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Groundwater and Surface-Water Interaction and Effects of Pumping in a Complex Glacial-Sediment Aquifer, Phase 2, East-Central Massachusetts

Scientific Investigations Report 2015–5174

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

Cover. Pod Meadow Pond, Framingham, Massachusetts. Ice-free area of open water in the foreground is caused by warmer groundwater discharging to the pond. Photograph by Jack Eggleston, U.S. Geological Survey, February 2011.

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SALLY JEWELL, Secretary

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

U.S. Geological Survey, Reston, Virginia: 2015

Suggested citation:
Eggleston, J.R., Zarriello, P.J., and Carlson, C.S., 2015, Groundwater and surface-water interaction and effects of pumping in a complex glacial-sediment aquifer, Phase 2, east-central Massachusetts: U.S. Geological Survey Scientific Investigations Report 2015–5174, 38 p., http://dx.doi.org/10.3133/sir20155174.

ISSN 2328-0328 (online)

Acknowledgments

This study was supported by the Town of Framingham, Massachusetts. The authors extend thanks to Peter Newton (Bristol Engineering Advisors, Inc.), Eric Johnson (Town of Framingham, Mass.), and Diane Stokes (City of Cambridge, Mass.), for sharing their knowledge of the study area and collecting and providing field information used in the study.

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Conversion Factors

[Inch/Pound to International System of Units]

Multiply	By	To obtain
	Length	
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
inch (in.)	2.54	centimeter (cm)
	Area	
square mile (mi^2)	2.590	square kilometer (km^2)
	Flow rate	
cubic foot per day (ft^3/d)	0.00001157	cubic foot per second (ft^3/sec)
cubic foot per second (ft^3/s)	0.6463	million gallons per day (Mgal/d)
million gallons per day (Mgal/d)	1.547	cubic foot per second (ft^3/s)
foot per day (ft/d)	0.032854	inch per year (in/yr)

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

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

‘Elevation’, as used in this report, refers to distance above the vertical datum (NAVD 88).

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

Abbreviations

EIR	Environmental Impact Report
EPA	U.S. Environmental Protection Agency
GPR	ground penetrating radar
GPS	global positioning system
HSPF	Hydrological Simulation Program—Fortran
Kh	horizontal hydraulic conductivity (feet per day)
Ks	streambed hydraulic conductivity (feet per day)
LAK	MODFLOW Lake package
MassDEP	Massachusetts Department of Environmental Protection
MassDCR	Massachusetts Department of Conservation and Recreation
MEPA	Massachusetts Environmental Policy Act
MODFLOW-NWT	modular groundwater flow model with Newtonian solver
MWRA	Massachusetts Water Resources Authority
NOAA	National Oceanic and Atmospheric Administration
NPS	National Park Service
NWIS	National Water Information System
PEST	parameter estimation software
SFR	MODFLOW Streamflow Routing package
Ss	specific storage (1/feet)
Sy	specific yield (percent)
t_{50}	time for 50 percent of streamflow depletion to take place (day)
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey

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Abstract

The U.S. Geological Survey, in cooperation with the Town of Framingham, Massachusetts, has investigated the potential of proposed groundwater withdrawals at the Birch Road well site to affect nearby surface water bodies and wetlands, including Lake Cochituate, the Sudbury River, and the Great Meadows National Wildlife Refuge in east-central Massachusetts. In 2012, the U.S. Geological Survey developed a Phase 1 numerical groundwater model of a complex glacial-sediment aquifer to synthesize hydrogeologic information and simulate potential future pumping scenarios. The model was developed with MODFLOW-NWT, an updated version of a standard USGS numerical groundwater flow modeling program that improves solution of unconfined groundwater flow problems. The groundwater model and investigations of the aquifer improved understanding of groundwater–surface-water interaction and the effects of groundwater withdrawals on surface-water bodies and wetlands in the study area. The initial work also revealed a need for additional information and model refinements to better understand this complex aquifer system.

In this second phase of the study, the original groundwater flow model was revised to improve representation of groundwater and surface-water hydrology, stabilize the model, and reduce model error. The model was simplified by reducing the number of layers from 5 to 3 and adding the MODFLOW lake package (LAK) to simulate Lake Cochituate and Pod Meadow Pond and better represent interaction between the lakes and the aquifer. Model revisions improved stability and shortened run times, allowing use of automated parameter estimation software (PEST) to further refine the model hydraulic parameters and reduce simulation errors.

Model simulations indicate that under average base-flow conditions, the Birch Road wells have a small effect on flow in the Sudbury River during most months, even at the maximum pumping rate of 4.9 ft^3/s (3.17 Mgal/d). Maximum percent streamflow depletion in the Sudbury River caused by simulated pumping takes place during simulated drought conditions, when streamflow decreased by as much as 21 percent under maximum continuous pumping. Simulations also indicate that groundwater withdrawals at the Birch Road site could be managed so that adverse streamflow impacts are substantially ameliorated. Under the most ecologically conservative simulated drought conditions, simulated streamflow depletion was reduced from 21 percent to 3 percent by pumping at the maximum rate for 6 months rather than for 12 months. Simulations that return 10 percent of the Birch Road well withdrawals to Pod Meadow Pond indicate a modest reduction in the Sudbury River streamflow depletion and provide a larger percentage increase to streamflow just downstream of the pond. The groundwater model also indicates that well locations can have a large effect on the sustainable pumping rate and so should be chosen carefully. The model provides a tool for evaluating alternative pumping rates and schedules not included in this analysis.

Introduction

The Town of Framingham in east-central Massachusetts operated several groundwater water-supply wells in the area known as the Birch Road site (fig. 1) from 1939 until about 1979. In 2009, in accordance with the Massachusetts Environmental Policy Act (MEPA), the Town of Framingham filed an Environmental Impact Report (EIR) with the Massachusetts Department of Environmental Protection to reactivate these supply wells (SEA Consultants, Inc., 2009). The growing recognition of the interconnection between groundwater and surface-water resources and the potential effects of groundwater withdrawals on surface water led to concerns raised by the National Park Service (NPS), the U.S. Fish and Wildlife Service (USFWS), the U.S. Environmental Protection Agency (the EPA), state and local agencies, and environmental interest groups and citizens. The impacts from groundwater withdrawals on the nearby Sudbury River and the downstream Concord River are of particular concern to Federal interests because of the Great Meadows National Wildlife Refuge and the Minute Man National Historical Park, respectively. In addition, the Sudbury River is designated by Congress as "Wild and Scenic," requiring special resource protection (U.S. Congress, 1999). In response to those concerns and notice by the U.S. Department of the Interior Office of the Solicitor to specifically address Federal interests (U.S. Department of Interior, 2009), the Town of Framingham and the U.S. Geological Survey (USGS) agreed to collaborate on a study of the aquifer system to better understand the potential effects of groundwater withdrawals on nearby surface-water resources.

In 2012, the USGS published a report (Eggleston and others, 2012) under the cooperative agreement with the Town of Framingham that characterized the complex glacial-sediment aquifer system in the area of the proposed pumping wells. The report also described a groundwater flow model (MODFLOW–NWT) developed to better understand the connection between groundwater and surface water that was used to simulate the effects of pumping on the aquifer and nearby surface-water resources. Simulations indicated about one third of the proposed withdrawal is from induced infiltration of water from Lake Cochituate and the rate of streamflow depletion in the Sudbury River downstream from the oxbow (fig. 1) is about equal to the rate of pumping under steady state conditions, but streamflow depletion changed quickly in response to changes in pumping. The report identified the need for additional data collection and refinement of the model, which could affect the present understanding of groundwater/surface-water interactions in the study area, and the report identified the need to reexamine withdrawal scenarios tested.

Purpose and Scope

This report describes the second phase of the cooperative study to further improve the understanding of the complex glacial-aquifer system and the effects of groundwater withdrawals on surface-water resources in the vicinity of the Birch Road well site. This study was designed to improve understanding of the shallow groundwater system in the vicinity of the Birch Road well site and to build a groundwater model for assessing the effects of proposed pumping on nearby surface-water features including Lake Cochituate and the Sudbury River. This report describes the new information obtained to characterize the aquifer and its interaction with the surface-water system and the modifications made to the previously developed groundwater flow model (MODFLOW-NWT) of the glacial-sediment aquifer in the Birch Road area in east-central Massachusetts (Eggleston and others, 2012). This report also presents the revised model (herein referred to as Phase 2) and simulation results used to further quantify groundwater and surface-water interaction under present conditions and under various pumping scenarios at the Birch Road site to as much as a maximum of 3.17 million gallons per day (Mgal/d). The model scenarios include alternative pumping schedules and return of a portion of the withdrawal, which could reduce the effects of groundwater withdrawals during seasonal low streamflows when stream ecosystems are most sensitive.

Study Area

The study area is 16 miles (mi) west of Boston in east-central Massachusetts mostly in the towns of Framingham and Wayland (fig. 1). The groundwater aquifer is a complex glacial-fill aquifer in a bedrock valley trending north to south with a large bedrock outcrop (bedrock island) near the center of the study area (fig. 1). Several large ponds, including Lake Cochituate, and numerous streams overlay the aquifer. The Sudbury River flows from the southwest toward the northeast through the study area. The active groundwater model area is about 5.5 square miles (mi^2) and includes about 1.7, 3.4, 0.4, and 0.01 mi^2 in the towns of Framingham, Wayland, Sudbury, and Natick, respectively. The active model area was reduced from about 6.1 mi^2 in the Phase 1 study to 5.5 mi^2 in the Phase 2 study to improve model stability as described in the Groundwater Model Modifications section.

The Sudbury River is the primary surface-water drainage feature and sets the natural base groundwater level in the study area (fig. 2). The Great Meadows National Wildlife Refuge includes lands adjacent to the Sudbury River north of the oxbow and wetland areas in the northern part of the study area, although most of the refuge is to the north of the study area (fig. 1). The north pond of Lake Cochituate covers about 0.3 mi^2 and is the largest surface-water feature in active model area. A mostly natural causeway forms a divide between the north pond and two ponds to the south. The northern pond of Lake Cochituate has a contributing drainage area of 17.5 mi^2 and drains to the Sudbury River through the 1.4 mi long Cochituate Brook. Additional surface-water bodies include Dudley Pond, Pod Meadow Pond, and Heard Pond, which drain to the Sudbury River.

The aquifer is a complex mix of stratified glacially deposited sediments, including melt water deltaic deposits and proglacial lake deposits that range in texture from clay to coarse gravel and boulders. Most of these deposits however are medium to fine sands, which form the primary aquifer. The glacial geomorphological sequences and depositional features are described in the Phase 1 study by Eggleston and others (2012).

Previous Investigations

Previous studies describing geology and hydrology of the area were summarized in the Phase 1 report by Eggleston and others (2012). The Phase 1 report provides additional information on the bedrock geology and depth to bedrock, interpretation of the stratigraphy of the surficial glacial and post-glacial deposits, surface-water features, and water use. The Phase 1 report also documents the groundwater flow model development and parameterization, which is the basis for the model presented in this Phase 2 study, and the simulated effects of select groundwater withdrawals on select surface-water features, which were reexamined in the Phase 2 study.

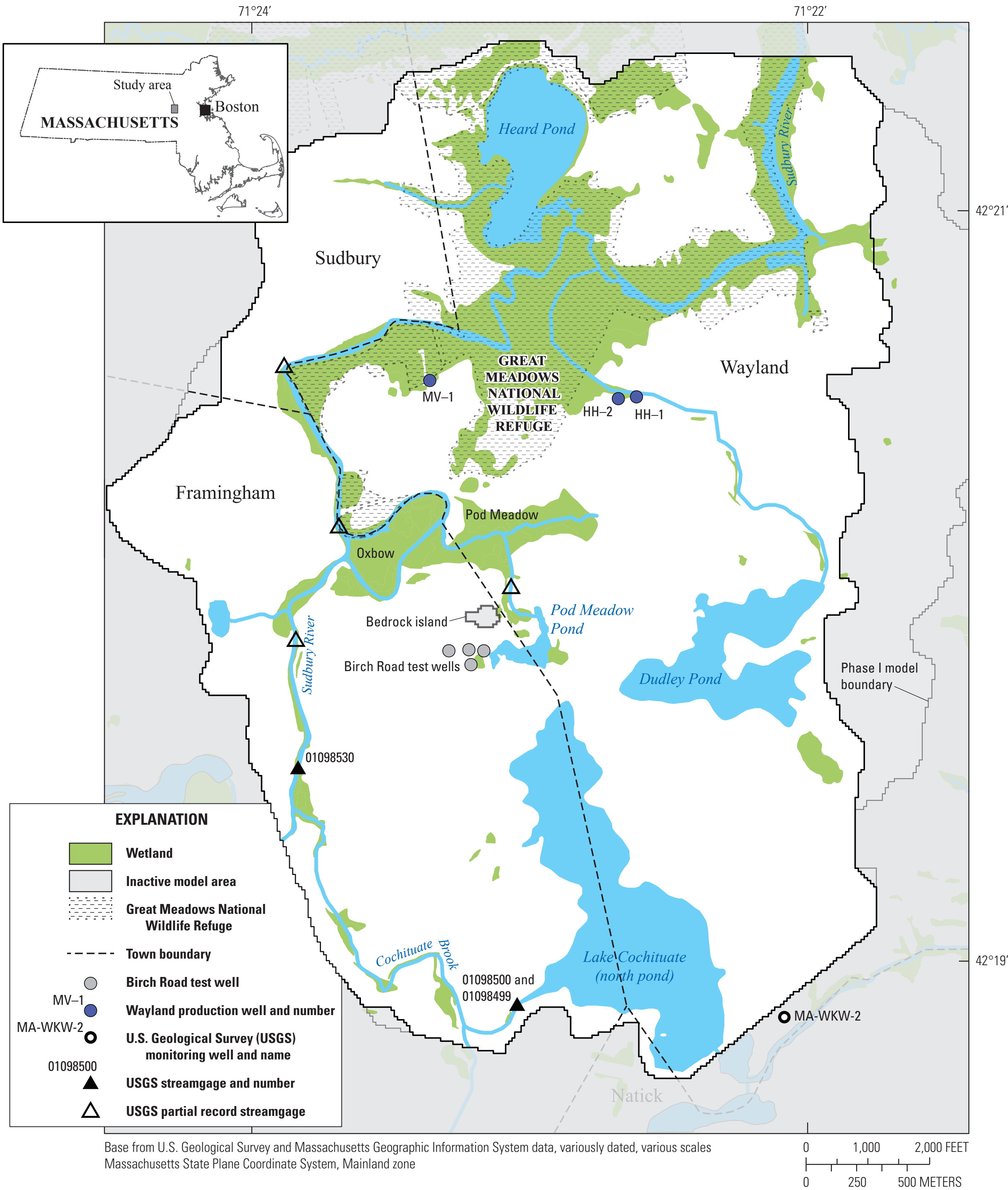


Figure 1. Glacial-sediment aquifer study area in east-central Massachusetts.

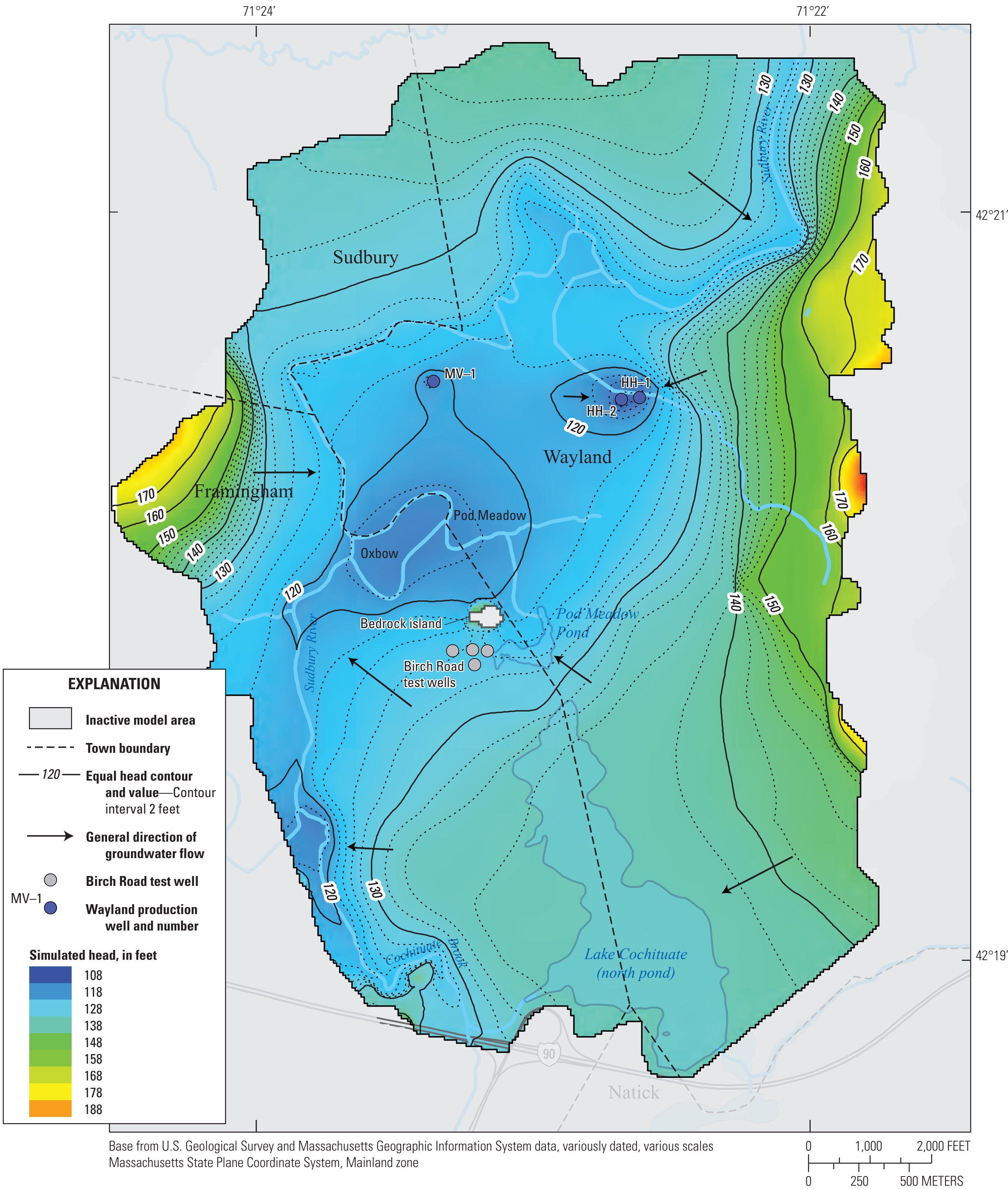


Figure 2. Groundwater levels (simulated heads) in the lower aquifer (model Layer 3) for the study area in east-central Massachusetts.

