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Simulation of Ground-Water Flow and Evaluation of Water-Management Alternatives in the Upper Charles River Basin, Eastern Massachusetts

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CONVERSION FACTORS, VERTICAL DATUM, AND ABBREVIATIONS

CONVERSION FACTORS

Multiply	By	To obtain
acre	0.4047	square kilometer
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
cubic foot per second per square mile [($ft^3/sec)/mi^2$]	0.02832	cubic meter per second per square kilometer
foot (ft)	0.3048	meter
foot per day (ft/d)	0.3048	meter per day
gallons per minute (gal/min)	0.0038	cubic meter per minute
inch (in.)	25.4	millimeter
inch per month (in/month)	25.4	millimeter
inch per year (in/yr)	25.4	millimeter per year
mile (mi)	1.609	kilometer
million gallons per day (Mgal/d)	0.04381	cubic meter per second
square foot per day (ft^2/d)	0.0929	square meter per day
square mile (mi^2)	2.590	square kilometer

VERTICAL DATUM

Sea level: In this report, "sea level" refers to the National Geodetic Vertical Datum of 1929 (NGVD of 1929)—a geodetic datum derived from a general adjustment of the first-order level nets of the United States and Canada, formerly called Sea Level Datum of 1929.

ABBREVIATIONS

CRPCD	Charles River Pollution Control District Treatment Facility
CRWA	Charles River Watershed Association
DEM	Digital elevation model
ET	Evapotranspiration
GIS	Geographic information system computer software
MADEM	Massachusetts Department of Environmental Management
MADEP	Massachusetts Department of Environmental Protection
MTF	Milford Treatment Facility
MOVE.1	Maintenance of Variance Extension, type 1, method of correlation
No.	Number
PART	Computer program for streamflow partitioning (Rutledge, 1993)
PWSS	Areas in the study area served by public water and septic systems
USGS	U.S. Geological Survey
WMA	Water Management Act

Simulation of Ground-Water Flow and Evaluation of Water-Management Alternatives in the Upper Charles River Basin, Eastern Massachusetts

By Leslie A. DeSimone, Donald A. Walter, John R. Eggleston, *and* Mark T. Nimiroski

Abstract

Ground water is the primary source of drinking water for towns in the upper Charles River Basin, an area of 105 square miles in eastern Massachusetts that is undergoing rapid growth. The stratified-glacial aquifers in the basin are high yield, but also are thin, discontinuous, and in close hydraulic connection with streams, ponds, and wetlands. Water withdrawals averaged 10.1 million gallons per day in 1989–98 and are likely to increase in response to rapid growth. These withdrawals deplete streamflow and lower pond levels. A study was conducted to develop tools for evaluating water-management alternatives at the regional scale in the basin. Geologic and hydrologic data were compiled and collected to characterize the ground- and surface-water systems. Numerical flow modeling techniques were applied to evaluate the effects of increased withdrawals and altered recharge on ground-water levels, pond levels, and stream base flow. Simulation-optimization methods also were applied to test their efficacy for management of multiple water-supply and water-resource needs.

Steady-state and transient ground-water-flow models were developed using the numerical modeling code MODFLOW-2000. The models were calibrated to 1989–98 average annual conditions of water withdrawals, water levels, and stream base flow. Model recharge rates were varied spatially, by land use, surficial geology, and septic-tank return flow. Recharge was changed during model calibration by means of parameter-estimation techniques to better match the estimated average annual base flow; area-weighted rates averaged 22.5 inches per year for the basin. Water withdrawals accounted for about 7 percent of total simulated flows through the stream-aquifer system and were about equal in magnitude to model-calculated rates of ground-water evapotranspiration from wetlands and ponds in aquifer areas. Water withdrawals as percentages of total flow varied spatially and temporally within an average year; maximum values were 12 to 13 percent of total annual flow in some subbasins and of total monthly flow throughout the basin in summer and early fall.

