals	25.1 (59)	3.4 (8)	2.6 (6)	1.4 (3)	2.4 (6)	7.7 (18)	42.6
3		30-percent decrease in mean 2003–13 groundwater withdrawals	26.7 (57)	3.6 (8)	3.0 (6)	2.7 (6)	2.5 (5)	8.4 (18)	46.9
4	5-percent-precipitation decrease from 2003–13 mean (30.7 in/yr)	2003–13 mean groundwater withdrawals	23.6 (64)	3.0 (8)	1.9 (5)	0.0 (0)	2.1 (6)	6.4 (17)	37.0
7		30-percent increase in mean 2003–13 groundwater withdrawals	22.8 (64)	2.9 (8)	1.7 (5)	0.0 (0)	2.0 (6)	6.1 (17)	35.5
1		30-percent decrease in mean 2003–13 groundwater withdrawals	24.4 (63)	3.1 (8)	2.1 (6)	0.1 (0)	2.1 (6)	6.7 (17)	38.5

Table 12. Simulated surface and river leakage in the 2003–13 mean and different hypothetical groundwater-withdrawal scenarios using the calibrated Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.—Continued

[UZF, MODFLOW unsaturated-zone flow package; RIV, MODFLOW River package; ; Mft³/d, million cubic foot per day; LAK, MODFLOW Lake package; ft, foot; in/yr, inch per year]

Model number	Precipitation conditions	Groundwater-withdrawal conditions	Simulated with RIV package: Simulated net leakage to groundwater system from lakes in RIV package (Mft³/d)[2]	Simulated river leakage out of groundwater system (Mft³/d)[3]: Net leakage to major rivers	Leakage to streams	Net river leakage from groundwater system	Total river leakage from groundwater system (Mft³/d)[4]
5	2003–13 mean (32.3 in/yr)	2003–13 mean groundwater withdrawals (calibrated model)	14.7	22.7 (88)	3.0 (12)	25.7	11.0
		Groundwater-withdrawal scenarios					
8		30-percent increase in mean 2003–13 groundwater withdrawals	15.3	22.3 (88)	2.9 (12)	25.2	9.9
2		30-percent decrease in mean 2003–13 groundwater withdrawals	14.0	23.1 (89)	3.0 (11)	26.1	12.1
		Precipitation and groundwater-withdrawal scenarios					
6	5-percent-precipitation increase from 2003–13 mean (33.9 in/yr)	2003–13 mean groundwater withdrawals	13.7	23.3 (88)	3.2 (12)	26.5	12.8
9		30-percent increase in mean 2003–13 groundwater withdrawals	14.3	22.9 (88)	3.1 (12)	26.0	11.7
3		30-percent decrease in mean 2003–13 groundwater withdrawals	13.0	23.7 (88)	3.2 (12)	26.9	13.9
4	5-percent-precipitation decrease from 2003–13 mean (30.7 in/yr)	2003–13 mean groundwater withdrawals	16.4	21.4 (89)	2.6 (11)	24.0	7.6
7		30-percent increase in mean 2003–13 groundwater withdrawals	17.1	20.9 (89)	2.6 (11)	23.5	6.4
1		30-percent decrease in mean 2003–13 groundwater withdrawals	15.7	21.8 (89)	2.7 (11)	24.5	8.8

[1]Values in parenthesis are percentages of total surface leakage.

[2]Simulated net leakage to groundwater system from lakes in RIV package is equal to the total leakage to the groundwater system minus the total leakge from the groundwater system to lakes in the RIV package.

[3]Values in parenthesis are percentages of net river leakage.

[4]Total river leakage from groundwater system is equal to the net river leakage from the groundwater system minus the simulated net leakage into groundwater system from lakes in RIV package.

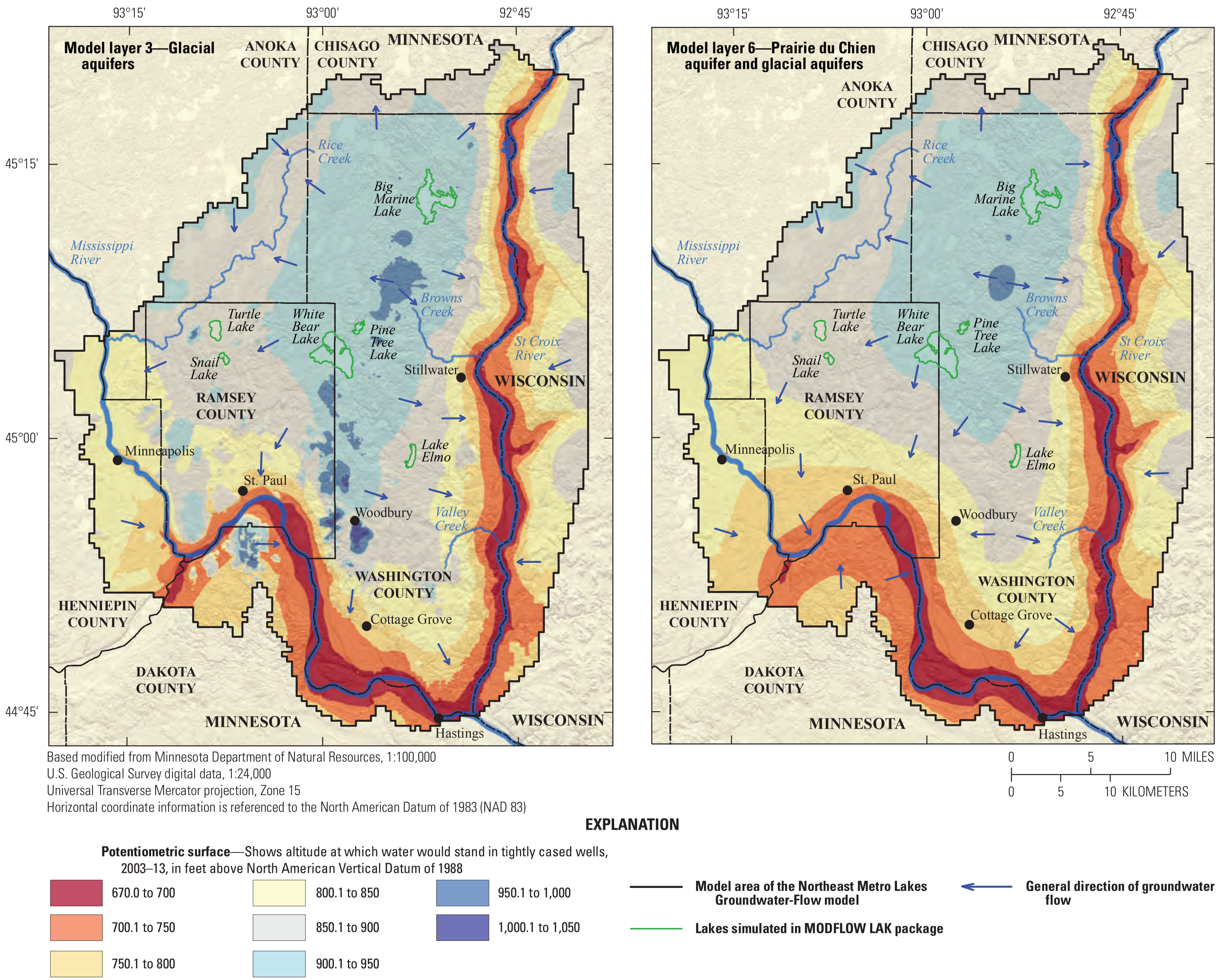


Figure 16. Simulated potentiometric surface for 2003–13 for model layer 3, glacial aquifers, and model layer 6, Prairie du Chien aquifer and glacial aquifers in the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota. [MODFLOW, modular finite-difference groundwater-flow model; LAK, Lake]

River on the southwest side and to the St. Croix River on the east side (fig. 16). In the southern part of the model area, the highest potentiometric surfaces are in glacial aquifers overlying the Decorah-Platteville-Glenwood confining unit (fig. 3). These high groundwater levels resulted from the confining unit restricting the downward movement of groundwater in the overlying glacial aquifers. Simulated groundwater levels typically drop more than 100 ft from glacial aquifers to bedrock aquifers below the Glenwood Formation in areas where the Decorah-Platteville-Glenwood confining unit is present. Groundwater levels gradually drop with proximity to the confluence of the Mississippi and St. Croix Rivers (fig. 16) with groundwater levels maintained in areas underlain by bedrock outcrops near the bluffs of the St. Croix River.

The simulated groundwater levels indicate low groundwater levels near wells, exhibiting the effects of groundwater withdrawals on localized groundwater flow; for example, groundwater withdrawals from the Prairie du Chien aquifer south of White Bear Lake (fig. 16) result in groundwater levels at wells about 10 ft lower than groundwater levels further from the wells. These simulated low groundwater levels are likely partially because of the confining conditions in an area where the overlying Decorah-Platteville-Glenwood confining unit to the south of White Bear Lake (fig. 4) prevents infiltration at the land surface from recharging the Prairie du Chien aquifer in this area. The simulated potentiometric surfaces in the White Bear Lake area generally are similar to potentiometric surfaces determined by Jones and others (2013) for groundwater synoptic studies in March and August 2011. Groundwater levels in the glacial and Prairie du Chien aquifers also were low near high-capacity wells in the southern Washington County near the cities of Cottage Grove and Woodbury.

Simulated water budgets for closed-basin lakes of Big Marine Lake, Snail Lake, and White Bear Lake indicated that groundwater inflow was not the major inflow component to the simulated lakes (table 13). Simulated groundwater inflow accounted for 14 to 25 percent of the total simulated inflow to these three lakes (table 13). Precipitation on the lake was the largest contributor to Big Marine Lake and White Bear Lake, whereas Snail Lake obtains an equal percentage of total water inflow from precipitation and surface-water runoff (42 percent, table 13). Direct precipitation is the largest inflow component for Big Marine Lake and White Bear Lake because they are large lakes, and the ratios of lake watershed area to lake surface area for the lakes are small (table 4), resulting in the other inflow components being small. Large lakes typically obtain much of their water from precipitation (Wilcox and others, 2007; Watras and others, 2013). Surface-water runoff accounted for 30 and 24 percent of the total inflows to Big Marine Lake and White Bear Lake, respectively (table 13).

Groundwater inflow was the major inflow component to the flow-through lake of Lake Elmo, accounting for 60 percent of the total inflow to the lake (table 13). Lake Elmo is a very deep (maximum depth of 137 ft), narrow lake with steep slopes (Jones and others, 2016), which results in a large surface area for groundwater and surface-water exchanges. Because the lake is narrow and the local watershed is small (Valley Branch Watershed District, 2015c), precipitation and surface-water runoff to the lake only accounts for 21 and 19 percent, respectively, of the total inflow to Lake Elmo.