Data

In response to data needs identified in Phase 1, data were collected to fill data gaps to the extent possible and to support work in Phase 2. Data collected in Phase 2 include a marine seismic survey, specific conductivity measurements of Lake Cochituate and Pod Meadow Pond, water-level measurements from newly installed wells, and flow from Pod Meadow Pond. In addition, groundwater levels used to calibrate the Phase 1 model were adjusted to better reflect steady-state conditions.

Seismic Survey

Seismic reflection profiling was completed in Lake Cochituate on May 30, 2013, to better characterize lake bottom sediments and determine the extent of fine-grained lake bed sediment deposits (gyttja). The extent of gyttja deposits is important to know because gyttja has low hydraulic conductivity and can reduce the rate of flow between the lake and underlying aquifer. Previous attempts to characterize the lake sediments by use of ground penetrating radar (GPR), which is generally preferable for mapping coarse-grained lake bed sediments in fresh water systems, were unsuccessful because the high specific conductivity of the lake water quickly attenuated the radar signal. Marine (underwater) seismic waves scatter in response to gaseous fine-grained lake bed sediment deposits on the bottom of glacial ponds so they were used as a surrogate to map the extent of the gyttja deposits in the north pond of Lake Cochituate.

Seismic response was measured with a chirp profiling system (EdgeTech XSTAR SB-216) that emits an acoustic signal in the water at regular intervals with a 4–20 kilohertz operating frequency. A towfish pulled behind a boat emits an acoustic energy, which is reflected from different boundaries and recorded by a hydrophone. The return signal is digitally recorded for later processing using DelphSeismic software. Nineteen transects made on May 30, 2013, generally traversed the lake from north to south and east to west (fig. 3). The profiles covered about 6 miles and a total of 18,732 seismic traces were collected. The position of the tow fish was tracked continuously during the profiling using a global positioning system (GPS). The profile transects were played back with Edge Tech Corp software (v. 3.52) from which areas of gaseous deposits were identified by the highly scattered seismic traces. Examples of the profile transects and interpreted gaseous deposits are shown for lines 11 (east-west line) and 18 (north-south line) in figure 4. The extent of the gaseous deposits were mapped and used as an approximation of gyttja deposits extent.

The seismic profiles indicated an extensive area of gaseous gyttja deposits in the widest and deepest part of the north pond of Lake Cochituate (fig. 3). Smaller areas of gyttja deposits were interpolated in the northern part of the lake between underwater mounds or ridges. The mounds or ridges were most pronounced in an east–west direction where the lake shore narrows, however some mounds also appear to traverse in a north–south direction. One interpretation of how these underwater mounds formed can be made from the sequence of the glacial retreat (Clapp, 1904). Lake Cochituate formed as a kettle pond where a large block of ice persisted as the main glacier retreated and paused farther to the north. The ice block limited sediments from being deposited in proglacial Lake Charles and, when it eventually melted, it left a depression that formed Lake Cochituate (Gay, 1985). During this process the ice block that later created the lake was likely fractured where melt-water sediments collected. When the ice block finally melted these sediments formed the underwater mounds or ridges that are likely a mix of fine and coarse grain sediments, the locations of these deposits are inferred from sediment cores and geologic interpretation, but not known exactly. Hydraulically, the mounds or ridges may create preferential flow paths between the lake and underlying aquifer especially compared to the gyttja deposits that cover about 30 percent of the lake bottom. Because gyttja deposits do not cover most of the lake bottom they are unlikely to substantially limit flow between the aquifer and the lake, so values of lakebed conductance were not constrained to low values in the model.

Specific Conductivity Measurements

The Phase 1 study identified groundwater seepage into Pod Meadow Pond as a potentially important hydraulic exchange with Lake Cochituate. To further investigate possible groundwater flow from Lake Cochituate to Pod Meadow Pond, specific conductivity profile measurements were made on May 30, 2013, at four locations in Lake Cochituate at 5 to 10 foot (ft) vertical intervals from near the water surface to near the lake bottom (fig. 5 and table 1). Values ranged from 499 to 545 microsiemens per centimeter (µS/cm) at 25 degrees Celsius with a mean of 519 µS/cm that decreased slightly with depth, and were generally consistent at the four measurement locations.

Additional specific conductivity measurements were made along the edge of Pod Meadow Pond (table 2) on June 5, 2013. The specific conductance values in Pod Meadow Pond were similar to those in Lake Cochituate although slightly lower at most locations (fig. 5 and tables 1–2). Anomalously lower specific conductance values were measured at sites 4 and 5 (312 and 140 µS/cm, respectively) and were 30 and 70 percent lower than the median specific conductance measurement of 435 µS/cm at Pod Meadow Pond. The lower specific conductance at sites 4 and 5 may provide another indication that groundwater discharge may be focused in the south-western part of the pond. Other indicators of focused

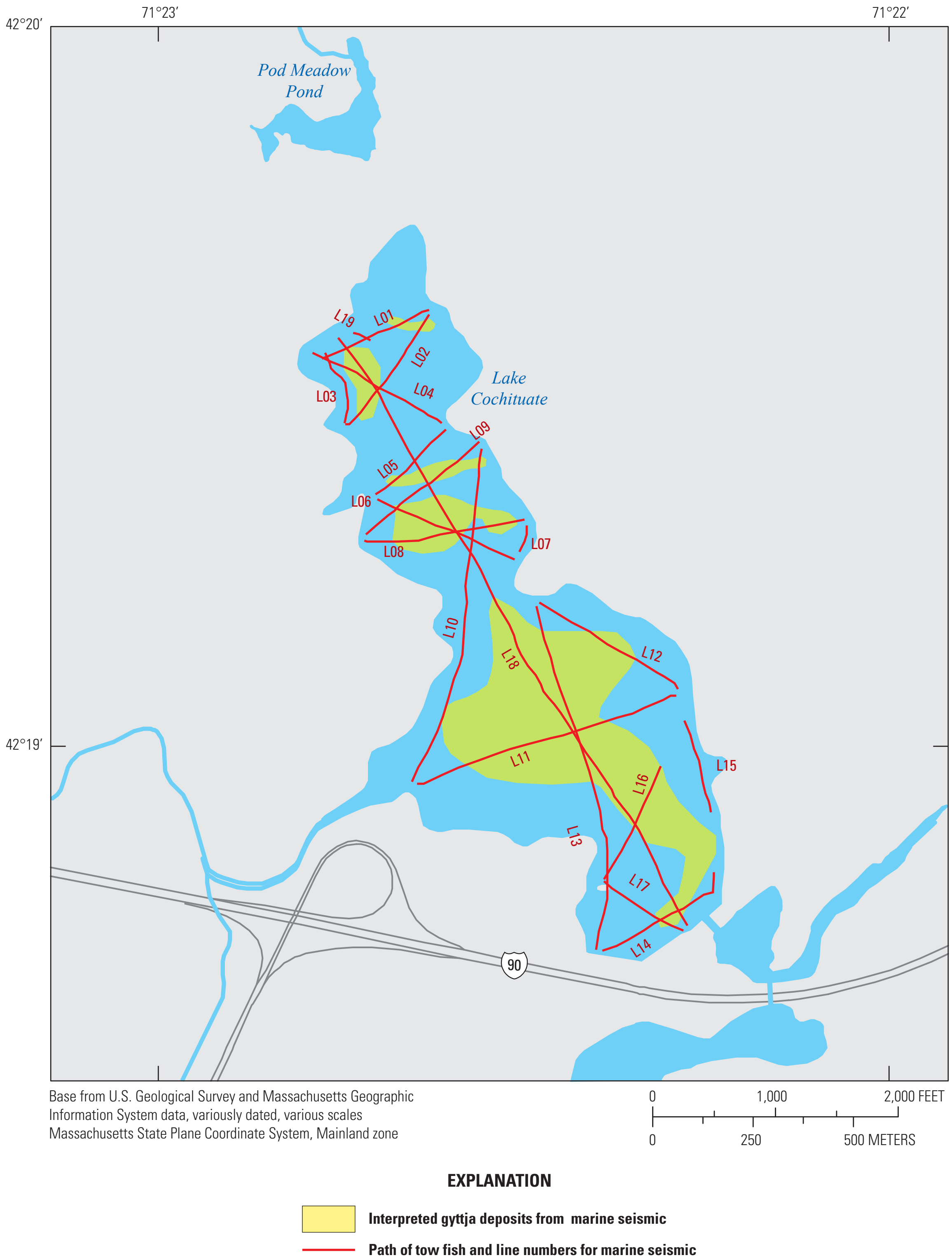


Figure 3. Seismic reflection transect lines made on May 30, 2013, Lake Cochituate (north pond), east-central Massachusetts.

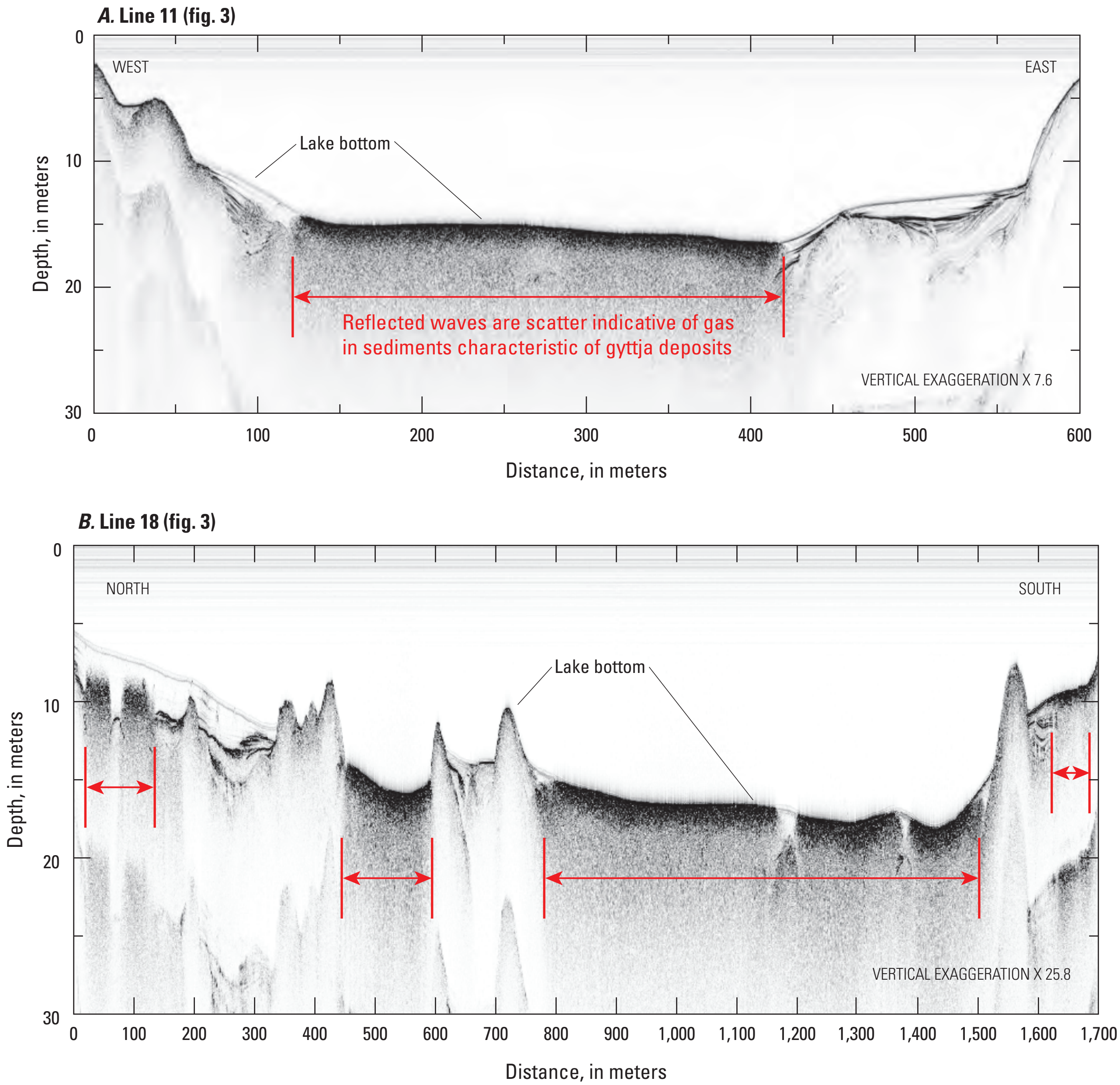


Figure 4. Example of marine transect profiles (lines 11 and 18, figure 3) of Lake Cochituate (north pond), east-central Massachusetts.

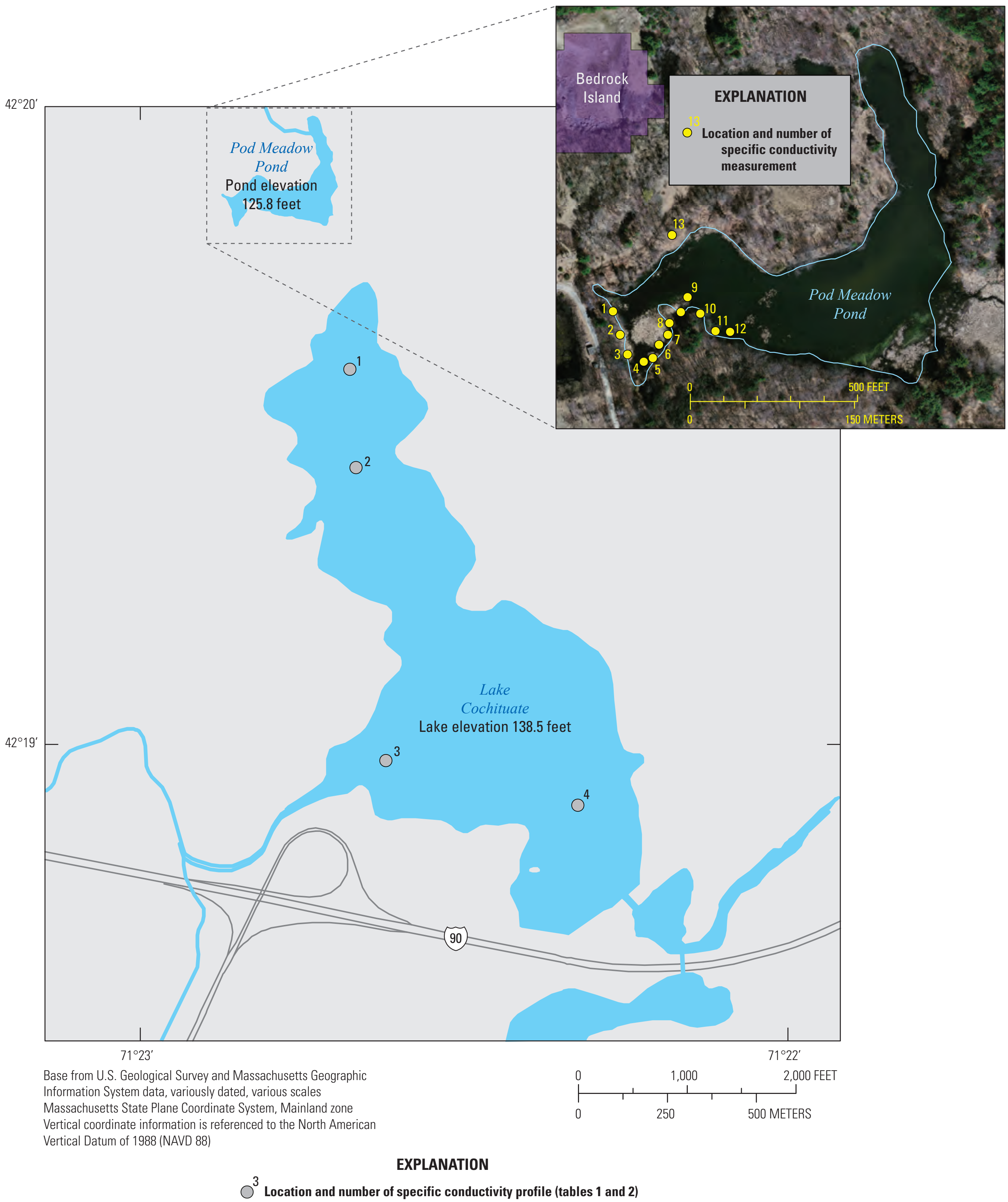


Figure 5. Locations of water specific conductance measurements made in Lake Cochituate and near shore of Pod Meadow Pond in east-central Massachusetts.

Table 1. Specific conductivity profile measurements made on May 30, 2013, in Lake Cochituate, east-central Massachusetts.

[ft, feet; WT, water temperature; °C, degrees Celsius, SC, specific conductivity; µS/cm, microsiemens per centimeter at 25 degrees Celsius]

Site (fig. 5)	Latitude	Longitude	Water depth (ft)	WT (°C)	SC (µS/cm)	Remarks
1	42°19′33.7″	-71°22′55.0″	1	20.0	526	Northern end of lake.
			7	17.8	535	
			17	13.0	512	
			27	8.9	499	
2	42°19′22.4″	-71°22′52.8″	1	22.0	528	
			7	18.0	534	
			13	16.6	531	
			19	11.8	511	
			25	8.9	505	
			31	8.2	506	
3	42°18′58.6″	-71°22′48.9″	1	21.0	532	Near lake outlet.
			6	18.0	534	
			12	16.6	535	
			19	13.0	511	
			25	8.3	507	
			31	7.5	505	
			37	7.4	501	
4	42°18′56.2″	-71°22′26.7″	1	21.2	537	Not at lake bottom.
			6	20.0	536	
			12	17.8	545	
			18	14.5	521	
			24	9.7	508	
			30	8.0	505	
			36	7.6	503	
					516	Median.