Water-management alternatives were evaluated by simulating hypothetical scenarios of increased withdrawals and altered recharge for average 1989–98 conditions with the flow models. Increased withdrawals to maximum State-permitted levels would result in withdrawals of about 15 million gallons per day, or about 50 percent more than current withdrawals. Model-calculated effects of these increased withdrawals included reductions in stream base flow that were

greatest (as a percentage of total flow) in late summer and early fall. These reductions ranged from less than 5 percent to more than 60 percent of model-calculated 1989–98 base flow along reaches of the Charles River and major tributaries during low-flow periods. Reductions in base flow generally were comparable to upstream increases in withdrawals, but were slightly less than upstream withdrawals in areas where septic-system return flow was simulated. Increased withdrawals also increased the proportion of wastewater in the Charles River downstream of treatment facilities. The wastewater component increased downstream from a treatment facility in Milford from 80 percent of September base flow under 1989–98 conditions to 90 percent of base flow, and from 18 to 27 percent of September base flow downstream of a treatment facility in Medway. In another set of hypothetical scenarios, additional recharge equal to the transfer of water out of a typical subbasin by sewers was found to increase model-calculated base flows by about 12 percent of model-calculated base flows. Addition of recharge equal to that available from artificial recharge of residential rooftop runoff had smaller effects, augmenting simulated September base flow by about 3 percent.

Simulation-optimization methods were applied to an area near Populatic Pond and the confluence of the Mill and Charles Rivers in Franklin, Medway, and Norfolk. Water is withdrawn from six supply wells for three towns in this area. Management objectives in this analysis were to develop pumping schemes that (1) maximized water withdrawals while imposing specified constraints on streamflow depletion and (2) minimized streamflow depletion while meeting minimum requirements for water supply. Application of the optimization techniques indicated that hydrologic responses of pond levels and streamflow to pumping at different supply wells in the Populatic Pond area vary in time and duration. This variability suggests that water withdrawals could be managed to minimize the effects of increased withdrawals on streams and ponds. Simulation of several preliminary scenarios indicated the possibility that, with active management of water-supply sources, water withdrawals could be substantially increased from existing and proposed sources while base flow in the Charles and Mill Rivers was maintained above minimum average monthly flow requirements. Alternatively, base flow could be increased in the Charles River during low-flow periods while existing and proposed withdrawals were met. Finally, results from these scenarios indicated that collaborative management of water sources by towns could reduce the effect of withdrawals on stream base flow, and, by implication, would allow greater withdrawals from the Populatic Pond area without exceeding specified limits on streamflow depletion.

INTRODUCTION

Ground water is the primary source of drinking water for eight towns in the Upper Charles River Basin (fig. 1), an area of eastern Massachusetts that is undergoing rapid growth and development. More than 30 public-supply wells withdraw water from sand and gravel aquifers that are high yield, but thin and discontinuous. The aquifers are in direct hydraulic connection with the Charles River and its tributaries, ponds, and wetlands, and often cross municipal boundaries. Along with one surface-water withdrawal and private sources, these wells serve a population of about 100,000, which, on average, has increased by about 15 percent, and by more than 30 percent in a few towns, during the past decade. Some of these towns also have had large increases in commercial and industrial development in recent years. Continued rapid growth is projected, driven by an expanding metropolitan Boston area and the technology-based economic opportunities along Interstate 495 (Bouck, 1998; Massachusetts Technology Collaborative, 1998; Metropolitan Area Planning Council, 2001). Towns, State agencies, business leaders, and residents are increasingly recognizing that water resources in the basin may fail to meet the needs of this continued development without adverse ecological and economic effects unless planning and management alternatives are investigated and implemented (Bouck, 1998; Massachusetts Technology Collaborative, 1999).