Simulated lake-water outflow to the groundwater system was the major outflow component in the simulated water budgets for Lake Elmo, Snail Lake, and White Bear Lake, indicating that these lakes were recharging water to the glacial aquifer systems. Simulated lake-water outflows in these three lakes ranged from 46 to 64 percent of the total outflows from the lakes (table 13). Lake-water flow to groundwater also was a major outflow component to the water budget for Big Marine Lake, accounting for 45 percent of the total outflow from the lake. Simulated evapotranspiration was the major outflow component (52 percent of total outflows) for Big Marine Lake (table 13), likely because it is a large lake with a large surface area for evaporation. Evapotranspiration also accounted for 44 percent of the total outflow from White Bear Lake (table 13), a large lake with a large surface area. Surface-water outflow was less than 5 percent of total outflows for the lakes, except for Lake Elmo (table 13). Lake Elmo is the only flow-through lake of the four lakes with substantial simulated surface-water outflow, which accounted for 35 percent of the total simulated outflow from the lake (table 13).

The simulated water budgets for Big Marine Lake, Lake Elmo, and White Bear Lake (table 13) are similar to water budgets determined in water-budget studies that measured groundwater inflows to the lakes. Alexander and others (2001) determined that groundwater-inflow contributions to Big Marine Lake (fig. 9) were small (less than 10 percent) with most of the water entering the lake coming from precipitation. Simulated groundwater inflow to Big Marine Lake in the NMLG model also was a small component of the lake's water budget, which only accounted for 14 percent of the total inflow to the lake (table 13). The Metropolitan Council (1983) estimated that about 80 percent of the total inflow to Lake Elmo came from groundwater inflow and direct precipitation. Simulated groundwater inflow and direct precipitation in the 2003–13 mean steady-state simulation accounted for 81 percent of the total inflow to the lake (table 13). Jones and others (2013) estimated that the percentage of groundwater inflow of the total water inflow to White Bear Lake from glacial aquifers in August 2011 was 20 percent, whereas the simulated water budgets using the NMLG model determined that 25 percent of the total inflow to the lake was from groundwater inflow (table 13). No known water-budget studies were done for Snail Lake.

Hypothetical Scenarios

Total water budgets for the model, potentiometric surfaces, lake-water budgets, and lake-water levels for the two 30-percent changes in groundwater-withdrawal scenarios and for the calibrated mean 2003–13 steady-state model were compared to assess the effects of changes in groundwater withdrawals. Total water budgets and lake-water levels for

Table 13. Simulated mean 2003–13 water budgets for four lakes simulated with the U.S. Geological Survey's modular finite-difference groundwater-flow model (MODFLOW) Lake (LAK) package in the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[Values in parantheses represent the percent of total inflow or outflow. MNDNR, Minnesota Department of Natural Resources; ID, identification number; Mft^3/d, million cubic foot per day]

MNDNR lake ID	Lake name	Lake type[1]	Simulated 2003–13 mean values							
			Lake inflow (Mft^3/d)				Lake outflow (Mft^3/d)			
			Precipitation	Surface-water inflow (runoff)	Groundwater inflow	Total inflow	Evapotranspiration	Surface-water outflow	Outflow to groundwater system	Total outflow
62007300	Snail (62–73 P)	Closed basin	0.05 (42)	0.05 (42)	0.02 (16)	0.12	0.04 (36)	0.00 (0)	0.07 (64)	0.11
82005200	Big Marine (82–52 P)	Closed basin	0.52 (56)	0.28 (30)	0.13 (14)	0.93	0.48 (52)	0.03 (3)	0.41 (45)	0.92
82010600	Elmo (82–106 P)	Flow through	0.08 (21)	0.07 (19)	0.22 (60)	0.37	0.07 (19)	0.13 (35)	0.17 (46)	0.37
82016700	White Bear (82–167 P)	Closed basin	0.72 (51)	0.34 (24)	0.35 (25)	1.41	0.62 (44)	0.01 (1)	0.77 (55)	1.4

[1]As defined in Jones and others, 2016.

the hypothetical precipitation and groundwater-withdrawal scenarios and for the calibrated mean 2003–13 steady-state model were compared to assess the effects of precipitation and combined effects of changes in groundwater withdrawals and precipitation.

Water levels in three closed-basin lakes simulated with the LAK package (Big Marine Lake, Snail Lake, and White Bear Lake) were evaluated in hypothetical scenarios where the simulated lake-water levels were below the lake's surface-water outlet elevations and the maximum specified bathymetric elevations for the lake in the LAK package. Simulated water levels for Lake Elmo were not assessed in the hypothetical scenarios because simulated lake-water elevations were above the lake's surface-water outlet elevation in all these simulations. Lake-water levels above the lake's surface-water outlet elevation were not evaluated because actual surface-water outflows at these elevations would differ from the 2003–13 mean surface-water outflows used in the calibrated model. This happens in 7 of the 24 simulated total water levels for Big Marine Lake, Snail Lake, and White Bear Lake in the hypothetical scenarios (table 14). The 2003–13 mean surface-water outflows for the lakes were not changed in the hypothetical simulations, and therefore the simulated surface-water outflows from the lakes in these scenarios would not represent actual surface-water outflows for lake-water elevations above the outlet elevations. Lake-water levels above the maximum specified bathymetric elevation for each lake in the LAK package were not evaluated because precipitation and evaporation rates on the lakes could not be determined for elevations above the maximum specified bathymetric elevations. Lake-water level results for Turtle Lake and Pine Tree Lake were not included in this report because their simulated mean 2003–13 lake-water levels did not fall between maximum and minimum measured lake-water levels recorded in the lakes during 2003–13 (table 4).

Groundwater-Withdrawal Scenarios

A 30-percent increase and a 30-percent decrease in mean 2003–13 groundwater withdrawals from wells in the NMLG model resulted in only small changes in the other components of the total water budget for the model (model numbers 2 and 8, table 11). The total amount of groundwater withdrawals (wells) from the model for the 30-percent increase and a 30-percent decrease in mean 2003–13 groundwater-withdrawal scenarios account for 16 and 10 percent, respectively, of the total outflow from the model simulations (model numbers 8 and 2, respectively, table 11). These percentages are 3-percent increases and decreases from the 13 percent of the total outflow in the 2003–13 mean steady-state simulation (model 5, table 11). In addition to the changes in groundwater withdrawals, the largest changes in the total water budgets with the 30-percent changes in groundwater withdrawals, although small, happened in changes in (1) inflows from constant-head cells along the model boundary; (2) inflow and outflows from river cells representing rivers, streams, and lakes in the model; and (3) outflow as surface leakage (table 11). Changes in these components of the total water budgets are because of the changes in groundwater levels associated with the increase or decrease in groundwater withdrawals; groundwater levels declined in areas near wells when groundwater withdrawals were increased, whereas groundwater levels rose in areas near wells when groundwater withdrawals were decreased. Changes in the inflow from the constant-head cells along the model boundaries occurred because high-capacity wells withdrawing groundwater from the simulated aquifers are near the model boundary (fig. 8). Increases in groundwater withdrawals in wells near the model boundary resulted in lower groundwater levels at the wells, resulting in larger hydraulic gradients between the boundaries and the wells and larger inflows from the constant-head boundaries where inflows occurred (table 11). Decreases in groundwater withdrawals in wells near the model boundary resulted in higher groundwater levels at the wells, resulting in smaller hydraulic gradients between the boundaries and the wells and smaller inflows from the constant-head boundaries where inflow occurred (table 11). Increasing and decreasing groundwater withdrawals by 30 percent resulted in 0-percent changes, respectively, in the amounts of inflow and outflows from constant-head cells along the model boundary. These small percentages indicate that the effects of 30-percent changes in groundwater withdrawals on the groundwater system were barely affected by the constant-head boundary conditions. Changes in river leakage inflows and outflows were less than 2 percent with the 30-percent changes in groundwater withdrawals (table 11). Similar to the changes in inflows at the constant-head boundary, change in the river leakage resulted from changes in groundwater levels associated with the changes in groundwater withdrawals. Changes in groundwater outflows to the land surface (surface leakage) and river leakage from river cells accounted for most of the changes in total outflows with the 30-percent changes in groundwater withdrawals (table 11). Lower groundwater levels resulting from the 30-percent groundwater-withdrawal increase resulted in less groundwater being discharged to the land surface and to river cells, whereas higher groundwater levels resulting from the 30-percent groundwater-withdrawal decrease resulted in more groundwater being discharged to the land surface and river cells (table 11).

Simulated potentiometric surfaces for the 30-percent increase and decrease in groundwater withdrawals indicated areas of groundwater-level declines and rises, respectively, from the 2003–13 mean (figs. 17, 18, respectively) in the western and southern parts of northeast Twin Cities Metropolitan Area; for example, groundwater-level declines resulting from the 30-percent increase in groundwater withdrawals ranged from 0 to 20.4 ft in the glacial aquifers (model layer 3) and from 0 to 26.5 ft in the Prairie du Chien aquifer and glacial aquifers (model layer 6, fig. 17). In general, groundwater-level declines in the glacial and Prairie du Chien aquifers associated with increased groundwater withdrawals were greater in areas of higher well

Table 14. Summary of simulated lake-water levels and lake-water level differences for the 2003–13 mean and different hypothetical groundwater-withdrawal and precipitation scenarios using the calibrated Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[Values in parentheses are model numbers (model 5 is the calibrated model; models 1–4 and 6–9 are the hypothetical scenarios). Numbers in bold represent values determined with the calibrated model. NAVD 88, North America Vertical Datum of 1988; in/yr, inch per year; --, no difference]