Table 2. Specific conductivity measurements made on June 5, 2013, near the shoreline of Pod Meadow Pond, east-central Massachusetts.

[WT, water temperature; °C, degrees Celsius; SC, specific conductivity; µS/cm, microsiemens per centimeter at 25 degrees Celsius; ft, feet; -- not recorded]

Site (fig. 5)	Time	WT (°C)	SC (µS/cm)	Remarks
1	11:22	22.5	424	shaded, 6 ft offshore.
2	11:26	22.6	428	shaded, 1 ft depth.
3	11:29	20.0	415	shaded.
4	11:38	13.2	312	shaded, very shallow, with algae.
5	11:40	14.2	140	in 1 ft depth hole dug at shore.
6	11:43	11.3	477	active groundwater discharge at pond edge.
7	11:50	11.4	501	active groundwater discharge, 6 ft offshore.
8	11:51	11.3	502	active groundwater discharge.
9	11:55	19.1	435	shaded, 8 ft offshore.
10	11:56	12.0	383	active groundwater discharge in hole dug 1 ft deep at shore.
11	11:58	15.5	450	6 ft offshore.
12	12:01	22.0	436	4 ft offshore.
13	--	28.8	435	north shore of pond.
			435	Median.

groundwater discharge are open water observed in the same area in February 2011, when other areas of the pond were frozen (Eggleston and others, 2012), and groundwater seepage noted along the shore of the pond during the June 2013 measurements. If the south-western part of the pond is an area of focused groundwater discharge, there are several possible contributing causes. The horizontal distance between the north shore of the lake and the south shore of the pond is small, only about 700 ft, whereas the difference in lake and pond surface elevations, about 12 ft, providing a relatively steep hydraulic gradient to drive groundwater flow. The bedrock island just to the north of the pond (fig. 1) and the rise in the bedrock valley to the north may force groundwater flow upward in this area.

The similarity between specific conductivity values measured at the bottom of Lake Cochituate near the north end (499) and at areas of observed groundwater discharge in Pod Meadow Pond (average of 466 µS/cm for sites 6–8, 10 in table 2 further support the hypothesis of rapid flow of recharge from Lake Cochituate to the aquifer that subsequently discharges to Pod Meadow Pond.

Streamflow Measurements

To better understand rates of groundwater discharge to the Sudbury River, a series of streamflow measurements, referred to as seepage measurements, were made on July 13, 2012, along a 1.3-mi reach of the Sudbury River. Flows in the Sudbury River at Saxonville (0109850) were then at about the 90-percent flow duration based on daily streamflow records, indicating base-flow conditions when groundwater is the primary source of water to the river. Four measurements were made in the Sudbury River (fig. 1) beginning at streamgage (01098530) at Saxonville (0 miles [mi]), about halfway between the streamgage and the oxbow (0.35 mi), at the oxbow (0.75 mi), and near where the river bends sharply to the east (1.3 mi). Streamflow at these locations were 16.1, 15.5, 16.5, and 15.2 ft^3/s, respectively, and were rated good (plus or minus 5 percent) to fair-poor (plus or minus 8 percent) because of the very shallow depths. Seepage measurements indicate a loss between the most upstream and downstream sites ranging from -0.6 to 1.3 ft^3/s but, given the potential measurement error, this may not be an accurate measure of groundwater discharge to the river.

Streamflow and groundwater-level measurements have continued to be collected since the Phase 1 study ended. Daily streamflow records have been collected for Cochituate Brook below Lake Cochituate at Framingham, Massachusetts (USGS station 01098500) and daily stage records have been collected for Lake Cochituate at Framingham, Mass., (USGS station 01098499). Both stations are operated and maintained in a cooperative agreement with the Town of Framingham and the USGS. Streamflow exiting Pod Meadow Pond was measured on a monthly basis through December 2012 to extend the flow record and improve descriptive flow statistics (table 3). The additional Pod Meadow Pond outlet streamflow measurements were very important because average Pod Meadow Pond outlet flow value is the only flux observation used in model calibration, as described later (Model Calibration section).

Table 3. Discharge measurements at the outflows from Pod Meadow Pond and Dudley Pond, east-central Massachusetts.

[ft^3/s, cubic feet per second]

Date	Discharge (ft^3/s)
	Pod Meadow Pond
03-23-2011	1.18
05-09-2011	0.79
06-14-2011	0.65
07-12-2011	0.43
08-01-2011	0.54
09-12-2011	0.74
10-07-2011	0.45
11-10-2011	[1]0.14
11-29-2011	[1]0.12
12-14-2011	0.58
01-09-2012	0.86
02-08-2012	0.68
03-08-2012	0.75
04-10-2012	0.63
05-07-2012	0.70
06-01-2012	0.65
07-05-2012	0.79
08-09-2012	0.60
09-11-2012	0.49
10-05-2012	0.44
11-09-2012	0.71
12-07-2012	0.54
Mean	[2]0.66
Standard deviation	[2]0.17

[1]Discharge affected by upstream beaver activity.

[2]November 2011 measurements excluded because of upstream beaver activity.

Observation Wells

Although many boreholes exist in the study area, more lithological information was needed in a few locations to better define the extent of a clay layer under Lake Cochituate and to determine depth to bedrock. Boreholes were drilled at five sites and seven observation wells installed at four of these sites by Bristol Engineering Advisors, Inc. (Bristol) on the behalf of the Town of Framingham for use in this study (fig. 6 and table 4). The sites were strategically located in accessible areas around the Birch Road wells where additional lithology and water-level information were determined to be of greatest value. Lithologic results from the boreholes were incorporated into model layering and representation of low hydraulic conductivity clay in the model, as discussed later (Groundwater Model Modifications section).

A series of seasonal water-level measurements at the newly installed observation wells were made by Bristol on October 1, 2011; February 15, 2012; June 19, 2012; and August 21, 2012. Observation well SB-3 was not measured because it was not screened and was used only for determining the depth to bedrock. The minimum water levels were measured near the end of August and ranged from 0.69 to 1.69 ft lower than the maximum water levels measured in October or in February. Water-level measurements made at these wells had an average standard deviation of 0.44 ft reflecting relatively small seasonal variations. Differences in water levels at paired wells finished in the upper and lower aquifer (SB-1, SB-4, and SB-5) were small at SB-1 and SB-5 (averaged 0.45 and 0.16 ft, respectively), but were relatively large at SB-4 (averaged 27.2 ft higher in the upper aquifer than the lower aquifer). The high groundwater level in the shallow well at SB-4 is similar to high levels in nearby shallow wells and likely indicates a perched upper aquifer that could be caused by the presence of the low hydraulic conductivity clay layer present in this area. Because a perched aquifer would be separated by an unsaturated zone from the deeper aquifer, high groundwater levels observed in shallow wells in this area were not used as calibration targets for the model. Water-level measurements from the other newly installed wells were used for model calibration.

Groundwater Elevations

Groundwater elevations from 63 observation wells were used to calibrate the Phase 2 groundwater model. The number of water-level measurements at these wells ranged from 1 to 16 with a median of 3. The measurements spanned from 1946 through 2012, but the time span varied among the observation wells and typically was grouped in clusters, such as around a 2006 aquifer pumping test. For calibration of a steady-state model meant to represent average conditions, it is important that groundwater levels used as calibration targets not be strongly affected by seasonal or climatic variations lasting 1–2 years or less. Typically, a mean groundwater level is compiled for each site having multiple measurements; however, when there are few measurements or when the measurements are temporally clustered around a period of relatively high or low water levels, a simple average may not be representative of average conditions over 10 years or more.

In the Phase 1 model calibration, a simple average of the water-level measurements at each observation well was used for calibration. To minimize bias in Phase 2 head calibration targets, the measured groundwater levels were adjusted by referencing them to water level measurements at USGS well MA-WKW-2 (421852071220501) (fig. 1), which has monthly data from 1965 to 2010 and continuous data thereafter. Deviations of the daily water level in MA-WKW-2 (interpolated

Figure 6. Locations of borehole drilling sites and wells installed in 2011 in the Birch Road well area, east-central Massachusetts.

Table 4. Observation wells installed in 2011 in the Birch Road well aquifer area, east-central Massachusetts.

[Lat., latitude, in decimal degrees; Lon., longitude, in decimal degrees; elevation (approximate) in feet NAVD 88; feet, feet below land surface; --, not applicable]

Site (fig. 6)	Lat.	Lon.	Land surface elevation (feet)	Well depth (feet)	Top of bedrock (feet)	Well screen elevation (feet)				Aquitard elevation (feet)		Clay present
						Shallow well		Deep well				
						Top	Bottom	Top	Bottom	Top	Bottom	
SB–1	42.3314	-71.3847	125	61.5	60	21.2	26.2	51.5	55.5	--	--	no
SB–2	42.3265	-71.3845	186	176	176	50	55	--	--	55	80	yes
SB–3	42.3310	-71.3853	125	45	38	--	--	--	--	--	--	no
SB–4	42.3254	-71.3946	180	124	124	64	69	99	104	70	90	yes
SB–5	42.3351	-71.3827	125	170	170	43	48	90	95	48	58	no

when only monthly data were available) from mean water level for the full period of record were used to adjust water-level measurements at observation wells used in the Phase 2 study to better account for seasonal water-level conditions.

MA-WKW-2 is located just outside the active model boundary to the south and about 1,060 ft east of Lake Cochituate (fig. 1) and is finished in coarse grain material at about 32 ft depth below land surface. The groundwater-level adjustment at observation wells used to calibrate the Phase 2 steady-state model assumes water-level fluctuations at MA-WKW-2 are representative of fluctuations in the observation wells used in the steady-state calibration. The validity of this assumption was tested by comparing water-level fluctuations at MA-WKW-2 with the few observation wells in the study area with sufficient data to compare against. Observation wells WKW-119, WKW-117, and F1W-84, finished in the upper aquifer had about 1 year of monthly data and generally followed the same pattern of water-level fluctuations observed in MA-WKW-2. Observation wells MW-4 and MW-5, finished in the lower aquifer had less than 1 year of data and had more sporadic observations. Water-level fluctuations in these wells did not match the fluctuation in MA-WKW-2 as closely as wells finished in the upper aquifer. Although the adjustment method helps water level observations better reflect appropriate mean values, the limited groundwater level measurement data, particularly in the lower aquifer, are still a source of model calibration error and uncertainty.

Eight observation wells used in the Phase 1 model calibration were not included in the Phase 2 calibration because these wells each had only one measurement made before 1978 when water levels may have been affected by the then active Birch Road supply wells. In addition, other nearby observation wells were available that had more measurements when the Birch Road wells were inactive. The total number of wells used in the Phase 2 calibration included 57 wells used in the Phase 1 calibration, plus 6 of the new wells installed in 2011 by Bristol. The revised water-level calibration dataset has an average of 6.4 measurements per well and just 7 wells with only 1 measurement.

Observation wells in the USGS NWIS database (USGS, 2014) (18 of the 63 wells) used in this study are in the National Geodetic Vertical Datum of 1929 (NGVD 29) datum and were corrected to the North American Vertical Datum of 1988 (NAVD 88) (-0.774 ft correction). With the exception of the seven recently installed wells (2011) and the observation wells in NWIS, the datum of other observation wells is unknown. Well installation documentation refers to the well elevation in feet above mean sea level only, which was assumed to be the NGVD 29 datum. To bring all well coordinates into the NAVD 88 datum a -0.774 ft correction was applied to groundwater-level measurements suspected of being in NGVD 29 datum.

Groundwater Model Modifications

In the Phase 1 study (Eggleston and others, 2012), a groundwater model was developed using MODFLOW-NWT (Niswonger and others, 2011) to simulate groundwater flow, stream base flow, and the effects of groundwater pumping on surface water. MODFLOW-NWT is a Newton-Raphson formulation for MODFLOW-2005 (Harbaugh, 2005) to solve problems involving drying and rewetting nonlinearities of the unconfined groundwater flow equation (http://water.usgs.gov/ogw/modflow-nwt/). The original groundwater flow model was discretized into 360 rows and 280 columns of cells 50 ft on a side (Eggleston and others, 2012, fig. 9) and discretized into 5 layers of variable thickness including explicitly representing bedrock as a layer (Eggleston and others, 2012, fig. 11). The original model was surrounded by no-flow boundary conditions (Eggleston and others, 2012, fig. 9). Model cells representing Dudley and Pod Meadow Ponds, and Lake Cochituate were modeled with high hydraulic conductivity. The original model was run in both steady-state and transient modes; the transient model had 60 monthly stress periods representing 5 years of average monthly conditions. The original model was calibrated by trial-and-error, with the resulting hydraulic parameter distribution described in Eggleston and others (2012). The original model had some issues with numerical instability and was sensitive to hydraulic conductivity under Lake Cochituate and around the pumping wells (Eggleston and others, 2012). Several issues such as these were identified in the original report that, if resolved, could help refine simulation of the area. This section describes implementation of changes to the Phase 1 groundwater flow model. Changes made to the groundwater model in Phase 2 include the following:

- Combining Phase 1 model Layers 1 and 2 (surface and near surface layers) and deleting Layer 5 (bedrock layer);
- Representing Lake Cochituate with the MODFLOW lake (LAK) package;
- Removing Dudley Pond from the model;
- Inactivating cells with shallow depths to bedrock; and
- Adjusting model parameterization using automated methods.

Changes to the model reduced run times (by about a factor of 30) and improved model stability, allowing automated parameter estimation to be applied.

Layering and Active Model Area

Model layering was simplified from the 5-layer model developed in Phase 1 to a 3-layer model in Phase 2 (fig. 7). Layers 1 and 2 of the Phase 1 model were combined to form Layer 1 of the Phase 2 model. In Phase 1, Layer 1 represented a thin (10 ft thick or less) unconsolidated sediment layer that became mostly dry during steady-state simulation (fig. 12*A* in Eggleston and others, 2012). Because this layer had little effect on the simulations it was combined with the underlying Layer 2 to form the uppermost layer (Layer 1) in the Phase 2 model, representing surficial sediment and underlying sand. Layer 2 of the Phase 2 model represents silt and clay deposits present in the north and south of the study area. Where buried silt and clay deposits do not exist in the study area, model Layer 2 was assigned a thickness of 0.5 ft. Layer 3 of the Phase 2 model represents the deeper sand and gravel aquifer in which the simulated supply wells are screened. Layer 5 in the Phase 1 model (top 80 ft of bedrock) was removed as it was determined to be relatively unimportant in model simulations relative to the more permeable unconsolidated deposits and little is known about the hydraulic properties of bedrock in the study area.

The active model area was reduced from about 6.1 mi^2 in the Phase 1 model to 5.5 mi^2 in the Phase 2 model by inactivating cells, mostly along the eastern model boundary and in other areas where bedrock was close to the surface (fig. 1). A notable change was the small bedrock island (fig. 1) just north of the Birch Road area, which was made inactive in the Phase 2 model. Other structural characteristics of the Phase 2 model remained unchanged relative to the Phase 1 model. These structural characteristics include total depth of the aquifer above bedrock and a uniform spatial discretization of 2,500 ft^2 cells (50 by 50 feet) with 360 rows (north–south) and 280 columns east-west.

Boundary Conditions and Surface-Water Body Representation

High horizontal hydraulic conductivity (Kh) zones in the Phase 1 model aquifer that represent Pod Meadow Pond and Lake Cochituate were removed in the Phase 2 model and replaced with surface boundary lakes using the Lake Package (LAK) in MODFLOW (Merritt and Konikow, 2000). LAK boundary cells control the flux between the water body and the top layer of the aquifer, providing greater flexibility in assignment of lakebed conductivity between open water and the aquifer. The depth of Pod Meadow Pond was set to 9.5 ft, whereas the depth of Lake Cochituate varied spatially according to bathymetric data. Inflows to Lake Cochituate and Pod Meadow Pond were assigned with the stream package (SFR), but precipitation to and evapotranspiration from the two lake surfaces were not simulated. Dudley Pond was removed from the Phase 2 model because the extensive thick gyttja deposits on the pond bottom limit it as a source of water to simulated pumping of the Birch Road wells. The simulated stream cells through Dudley Pond remain unchanged.

Specified flow boundaries representing pumping at Wayland water supply wells (HH-1, HH-2, and MV-1 on fig. 1) were unchanged from the Phase 1 model. Long-term average steady-state or monthly pumping rates were assigned at all 3 wells for all simulations with the Phase 2 model.

Streams represented by the streamflow-routing package (SFR2; Niswonger and Prudic, 2005) were assigned to appropriate cells in Layer 1 (fig. 8). SFR2 cells traversing Lake Cochituate or Pod Meadow Pond were removed to accommodate the LAK package in the Phase 2 model except for two SFR2 cells needed to interface with the LAK cells. A SFR2 cell was specified at the southern end of Lake Cochituate to receive inflow from SFR2 cells to the south and another SFR2 cell was specified at the outlet of Lake Cochituate to receive output from the lake at the head of Cochituate Brook. Similarly, SFR2 cells interface to LAK cells to account for indirect runoff to Pod Meadow Pond and output water from the LAK cells at the outlet. All other SFR2 cells in the Phase 2 model remain unchanged from the Phase 1 model.

Inflows were specified at stream reaches at the model boundary to account for upstream drainage area contributions to streamflow. For the steady-state model, inflows assigned to the Sudbury River, Lake Cochituate, and the tributary to Cochituate Brook were the same as those specified in the Phase 1 model; 107.5, 11.6, and 1.4 ft^3/s, respectively (fig. 8). These values are based on the median daily flow (144 ft^3/s) observed at the Sudbury River at Saxonville (01098530) streamgage from January 1980 through December 2010 proportionally distributed based on the drainage area upstream from each boundary and the Saxonville streamgage, less a small amount (23.5 ft^3/s) to account for the intervening area between the streamgage and the model boundaries.