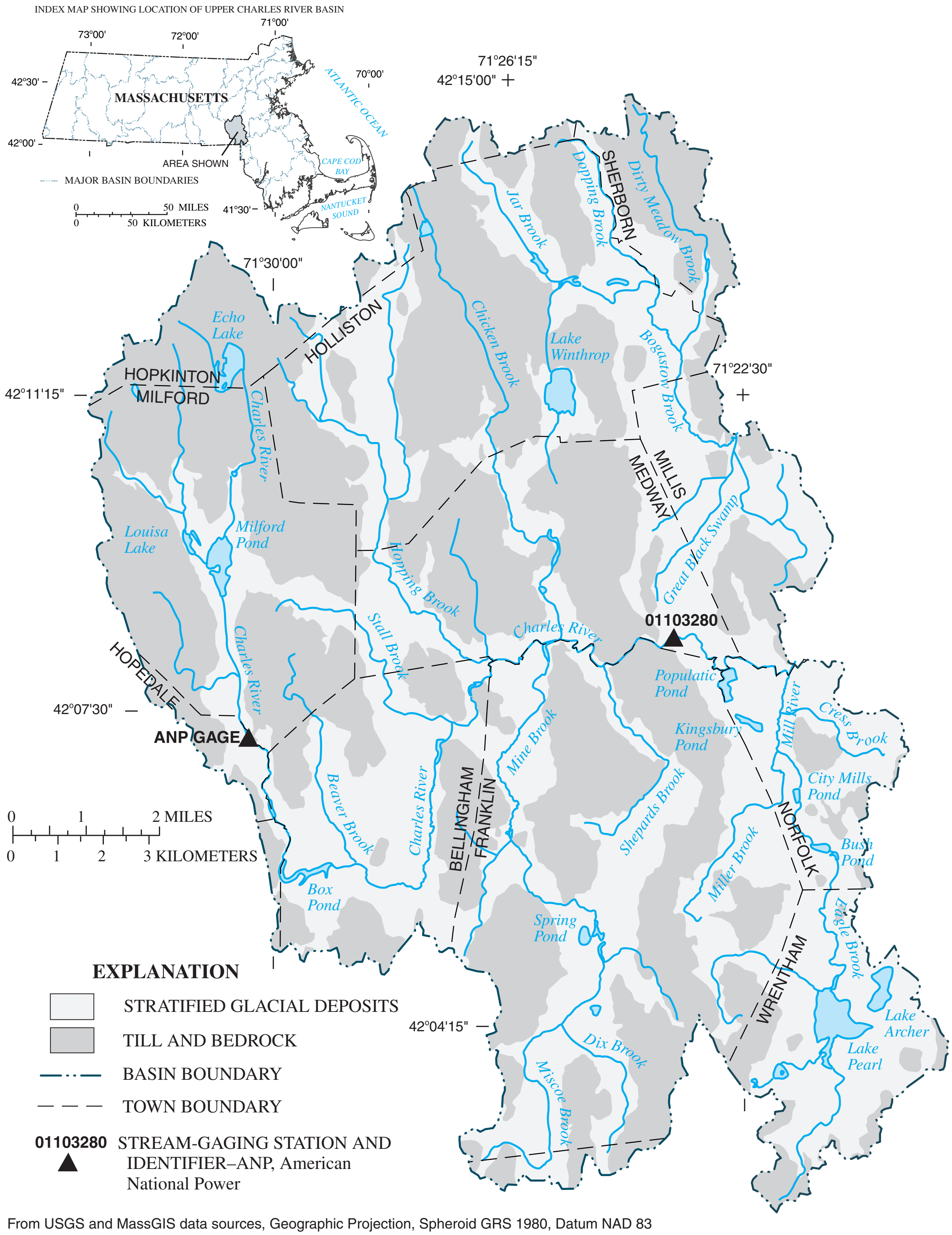

Figure 1. Location and surficial geology of the study area, upper Charles River Basin, eastern Massachusetts.

Increased water withdrawals, combined with out-of-basin or downstream transfers of wastewater and decreased natural recharge, which are associated with development, contribute to streamflow depletion and lowered water levels in ponds and wetlands in the upper Charles River Basin. Streamflow depletion, which is caused by reduced ground-water inflow, induced infiltration, in- and near-stream withdrawals, and pond-level changes, can adversely affect aquatic and riparian communities by reducing available water and altering habitat. Streamflow depletions also may have adverse effects on stream-water quality, because there is less ground-water discharge (base flow) to dilute contaminants from wastewater discharges and non-point pollution sources. These conditions and potential problems are typical of many suburban and rural communities in eastern Massachusetts and adjacent States.