Lake	Groundwater-withdrawal conditions (calibrated model and hypothetical scenarios)	Precipitation conditions					
		Simulated lake-water elevations (ft NAVD 88)			Simulated lake-water-level difference from base simulation (ft)		
		5-percent-precipitation decrease from 2003–13 mean (30.7 in/yr)	2003–2013 mean (32.3 in/yr)	5-percent-precipitation increase from 2003–13 mean (33.9 in/yr)	5-percent-precipitation decrease from 2003–13 mean (30.7 in/yr)	2003–2013 mean (32.3 in/yr)	5-percent-precipitation increase from 2003–13 mean (33.9 in/yr)
Big Marine Lake	30-percent decrease in mean 2003–13 groundwater withdrawals	938.14 (1)	940.76 (2)	[1]943.16 (3)	-2.60 (1)	0.02 (2)	[1]2.42 (3)
	2003–2013 mean (calibrated model)	938.13 (4)	**940.74 (5)**	[1]943.42 (6)	-2.61 (4)	--	[1]2.68 (6)
	30-percent increase in mean 2003–13 groundwater withdrawals	938.11 (7)	940.73 (8)	[1]943.45 (9)	-2.63 (7)	-0.01 (8)	[1]2.71 (9)
Snail Lake	30-percent decrease in mean 2003–13 groundwater withdrawals	881.15 (1)	[2]883.72 (2)	[1,2]885.19 (3)	0.34 (1)	[2]2.91 (2)	[1,2]4.38 (3)
	2003–2013 mean (calibrated model)	878.30 (4)	**880.81 (5)**	882.47 (6)	-2.51 (4)	--	1.66 (6)
	30-percent increase in mean 2003–13 groundwater withdrawals	875.27 (7)	878.09 (8)	879.79 (9)	-5.54 (7)	-2.72 (8)	-1.02 (9)
White Bear Lake	30-percent decrease in mean 2003–13 groundwater withdrawals	921.63 (1)	924.48 (2)	[1]926.36 (3)	-1.36 (1)	1.49 (2)	[1]3.37 (3)
	2003–2013 mean (calibrated model)	919.98 (4)	**922.99 (5)**	[1]925.05 (6)	-3.01 (4)	--	[1]2.06 (6)
	30-percent increase in mean 2003–13 groundwater withdrawals	918.17 (7)	921.45 (8)	924.05 (9)	-4.82 (7)	-1.54 (8)	1.06 (9)

[1]Simulated lake-water level or lake-water level difference likely is not accurate because the simulated lake-water level is higher than surface-water outlet elevation, and the model did not simulated surface-water outflow.

[2]Simulated lake-water elevation likely is not accurate because the simulated lake-water elevation is higher than maximum baythymetric elevation specified in MODFLOW Lake (LAK) packge.

density and greater withdrawals in the central to southern parts of the model area (figs. 8, 17) and approached zero in outlying areas having fewer wells and lower withdrawals, such as in the northern part of the NMLG model area. Groundwater-level differences relative to the calibrated model in the glacial aquifers and Prairie du Chien aquifer as a result of a 30-percent increase in groundwater withdrawals were most prevalent in southern Washington County, near the cities of Cottage Grove and Woodbury (fig. 17). These groundwater-level differences between the calibrated model and the 30-percent increase scenario spanned much of the aquifers between the Mississippi River and St. Croix River, and expand to the north towards White Bear Lake (fig. 17). Notable groundwater-level differences between the 30-percent groundwater-withdrawal increase scenario and the calibrated model occur in wells open to the Prairie du Chien aquifer in the cities of New Brighton (–4.0 ft), St. Paul (–4.3 ft), South St. Paul (–8.6 ft), and White Bear Lake (–7.7 ft). The Prairie du Chien aquifer is the shallowest bedrock aquifer in south Washington County and other parts of the Twin Cities Metropolitan Area, including near some of these wells (figs. 4, 17), and likely is hydraulically connected to glacial sediments in these areas. Lakes are not abundant in south Washington County to provide water to shallow aquifers compared to other parts of the northeast Twin Cities Metropolitan Area (fig. 1). As a result, groundwater withdrawals from the Prairie du Chien aquifer and Jordan aquifer, which can be hydraulically connected to the Prairie du Chien aquifer in the northeast Twin Cities Metropolitan Area, can result in groundwater-level declines in the glacial aquifer. Groundwater levels declined as much as 16.8 ft in the simulated glacial aquifer northeast of Cottage Grove (fig. 17) as a result of 30-percent increase in groundwater withdrawals from the Prairie du Chien and Jordan aquifers, indicating a strong connection between the glacial, Prairie du Chien, and Jordan aquifers in this area. Lower groundwater-level declines in the glacial and Prairie du Chien aquifers in the western part of the northeast Twin Cities Metropolitan Area between the cities of New Brighton and White Bear Lake (fig. 17) could be because (1) groundwater is extracted from a larger variety of glacial and bedrock aquifers in the area, and (2) more lakes are present to provide water to the aquifers in the area compared to the southern part of the model area. Groundwater-level declines in the glacial aquifer were lower with the 30-percent increase in groundwater withdrawals in areas where the Decorah-Platteville-Glenwood confining unit was present below the glacial aquifer (figs. 4, 17), limiting the effects of increased groundwater withdrawals from aquifers below the confining unit on groundwater levels in the glacial aquifer. In contrast, groundwater levels in the Prairie du Chien aquifer declined more than 3.0 ft near the city of Minneapolis because groundwater withdrawals were high (fig. 8), and the Decorah-Platteville-Glenwood confining unit is present through much of the area in and near Minneapolis, limiting the amount of groundwater exchange between glacial aquifer and the Prairie du Chien aquifer.

The simulation of the 30-percent decrease in groundwater withdrawals resulted in groundwater-level rises near high-capacity pumping wells in the glacial and Prairie du Chien aquifers in the same areas (near Cottage Grove, central and south parts of the model area, near Minneapolis) where the groundwater levels declined with the 30-percent increase simulation (figs. 17, 18). The magnitude of these groundwater-level rises were similar in magnitude to the declines simulated in the 30-percent increase scenario. In general, the largest groundwater-level rises in the glacial and Prairie du Chien aquifers in response to the 30-percent decrease in groundwater withdrawals occurred in areas of higher well density and greater withdrawals in the central and southern parts of the model area (figs. 8, 18), and approached zero in outlying areas having fewer wells and lower withdrawals, such as in the northern part of the NMLG model area. This response is to be expected because areas with more wells with higher groundwater withdrawals would have the greatest changes in water volumes and levels in the aquifers with the 30-percent changes in groundwater withdrawals. The simulation of the 30-percent decrease scenario resulted in a 10-ft increase in groundwater levels in the Prairie du Chien aquifer throughout much of southern Washington County (figs. 1, 18), with the maximum increase being 21.4 ft. Groundwater levels in the glacial aquifer rose as much as 18.4 ft near Cottage Grove and east of Woodbury (fig. 18) with the 30-percent decrease in groundwater withdrawals.

Increased and decreased groundwater-withdrawals scenarios that were simulated using the NMLG model affected simulated lake-water levels for Big Marine Lake, Snail Lake, and White Bear Lake, with the relative location of wells to lakes and the amount of groundwater withdrawn from the wells being important factors in the response of lake-water levels to changes in groundwater withdrawals. Simulated lake-water-level-declines ranged from 0.01 to 2.72 ft in the three lakes with a 30-percent increase from the mean 2003–13 groundwater withdrawals in the high-capacity wells (fig. 19; model number 8, table 14). The largest simulated lake-water-level decrease (2.72 ft) occurred in Snail Lake. Several high-capacity wells screened in glacial aquifers or open to the Jordan aquifer are less than 1 mi north and northwest of the Snail Lake (fig. 8). The Jordan aquifer is the shallowest bedrock aquifer near Snail Lake, and simulated increases in groundwater withdrawals from the aquifer near the lake (fig. 8) decreased simulated water levels in the lake and in the glacial aquifer near the lake (fig. 17). Only three high-capacity wells are near Big Marine Lake (fig. 8), which had the smallest change in simulated lake-water level (0.01 ft) with the 30-percent increase in groundwater withdrawals. Mean 2003–13 groundwater withdrawals for each of these three high-capacity wells near Big Marine Lake were less than 400 ft^3/d (fig. 8). Lake-water levels in White Bear Lake declined by 1.54 ft with the 30-percent increase in groundwater withdrawals (model number 8, table 14).

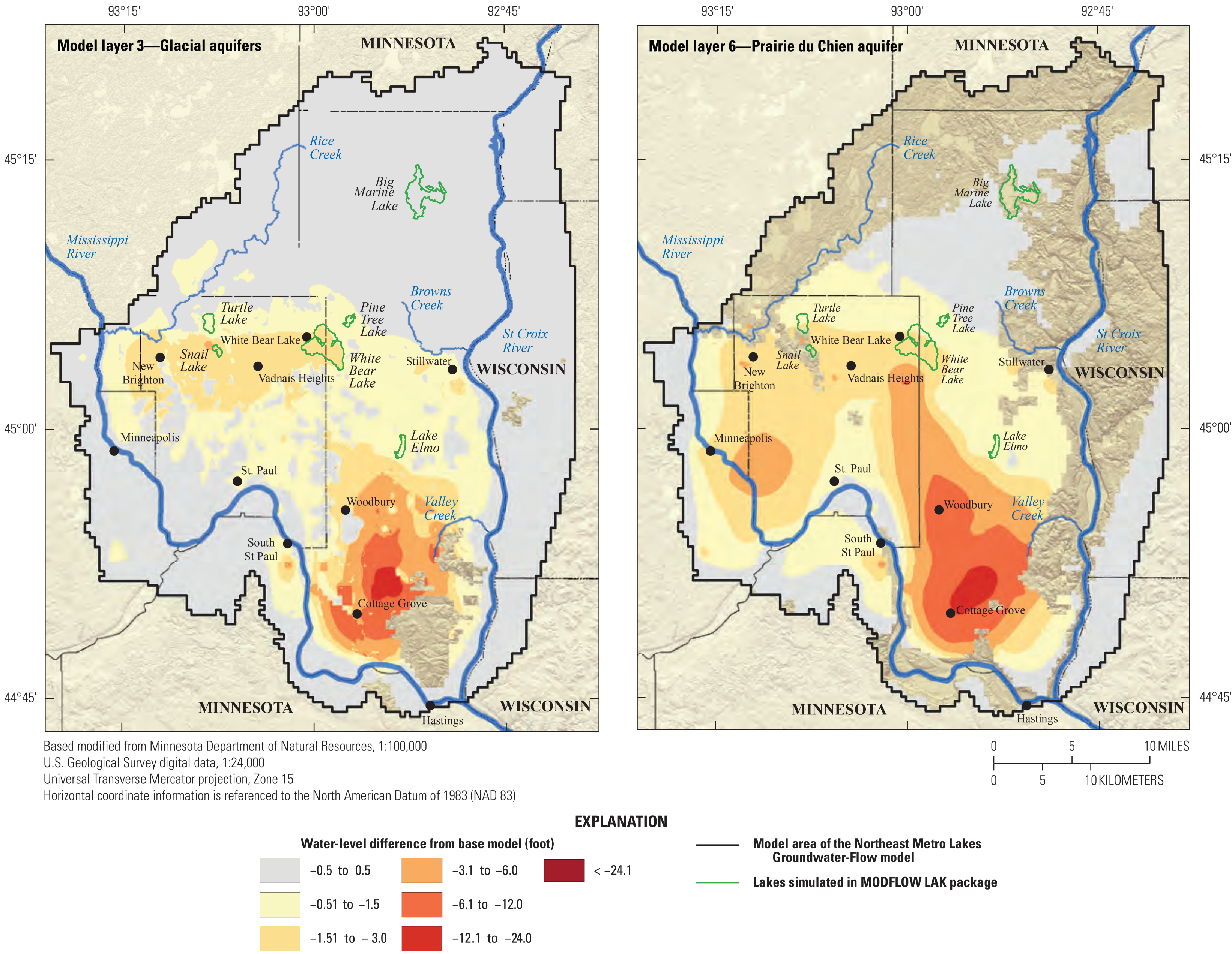


Figure 17. Simulated groundwater-level changes using a hypothetical scenario of a 30-percent increase in groundwater withdrawals from mean 2003–13 groundwater withdrawals for model layer 3, glacial aquifers, and model layer 6, Prairie du Chien aquifer in the Northeast Metro Lakes Groundwater-Flow model, in northeast Twin Cities Metropolitan Area, Minnesota. [MODFLOW, modular finite-difference groundwater-flow model; LAK, Lake]