Transient simulations specified monthly inflows at the stream boundaries. For transient simulations of average monthly conditions, the monthly inflows were held the same as those specified in the Phase 1 model based on the 25th-percentile daily flows during each month. The 25th-percentile flows are equivalent to the monthly 75-percent flow duration, which is the flow value that is exceeded 75 percent of the time. The monthly 25th-percentile flows were computed from observed flows at the Sudbury River at Saxonville (01098530), from November 1979 through November 2011, and were apportioned by drainage area at the Sudbury River boundary (80.2 percent) and the Lake Cochituate boundary (17.8 percent), less a small percent of the drainage area accounting for by the active model area between the streamgage and the stream cell boundary cells.

For transient simulations of dry conditions, monthly inflows at the stream boundaries were specified as 10th-percentile daily flows for each month (table 5). The 10th-percentile flows are equivalent to the monthly 90-percent flow duration, which is the daily flow value exceeded 90 percent of the time, and are representative of drought condition, especially when sustained for more than a 12-month period.

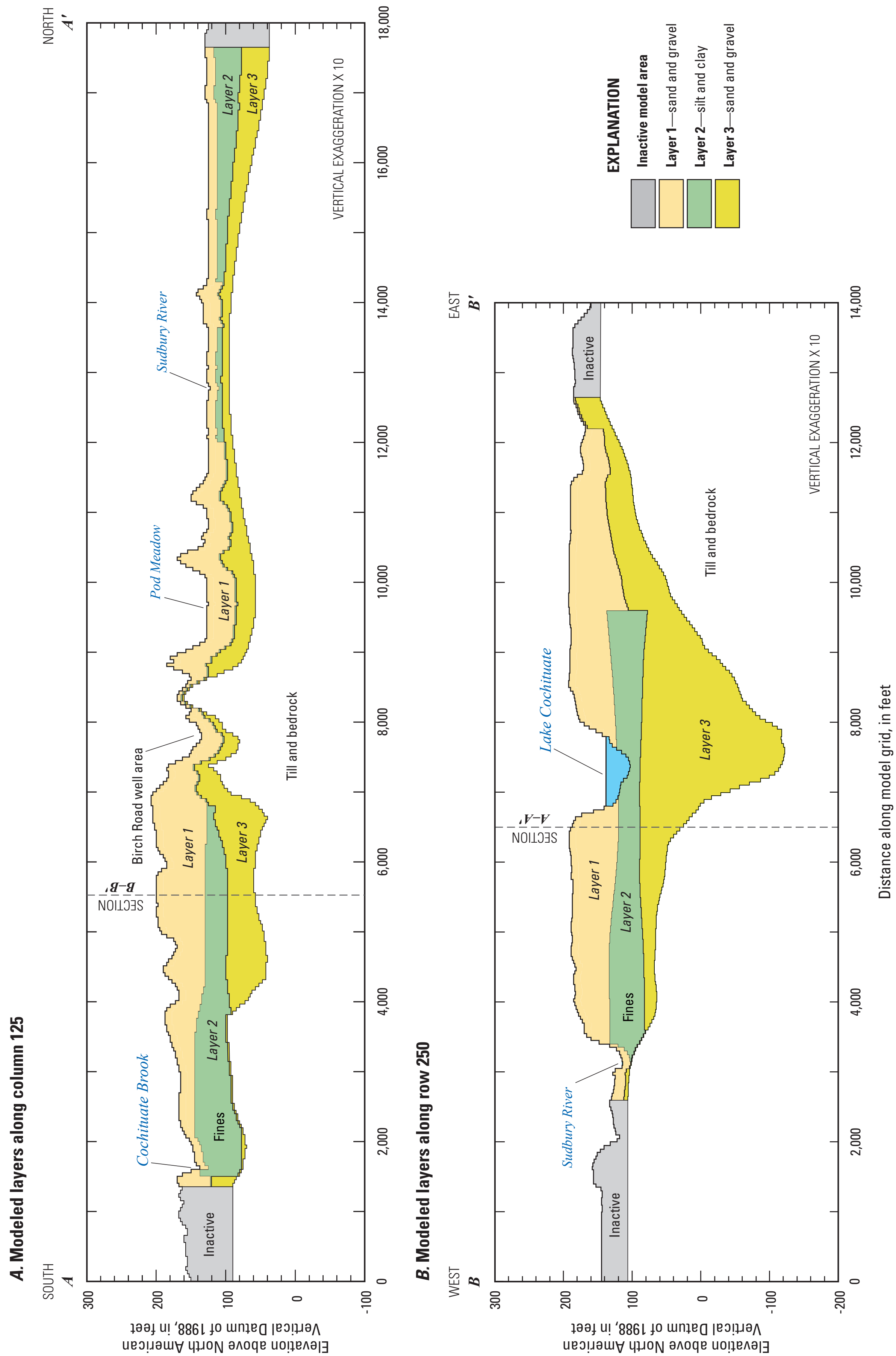
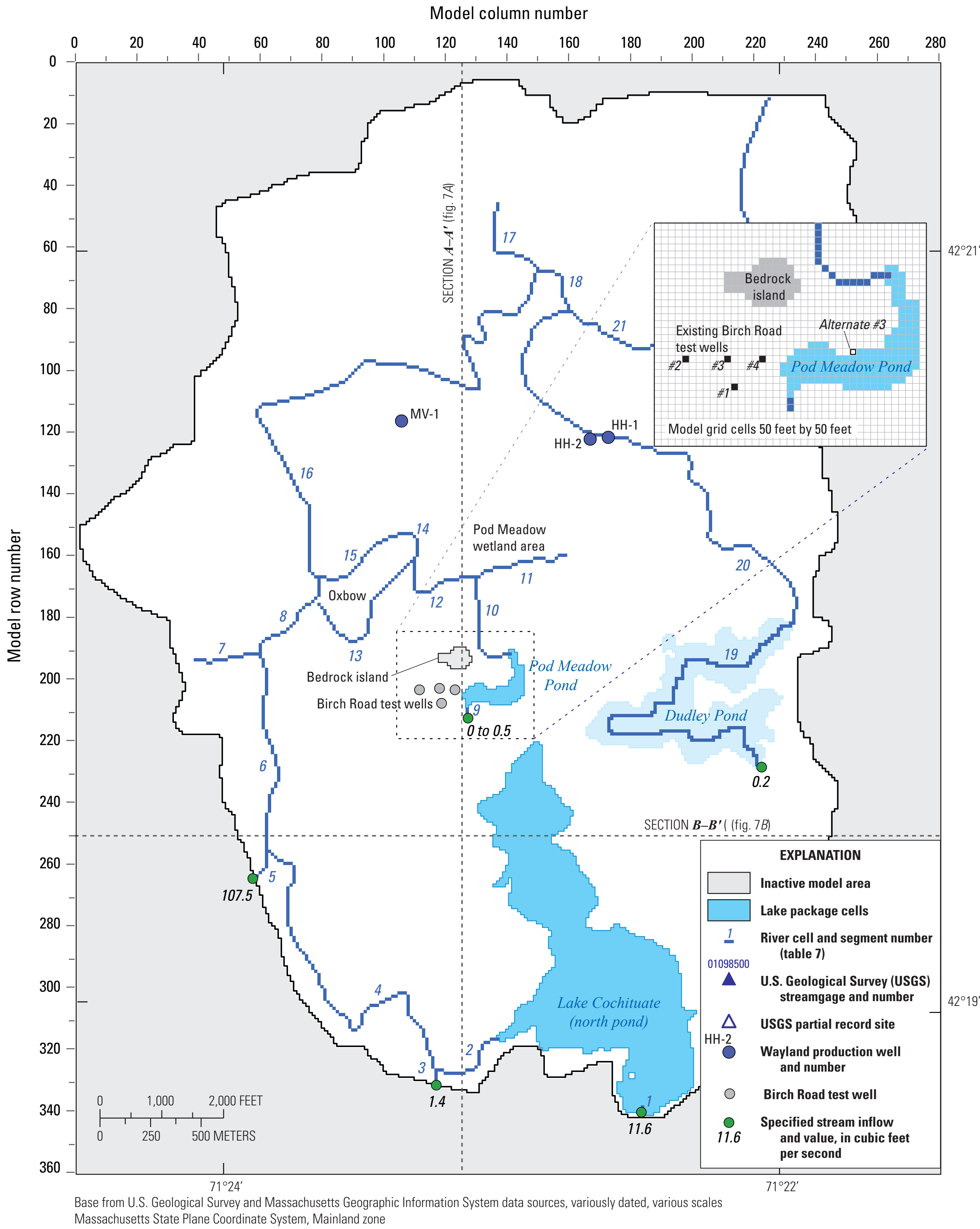


Figure 7. Vertical cross sections of groundwater model layers *A*, north-south and *B*, east-west.



Base from U.S. Geological Survey and Massachusetts Geographic Information System data sources, variously dated, various scales
Massachusetts State Plane Coordinate System, Mainland zone

Figure 8. Boundary conditions in the Phase 2 groundwater model.

Table 5. Assigned monthly stream inflow to the model, in cubic feet per second, under average (25th percentile) and dry (10th percentile) climatic conditions.

Month	Average conditions			Dry conditions		
	Lake Cochituate	Sudbury River	Total	Lake Cochituate	Sudbury River	Total
	Stream segment 1	Stream segment 5		Stream segment 1	Stream segment 5	
January	24.8	103.2	128.0	14.5	60.5	75.0
February	26.8	111.9	138.8	17.0	71.0	88.0
March	42.2	175.8	218.0	22.7	94.9	117.6
April	37.9	158.1	196.0	23.8	99.3	123.1
May	23.0	95.8	118.8	13.9	58.1	72.0
June	10.8	45.2	56.0	7.4	30.7	38.1
July	5.6	23.4	29.0	3.3	13.7	17.0
August	4.0	16.7	20.8	1.9	7.8	9.7
September	3.5	14.5	18.0	1.7	7.3	9.0
October	6.8	28.2	35.0	2.3	9.7	12.0
November	14.5	60.5	75.0	8.3	34.7	43.0
December	23.2	96.8	120.0	14.2	59.4	73.6

The 10th-percentile daily flows for each month observed at the Saxonville streamgage (January 1980 through December 2014) were apportioned again by applying 80.2 percent of the flow to the Sudbury River boundary and 17.8 percent of the flow to the Lake Cochituate boundary. The monthly 10th-percentile flows decreased by 34 to 68 percent compared to the 75th-percentile flows with an average monthly decrease of 56 percent. There were no inflows to the tributary to Cochituate Brook for the dry-condition simulations.

Recharge rates in the Phase 2 model were assigned as specified fluxes to the top of the model at rates equal to the Phase 1 model, except for formerly active model areas that were made inactive in the Phase 2 model had recharge reassigned to the nearest active model cells in proportion to the area made inactive. For steady-state simulations, recharge was assigned a spatially uniform value of 0.00502 feet per day (ft/d) (22 inches per year [in/yr]). For the transient model, recharge rates were varied monthly at the same rates as in the Phase 1 model (table 6, Eggleston and others, 2012), including for transient simulations of dry conditions in which recharge was turned off during July, August, and September. Recharge was not calibrated or treated as an adjustable parameter during the model calibration process because the rate is relatively well known and constrained by previous studies (Zarriello and others, 2010; DeSimone, 2004; DeSimone and others, 2002).

Hydraulic Parameter Assignment

A major objective of the Phase 2 model was to improve the representation of aquifer hydraulic parameters. To achieve this objective, the Phase II model was calibrated by automated parameter estimation, which provides an optimal set of parameter values for the model design and observation dataset, in contrast to the Phase I study in which the model was manually calibrated. Calibrated hydraulic parameter values are described in this section of the report and compared to Phase 1 model values from Eggleston and others (2012). Automated parameter estimation techniques used to calibrate the Phase 2 model are also described in this section of the report. Model error and sensitivity are discussed later in the report (Model Uncertainty and Sensitivity Analysis section).

Horizontal hydraulic conductivity (Kh) was adjusted using automated parameter estimation techniques only within 2,500 ft of the Birch Road test wells (fig. 8) because the many boreholes in that area reveal a complicated alluvial sediment structure and provide numerous water-level observations for detailed model calibration. Beyond 2,500 ft from the Birch Road test wells, calibration data are sparse and automated calibration of Kh outside the 2,500-ft radius tended to produce unreasonably high and low values, based on previous studies of similar hydrogeologic settings in Massachusetts (DeSimone and others, 2002; Masterson and others, 2009; Eggleston and others, 2012). Hence, Phase 2 model Kh values outside of the 2,500-ft radius were assigned according to the conceptualized hydrogeology described in Phase 1 of the study. One change to the conceptualized hydrogeology was made by adding a zone of sand and gravel (fig. 9) around the Happy Hollow wells (HH-1 and HH-2 in fig. 9). This zone encompassed 672 cells where Kh was increased from 10 or 20 ft/d to 200 ft/d in layers 2 and 3. The Kh increase prevented unrealistically high simulated groundwater-level drawdowns in response to pumping at those wells and is consistent with descriptions of highly conductive sand and gravel deposits in that area from Fortin (1981).

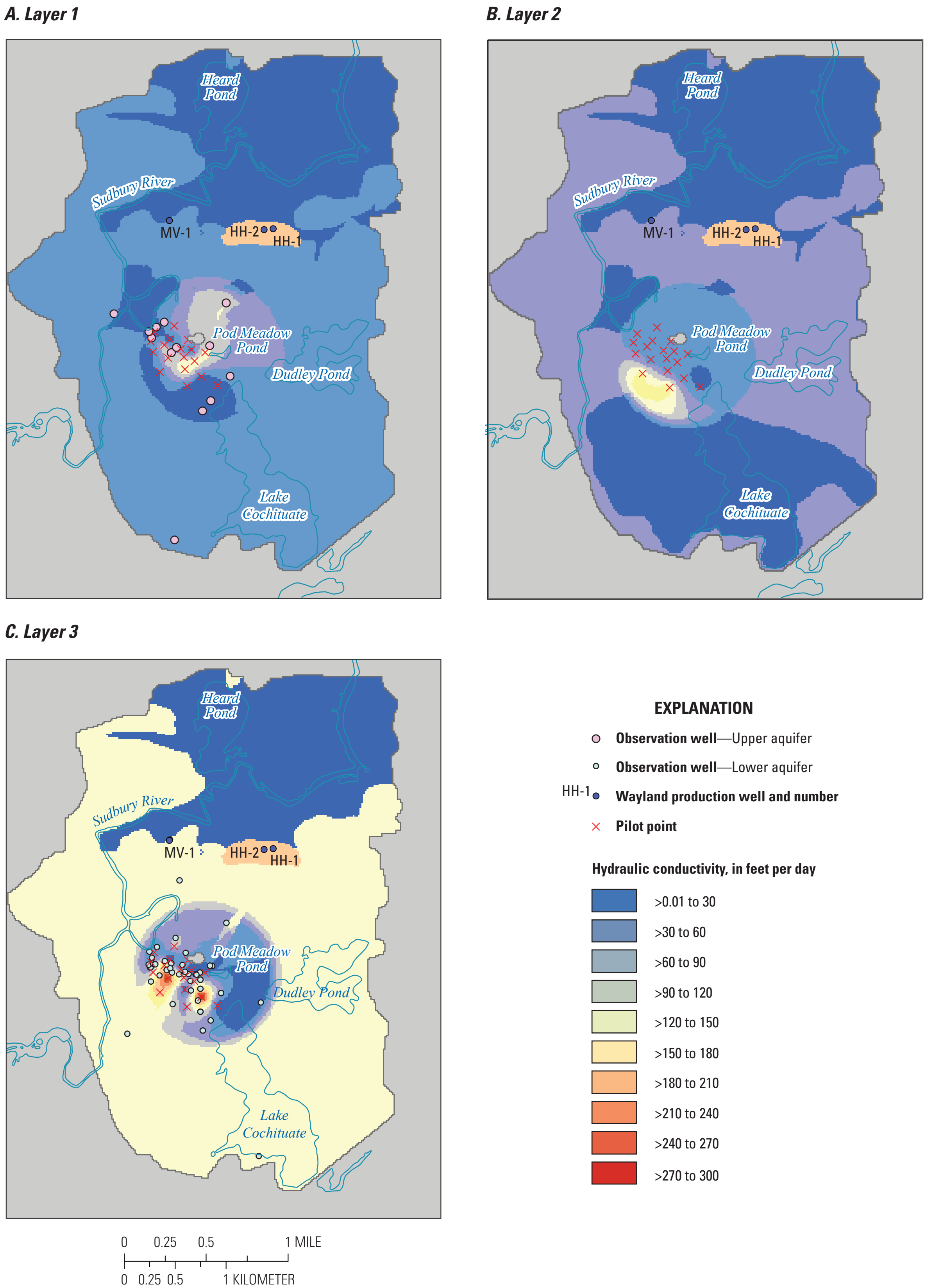


Figure 9. Horizontal hydraulic conductivity (Kh) of groundwater model for *A*, Layer 1, *B*, Layer 2, and *C*, Layer 3.

Parameterization was augmented by use of PEST software (Doherty, 2010) to minimize model error by automated adjustment of selected model parameters. Parameters calibrated using PEST include horizontal and vertical hydraulic conductivity, streambed hydraulic conductivity, lakebed conductance, specific storage, and specific yield (table 6). The initial value of each model parameter was assigned based on Phase 1 values and then adjusted by PEST. PEST repeatedly runs MODFLOW-NWT using iteratively adjusted parameter values until the objective function is satisfactorily minimized and further parameter value changes are small based on user-specified convergence criteria (Doherty and Hunt, 2010). The final optimized parameter values (tables 6 and 7) were then used to run transient model scenarios discussed later (Effects of Pumping section).

A pilot point process (Doherty and others, 2010) was used to assign Kh values within a 2,500-ft radius of the pumping wells. Instead of assigning Kh by zone or by layer, Kh was interpolated from Kh values assigned at 54 pilot points, 18 in each layer (fig. 9). Pilot point locations were selected subjectively with the goals of locating points in areas of observation wells, keeping pilot points separated from one another, and keeping the number of pilot points low enough to allow reasonable run times. Each pilot point Kh value is treated by PEST as a separate parameter to be fitted based on how changes to that pilot point Kh value affect simulated heads and streamflows. The calibrated pilot point Kh values are spatially interpolated with kriging to assign Kh values to model cells within a 2,500-ft radius of the Birch Road test wells. Model cells outside of the 2,500-ft radius were assigned Kh values equal to Phase 1 values (except in the zone around the Happy Hollow wells previously mentioned). Model cells from 2,000 to 2,500 ft from the pump test wells were assigned a distance-weighted average Kh value between the kriged

Table 6. Calibrated hydraulic parameter values in the Phase 2 model, with comparison to Phase 1 values.

[Ss, specific storage in 1/feet; E, exponent (for example, 1.5E3 means "1.5 X 10^3"); Sy, specific yield in fractional percent; hydraulic conductivity in feet per day; --, not applicable]

Parameter	Phase	Model layers		
		Upper	Middle	Lower
	Phase 1	1 and 2	3	4
	Phase 2	1	2	3
Hydraulic conductivity[1]				
Horizontal	Phase 1	53.5	51.6	139.2
	Phase 2	57.5	57.1	82.6
Vertical (expressed as anisotropy[2])	Phase 1	10.0	20.0	10.0
	Phase 2	52.2	1.0	19.2
Storage				
Ss	Phase 1	1.00E-05	1.00E-05	1.00E-05
	Phase 2	4.95E-07	4.20E-07	2.84E-06
Sy	Phase 1	0.15	0.15	0.15
	Phase 2	0.14	0.10	0.10
Lakebed leakance[3] (1/feet)				
Pod Meadow Pond		0.05	--	--
Lake Cochituate		0.20	--	--

[1]Horizontal hydraulic conductivity values are average for model cells within 2,500 feet of the Birch Road test wells.

[2]Anisotropy is the ratio of horizontal to vertical hydraulic conductivity.

[3]Lake (LAK) package used only in Phase 2 model.

Table 7. Calibrated streambed hydraulic conductivity values and comparison to Phase 1 values.

[Ks, vertical streambed hydraulic conductivity in feet per day; Ratio, Phase 2 Ks/Phase 1 Ks; Sensitivity, change in PEST objective function divided by change in model parameter value]

Segment (fig. 8)	Phase 1 Ks	Phase 2 Ks	Ratio (percent)	Sensitivity (10^{-5})
1	20	15.2	76	0.95
2	20	15.2	76	0.95
3	20	15.2	76	0.95
4	20	15.2	76	0.95
5	20	15.2	76	0.95
6	20	15.2	76	0.95
7	20	15.2	76	0.95
8	20	2.7	14	250
9	10	175.4	1,754	27
10	20	175.4	877	27
11	20	2.7	14	250
12	20	2.7	14	250
13	20	2.7	14	250
14	20	2.7	14	250
15	20	2.7	14	250
16	200	0.5	0. 25	0.84
17	200	0.5	0. 25	0.84
18	200	0.5	0. 25	0.84
19	2	3.5	177	0.84
20	2	0.5	25	0.84
21	200	0.5	0.25	0.84
Mean	52.1	22.8	164	8.2

values and Phase 1 Kh values. The pilot point process was not applied to parameters other than Kh.

PEST calibrated Kh values within 2,500 ft of the test wells indicate a complex and spatially heterogeneous aquifer system. Spatial variation of Kh created by the automated calibration process is greater and more detailed than the spatial variation of Kh in the Phase 1 model resulting from assignment by hydrogeologic zone. Within the 2,500-ft radius, calibrated Kh values in Layer 1 (upper aquifer) ranged from 0.1 to 156 ft/d with an area of high Kh near the simulated Birch Road wells and an area of very low Kh values just south of the high Kh values (fig. 9). This contrasts with the corresponding Phase 1 Kh values in the upper aquifer (Phase 1 Layer 2) that were 60 ft/d except in wetland areas where they were 10 ft/d. Layer 2 (semiconfining unit) Kh values within 2,500 ft of the test wells are 30–40 ft/d immediately around the Birch Road wells and increase to as much as 160 ft/d just to the south of the test wells. This contrasts with the corresponding Phase 1 Kh values in the semiconfining unit (Phase 1 Layer 3), which were assigned lower Kh values of 10 ft/d in wetland areas to the north of the test wells and in the area south of the test wells that may contain less permeable silts and clays and a value of 70 ft/d elsewhere. In Layer 3 (lower aquifer) of the Phase 2 model, PEST calibrated Kh values are as high as 300 ft/d to the south of the test wells but as low as 2 ft/d elsewhere. In contrast, corresponding Phase 1 Kh values in Layer 4 (lower aquifer) were a uniform 140 ft/d. The spatial variability of Kh determined by PEST within 2,500 ft of the test wells is supported by variability in the lithology of many well logs in that area. The spatial variability suggests that simplified conceptual models of aquifer geology may not be sufficiently detailed to accurately assign aquifer properties to a groundwater model of a glacial sediment aquifer with the complexity and heterogeneity seen in this study.

Hydraulic conductivity anisotropy (ratio of horizontal/vertical hydraulic conductivity), specific storage, and specific yield were calibrated using PEST but were assigned by layer rather than by cell and were reassigned everywhere in the model rather than just within 2,500 ft of the test wells (table 6). Vertical hydraulic conductivity for each cell in the Phase 2 model was determined by dividing the cell's Kh value by an anisotropy factor assigned by layer. Phase 2 model anisotropy values indicate greater variability and are higher in the upper and lower aquifer and lower in the middle semiconfining unit than Phase 1 values. Specific storage and specific yield are lower in the Phase 2 model than in the Phase 1 model. Lake-bed conductivity, a parameter not used in the Phase 1 model, was uniform for each of the two lakes and these values, 0.20 (1/day) for Lake Cochituate and 0.05 (1/day) for Pod Meadow Pond, were calibrated with PEST.

Streambed hydraulic conductivity (Ks) was recalibrated in the Phase 2 model and values varied considerably from Phase 1 values (table 7). Stream segments were grouped for calibration according to morphology, observed bed characteristics, and position relative to the Birch Road well site. For example, stream segments 1–7 (locations shown in fig. 8) are upstream from the Birch Road site, and generally have greater slopes and gravel beds, whereas stream segments 11–15 are in the Pod Meadow area downstream from the Birch Road well site and generally have lower slopes and mud beds. The greatest increases in streambed conductivity were for stream segments 9 and 10, representing the inlet to and immediate downstream drainage from Pod Meadow Pond. Model results were most sensitive to streambed conductivity of the reaches in the Pod Meadow wetlands area (segments 8 and 11 to 15). The greatest decrease in streambed conductivity was for downstream parts of the Sudbury River; however, no measurements of streambed hydraulic conductivity were available to guide parameter estimation and streambed hydraulic conductivity had low sensitivity in the model and little effect over model error.

Model Calibration

The Phase 2 groundwater model was calibrated by adjusting model input parameter values, with the use of PEST, to minimize the difference between simulated and observed water levels and stream base flow (table 8). For automated parameter estimation using PEST, the model was run with 60 stress periods, the first a steady-state stress period providing simulated values for comparison to steady-state calibration targets (water-levels and outflow from Pod Meadow Pond). After the initial steady-state stress period the model simulated 39 monthly stress periods to establish dynamic seasonal equilibrium (monthly heads and flows consistent from year to year), then 20 daily stress periods to simulate the April 26 through May 15, 2006, pumping test. All observation data including steady-state water levels, Pod Meadow Pond outflow, and transient drawdowns were used collectively in the objective function, rather than calibrating the steady-state and transient models separately. Parameters were calibrated sequentially with PEST to achieve reasonable calibration run times. The sequential calibration process was repeated until a stable suite of calibrated values was achieved that provided the desired match between simulated values and observations.

Calibration Targets and Weights

Targets for model calibration were observations seasonally adjusted average groundwater levels, outflow from Pod Meadow Pond, just downstream from the pond, and measured drawdowns during an aquifer pumping test (table 8). The average groundwater level values differed somewhat from the values used to calibrate the Phase 1 model (Eggleston and others, 2012, tables 2 and 8), because they included additional data and were adjusted to remove seasonal bias, as described earlier (Groundwater Elevations section). To calibrate aquifer storage properties, drawdown data from an aquifer pumping test (SEA Consultants, Inc., 2008) were included as calibration targets (table 9).

Each observation target is assigned a weight that is used by PEST in the parameter estimation process. Weights are useful for emphasizing some observations or groups of observations more than others and for normalizing observations of different units or magnitudes. Weights were adjusted for different PEST runs during the calibration process as different sets of parameters were calibrated, but each set of weights was assigned following similar guidelines. The first guideline was to emphasize Pod Meadow Pond outflow as an observation target relative to groundwater elevation observations, because it was the only flux observation. The second guideline was to assign roughly equal weights to three defined groups of groundwater elevation observations; groundwater level observations in Layer 1 (ssgwlvlslay1), groundwater level observations in Layer 3 (ssgwlvlslay3), and pumping test drawdown observations (pumpgwlevels). These groups of observations served different purposes in the calibration: ssgwlvlslay1 observations had the strongest control over hydraulic characteristics of the upper aquifer and interactions between groundwater and surface water, ssgwlvlslay3 observations had the strongest control over hydraulic characteristics in the lower aquifer and flow to pumping wells, whereas pumpgwlevels observations had the strongest control over aquifer storage coefficients and the timing of delays between groundwater pumping and streamflow depletion. A further rule was to adjust observation weights individually by well within observation groups ssgwlvlslay1 and ssgwlvlslay3. The individual weights were assigned to reflect confidence in the average observed groundwater elevation value and the relatively lower importance of spatially clustered observations. Relative weight of an average groundwater elevation was based on the number of observations for a given well (50 percent of variation), proximity to other observation wells (30 percent), and other sources of uncertainty such as accuracy of XYZ coordinate locations (20 percent). Representative weights used in later PEST runs are shown in table 10.

Model Uncertainty and Sensitivity Analysis

Groundwater models are inherently uncertain because they are simplifications of complex natural systems. To better understand the effects of uncertainty on the soundness of conclusions drawn from model results it helps to understand which model parameters have the largest effects on simulation outcomes. For this Phase 2 model uncertainty and sensitivity are analyzed in four separate modes, as follows:

1. PEST determined parameter sensitivities,
2. numerical instabilities,
3. model error, and
4. hydraulic response to changes in hydraulic parameter values.

PEST Parameter Sensitivities

Sensitivity to model parameters is calculated by PEST as part of the optimization process. Any given parameter sensitivity is measured as the weighted change in the objective function divided by the change in that model parameter value. Doherty and Hunt (2010) provides detailed descriptions of the calculation and meaning of model parameter sensitivities.

PEST run times were prohibitively long when all parameters in the Phase 2 model were calibrated simultaneously and in some cases when particular parameter value sets caused difficulty for MODFLOW-NWT to reach a solution. To avoid such problems, parameters were calibrated sequentially with PEST; sensitive parameters were calibrated first, whereas less sensitive parameters were fixed at likely values. Then the more sensitive parameters were fixed at calibrated values, whereas less sensitive parameters were calibrated. This process was iterated until parameter values indicated little change between iterations and simulation errors fell below target maximums. Some parameters were grouped for PEST calibration to reduce PEST run times. Storage parameters (Ss and Sy) and vertical hydraulic conductivity were assigned by layer and stream segments were lumped into five groups representing different reaches for determination of streambed conductivity.

Grouping like parameters together, the parameter groups in order of greatest sensitivity were specific storage, specific yield, streambed hydraulic conductivity, pilot point horizontal hydraulic conductivity, aquifer vertical hydraulic conductivity, and lakebed conductance. Sensitivity values shown (table 11) are for one of the later PEST runs, prun23, with sensitivities of streambed hydraulic conductivity included from prun26 because streambed hydraulic conductivity was not an adjustable parameter in prun23.

Numerical Instabilities

In some cases the calibrated Phase 2 model did not converge if groundwater withdrawals were set too high or if recharge or aquifer hydraulic conductivity were set too low. In these cases the model convergence problems are usually caused by model cells that contain pumping wells going drying and rewetting from one solver iteration to the next, creating a numerical instability. MODFLOW-NWT automatically reduces pumping rates to prevent cells from drying, which can cause simulated pumping rates to be lower than the target rates.

Alternative pumping rates and locations were therefore used to help the model converge during assessment of model sensitivities. Hypothetical locations of future supply wells in the Birch Road area were varied from actual locations to improve model convergence. By trial and error it was determined that moving the model location of Test Well 3 away from the other test wells into an area of greater aquifer transmissivity improved model stability. The best location is labeled “alternate #3” on the figure 8 inset map. Rates of pumping at four hypothetical supply wells also were varied to provide additional stability in the model under drought

Table 8. Average observed and simulated groundwater levels and flow just downstream from the Pod Meadow Pond outlet for the calibrated groundwater model of an aquifer in east-central Massachusetts.

[ft^3/d, cubic feet per day; ft^3/s, cubic feet per second; %, percent; Average groundwater level applies to observed only; feet MSL, feet below mean sea level]

Site	Observed	Simulated	Error = simulated – observed
Stream base flow (ft^3/d)			
Pod Meadow Pond outflow	57,024 (0.660 ft^3/s)	57,395 (0.6643 ft^3/s)	371 (0.7%)
Average groundwater level in wells (feet MSL)			
F1W–64	124.2	120.4	-3.8
F1W–84	138.7	134.9	-3.8
F1W–92	126.3	128.7	2.4
WKW–119	131.9	130.8	-1.1
MW–10	120.0	120.5	0.6
MW–12	117.0	118.4	1.4
MW–15	126.2	125.1	-1.0
MW–16	129.4	124.1	-5.3
MW–8	119.4	117.0	-2.5
MW–9	116.7	117.4	0.7
USGS–F1S	122.1	122.8	0.7
USGS–F2	140.4	135.9	-4.5
USGS–F5S	117.4	120.5	3.2
1–90	124.6	126.4	1.8
2–90	124.7	126.3	1.6
3–90	126.4	126.2	-0.2
5–90	126.8	126.0	-0.8
7–90	119.1	120.6	1.4
8–90	126.4	128.0	1.7
F1W–74	137.4	136.5	-0.9
F1W–88	126.5	130.7	4.2
F1W–89	126.6	130.7	4.1
F1W–90	132.1	133.4	1.3
F1W–91	133.0	133.4	0.5
F1W–93	126.2	128.9	2.7
F1W–94	126.2	128.9	2.7
WKW–123	138.0	135.9	-2.1
WKW–030	118.0	119.1	1.0
WKW–052	117.2	115.8	-1.5
MW–01	126.1	125.1	-1.0
MW–11	123.5	121.2	-2.3

Table 8. Average observed and simulated groundwater levels and flow just downstream from the Pod Meadow Pond outlet for the calibrated groundwater model of an aquifer in east-central Massachusetts.—Continued

[ft^3/yr, cubic feet per year; ft^3/s, cubic feet per second; %, percent; Average groundwater level applies to observed only; MSL, mean sea level]

Site	Observed	Simulated	Error = simulated – observed
Average groundwater level in wells (feet MSL)—Continued			
MW–13	126.3	124.3	-2.0
MW–14	125.8	124.2	-1.6
MW–14R	126.1	124.2	-1.9
MW–15D	126.5	124.6	-1.9
MW–02	125.5	122.7	-2.8
MW–2B	125.8	122.7	-3.2
MW–2D	125.3	122.7	-2.6
MW–03	124.0	121.9	-2.0
MW–04	123.1	122.5	-0.5
MW–05	124.7	122.5	-2.2
MW–5D	125.4	122.5	-2.9
MW–06	131.4	125.2	-6.2
MW–07	129.9	122.6	-7.3
MW–8D	120.6	119.0	-1.6
MW–9D	118.2	118.6	0.5
SEA–10	127.1	129.5	2.5
SEA–11	128.7	128.1	-0.6
SEA–12	121.0	123.6	2.7
SEA–13	127.0	124.1	-2.8
SEA–14	121.0	124.3	3.3
SEA–15	116.8	118.4	1.6
SEA–16	130.8	133.0	2.2
SEA–17	127.0	125.6	-1.4
SEA–18	126.8	127.7	0.9
SEA–02	124.7	125.9	1.2
SEA–03	121.5	122.8	1.4
SEA–04	126.7	126.6	-0.1
SEA–07	127.0	125.7	-1.3
SEA–08	126.1	126.2	0.2
USGS–F1RD	121.8	122.9	1.1
USGS–F4D	127.1	125.1	-2.0
USGS–F5D	117.2	121.1	3.9
Mean	125.5	125.1	-0.4

Table 9. Observed and simulated maximum groundwater-level drawdown during 2006 aquifer pumping test, Phase 2 groundwater model of the aquifer in east-central Massachusetts.

Observation well	Groundwater-level drawdown at end of test (feet)		
	Observed	Simulated	Error = simulated – observed
7–90	1.3	2.4	1.2
MW–1	5.9	4.3	-1.7
SEA–10	6.1	3.9	-2.2
SEA–11	2.9	3.1	0.1
SEA–15	0.4	1.0	0.6
SEA–3	0.9	1.7	0.8
Average	2.9	2.7	-0.2

Table 10. Calibration target groups used in automated parameter estimation, with representative assigned weights by group.

[--, not applicable]

Calibration target group	Description	Number of observations	Observation weights	
			Average	Range
streamflow	Pod Meadow Pond outflow, steady state (cubic feet per day)	1	0.00004	--
ssgwlvlslay1	Groundwater level observation, steady-state, Layer 1 (feet)	14	0.02786	0.0166–0.333
ssgwlvlslay3	Groundwater level observation, steady-state, Layer 3 (feet)	50	0.01894	0.0096–0.032
pumpgwlevels	Observed groundwater level drawdown 2006 pumping test (feet)	30	0.12100	0.121–0.121

conditions and higher pumping rates, unlike in the Phase 1 model where all four wells were assigned equal pumping rates (Eggleston and others, 2012). Phase 2 simulations using a variety of hypothetical well locations and pumping rates indicated that 3.17 Mgal/d of total pumping is close to the maximum possible steady-state withdrawal from four wells in the Birch Road area. Model cells are less likely to go dry if (1) pumping wells are spaced farther apart, (2) pumping wells are placed away from the bedrock island (fig. 8), and (3) pumping rates are higher at well locations having relatively greater transmissivity. Any siting of future supply wells should take these findings into account so that greater flexibility in pumping rates and schedules is achieved.