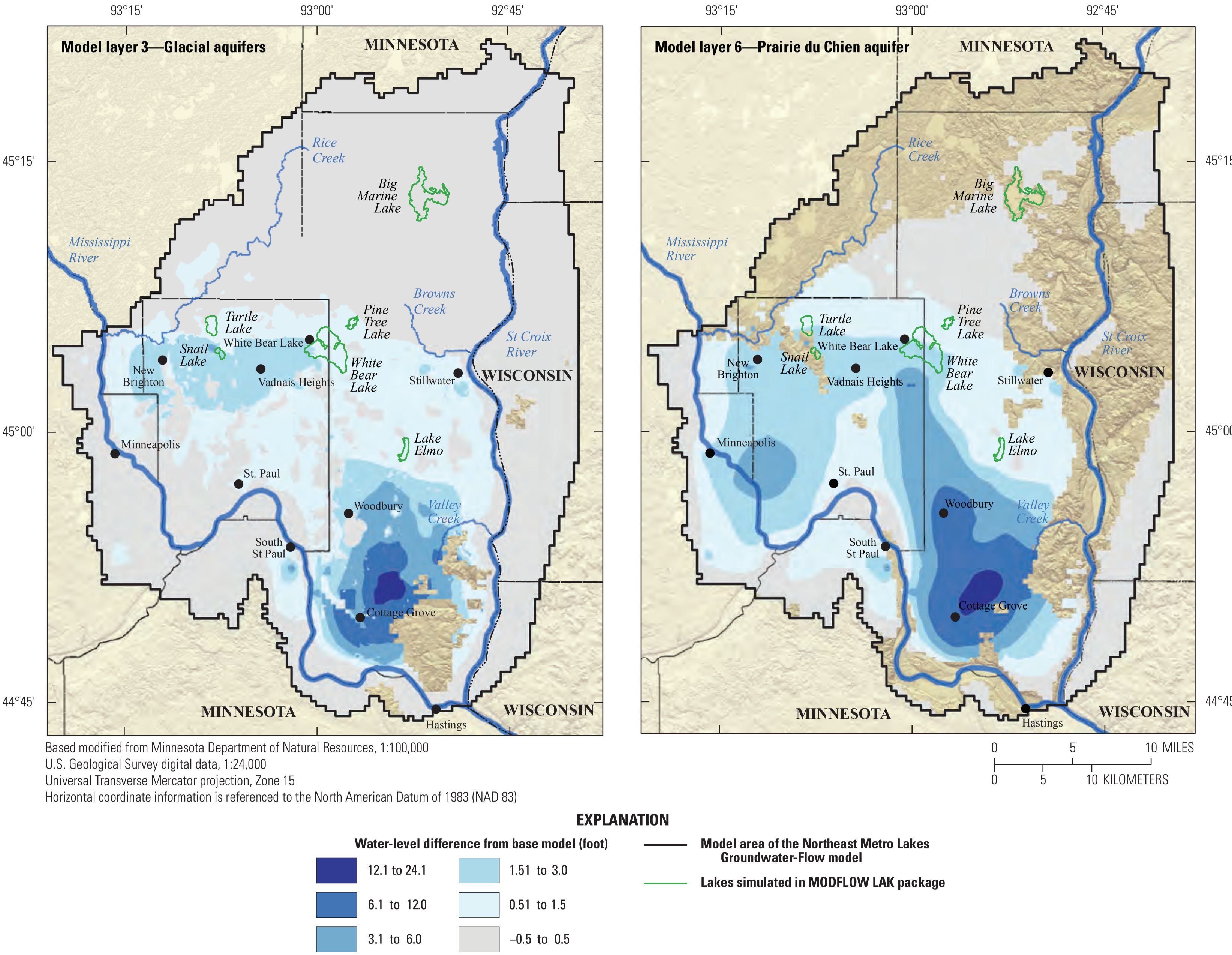


Figure 18. Simulated groundwater-level changes using a hypothetical scenario of a 30-percent decrease in groundwater withdrawals from mean 2003–13 groundwater withdrawals for model layer 3, glacial aquifers, and model layer 6, Prairie du Chien aquifer in the Northeast Metro Lakes Groundwater-Flow model, in northeast Twin Cities Metropolitan Area, Minnesota. [MODFLOW, modular finite-difference groundwater-flow model; LAK, Lake]

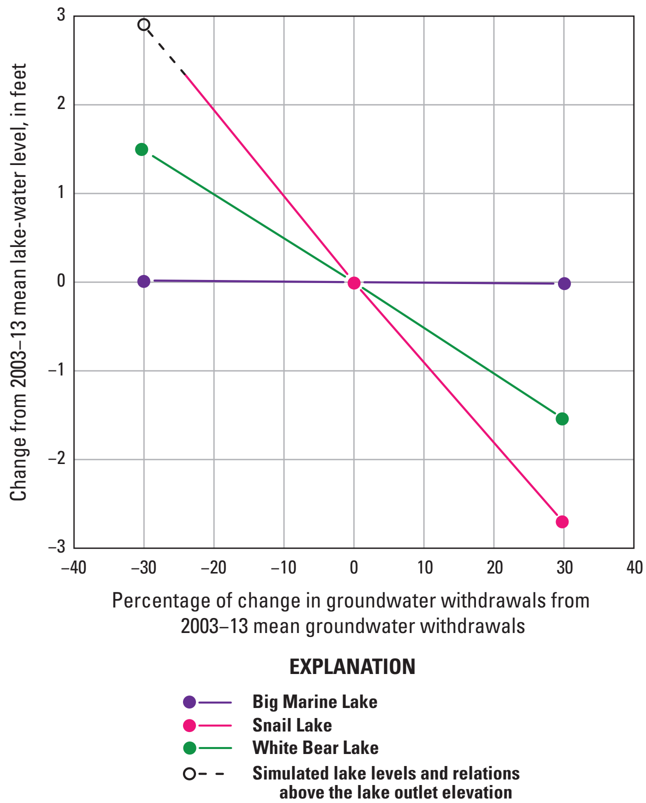


Figure 19. Simulated mean 2003–13 lake-water levels for Big Marine Lake, Snail Lake, and White Bear Lake under different hypothetical groundwater-withdrawal scenarios using the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

Water budgets for Big Marine Lake, Snail Lake, and White Bear Lake slightly differed between the 2003–13 mean simulation and hypothetical scenarios simulated using the 30-percent groundwater-withdrawal change (table 15). Total inflows and outflows for the lakes varied by less than 0.05 Mft^3/d from the inflows and outflows in the calibrated model (2003–13 mean simulation, table 15). These changes occurred in the amount of direct precipitation and groundwater inflow to the lake, and the evaporation and lake-water outflow to the groundwater system with the lake-water-level changes resulting from the increased groundwater withdrawals. The total water budget changes were the smallest in Big Marine Lake because the lake-water-level changes in the lake were small (0.01 ft, table 14). Differences in precipitation to and evaporation from White Bear Lake were the largest because the surface area of the lake is large (table 4) and the lake-water-level change was larger than the change in Big Marine Lake (table 14). Surface-water inflows to and outflows from the lakes did not vary because the surface-water inflows (runoff) and outflows for the lakes in the calibrated model were used in the groundwater-withdrawal simulations.

The simulated 30-percent decrease in mean 2003–13 groundwater withdrawals in the high-capacity wells resulted in higher lake-water levels of similar magnitudes as the lower lake-water levels that were simulated for the 30-percent increase in groundwater withdrawals (fig. 19). Simulated lake-water-level rises resulting from differences between mean 2003–13 groundwater withdrawals and the 30-percent groundwater withdrawal decreases ranged from 0.02 to 2.91 ft (fig. 19). The largest change in lake-water levels was simulated for Snail Lake, which likely is because of decreased groundwater withdrawals from three high-capacity wells open to the Jordan aquifer north of the lake (fig. 8). The smallest lake-water-level change was simulated for Big Marine Lake, which has only three nearby high-capacity wells (fig. 8).

Precipitation and Groundwater-Withdrawal Scenarios

The 5-percent changes in mean 2003–13 precipitation used in the calibrated NMLG model resulted in larger differences in the total water budget from the calibrated model compared to hypothetical scenarios simulated using 30-percent changes in groundwater withdrawals. The 5-percent increase and decrease in precipitation (models 6 and 4, respectively, in table 11) resulted in about 5-percent increases and decreases, respectively, in total inflow and outflow from the simulated aquifers. Results from the sensitivity analyses indicated that recharge was the most sensitive parameter in the NMLG model, and the larger differences in the total water budget for the model with 5-percent changes in precipitation further indicated the importance of recharge in the model. The total amount of areal groundwater recharge to the model for the 5-percent increase and decrease in precipitation from the mean 2003–13 conditions (model number 5, table 11) account for 44 and 35 percent, respectively, of the total inflow for the model simulations (model numbers 6 and 4, respectively, table 11). These percentages represent 3-percent increases and 6-percent decreases from the 41 percent of the total inflow in the 2003–13 mean steady-state simulation (model number 5, table 11). Inflows from constant-head cells and river cells (leakage) remained important sources of water to the simulated groundwater system in the precipitation and groundwater-withdrawal scenarios, accounting for 35 and 15 percent, respectively, of the total inflow in the 5-percent precipitation increase simulations (model number 6, table 11) and 41 and 19 percent, respectively, of the total inflow in the 5-percent precipitation decrease simulations (model number 4, table 11).