One set of well locations was determined (locations #1, #2, #4, and alternate #3 in fig. 8) that allowed a steady-state pumping rate of 3.17 Mgal/d. Note that the supply wells are unlikely to be operated at this full maximum pumping rate on a steady-state or constant basis, which could affect site selection for future groundwater supply wells. Alternative site analysis was not exhaustive and, with realistic pumping schedules, other well configurations and pumping rates would likely yield 3.17 Mgal/d. In addition, the drilling of wells for the pumping tests in 2006 (SEA Consultants, Inc., 2008) indicated that the aquifer contains localized pockets of cobbles and boulders with exceptionally high hydraulic conductivity. Although there is insufficient field data to accurately support inclusion of these localized high-Kh deposits in the groundwater model, locating wells in such deposits would reduce well drawdown during pumping that could yield pumping rates greater than that simulated.

Model Error

Simulated steady-state heads show good agreement with observed heads (figs. 10 and 11). Errors are lower for the Phase 2 model that for the Phase 1 model. Phase 2 model simulated steady-state water levels have an average error of -0.4 ft (table 12), average absolute error of 2.1 ft, and range of -7.3 to 4.2 ft. Simulated maximum drawdowns in six wells during the 2006 aquifer pumping test have a mean error of -0.3 ft (table 12). Simulated Pod Meadow Pond outflow in the Phase 2 model matched the observed outflow (table 12). All of these error measures improve on Phase 1 model calibration results.

The Phase 2 model, like the Phase 1 model, provides an acceptable spatial representation of average groundwater levels, average Pod Meadow Pond outflow, and pumping test drawdowns. The Phase 2 model, however, provides better representation of the spatial heterogeneity in aquifer hydraulic conductivity and more realistic simulation of lake boundary effects, which improved the model calibration and stability compared to the Phase 1 model. Overall, the Phase 2 model better reflects the heterogeneity of the aquifer and provides an improved basis for comparing model results and uncertainties.

Effect of Parameter Values on Streamflow Timing and on Induced Recharge from Lake Cochituate

Hydraulic parameters in the model affect the timing of streamflow response to pumping and rates of recharge from Lake Cochituate to the aquifer induced by pumping. The sensitivity of these hydraulic responses was tested by individually varying a selection of model parameters and calculating the resulting change in the timing of streamflow response and in the rate of induced recharge from Lake Cochituate. Model hydraulic parameters were varied for steady-state simulations to determine sensitivity of induced recharge and for transient simulations to determine sensitivity of streamflow response times (table 13). Hydraulic parameters varied included Ks, Kh, hydraulic conductivity anisotropy, lakebed leakance, specific yield, and specific storage. For simplicity, the model parameter changes were made uniformly throughout the model by factors of 50 and 200 percent.

Sensitivity of induced recharge to parameter values was determined for the parameters listed in table 13 using steady-state simulations. Induced recharge is the simulated difference in net recharge from Lake Cochituate to the aquifer, caused by pumping at the Birch Road wells, as was done for the Phase 1 model sensitivity tests (Eggleston and others, 2012). A steady-state pumping rate of 423,459 ft^3/d (equal to 3.17 Mgal/d or 4.9 ft^3/s) was simulated at the existing well locations (#1 through #4 in the inset map on fig. 8), to evaluate induced recharge from Lake Cochituate. Variation of induced recharge caused by parameter value changes was calculated relative to the calibrated model induced recharge rate of 152,807 ft^3/d (SsCalb102 in table 13). Induced recharge from Lake Cochituate was equal to 36 percent of the simulated pumping, as compared to 32 percent in the Phase 1 model. Results (table 13) show that induced recharge from Lake Cochituate is most sensitive to Kh, which decreased induced recharge by 19.8 percent when Kh was decreased by 50 percent and increased induced recharge by 9.1 percent when Kh was doubled (SsCalb105 and SsCalb106). Independently halving and doubling streambed hydraulic conductivity (SsCalb103 and SsCalb104) and anisotropy (SsCalb107 and SsCalb108) had little effect on induced lake recharge, and lakebed leakance (SsCalb113 and SsCalb114) caused only minor changes in induced recharge from Lake Cochituate (changes ranged from -3.3 to +1.8 percent). Lakebed leakance has little effect on rates of flow between lake and aquifer induced by pumping, increased lakebed leakance slightly decreases the induced recharge rate. This somewhat unexpected result is likely due to lakebed conductance affecting not only rates of flow from the lake but also rates of flow into the lake. Under conditions of no groundwater pumping, Lake Cochituate has a simulated net groundwater inflow, while under conditions of 3.17 Mgal/d continuous pumping at the Birch Road site, Lake Cochituate has simulated net groundwater outflow. In the scenario with higher lakebed leakance (SsCalb114), discharge from the aquifer to the lake and recharge from the lake to the aquifer both increase, but the net groundwater inflow to the lake increased more. The sensitivity of lake recharge to Kh indicates that the complexity of aquifer sediment textures, reflected in the well log information, has large potential to affect model accuracy and uncertainty. In addition, the variable lake-bottom sediments indicated in the marine seismic survey also affects the lake-aquifer interaction.

The sensitivity of surface-water response time was calculated using monthly average transient model simulations (TransCalb210–TransCalb222). Response time was defined as the number of days for 50 percent of streamflow depletion (t_{50}) to take place following 1 month of pumping at 2.3 ft^3/s (1.5 Mgal/d), distributed equally among the existing Birch Road wells (#1 through #4; inset map on fig. 8). A lower rate of pumping, 2.1 ft^3/s (1.3 Mgal/d), was imposed for scenario TransCalb213 to enable model convergence. The transient model was run for 5 years (60 months) with no pumping, except in January of year 4 (month 37), which simulated pumping of 2.3 ft^3/s for 1 month and then shut off for the remainder of the 5-year simulation. The transient simulations allowed the models with different parameter values to stabilize during the first 4 years then continued through month 60 to ensure streamflow depletion, measured at the Sudbury River exit from the model, was fully accounted for. Depletion was calculated by running each scenario with and without pumping and subtracting the streamflow simulated by the two runs. Because the transient simulations had monthly stress periods, linear interpolation was used to interpolate between monthly responses and estimate the number of days to reach the t_{50} target. For example, if total streamflow depletion at the end of pumping in January was 40 percent, and at the end of February was 60 percent t_{50} was calculated to take place halfway through February, 45 days (31 +14 days) after pumping started. Some variation in t_{50} could be expected if pumping were simulated in a different month, other than January of year 4 in which groundwater recharge is relatively high. For example, if pumping were simulated in August when recharge is at an annual low, then t_{50} would be longer; however, all simulations are relative to the same pumping period (January of year 4) to assess model sensitivities.

Timing of streamflow response was most sensitive to specific yield (Sy). Halving and doubling Sy decreased t_{50} by 32 percent and increased t_{50} by 74 percent, respectively. The parameters Kh and anisotropy also affected the timing of streamflow response. Halving and doubling Kh increased t_{50}

Table 11. Parameter sensitivity values determined by PEST for the calibrated Phase 2 groundwater model of a shallow aquifer in east-central Massachusetts.

[E, exponent (for example, 1.5E3 means "1.5 X 10^3"); +, plus]

Parameter group		Parameter		
Name	Relative sensitivity	Name	Description	Sensitivity
Specific storage	1.0E+00	sslay1	Layer 1	6.2E-01
		sslay2	Layer2	4.2E-02
		sslay3	Layer 3	9.2E-01
Specific yield	1.4E-02	sylay1	Layer 1	1.7E-02
		sylay2	Layer2	2.6E-04
		sylay3	Layer 3	5.0E-03
Streambed hydraulic conductivity	1.1E-02	strcon1	Segments 1–7	9.2E-05
		strcon2	Segments 8, 11–15	1.4E-02
		strcon3	Segments 9–10	7.4E-05
		strcon4	Segments 16–18, 20–21	2.3E-03
		strcon5	Segments 19	9.3E-04
Horizontal hydraulic conductivity pilot points	1.8E-04	ptl1kx1	Layer 1, pilot point 1	7.6E-06
		ptl1kx2	Layer 1, pilot point 2	4.8E-06
		ptl1kx4	Layer 1, pilot point 3	5.0E-06
		ptl1kx5	Layer 1, pilot point 4	5.1E-06
		ptl1kx6	Layer 1, pilot point 5	7.4E-06
		ptl1kx7	Layer 1, pilot point 6	1.1E-05
		ptl1kx8	Layer 1, pilot point 7	4.9E-06
		ptl1kx9	Layer 1, pilot point 8	4.9E-06
		ptl1kx10	Layer 1, pilot point 9	4.9E-06
		ptl1kx11	Layer 1, pilot point 10	4.9E-06
		ptl1kx13	Layer 1, pilot point 11	4.8E-06
		ptl1kx14	Layer 1, pilot point 12	4.8E-06
		ptl1kx15	Layer 1, pilot point 13	4.9E-06
		ptl1kx21	Layer 1, pilot point 14	4.9E-06
		ptl1kx22	Layer 1, pilot point 15	5.5E-06
		ptl1kx24	Layer 1, pilot point 16	5.1E-06
		ptl1kx30	Layer 1, pilot point 17	5.3E-06
		ptl1kx31	Layer 1, pilot point 18	4.8E-06
		ptl2kx1	Layer 2, pilot point 1	4.8E-06
		ptl2kx2	Layer 2, pilot point 2	4.8E-06
		ptl2kx4	Layer 2, pilot point 3	5.0E-06
		ptl2kx5	Layer 2, pilot point 4	4.8E-06
		ptl2kx6	Layer 2, pilot point 5	4.8E-06
		ptl2kx7	Layer 2, pilot point 6	4.8E-06
		ptl2kx8	Layer 2, pilot point 7	4.8E-06
		ptl2kx9	Layer 2, pilot point 8	4.8E-06
		ptl2kx10	Layer 2, pilot point 9	4.8E-06
		ptl2kx11	Layer 2, pilot point 10	5.0E-06

Table 11. Parameter sensitivity values determined by PEST for the calibrated Phase 2 groundwater model of a shallow aquifer in east-central Massachusetts.—Continued

[E, power of 10 exponent; +, plus]

Parameter group		Parameter		
Name	**Relative sensitivity**	**Name**	**Description**	**Sensitivity**
Horizontal hydraulic conductivity pilot points—Continued	1.8E-04	ptl2kx13	Layer 2, pilot point 11	4.8E-06
		ptl2kx14	Layer 2, pilot point 12	4.8E-06
		ptl2kx15	Layer 2, pilot point 13	4.8E-06
		ptl2kx21	Layer 2, pilot point 14	4.8E-06
		ptl2kx22	Layer 2, pilot point 15	4.8E-06
		ptl2kx24	Layer 2, pilot point 16	4.8E-06
		ptl2kx30	Layer 2, pilot point 17	4.8E-06
		ptl2kx31	Layer 2, pilot point 18	4.8E-06
		ptl3kx1	Layer 3, pilot point 1	6.2E-06
		ptl3kx2	Layer 3, pilot point 2	6.7E-06
		ptl3kx4	Layer 3, pilot point 3	6.8E-06
		ptl3kx5	Layer 3, pilot point 4	6.7E-06
		ptl3kx6	Layer 3, pilot point 5	5.8E-06
		ptl3kx7	Layer 3, pilot point 6	5.8E-06
		ptl3kx8	Layer 3, pilot point 7	6.4E-06
		ptl3kx9	Layer 3, pilot point 8	5.4E-06
		ptl3kx10	Layer 3, pilot point 9	5.2E-06
		ptl3kx11	Layer 3, pilot point 10	5.4E-06
		ptl3kx13	Layer 3, pilot point 11	5.2E-06
		ptl3kx14	Layer 3, pilot point 12	5.0E-06
		ptl3kx15	Layer 3, pilot point 13	5.5E-06
		ptl3kx21	Layer 3, pilot point 14	5.0E-06
		ptl3kx22	Layer 3, pilot point 15	5.2E-06
		ptl3kx24	Layer 3, pilot point 16	5.3E-06
		ptl3kx30	Layer 3, pilot point 17	5.3E-06
		ptl3kx31	Layer 3, pilot point 18	5.3E-06
Vertical hydraulic conductivity	3.2E-05	vklay1	Layer 1	2.6E-05
		vklay2	Layer2	1.4E-05
		vklay3	Layer 3	9.7E-06
Lakebed conductance	1.4E-16	lakek_coch	Lake Cochituate	2.2E-16
		lakek_pmp	Pod Meadow Pond	1.1E-18

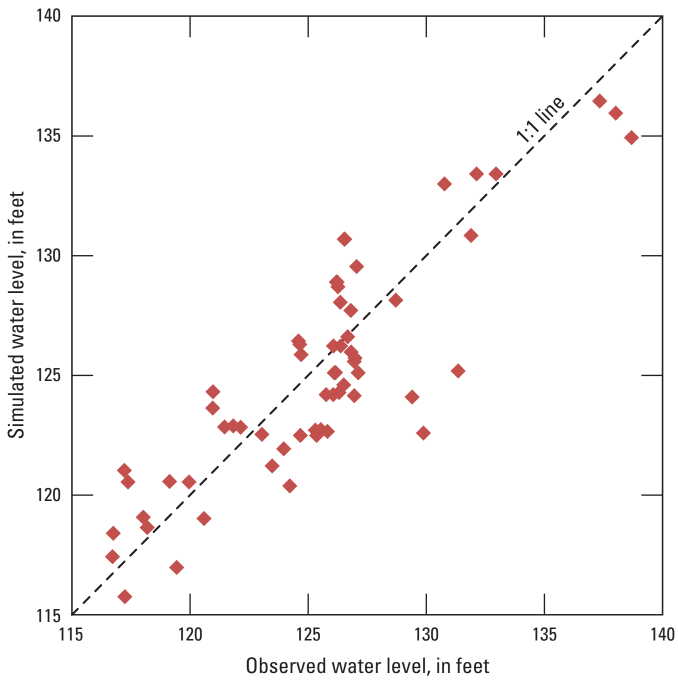


Figure 10. Observed water level compared to simulated steady-state groundwater levels (heads) for the calibrated Phase 2 groundwater model.

by 58 percent and decreased t_{50} by 29 percent, respectively. Halving and doubling anisotropy increased t_{50} by 28 percent and decreased t_{50} by 14 percent, respectively. Small changes in t_{50} take place when streambed hydraulic conductivity (1.3 to 4.5 percent) or lakebed leakance (-4.4 to 7.7 percent) are varied. Varying specific storage has no effect on t_{50}.

Induced recharge from Lake Cochituate was most sensitive to Kh of the aquifer, reducing Kh by one-half caused a 38-percent decrease in induced recharge. Streamflow response time was most sensitive to changes in Sy. Doubling Sy caused t_{50} to increase by 23 days, whereas a 50-percent decrease in Sy caused t_{50} to decrease by 10 days. The t_{50} streamflow response time of the Phase 2 model (31 days) was nearly four times greater than that of the Phase 1 model (8.3 days).

Model Limitations

The Phase 2 groundwater model provides some notable changes and improvements over the Phase 1 model but, like all models, representation of the aquifer system is still imperfect. As such, users should be aware of the model limitations. The model simulates only parts of Lake Cochituate's hydrology, most importantly the exchange of lake water with the aquifer and the effects of groundwater pumping on this exchange. The groundwater model does not simulate the surface runoff that strongly affects Lake Cochituate's water budget and so, by itself, cannot accurately simulate lake levels. Any changes simulated are therefore best viewed as a relative change compared to other conditions simulated by the groundwater model. Similarly, the groundwater model simulates stream base flow but does not simulate surface runoff and so cannot accurately simulate streamflow. Therefore simulated changes in streamflow also should be viewed as a relative change compared to other conditions simulated by the groundwater model.

The model has limitations in its ability to simulate the exchange of water between the lakes and the aquifer. Although the new marine seismic data improve understanding of where Lake Cochituate's bed is covered with gyttja, the low hydraulic conductivity mud that likely reduces flow, the hydraulic conductivity of Lake Cochituate bed sediments is still not well known. Simulated net recharge of groundwater from Lake Cochituate to the aquifer is not sensitive to the lakebed leakance parameter because leakance simultaneously controls groundwater discharge to the lake and groundwater recharge from the lake. Limitations in the model's ability to simulate aquifer-lake interaction also apply to Pod Meadow and Pod Meadow Pond. Although the Pod Meadow Pond stream outflow is accurately simulated under existing (no Birch Road well pumping) conditions, it is not known if simulations of the pond under pumping conditions will be accurate and whether the model can accurately answer questions such as what rates of pumping might cause Pod Meadow Pond to go dry.

Using PEST to automatically calibrate hydraulic conductivity and other key parameter values reduced model error; however, model error still exists because of the many unknown properties of the aquifer and the limitations of the head and flow data available for model calibration. Few groundwater elevation measurements are available outside the 2,500-ft radius around the Birch Road well site, so hydraulic conductivity and storativity are likely not well calibrated there. The resulting uncertainties restrict the applicability of detailed head simulations. Outflow from Pod Meadow Pond and groundwater fluxes in the Pod Meadow wetlands also are affected by parameter uncertainty.

Other important considerations are the starting conditions and the time-steps of the model. Stresses of greatest concern often take place at a shorter time scale than a month, such as the 7-day low flow. Likewise, pumping schedules in the model were simulated on a monthly basis. The Phase 2 model as described in this report is not designed to simulate finer temporal or spatial resolutions that may be important to future day-to-day operation and management of withdrawals.