Differences in the water budgets between the 2003–13 mean groundwater withdrawals (model number 5, table 11) and the 5-percent precipitation change simulations (model numbers 6 and 4, table 11) reflect the changes in groundwater levels associated with the changes in areal groundwater recharge. The 5-percent increase and decrease in areal groundwater recharge result in higher and lower simulated groundwater levels, respectively. These higher and lower groundwater levels resulted in an 3.6 Mft^3/d increase and a 3.9 Mft^3/d decrease, respectively, in surface leakage to major rivers, streams, lakes, and wetlands (model numbers 6 and 4, respectively, tables 11, 12). Inflows from constant-head cells and river leakage decreased with a 5-percent precipitation increase

Table 15. Summary of simulated water budgets for Big Marine Lake, Snail Lake, and White Bear Lake for the 2003–13 mean and different hypothetical groundwater-withdrawal scenarios using the calibrated Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota.

[Values in parentheses represent the percent of total inflow or outflow. MNDNR, Minnesota Department of Natural Resources; ID, identification number; Mft^3/d, million cubic foot per day]

MNDNR lake ID	Lake name	Groundwater-withdrawal conditions (calibrated model and hypothetical scenarios)	Lake inflow (Mft^3/d)				Lake outflow (Mft^3/d)			
			Precipitation	Surface-water inflow (runoff)	Groundwater inflow	Total inflow	Evapotranspiration	Surface-water outflow	Outflow to groundwater system	Total outflow
62007300	Snail (62–73 P)	2003–13 mean groundwater withdrawals (calibrated model)	0.05 (42)	0.05 (42)	0.02 (17)	0.12	0.04 (36)	0.00 (0)	0.07 (64)	0.11
		30-percent increase in mean 2003–13 groundwater withdrawals	0.03 (30)	0.05 (50)	0.02 (20)	0.10	0.03 (30)	0.00 (0)	0.07 (70)	0.10
		30-percent decrease in mean 2003–13 groundwater withdrawals	[1]0.06 (38)	[1]0.05 (31)	[1]0.05 (31)	[1]0.16	[1]0.06 (37)	[1]0.00 (0)	[1]0.10 (63)	[1]0.16
82005200	Big Marine (82–52 P)	2003–13 mean groundwater withdrawals (calibrated model)	0.52 (56)	0.28 (30)	0.13 (14)	0.93	0.48 (52)	0.03 (3)	0.41 (45)	0.92
		30-percent increase in mean 2003–13 groundwater withdrawals	0.51 (56)	0.28 (31)	0.12 (13)	0.91	0.48 (52)	0.03 (3)	0.41 (45)	0.92
		30-percent decrease in mean 2003–13 groundwater withdrawals	0.52 (56)	0.28 (30)	0.13 (14)	0.93	0.48 (52)	0.03 (3)	0.41 (45)	0.92
82016700	White Bear (82–167 P)	2003–13 mean groundwater withdrawals (calibrated model)	0.72 (51)	0.34 (24)	0.35 (25)	1.41	0.62 (44)	0.01 (1)	0.77 (55)	1.4
		30-percent increase in mean 2003–13 groundwater withdrawals	0.68 (49)	0.34 (25)	0.36 (26)	1.38	0.59 (43)	0.01 (1)	0.77 (56)	1.37
		30-percent decrease in mean 2003–13 groundwater withdrawals	0.76 (53)	0.34 (24)	0.34 (23)	1.44	0.66 (46)	0.01 (1)	0.76 (53)	1.43

[1]Simulated lake-water budget likely is not accurate because the simulated lake-water elevation is higher than maximum baythymetric elevation specified in LAK packge.

(model number 6, table 11) because the water-level differences between the higher groundwater levels and constant-head cells and river cells where inflows occurred were less, or the hydraulic gradients were lower. Outflows to the constant-head and river cells where outflow occurred increase with the 5-percent precipitation increase because hydraulic gradients increased near these cells as groundwater levels rose. Opposite effects occurred with the 5-percent precipitation decrease simulation (model number 4, table 11); as groundwater levels declined with less precipitation, inflows from constant-head cells and river leakage increased where inflow occurred and outflows to the constant-head and river cells decreased where outflow occurred. Seepage from the lakes simulated with the LAK package in the groundwater system also increased with the 5-percent increase in precipitation (table 11) because the rise in lake-water levels was more than the rise in groundwater levels around the lakes in response to the additional precipitation, increasing the hydraulic gradient between the lakes and groundwater system where inflows occurred. Outflows from the simulated groundwater system to the lakes decreased because the hydraulic gradients between the lakes and the groundwater system were lower where these outflows occurred.

Differences in total water budgets between the 5-percent precipitation change scenarios and the combination of 5-percent precipitation change and 30-percent groundwater-withdrawal change scenarios indicated that changes in precipitation have a larger effect on the simulated model water budget than changes in groundwater withdrawals. Only small differences were present in the total water budgets between the 5-percent precipitation change simulations (model numbers 4 and 6, table 11) and the combination of 5-percent precipitation change and 30-percent groundwater-withdrawal change simulations (model numbers 1, 3, 7, and 9, table 11), indicating that the effects of the 30-percent changes in the groundwater withdrawals had less of an effect than the 5-percent precipitation changes. These water-budget differences mainly occurred in the same water-budget components (inflow from constant-head cells, inflow and outflow from river leakage, and outflow from surface leakage) that differed between the 30-percent groundwater-withdrawal changes (model numbers 2 and 8, table 11) and the calibrated model (model number 5, table 11).

Lake-water levels in Big Marine Lake, Snail Lake, and White Bear Lake simulated in the hypothetical scenarios varied in their responses to changes in groundwater withdrawals and precipitation (fig. 20; model numbers 1–4 and 6–9, table 14), and the lake-water levels generally affected more by the 5-percent precipitation changes than the 30-percent groundwater-withdrawal changes. Compared to the calibrated model simulation (model number 5), changes in groundwater recharge and surface-water runoff associated with a 5-percent precipitation decrease resulted in lake-water-level declines ranging from 2.51 to 3.01 ft for the three lakes (model number 4, table 14). Conversely, 30-percent increases from 2003–13 mean groundwater withdrawals (model number 8) resulted in lake-water-level declines ranging from 0.01 to 2.72 ft for the three lakes (table 14). With lower lake-water levels resulting from the 30-percent groundwater-withdrawal increases, the surface area of the lakes generally decreased, resulting in reductions in the amount of precipitation falling directly onto the three lakes and evaporation off of the lakes was reduced. This occurred to a larger extent in White Bear Lake (table 15). A similar response would occur with the lake-water-level declines associated with the 5-percent precipitation decline.

The largest lake-water-level declines in Big Marine Lake, Snail Lake, and White Bear Lake occurred with the 5-percent precipitation decrease and the 30-percent groundwater-withdrawal increase (fig. 20; model number 7, table 14). The largest lake-water-level change (–5.54 ft) from the 2003–13 mean conditions occurred in Snail Lake in the 5-percent precipitation decrease and 30-percent groundwater-withdrawal increase (fig. 20; model number 7, table 14), whereas the smallest lake-water-level change (–0.01 or 0.02 ft) occurred in Big Marine Lake when only the groundwater withdrawals were increased or decreased, respectively, by 30 percent (model numbers 8 and 2, respectively, table 14). The water levels in Big Marine Lake changed to a larger extent in the 5-percent precipitation decrease simulation (model number 4) than in the groundwater-withdrawal change simulations (model numbers 2 and 8), and the 5-percent decrease simulation resulted in a 2.62 ft decline in the water level of Big Marine Lake (table 14). Changes in groundwater withdrawals had little effect on the lake-water levels for Big Marine Lake because only three high-capacity wells withdraw groundwater from aquifers near the lake at relatively small volumes (fig. 8).

Lake-water-level changes simulated for Snail Lake using the hypothetical scenarios indicated that the 5-percent precipitation decrease had about the same effect as the 30-percent groundwater-withdrawal increase on the water level of Snail Lake (fig. 20). By decreasing the precipitation by 5 percent and increasing groundwater withdrawals by 30-percent (model number 7), the lake-water level in Snail Lake was an additional 2.83 ft lower than the water level for the lake in comparison to the simulated lake-water level for the 30-percent groundwater-withdrawal-increase scenarios (–2.71 ft, model number 8, table 14). The lake-water level in Snail Lake declined from the 2003–13 mean water level by 2.50 ft with the 5-percent precipitation decrease (model number 4, table 14).

Lake-water levels in White Bear Lake were lower with the sole 5-percent decrease in precipitation than the sole 30-percent increase in groundwater withdrawals. Decreasing precipitation by 5 percent from the mean 2003–13 mean precipitation lowered the water level in White Bear Lake by 3.01 ft (model number 4, table 14), whereas increasing groundwater withdrawals by 30 percent lowered the lake's water level by only about one-half as much (1.54 ft, model number 8, table 14); therefore, decreasing precipitation by 5 percent over the model area had a greater effect, almost double the effect, on White Bear Lake's water level than increasing the groundwater withdrawals by 30 percent. This is likely because more water was removed from the lake and

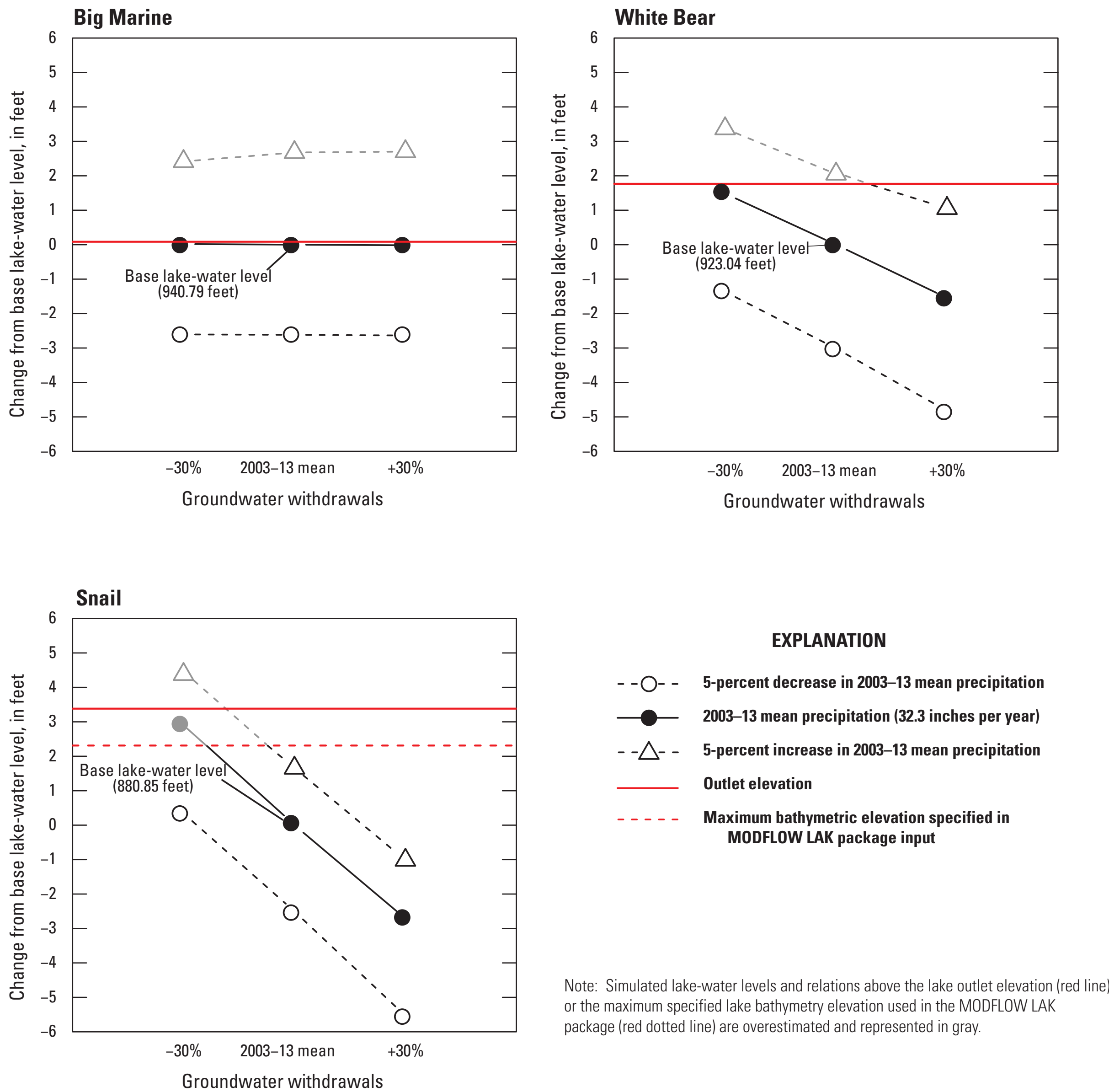


Figure 20. Simulated mean 2003–13 lake-water levels for Big Marine Lake, Snail Lake, and White Bear Lake under different hypothetical precipitation and groundwater-withdrawal scenarios using the Northeast Metro Lakes Groundwater-Flow model, northeast Twin Cities Metropolitan Area, Minnesota. [MODFLOW, modular finite-difference groundwater-flow model; LAK, Lake]

its watershed with the 5-percent precipitation decrease than by the 30-percent increase in groundwater withdrawals.

Simulation results from the hypothetical scenarios indicated that the 5-percent precipitation changes had a larger effect on total model water budgets and lake-water levels for Big Marine Lake and White Bear Lake than the 30-percent changes in groundwater-withdrawal rates, and the largest changes occurred when precipitation was decreased and groundwater withdrawals were increased; for example, the mean decline in lake-water levels for the three lakes (Big Marine Lake, Snail Lake, and White Bear Lake) in the sole 5-percent precipitation decrease simulation (model number 4) was 2.71 ft, whereas the mean lake-water-level decline in the sole 30-percent groundwater-withdrawal-increase simulation (model number 8) was 1.42 ft. Snail Lake was the only lake of the three lakes with a slightly larger lake-water-level decline (2.71 ft) simulated for the 30-percent increase in groundwater withdrawals than simulated for the 5-percent precipitation decrease (2.50 ft). Precipitation likely had a larger effect because (1) the 5-percent precipitation changes were effective over the model area whereas the effects of groundwater withdrawal varied with the distribution of high-capacity wells and their groundwater-withdrawal rates, and (2) the volume of water changed in the total water budget with the 5-percent precipitation changes was larger than the volume of water changed with the 30-percent change in groundwater withdrawals. The combined effect of decreasing precipitation in conjunction with increasing groundwater-withdrawal rates in a single simulation (model number 7) resulted in the largest lake-water-level declines from the mean 2003–13 lake-water levels (table 14). For Snail Lake and White Bear Lake, these declines (5.54 and 4.82 ft, respectively, model number 7 in table 14) were larger than the additive lake-water-level declines from simulations in which the 5-percent precipitation decrease (model number 4) and 30-percent groundwater-withdrawal increase (model number 8) were varied independently (5.21 and 4.55 ft, respectively, summation of lake-water-level changes in model numbers 4 and 8).

Model Accuracy and Limitations

The NMLG model was constructed and calibrated as a regional model for the northeast Twin Cities Metropolitan Area. The agreement between simulated and measured lake-water levels, groundwater levels, and streamflows varied across the model area with the spatial density and quality of the measured lake-water-level, groundwater-level, streamflow, and geologic data. Simulated results may be less representative of actual groundwater-flow conditions and groundwater and surface-water exchanges in local areas where little hydrologic or geologic data exist or at scales where the discretization of the model is insufficient to accurately represent phenomena or processes important to such exchanges; for example, the accuracy of recharge and runoff estimates determined by SWB model was limited because base flows used to calibrate the model were only from one streamgage (streamgage 05288580; table 2, fig. 6). Users should recognize the limitations of using this simulation for analyses of localized areas within the model area, particularly areas where hydrologic data were not available for model calibration. Areas east of Oakdale and Woodbury (fig. 1) had few observation wells (fig. 10), and therefore simulated results in these areas may be less representative of actual groundwater-flow conditions. These areas may have hydrologic features or processes that are locally important in the movement of groundwater but are not simulated in the NMLG model.

The NMLG model is a simplification of a complex glaciated and bedrock terrain and groundwater-flow system, representing mean hydrologic conditions during 2003–13. Glacial sediments in the NMLG model area were deposited through a series of glacial advances and retreats by different ice lobes and postglacial reworking of glacial tills, sands, and gravels. These glacial advances and retreats produced a complex horizontal and vertical distribution of particle sizes and heterogeneous compositions, resulting in a complex distribution of hydraulic properties that control groundwater movement in glacial aquifers. The representation of complex groundwater flow in the karst features of the bedrock units, such as the Shakopee Formation of the Prairie du Chien Group, cannot be simulated or verified with this model, which is built on the assumption of groundwater flow through homogeneous, porous media. Groundwater flows through karst features in the Prairie du Chien aquifer are known to occur in Hennepin, Ramsey, and Washington Counties (Tipping and others, 2015; Yingling, 2015). Model results in the karstic aquifers may be inaccurate because available maps of the karst features, and water-level and geologic data, are not detailed enough to distinguish differences in groundwater flow in the karstic and nonkarstic parts of the aquifers; therefore, the model should not be used to simulate transport of water or assess groundwater ages through the karstic bedrock units, such as the Shakopee Formation of the Prairie du Chien Group, and areas with more complex groundwater-flow conditions in glacial sediments.

Model simulations are limited to the accuracy, amount, distribution, and implementation of the data used to describe the hydrologic characteristics of the groundwater-flow system. These characteristics include the hydraulic properties of the aquifers and confining units, areal recharge rates, and hydrologic boundary conditions. Parameter values determined for the model during calibration are not unique, and different combinations of model parameters could produce similar results. Parameter uncertainty analysis was not done with the NMLG model, but uncertainty needs to be considered when applying the model to various hydrologic scenarios. Parameter uncertainty can be quantified by comparing calibrated parameter values to prior information, such as hydrologic parameter values determined from aquifer testing and previous groundwater-flow model simulations. Calibrated hydraulic conductivities for aquifers and confining units in the NMLG model were compared to values in the MM3 model (Metropolitan

Council, 2016) to generally assess the uncertainty of hydraulic conductivity values in the model, and a range in hydraulic conductivities determined in other hydrogeologic and geologic studies (Runkel and others, 2003; table 1) were used in PEST during the model calibration to limit uncertainty in the hydraulic conductivity values.

The calibrated model was produced using automated parameter estimation techniques using a comparison between simulated and measured lake-water levels, groundwater levels, and streamflows at multiple locations throughout the study area. Additional hydrologic features, data, or processes should not be incorporated into the model without considering if the model should be recalibrated; for example, evapotranspiration from the groundwater system was not simulated in the model, which can be an important hydrologic process near lakes and affect groundwater and lake-water levels. If evapotranspiration from the shallow aquifers was incorporated into the model, the model would need to be recalibrated.

The use of constant-head cells to represent hydrologic conditions along the lateral boundaries of the model allows for unlimited flow in the groundwater-flow system. This source of water can result in higher groundwater, river, stream, and lake-water levels, particularly near the boundaries. When these constant-head conditions were temporarily replaced with no-flow boundary conditions, the boundary conditions did not affect simulated lake-water levels for the six lakes in the LAK package in the NMLG model; only conditions near the model boundary were affected, and therefore, model results along the lateral boundaries should not be used for interpretation of lake-water level and other hydrologic changes.

The dimensions of each model cell (410 by 410 ft [125 by 125 m]) were assumed to appropriately simulate the hydrologic scales of sources and sinks represented in the model. When using MODFLOW, aquifer properties and other model parameters must be uniformly assigned to each model cell (Harbaugh, 2005); therefore, the assumption is that these properties and parameters represent the entire volume of the model cell. Similarly, surface-water features represented with the RIV package were assumed to occupy the entire model cell, even if the actual dimensions of the stream were much smaller (or larger) than the cell dimensions. Interaction of a lake with groundwater flow is simulated through calculations that represent the center of the model cell. The model should be applied to simulations of hydrologic features and processes that can be accurately represented in the model area at the scale of the model cells.

Steady-State Conditions

The NMLG model documented in this report and available from Trost and others (2017) was calibrated to mean 2003–13 groundwater levels, lake-water levels, and streamflows assuming steady-state condition during the period based on a conceptual model outlined in the “Conceptual Groundwater-Flow Model” section. Model results should not be used to assess transient changes in measured groundwater withdrawals, precipitation, lake-water levels, or other hydrologic variables because specific yield and storage, which are model parameters important to the simulation of these phenomena, were not incorporated into the model. The hypothetical model simulations cannot be used to address future or measured transient changes in lake-water levels but can be used to evaluate whether changes in mean groundwater withdrawals or mean precipitation would result in changes in lake-water levels during steady-state conditions for the three lakes (Big Marine Lake, Snail Lake, and White Bear Lake) during 2003–13.