Effects of Pumping

After calibration, the Phase 2 groundwater flow model was used to simulate the effects of pumping the Birch Road wells on groundwater flow and streamflow. Pumping scenarios (table 14) are based on design criteria for the proposed Birch Road wells that varied pumping rates as much as 4.9 ft^3/s (3.17 Mgal/d) (Peter Newton, Bristol, written commun., December 2014). All scenarios simulated the Wayland wells pumping at their average reported annual or monthly

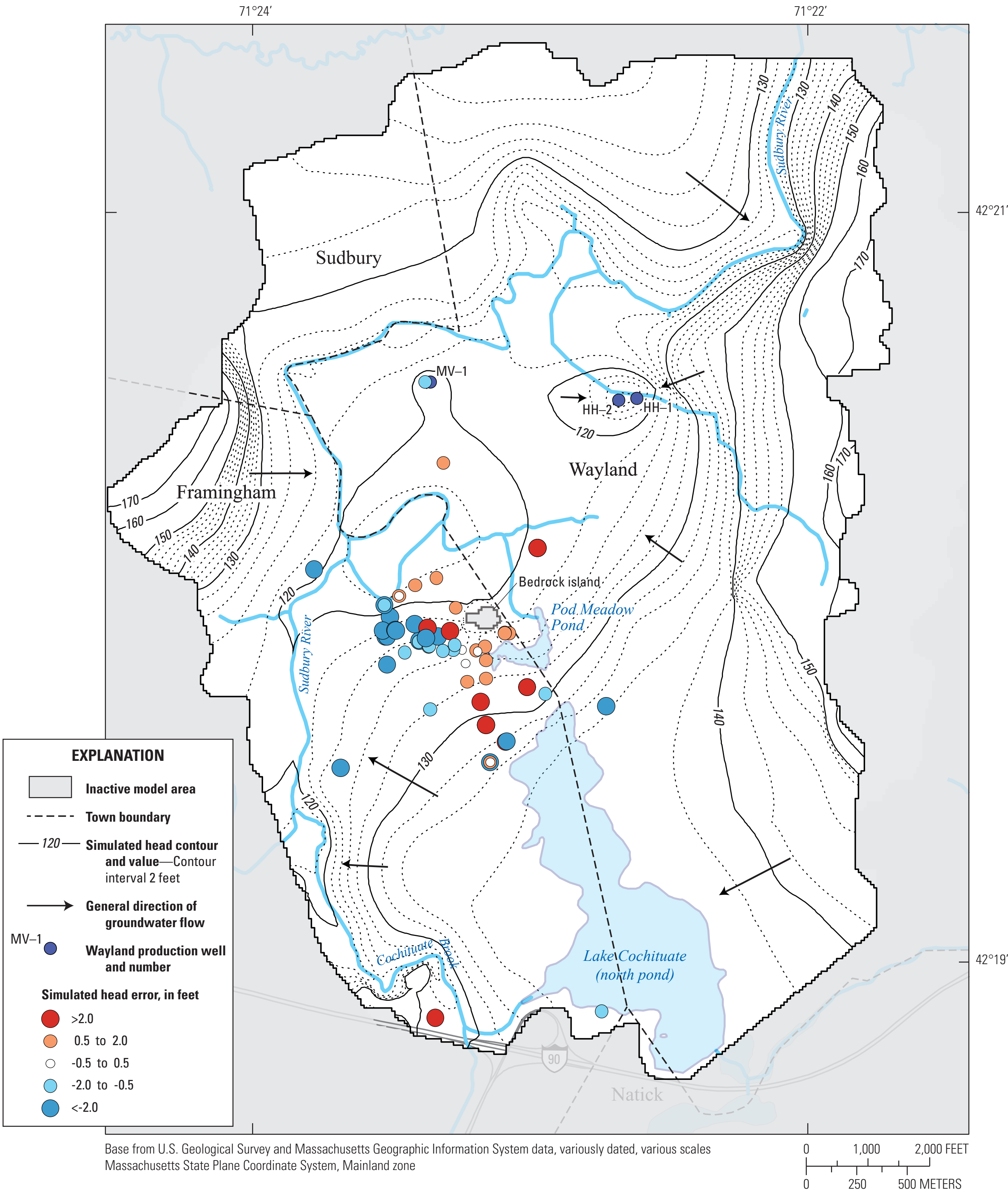


Figure 11. Simulated groundwater levels (heads) under existing conditions in model Layer 3 and error in simulated heads (observed minus simulated) in model Layers 1 and 3 for the study area in east-central Massachusetts.

Table 12. Summary of model errors.

[ft^3/s, cubic feet per second; ft, feet; Error = simulated – observed]

Calibration targets—average error	Observed	Simulated	Error
Pod Meadow Pond outflow (ft^3/s)	0.66	0.66	0.00
Mean steady-state heads[1] (ft)	125.5	125.1	-0.4
Mean pumping test drawdown[2] (ft)	2.9	2.6	-0.3

[1]Mean of all observed and simulated water levels used in Phase 1 and Phase 2 steady-state model calibrations. Note the number and the observed wells used differs slightly.

[2] Mean maximum drawdown at six observations simulated in Phase 1 and Phase 2 models (table 9 of this report).

withdrawal rates. References to scenarios with or without pumping only refer to pumping at the Birch Road wells. Notable changes from the Phase 1 model scenarios include (1) constant pumping throughout the year was reduced slightly (from 4.9 to 3.9 ft^3/s) during July through September in order for the model to converge to a solution, (2) more conservative drought conditions were imposed, and (3) an additional set of scenarios simulated 10 percent return flow from the pumped wells to Pod Meadow Pond.

Steady State Simulations

Water balances for steady-state simulations of existing conditions (SsCalb101) indicate that without pumping at the Birch Road wells, 9.4 ft^3/s of the total 10.4 ft^3/s inflow to the aquifer is from areal recharge from surface precipitation, whereas the remaining 1.0 ft^3/s is recharge from streams and lakes (table 15). With hypothetical pumping rates of 4.9 ft^3/s (3.17 Mgal/d) from the Birch Road wells (SsCalb102), the same amount of inflow recharges from surface precipitation (9.43 ft^3/s), but an additional 1.81 ft^3/s of recharge is induced from streams and lakes and groundwater discharge to streams and lakes decreases by 3.11 ft^3/s (table 15). Net induced recharge from Lake Cochituate caused by pumping, equal to inflow change minus outflow change, is 36.1 percent of the pumping (1.8 of 4.9 ft^3/s). The Phase 2 model water balance is similar to the Phase 1 model water balance, but in the Phase 2 model lake-aquifer exchange is explicitly simulated.

An estimate of how future pumping at the Birch Roads wells may affect Lake Cochituate water levels can be made by reconsidering surface-water modeling results from Zarriello and others (2010), in light of the groundwater modeling results presented here. The 2010 study used HSPF software (Bicknell and others, 2000) to simulate surface water in the Sudbury-Assabet basin and the reaction of Lake Cochituate to hypothetical pumping at a constant rate of 6.65 ft^3/s (4.3 Mgal/d) at the Birch Road well site. Four surface-water modeling scenarios were simulated assuming different proportions of water to the Birch Road wells came from induced infiltration from Lake Cochituate (table 14 and fig. 33 of Zarriello and others, 2010). The 3rd scenario (BIR-LC33), which assumes that one-third of the water (to hypothetical Birch Road supply wells) is from Lake Cochituate and two-thirds is from the Sudbury River, is the scenario that most closely matches the 36.1 percent induced recharge indicated by the steady-state groundwater modeling results. The HSPF scenarios simulated the 44-year period 1961–2004, using an hourly time step and average water use conditions for 1993–2003. The HSPF model, unlike the MODFLOW-NWT model presented in this study, performs detailed accounting of surface-water inflows and lake outflows (surface outflow, evaporation). The HSPF simulation results indicate that under these conditions (BIR-LC33 scenario) Lake Cochituate water elevations drop below the Cochituate Brook outfall elevation 32.3 days per year as compared to 17.0 days per year under existing conditions without Birch Road pumping (AVGWU scenario).

The number of days that Lake Cochituate water elevations are below the Cochituate Brook outfall simulated with the HSPF model would be less if the pumping rate decreased (4.9 compared to 6.65 ft^3/s), or ceased, or some combination thereof during the summer months. The MODFLOW-NWT groundwater flow model developed in this study could be used to refine the lake-groundwater interaction for application in the HSPF surface-water model and simulate the effect of alternate pumping schedules that would likely be used to maintain desired streamflows during the normal low-flow period. Coupling of the MODFLOW-NWT groundwater model to the existing HSPF surface-water model was beyond the scope of this study but a future study could address these questions.

Simulation scenarios SsCalb101 and SsCalb102 also indicate that hypothetical steady-state pumping at the Birch Road well site also decrease flows and water levels in Pod Meadow Pond, the downstream Pod Meadow wetland, and the Sudbury River. The question of how pumping schedules could be managed to reduce such surface-water effects during low-flow periods, is addressed using the transient flow model.

Table 13. Sensitivity of pumping-induced recharge from Lake Cochituate to the aquifer and of streamflow response time to select steady-state and transient model parameters.

[ft^3/d, cubic feet per day; T_{50}, time for 50 percent change in streamflow depletion; --, not computed]

Parameter	Parameter change factor (percent)	Induced recharge rate (steady-state)				Streamflow response time (transient)			
		Model run	Volume (ft^3/d)	Change relative to calibrated (ft^3/d)	Percent change	Model run	T_{50} (days)	Change relative to calibrated (days)	Percent change
Calibrated model									
	--	SsCalb102	152,807	--	--	TransCalb210	31.5	--	--
Sensitivity runs									
Streambed hydraulic conductivity (ft/d)	50	SsCalb103	153,238	431	0.3	TransCalb211	31.9	0.4	1.3
	200	SsCalb104	152,320	-487	-0.3	TransCalb212	32.4	0.9	3.0
Horizontal hydraulic conductivity (ft/d)	50	[1]SsCalb105	122,534	-30,273	-19.8	[2]TransCalb213	49.8	18.4	58
	200	SsCalb106	166,721	13,914	9.1	TransCalb214	22.5	-9.0	-29
Anisotropy—horizontal/vertical conductivity	50	SsCalb107	151,818	-989	-0.6	TransCalb215	40.2	8.7	28
	200	SsCalb108	153,261	454	0.3	TransCalb216	27.2	-4.3	-14
Specific yield (unitless)	50	--	--	--	--	TransCalb217	21.5	-10.0	-32
	200	--	--	--	--	TransCalb218	54.7	23.3	74
Specific storage (1/ft)	50	--	--	--	--	TransCalb219	31.5	0.0	0.0
	200	--	--	--	--	TransCalb220	31.5	0.0	0.0
Lakebed leakance (1/d)	50	SsCalb113	153,900	1,093	0.7	TransCalb221	33.9	2.4	7.7
	200	SsCalb114	150,944	-1,863	-1.2	TransCalb222	29.8	-1.7	-5.3

[1]Pumping reduced 50 percent to permit model convergence, rates then rescaled 2X for comparison.

[2]One month of pumping at 90,000 ft^3/d, as compared to 100,000 ft^3/d for the other transient scenarios.

Table 14. Groundwater model scenarios used to evaluate the aquifer and streamflow response to hypothetical withdrawals at the Birch Road wells, east-central Massachusetts.

[Yellow shaded cells highlight months of maximum pumping. Highlighted cells indicate pumping at 4.9 cubic feet per second (ft^3/s) (3.17 million gallons per day [Mgal/d]) or 3.9 ft^3/s (2.50 Mgal/d)]

Scenario	Hypothetical pumping of Birch Road wells (cubic feet per second)											
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Steady-state model, average recharge conditions												
SsCalb101	0											
SsCalb102	4.9											
Transient model, average recharge conditions												
TransCalb400	0	0	0	0	0	0	0	0	0	0	0	0
TransCalb401	4.9	0	0	0	0	0	0	0	0	0	0	0
TransCalb402	4.9	4.9	4.9	4.9	4.9	4.9	3.9	3.9	3.9	4.9	4.9	4.9
TransCalb403	4.9	4.9	4.9	4.9	4.9	4.9	0.1	0.1	0.1	4.9	4.9	4.9
TransCalb404	4.9	4.9	4.9	0.1	0.1	0.1	0.1	0.1	0.1	4.9	4.9	4.9
Transient model, dry (low recharge) conditions												
TransCalb410a	0	0	0	0	0	0	0	0	0	0	0	0
TransCalb411a	4.9	4.9	4.9	4.9	4.9	4.9	3.9	3.9	3.9	4.9	4.9	4.9
TransCalb412a	4.9	4.9	4.9	4.9	4.9	4.9	0.1	0.1	0.1	4.9	4.9	4.9
TransCalb413a	4.9	4.9	4.9	0.1	0.1	0.1	0.1	0.1	0.1	4.9	4.9	4.9
Transient model, dry (low recharge) conditions, 10 percent return flow[1]												
TransCalb411b	4.9	4.9	4.9	4.9	4.9	4.9	3.9	3.9	3.9	4.9	4.9	4.9
TransCalb412b	4.9	4.9	4.9	4.9	4.9	4.9	0.1	0.1	0.1	4.9	4.9	4.9
TransCalb413b	4.9	4.9	4.9	0.1	0.1	0.1	0.1	0.1	0.1	4.9	4.9	4.9

[1]Scenarios return 10 percent of groundwater withdrawals at the Birch Road wells to Pod Meadow Pond.

Table 15. Steady-state simulated water budgets with and without hypothetical Birch Road well withdrawals at a continuous rate of 4.9 ft^3/s, east-central Massachusetts.

[ft^3/s, cubic feet per second]

Flow component	Average existing conditions	With continuous Birch Road pumping of 4.9 ft^3/s
Inflows		
Recharge	9.43	9.43
Recharge from streams	0.53	0.58
Recharge from lakes	0.48	2.24
Total	10.44	12.26
Outflow		
Pumping	1.40	6.30
Discharge to streams	7.87	5.19
Discharge to lakes	1.21	0.78
Total	10.47	12.26
Numerical error	-0.03	0.00

Transient Simulations

The model was used to simulate monthly responses of groundwater and surface water to hypothetical Birch Road pumping under average and low-recharge conditions. In the first set of transient scenarios (TransCalb400 to TransCalb404, table 14), average recharge rates were assumed and, as described earlier (Boundary Conditions and Surface-Water Body Representation section), streamflow input to the model was assigned 25-percentile flow values to represent ecologically conservative base-flow conditions.

Simulated monthly water balances show seasonal hydrologic changes (fig. 12*A*). Under existing conditions in the warmer months of June through September, inflow to the aquifer is mostly from storage, as the water-table drops, whereas outflows are mostly groundwater discharge to streams. The colder months of October through March indicate inflows dominated by recharge from precipitation, whereas outflows are again mostly to streams with some outflow going into aquifer storage as groundwater levels rise. The small amount of pumping outflow under existing conditions is from the Wayland well withdrawals.

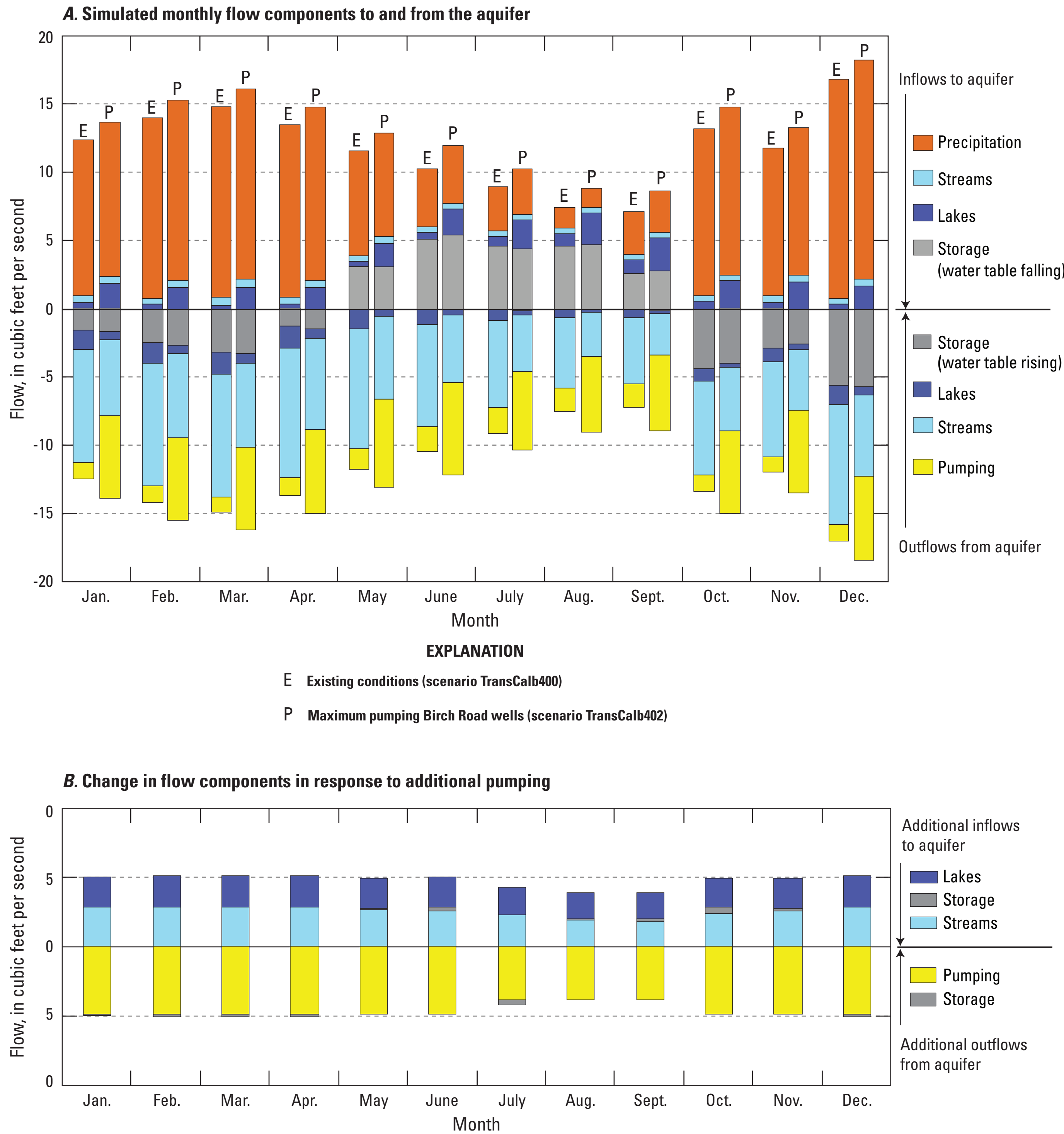


Figure 12. Simulated monthly *A*, flow components under existing average conditions and with additional pumping from Birch Road wells and *B*, changes in flows.