The application of steady-state simulations assumes that groundwater levels, lake-water levels, and streamflows generally were in equilibrium with mean hydrologic conditions during the period. This is not true for several lakes in the model area where water levels were declining during the period (Jones and others, 2013); for example, lake-water levels in White Bear Lake declined more than 5 ft during 2003–11 (Jones and others, 2013). It is not known if the 2003–13 steady-state simulation would properly reproduce hydrologic responses for hydrologic conditions beyond those that occurred during 2003–13. If precipitation conditions or groundwater withdrawals were substantially changed from the mean during the 2003–13 period, the model would likely need to be updated and recalibrated. Because the model was calibrated under the assumption of steady-state, groundwater-flow conditions, water storages were not calibrated in the model, and therefore any scenarios influenced by short-term, transient changes in water storage may be invalid. The model will most accurately reflect the effects of long-term (multiple-year) and not short-term stresses.

The calibrated, steady-state model is an appropriate tool for water-resources management decisions assuming future hydrologic conditions and groundwater withdrawals will be similar to the mean, steady-state 2003–13 conditions. Steady-state simulations may not accurately simulate conditions if future mean recharge or discharge greatly exceeds the values used in the calibrated model. The numerical model was calibrated under the assumption of steady-state groundwater-flow conditions; therefore, the model will most accurately reflect the effects of multiple-year stresses and not short-term transient conditions.

Recharge and Runoff

Several assumptions must be considered associated with using base flow, recharge, and runoff estimates determined from streamflow data from the Mississippi River, Rice Creek, Brown’s Creek, and Valley Creek to calibrate the NMLG model when evaluating additional applications of the model. These assumptions are (1) mean 2003–13 low streamflows represented base-flow conditions during the period, (2) the large drainage area and control structures and sewage discharge upstream from the Mississippi River streamgages did not affect the base-flow estimates determined from the HYSEP program and used in the calibration of the NMLG model, and (3) differences in the boundaries of the surface-watershed and groundwater contributing area for Brown's Creek and Valley Creek were not major factors in determining recharge and runoff estimates. Mean streamflows during low-flow periods were assumed to represent base-flow conditions as estimated using the SWB model (Westenbroek and others, 2010) and HYSEP software (Sloto and Crouse, 1996) to separate runoff from base flows (see the "Soil-Water-Balance Model" section). If measured low flows were greater than base flows, the model may represent conditions slightly higher than base flow. If the streamflows were representative of base-flow conditions, then the streamflow measurements approximate the groundwater discharge to and from the streams. The model should be used with simulations of base-flow conditions that are similar to low-flow periods during 2003–13.

The streamflows at and drainage areas upstream from the Mississippi River streamgages (table 5) were larger than recommended for use with stream-hydrograph-separation methods. Applications of stream-hydrograph-separation methods to determine recharge and runoff estimates are not recommended for drainage areas greater than 500 mi^2 (1,300 square kilometers [km^2]) because separation of surface-water flow, groundwater flow, and bank-storage effects is difficult (Rutledge, 1998; Scanlon and others, 2002). As a result, unrealistically high or low base flows may be determined from the application of stream-hydrograph-separation method and used to estimate runoff coefficients in the model. Also, the natural streamflow measured at the Mississippi River streamgages were affected by control structures, which can either result in high or low recharge and runoff estimates determined by stream-hydrograph-separation methods than would be expected under natural streamflow conditions. Also, discharge from the Metropolitan Wastewater Treatment Plant near Pigs Eye Lake (fig. 1) to the Mississippi River upstream from the streamgage at Prescott, Wisconsin (streamgage 05344500, fig. 6), likely resulted in higher estimates of recharge from the application of stream-hydrograph-separation methods. The application of stream-hydrograph-separation methods to large watersheds and regulated streamflows is cautioned in supporting documents for HYSEP (Sloto and Crouse, 1996) and in assessments of stream-hydrograph-separation methods (Rutledge, 1998; Scanlon and others, 2002).

Application of stream-hydrograph-separation methods are not recommended for hydrographs for streamgages where the contributing groundwater area does not coincide with the contributing surface-water watershed (Neff and others, 2005). The surface-water watersheds for the Brown's Creek and Valley Creek streamgages are smaller than their corresponding contributing groundwater watersheds (Almendinger, 2014; Brown's Creek Watershed District, 2012) because changes in the natural routing of surface water and differences in natural water-flow conditions exist. When the contributing groundwater area does not coincide with the surface-water watershed, the stream base flow may not equal the sum of the recharge of the contributing surface-water watershed minus evapotranspiration, which is an assumption of stream-hydrograph-separation methods (Neff and others, 2005). This issue can be minimized by determining mean recharge estimates from long-term streamflow records, which was done with the 11-year records for the Brown's Creek and Valley Creek streamgages used in this study (table 2).

The approach of using recharge values determined by the SWB model to represent recharge to the UZF package oversimplifies many aspects of flow dynamics between surface water and groundwater. Recharge produced with the SWB model does not account for shallow groundwater tables where they may be present, making it infeasible for recharge in these areas; therefore, the SWB model may overpredict recharge in these areas. The SWB model does not account for depressional storage in downgradient areas receiving runoff. In many cases, these depressions may store water that may infiltrate or evaporate during periods after precipitation. The SWB model water balance, however, is constrained to a single day so all water that accumulates in closed depressions from runoff routing will infiltrate, evaporate, or be rejected after one time step. Also, in many cases, these depressions will ultimately spill over during periods of excessive precipitation. These depressions, often with wetlands and small lakes, are common throughout the northeast Twin Cities Metropolitan Area; therefore, SWB-predicted recharge rates in these areas are highly uncertain and may be unreasonable. Also, the applied SWB calibration and simulation approach assumes spatial co-alignment of the contributing groundwater area and surface watershed, which could affect calibrated parameters of the SWB model and associated runoff and recharge values supplied to the NMLG model.

Groundwater Withdrawals, Rivers, and Lakes

The effects of groundwater withdrawals from domestic and other unpermitted wells were not simulated in the NMLG model because no known records exist regarding the amounts of groundwater withdrawn from individual domestic wells during 2003–13 in the model area (see the "Groundwater Withdrawals from Wells" section). Domestic wells generally are in nondeveloped lands in the northeast, east, and southeast parts of the model area (fig. 2), and do not withdraw as much groundwater as the high-capacity public supply wells. Also, much of the groundwater withdrawn by domestic wells from aquifers is returned to the aquifer through sanitary drain fields. The model may overpredict groundwater levels in areas where a large number of domestic wells are present because the model does not account for the domestic groundwater withdrawals.

The use of the RIV package to represent surface-water features is a simplistic approach to simulating interactions between surface water and the groundwater system, which does not route water through the stream network. Because of a lack of streamflow routing, river cells can recharge the groundwater system proportional to their stage, conductance, and the groundwater level of the model cell but without consideration for whether upstream river cells could actually supply this recharged water. In general, this approach has several implications for analysis of features simulated using river cells and model results. Streams and rivers were conceptually viewed as locations of groundwater discharge within the study area. By representing these features with the RIV package, groundwater discharged to a model cell containing a river feature is not included in subsequent calculations in downstream cells. In the case of streams, this may become important in the event that an upstream reach is in contact with the groundwater and a lower reach is not, thereby reducing stream leakage to the groundwater system in the lower reach.

Using the RIV package to represent closed, nonflowing surface-water bodies, such as lakes, also has limitations because these RIV cells can supply unlimited water to the groundwater model. Care was taken to ensure that leakage into the groundwater system from these cells was reasonable and adhered to the conceptual model of their function in the hydrologic system. It is recognized that these fluxes and the conductances governing them are highly uncertain; furthermore, the user must recognize that specified river stages are not affected by the changes in the stresses when using this model as a tool to demonstrate an effect of a change in stress on another feature within the model. That is, river stages for lakes were not adjusted for any of the hypothetical scenarios. If a user wishes to demonstrate the effects of increased groundwater withdrawals from high-capacity wells not described in this report on lake-water levels, they must recognize that stages in lakes represented with river cells will not be affected and therefore these river cells may introduce more water to the model as groundwater withdrawals increase and thus reduce the drawdown in lakes simulated in the LAK package. The effects of groundwater withdrawals on lake-water levels shown in figures 18 and 19 of this report may be underestimated because of the unlimited water available from the river cells.

Surface-water outflow was not determined in the hypothetical scenarios using the NMLG model for the lakes simulated with the LAK package. This outflow can be a major or minor component of the lake-water budget, depending upon changes to other components of the budget; therefore, lake-water levels and budgets for simulations where the lake-water levels are above the surface-water outlets likely are in error and should not be used or considered in any lake-water level or budget assessment.

Implications

Results from Jones and others (2016) and the NMLG model simulations indicate that (1) lake-water outflow to shallow and deep aquifers can be an important component in water budgets for northeast Twin Cities Metropolitan lakes and aquifers; (2) these exchanges can leave lakes vulnerable to groundwater withdrawals and other hydrologic factors, such as lower precipitation; (3) the response of lake-water levels to groundwater withdrawals varies with northeast Twin Cities Metropolitan Area lakes, depending on well, aquifer, and lake characteristics; and (4) the effects of groundwater withdrawals on lake-water levels are exacerbated under drier conditions for lakes affected by groundwater withdrawals. The lake sediments generally are permeable along the shores of northeast Twin Cities Metropolitan Area lakes, allowing the lakes to exist in part because of groundwater and surface-waters exchanges. These exchanges can leave lakes vulnerable to groundwater withdrawals and other variables that affect groundwater levels in shallow glacial aquifers that are important in these exchanges. Lake-water quality also has been determined to be affected by groundwater and surface-water exchanges where contaminated groundwater flows into northeast Twin Cities Metropolitan Area lakes (for example, Minnesota Pollution Control Agency, 2008). Model results indicating that lakes are providing water to underlying aquifers in the northeast Twin Cities Metropolitan Area supports the findings of Jones and others (2016) that water from White Bear Lake (fig. 1) and potentially other northeast Twin Cities Metropolitan Area lakes can reach underlying glacial and bedrock aquifers, recharging these important regional aquifer systems. Northeast Twin Cities Metropolitan Area lakes need to be considered as water sources to these aquifers and any wells withdrawing water from these aquifers.