The monthly aquifer water balance changes appreciably when the Birch road wells are simulated as pumping at maximum hypothetical rates (scenario TransCalb402) of 4.9 and 3.9 ft^3/s (fig. 12*B*). The additional outflow is balanced mostly by greater net inflows to the aquifer from lakes and less outflow to streams (fig. 12*B*).

Simulated streamflow in the Sudbury River is assessed at the downstream exit of the active model area where all pumping effects can be seen (fig. 8). Streamflow depletion is calculated by subtracting the no-pumping scenario streamflow from the pumping scenario streamflow. For example, streamflow depletion in response to 1 month of pumping is calculated by subtracting simulated Sudbury River streamflow in scenario TransCalb400 from simulated streamflow in scenario TransCalb401. The timing of streamflow depletion in response to pumping in the Phase 2 model is similar to timing in the Phase 1 model (fig. 13). Streamflow depletion as a percent of pumping for scenario TransCalb401, during the month of pumping, is slightly less in the Phase 2 model than in the Phase 1 model (37 and 43 percent, respectively), but for all succeeding months streamflow depletion is slightly greater in the Phase 2 model than in the Phase 1 model. Within 1 month after pumping has stopped 66 percent of all resulting streamflow depletion has taken place, within 2 months 82 percent of streamflow depletion has taken place, and within 4 months 94 percent of depletion has taken place. The rapid response of surface water to pumping stresses seen in both models indicates good potential for pumping to be managed so that streamflow depletions are acceptable during periods of low flow.

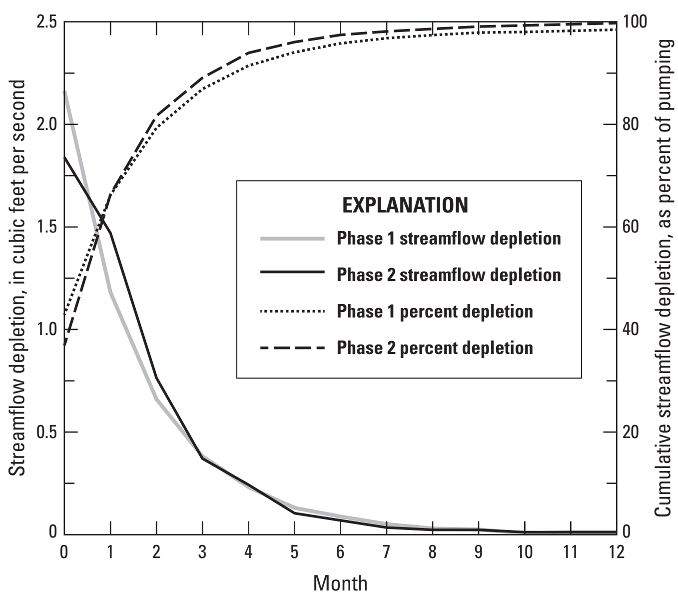


Figure 13. Simulated streamflow response of the Sudbury River after 1 month of pumping the Birch Road wells at 4.9 cubic feet per second (3.17 million gallons per day).

Simulations of dry climatic conditions (scenarios TransCalb410a to TransCalb413b) were made by decreasing recharge and streamflow specified at the model boundaries. As in the Phase 1 study, recharge was decreased by setting recharge to zero for the months of July, August, and September. In Phase 2 simulations, additional dry conditions were imposed by reducing streamflow inflow at the model boundaries from the monthly 25-percentile to the 10-percentile daily flow values as described earlier (Boundary Conditions and Surface-Water Body Representation section). The monthly 10-percentile flows that are low and are sustained more than 12 months represent drought conditions. The reduced stream inflow to the model, combined with the assigned recharge reductions for scenarios TransCalb410a-413b, impose very dry conditions that are more ecologically conservative than those used in the Phase 1 study.

The very dry conditions of scenarios TransCalb410a-413b made it difficult for the model to reach a solution because many cells dried. Scenarios TransCalb411a and TransCalb411b were not able to solve due to model cells going dry. To estimate streamflows for the conditions of these two scenarios (table 14), results from scenario TransCalb402, which applies the same pumping conditions, were instead used and reduced by the difference between the 25 percent and 10 percent duration inflows. This alternative estimation enforces the water balance and gives values that should be close to simulated values had the model been able to converge to a solution. All of the dry condition scenarios, TransCalb410a-413b, had some drier months with water mass balance errors of more than 1 percent. Various model adjustments, such as shortening time steps and adjusting convergence criteria, were made to reduce errors as much as possible. The resulting simulated values are generally representative of expected conditions but should be understood to have limitations in accuracy.

In scenarios TransCalb411a through TransCalb413b withdrawals from the Birch Road wells are assigned at various schedules and rates (table 14) to evaluate the effects of pumping on surface-water flows. Simulation results provide examples of managing and reducing streamflow depletion during average recharge conditions (TransCalb400 through TransCalb404) and during dry conditions (TransCalb410a through TransCalb413b). Scenarios TransCalb400 and TransCalb410 set a baseline condition with no pumping at the Birch Road wells under average and dry climatic conditions, respectively. For scenarios with pumping, pumping rates were assigned by month at one of several fixed rates (table 14); minimal pumping (0.1 ft^3/s), reduced-maximum pumping (3.9 ft^3/s), or maximum pumping (4.9 ft^3/s). The 0.1 ft^3/s pumping rate is the minimum required to keep a groundwater treatment facility operational (Peter Newton, Bristol Engineering Advisors, Inc., written commun., December 2014). The reduced maximum pumping rate of 3.9 ft^3/s was the maximum rate that the model could sustain without convergence failure. The maximum pumping rate of 4.9 ft^3/s corresponds to a proposed pumping rate in the Framingham permit application (SEA Consultants Inc., 2009). During actual operation of a future well field, pumping rates could be more finely adjusted and adjusted at finer time scales (daily or hourly for example) in response to changing conditions, such as drought conditions or operation of upstream reservoirs on the Sudbury River by the MWRA, to maximize pumping while minimizing effects on nearby surface-water resources.

Under average conditions, pumping causes relatively small changes to simulated stream base flows (table 16, fig. 14*A*). Changes in base flow range from about -3 percent during months of greatest recharge and streamflow to about -17 percent at the end of summer when recharge and streamflow were at seasonal lows (fig. 14*B*). The maximum streamflow depletion under average conditions is about 9 percent when pumping is for 9 months instead of 12 months and about 5 percent when pumping is for 6 months instead of 12 months. The maximum streamflow depletion takes place at different times of the year in response to alternate pumping schedules.

Percent changes to streamflow are more pronounced under drought conditions (fig. 14*C*). The largest percent streamflow depletions are caused by pumping for 12 months at maximum rates under dry climatic conditions (TransCalb411b), which causes streamflow during September to decrease by about 21 percent from 11.8 to 9.3 ft^3/s. Under these same conditions, if wells are pumped for 9 months rather than 12 months (TransCalb412b), streamflow depletion is decreased by about 13 percent, from 11.8 to 10.4 ft^3/s and if wells are pumped for 6 months (TransCalb413b) streamflow depletion is decreased by just 3.0 percent from 11.8 to 11.4 ft^3/s (table 16, fig. 14*A*).

Scenarios TransCalb411b–13b simulate the return of 10 percent of groundwater withdrawals from the Birch Road wells to Pod Meadow Pond, to maintain flow to and from the pond. Equivalent scenarios were not simulated in the Phase 1 study. In each month of pumping (12 months in TransCalb411, 9 months in TransCalb412, and 6 months in TransCalb413) 10 percent of groundwater withdrawals are diverted into Pond Meadow Pond (stream segment 9). These diversions increase streamflow in the stream that traverses the Pod Meadow wetlands (stream segments 9–12,14–15) and in the Sudbury River downstream from the Pod Meadow wetlands (stream segments 14–16, 18, 21). Streamflow is increased during the months of the diversions and also during the summer months when there are no diversions or pumping. At the Sudbury River exit from the model the gains from diverting 10 percent of the withdrawals to Pod Meadow Pond are small (fig. 14*C*), but account for a large percentage of the outflow from Pod Meadow without pumping.

Table 16. Simulated streamflow in the Sudbury River at the exit of the Phase 2 groundwater model.

[Highlighted cells indicate pumping at the Birch Road wells at 4.9 or 3.9 cubic feet per second]

Scenario	Sudbury River flow (cubic feet per second)											
	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Steady-state model, average recharge conditions												
SsCalb101	128.8											
SsCalb102	123.8											
Transient model, average recharge conditions												
TransCalb400	137.0	148.6	227.4	206.7	128.9	64.0	35.4	25.5	22.4	42.0	82.2	129.6
TransCalb401	135.2	147.1	226.6	206.3	128.7	63.8	35.3	25.5	22.3	42.0	82.2	129.6
TransCalb402	132.0	143.5	222.3	201.6	124.0	59.3	31.2	21.7	18.7	37.7	77.5	124.5
TransCalb403	132.8	144.0	222.7	201.9	124.2	59.4	32.2	23.4	21.0	39.6	78.9	125.5
TransCalb404	132.9	144.2	222.8	203.1	126.6	62.7	34.7	25.0	22.0	40.1	79.2	125.8
Transient model, dry (low recharge) conditions												
TransCalb410a	83.8	97.4	127.4	133.3	81.7	45.6	22.0	13.4	11.8	18.0	49.5	82.7
TransCalb411a	78.6	92.2	122.2	128.1	76.8	40.9	18.0	10.3	9.3	13.5	44.7	77.5
TransCalb412a	79.5	92.8	122.7	128.4	77.0	41.1	18.8	11.4	10.4	15.6	46.2	78.6
TransCalb413a	79.7	93.0	122.8	129.7	79.3	44.3	21.3	12.9	11.4	16.2	46.6	78.9
Transient model, dry (low recharge) conditions, 10 percent return flow[1]												
TransCalb411b	79.2	92.8	122.8	128.7	77.3	41.4	18.3	10.5	9.4	13.7	45.2	78.0
TransCalb412b	79.9	93.3	123.2	129.0	77.5	41.5	19.2	11.6	10.5	15.8	46.5	79.1
TransCalb413b	80.1	93.4	123.3	130.1	79.5	44.5	21.3	13.0	11.5	16.3	46.8	79.3

[1]10 percent of groundwater withdrawals at the Birch Road wells are returned to Pod Meadow Pond.

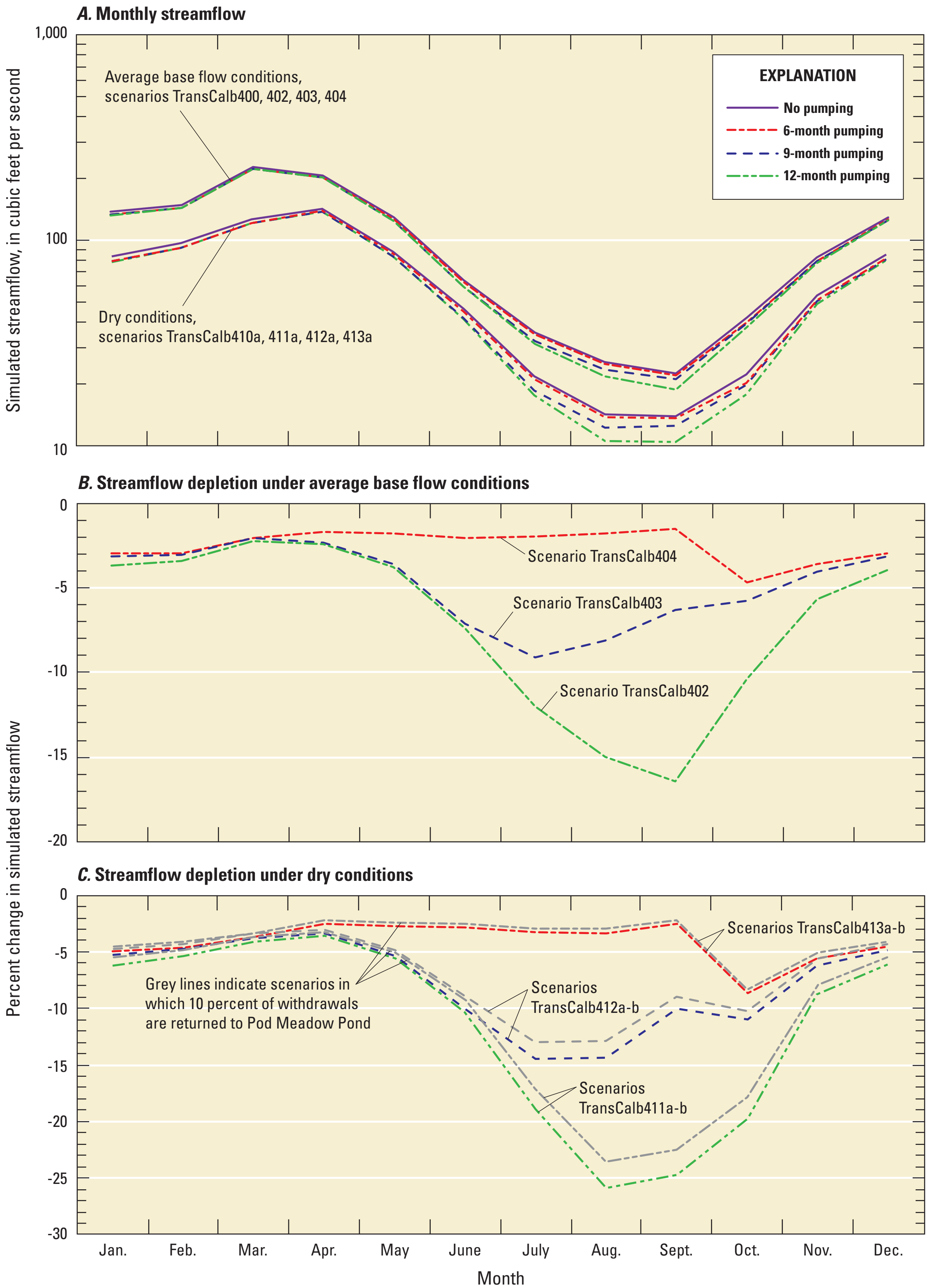


Figure 14. Simulated *A*, monthly streamflows and streamflow depletions in the Sudbury River at the model exit, with hypothetical groundwater withdrawals at the Birch Road wells, under *B*, average, and *C*, dry conditions.

Summary and Conclusions

The Town of Framingham has proposed increasing water supplies by reactivating the Birch Road well site near the Sudbury River and the adjacent towns of Wayland, Sudbury, and Natick. Questions have been raised about whether or not groundwater withdrawals from the Birch Road supply wells will affect hydrology of the ecologically important Sudbury River Basin in east-central Massachusetts; specifically whether or not withdrawals will reduce flow in streams, such as the Sudbury River, or adversely affect nearby State and Federal conservation areas, such as the Great Meadows National Wildlife Refuge, and nearby wetlands and surface-water bodies, particularly Lake Cochituate. To address these questions, the U.S. Geological Survey carried out this study in cooperation with the Town of Framingham. A Phase 2 groundwater flow simulation model was developed, based on the existing Phase 1 model, and used to improve understanding of groundwater surface-water interactions.

In the previously completed Phase 1 study (Eggleston and others, 2012) a numerical groundwater flow model was developed (with MODFLOW–NWT) that revealed a need for additional data and model refinements to better understand the complex aquifer system and the effects of groundwater withdrawals on surface-water bodies and wetlands in the study area. In this Phase 2 part of the study, the original groundwater flow model was revised to improve representation of groundwater and surface-water hydrology, stabilize the model, and reduce model error. The model was simplified by reducing the number of layers from 5 to 3 and adding the MODFLOW lake package (LAK) to simulate Lake Cochituate and Pod Meadow Pond and better represent interaction between the lakes and the aquifer. Model revisions improved stability and shortened run times, allowing use of automated parameter estimation software (PEST) to further refine the model hydraulic parameters and reduce simulation errors.

The calibrated Phase 2 groundwater flow model was applied to simulate hydrologic conditions under hypothetical pumping and climatic conditions. Model simulations indicate that under average base-flow conditions, the Birch Road wells have a small effect on flow in the Sudbury River during most months, even at the maximum proposed pumping rate of 4.9 ft^3/s (3.17 Mgal/d). Maximum percent simulated streamflow depletion in the Sudbury River at the outlet of the active model area caused by pumping takes place during seasonally dry periods and during severe drought conditions, decreasing streamflow by as much as 17 percent under average conditions and as much as 21 percent under dry conditions. By adjusting groundwater withdrawal rates on a monthly basis, reductions in simulated streamflow under sustained drought conditions, the most ecologically conservative assumptions, streamflow depletion changed from about 21 percent under 12 months of maximum pumping to 13 percent under 9 months of pumping and to 3 percent under 6 months of pumping. The maximum streamflow depletion took place in September under the maximum simulated pumping rates and seasonally low recharge and streamflow, but changed to other months of the year according to the pumping schedule simulated. Simulations that return 10 percent of the Birch Road well withdrawals to Pod Meadow Pond indicate a modest reduction in the Sudbury River streamflow depletion, but provide a larger percentage increase to streamflow at the pond outlet and in the stream reach in the downstream wetland.

Model simulations indicate that groundwater withdrawals at the Birch Road site could be managed to substantially reduce adverse streamflow impacts caused by groundwater withdrawals during seasonally dry conditions and severe drought conditions. Results of the study indicate that well locations can have a large effect on the sustainable pumping rate and so should be chosen carefully in any potential future reactivation of the well field. The simulation model is a useful tool for evaluating alternative pumping rates and schedules not included in this analysis.

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Eggleston and others—**Groundwater and Surface-Water Interaction and Effects of Pumping, Complex Glacial-Sediment Aquifer, Phase 2, Mass.**—SIR 2015–5174

ISSN 2328-0328 (online)
http://dx.doi.org/10.3133/sir20155174