The effect of groundwater withdrawals and precipitation on water levels in many northeast Twin Cities Metropolitan Area lakes varies because of natural and altered conditions of the lakes (Jones and others, 2016). Closed-basin lakes, such as White Bear Lake and Snail Lake (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 (Jones and others, 2016). Because they have limited sources of water, these closed-basin lakes can be more vulnerable to changes in groundwater withdrawals and precipitation than flow-through lakes where high-capacity wells are near the lakes. Their vulnerability to groundwater withdrawals depends on the permeability of lake sediments and glacial sediments near the lake, the number of high capacity wells near the lake, the withdrawal rates of nearby high-capacity wells, and the depths of wells relative to the lake depth. Many northeast Twin Cities Metropolitan Area lakes have been altered to be flow-through lakes, receiving surface water through inflow channels connected to upgradient lakes and discharging surface-water through channels to downgradient lakes and rivers (Jones and others, 2016).

Lake-water-level changes assessed in the NMLG hypothetical scenarios to changes in groundwater withdrawals are not representative of expected water-level responses in all lakes across the northeast Twin Cities Metropolitan Area. Simulated water level responses in Snail Lake and White Bear Lake to groundwater-withdrawal changes likely represent worst-case scenarios because both lakes generally were closed-basin lakes during the study period (2003–13) and were near high-capacity wells. Many of the northeast Twin Cities Metropolitan Area lakes are flow-through lakes, supported by surface-water and groundwater flow (Jones and others, 2016). Any effects of groundwater withdrawals on the water levels of these flow-through lakes likely would be dampened by the contribution of surface-water flow to the lakes. Precipitation may have a larger effect on lake-water levels for these lakes, particularly areas where groundwater withdrawals are relatively small. Also, many of the closed-basin lakes in the northeastern part of the model area, such as Big Marine Lake, are in areas where groundwater withdrawals are low. A similar small water-level response to groundwater withdrawals and greater response to precipitation changes would be expected for these closed-basin lakes in the northeastern part of the model area as was simulated for Big Marine Lake; however, groundwater withdrawals may increase as projected water demands increase in the north and eastern part of the study area (Metropolitan Council, 2015), potentially having a greater effect on lake-water levels in these lakes. The large groundwater withdrawals (fig. 8) and simulated groundwater-level declines in the southeast part of the study area (fig. 17) could indicate that any closed-basin lakes in that area could be more vulnerable to groundwater withdrawals.

Results from the NMLG model indicated that lakes overlying buried bedrock valleys may be more vulnerable to groundwater withdrawals in deeper bedrock aquifers. Many northeast Twin Cities Metropolitan Area lakes overlie deep buried bedrock valleys that are filled with permeable glacial sediments and glacial tills (Bauer, 2016; Winter and Pfannkuch, 1976). The shallowest bedrock units in these valleys commonly are the Prairie du Chien Group and Jordan Sandstone, which compose aquifers simulated in the model. Groundwater levels simulated with the NMLG model indicated that groundwater withdrawals in these two bedrock aquifers can result in lower groundwater levels in glacial aquifers and lower water levels in lakes (Snail Lake and White Bear Lake) above the bedrock aquifers in buried valleys.

The NMLG model developed in this study simulates the hydrogeology on the scale of the northeast Twin Cities Metropolitan Area for 2003–13. This NMLG model is a tool that allows assessment of the hydrologic effects of precipitation, groundwater withdrawals, and other hydrologic variables on the magnitude of exchanges between groundwater and lakes. Regional groundwater-flow models, such as the NMLG model, can provide a basis for assessing the general effects of groundwater withdrawals on lake-water levels over a study area; however, because of the high complexity of surface-water flow conditions in many flow-through lakes, more detailed, coupled surface-water and groundwater flow models might be needed to assess water-level changes in these types of lakes. Many hydrologic and geologic variables, in addition to groundwater withdrawals and precipitation, should be considered when assessing the magnitude of potential lake-water-level changes resulting from increases in groundwater withdrawals with expanded urban development. These variables include changes in the number, location, and groundwater-withdrawal rates of high-capacity wells, and changes in recharge rates associated with land-use changes with urban development.

Evaluating specific water-management scenarios (for example, conservation and augmentation) was beyond the scope of this study; however, the NMLG model is a substantial advancement in providing a mechanistically based tool for resource managers to assess different water-management scenarios. As water-resource management grows in importance with increasing pressures from urban development, future enhancements and refinements of this tool may be desirable. The NMLG model would benefit from more complete assessments of the hydrogeology in which the lakes and aquifers are set; furthermore, this model was a steady-state model, with each model run using one set of model parameters. Development of a transient model would allow more detailed assessment of lake and aquifer levels in relation to temporally dynamic (changing) model parameters.

Summary

Water levels during 2003 through 2013 were less than mean water levels for the period 1925–2013 for several lakes in the northeast Twin Cities Metropolitan Area in Minnesota. Since 2013, water levels in some northeast Twin Cities Metropolitan Area lakes have recovered to near their ordinary water levels, whereas other lakes have not. Previous periods of low water levels in lakes in the study area generally correlate with periods of below-normal precipitation. Increases in groundwater withdrawals and land-use changes have put into question whether water-level declines in lakes solely are due to declines in precipitation. A previous U.S. Geological Survey (USGS) study indicated that a combination of decreased precipitation and increased groundwater withdrawals could explain the lake-water-level changes of White Bear Lake; however, this lake-water-level analysis did not account for the spatial effects of groundwater withdrawals on lake-water levels. A thorough understanding of groundwater and surface-water exchanges was needed to assess the effect of water-management decisions on lake-water levels. To address this need, the USGS, in cooperation with the Metropolitan Council and Minnesota Department of Health, developed a three-dimensional, steady-state groundwater-flow model, called the “Northeast Metro Lakes Groundwater-Flow (NMLG) model,” to assess effects of groundwater withdrawals and precipitation on lake-water levels and groundwater and surface-water exchanges.

The NMLG model was calibrated by comparing simulated to measured 2003–13 mean groundwater levels, lake-water levels, and base flows. The USGS modular finite-difference groundwater-flow model (MODFLOW), Newton formulation (MODFLOW–NWT, version 1.0.8), was used to simulate mean 2003–13 steady-state, groundwater-flow conditions in an about 1,000-square-mile area of the northeast Twin Cities Metropolitan Area and parts of western Wisconsin. The model overall adequately reproduced measured 2003–13 mean data as indicated by the good model fit to measured data and the relative even spatial distribution of error for groundwater levels, lake-water levels, and groundwater discharge to streams and rivers (base flow). Six lakes (Big Marine Lake, Lake Elmo, Pine Tree Lake, Snail Lake, Turtle Lake, and White Bear Lake) were represented in the model using the MODFLOW Lake package for detailed analyses of groundwater and surface-water exchanges. Simulated lake-water levels were within acceptable limits for four of the six lakes simulated with the MODFLOW Lake package. Lake-water budgets for the calibrated model indicated that lakes in the northeast Twin Cities Metropolitan Area are providing water to underlying aquifers. Lake-water outflow to the simulated groundwater system was a major outflow component for Big Marine Lake, Lake Elmo, Snail Lake, and White Bear Lake, accounting for 45 to 64 percent of the total outflows from the lakes.

The calibrated model was used to simulate eight hypothetical scenarios to assess the effects of groundwater withdrawals and precipitation on water budgets and levels in Big Marine Lake, Snail Lake, and White Bear Lake. These three lakes were chosen from other lakes in the model area based on the number of water-level measurements made in the lakes during 2003–13, their relatively large size and depth, and their relative distance (more than 3 miles) from model boundaries. Simulated water budgets and lake-water levels for two of the six lakes simulated with the MODFLOW Lake package (Turtle Lake and Pine Tree Lake) were not included in the analysis of the eight hypothetical scenarios because their simulated mean 2003–13 lake-water levels did not fall between maximum and minimum measured lake-water levels recorded in the lakes during 2003–13. Groundwater withdrawals and precipitation in the hypothetical scenarios were increased and decreased by 30 and 5 percent, respectively, from the 2003–13 mean values used in the calibrated model. The groundwater-withdrawal changes were done in the 838 high-capacity wells in the calibrated model; no additional wells were added to the model. Increases and decreases in groundwater withdrawals by 30 percent from the 2003–13 mean groundwater withdrawals changed the total amount of water withdrawn from the groundwater system by about 3.5 percent of the total outflow from the calibrated model. Changes in groundwater withdrawals mostly resulted in changes in surface leakage to the land surface as groundwater levels changed. In comparison, increases and decreases in the 2003–13 mean precipitation by 5 percent changed the total amount of recharge simulated in the model by about 4.7 percent of the total outflow from the calibrated model. These hypothetical changes in groundwater withdrawals and precipitation were simulated separately or in combination. Simulated lake-water budgets and levels were compared between the eight hypothetical scenarios and the calibrated model to assess the effects of groundwater withdrawals and precipitation conditions on lake-water levels.

Lake-water levels and water budgets simulated for the hypothetical scenarios indicated that lake-water levels for Snail Lake and White Bear Lake are affected by changes in precipitation and groundwater withdrawals in the area, whereas Big Marine Lake is mainly affected by precipitation changes. The effects of groundwater withdrawals on the lakes depend on the number of high-capacity wells and amount of withdrawals from wells near the lakes. The largest lake-water-level declines in Big Marine Lake (2.64 feet), Snail Lake (5.54 feet), and White Bear Lake (4.82 feet) from the 2003–13 mean water-levels occurred when precipitation was decreased and groundwater withdrawals were increased. The sole 5-percent precipitation decrease had a larger effect on total model water budgets and lake-water levels for Big Marine Lake and White Bear Lake than the sole 30-percent increase in groundwater-withdrawal rates. The 5-percent precipitation decrease resulted in lower water levels in Big Marine Lake from the 2003–13 mean water-level by 2.62 feet, whereas increasing groundwater withdrawals by 30 percent only resulted in lower water levels in the lake by 0.01 foot. The water level in White Bear Lake decreased by about twice as much (3.01 feet from the 2003–13 mean water level) with a 5-percent precipitation decrease than with only a 30-percent groundwater-withdrawal increase (1.54-foot). The 5-percent precipitation decrease had about the same effect as the 30-percent groundwater-withdrawal increase on the water level of Snail Lake.

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Back cover. Bald Eagle Lake, September 1, 2011 (photograph by Perry M. Jones, U.S. Geological Survey).

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