In the upper Charles River Basin, streamflow depletion and altered pond levels already are problems in several areas (Bouck, 1998; Higgins, 2000; Fiorentino and others, 2000). Reduced streamflows due to municipal withdrawals in the upper reaches of the mainstem Charles River in Milford (fig. 1) have resulted in poor habitat quality for aquatic life. Stratified glacial aquifers along Mine Brook in Franklin and Mill River in Norfolk are considered stressed by withdrawals from municipal supply wells; these withdrawals result in base-flow reductions in the rivers and lowered pond levels. The flow in Dopping Brook and Jar Brook in Holliston also may be reduced by future pumping. Nutrient enrichment in the Charles River and in-stream ponds occurs downstream of wastewater-treatment facilities in Milford and Medway. Lowered pond levels in Kingsbury Pond, a kettle-hole pond in Norfolk, have been a problem since the 1960s and have been associated with municipal pumping (Williams, 1967). The stresses associated with existing water-management practices are likely to intensify, and new problem areas may develop, as a result of implementation of the multiple projects that currently are proposed. These projects include new supply wells in Franklin, Holliston, Norfolk, and Wrentham, a new surface-water withdrawal in Milford, and sewer-installation projects in Bellingham and elsewhere. Nontraditional management alternatives, such as artificial recharge projects for wastewater and stormwater, also are proposed (DiBona and Eleria, 1999; Earth Tech, 1999; SEA Consultants, 1999). The location of some productive stratified glacial aquifers in the upper Charles River Basin along town boundaries makes implementation of some of these projects particularly contentious, leading to conflicts between towns over water rights and the adverse effects of withdrawals and wastewater discharges.

A regional approach for evaluating water-resource development and management alternatives is needed in the upper Charles River Basin. Evaluation of proposed projects on a case-by-case basis does not adequately determine cumulative effects on the stream-aquifer system, nor fully assess effects on downstream uses. In some cases, regional water-management practices may be reasonable alternatives for meeting water demands while minimizing adverse effects on water resources. A numerical ground-water-flow model is one tool that can be implemented at the basin-wide scale and used to assess the cumulative effects of large- or small-scale changes in the stream-aquifer system on ground-water levels, pond levels, and streamflow depletion. When combined with optimization techniques for conjunctive management of surface- and ground-water systems, numerical simulation can be used to identify pumping strategies or other management schemes that simultaneously minimize adverse effects on surface waters and maximize ground-water yield (Barlow and Dickerman, 2001).

Concern by State agencies and others over the effects of population growth and development on water resources in the upper Charles River Basin and the need for a regional approach to evaluate water management and development alternatives prompted a study by the U.S. Geological Survey (USGS) in cooperation with the Massachusetts Departments of Environmental Management (MADEM) and Environmental Protection (MADEP) and the Charles River Watershed Association (Charles River Watershed Association). The objectives of this investigation were to define water-resource conditions within the sand and gravel aquifers and to evaluate the effects of selected ground-water development and return-flow alternatives on water resources in the basin through the development and application of ground-water-flow models. To ensure that the investigation adequately addressed issues of concern for water use and resources in the basin, representatives from towns, a water supplier, water dischargers, and a major water user in the study area, along with MADEM and MADEP, participated in a Technical Advisory Committee that was coordinated by CRWA for the study. The water-use and management issues investigated in this study for the upper Charles River Basin are common to many other basins in eastern Massachusetts and adjacent States, where communities are striving to develop sustainable water-use practices.

Purpose and Scope

This report describes the development and application of numerical ground-water-flow models of the stream-aquifer system in the upper Charles River Basin in eastern Massachusetts. The study area of 105 mi^2 encompasses most or all of Franklin, Holliston, Medway, and Milford, and substantial parts of Bellingham, Millis, Norfolk, and Wrentham (fig. 1). Steady-state and transient models were developed to evaluate regional effects of changes in the stream-aquifer system associated with increased water demands and development. Separate models were developed for aquifers in western and eastern parts of the study area. The models were developed and calibrated on the basis of hydrogeologic data collected during this and previous investigations. These data include lithologic information for aquifers; hydraulic properties of and recharge to aquifers; water levels measured in wells and ponds; streamflow measurements for the Charles River and its tributaries; and ground- and surface-water withdrawal rates and wastewater-discharge rates to the aquifers and the Charles River. Several of these data sets were compiled in cooperation with the CRWA. The models are representative of average withdrawal and hydrologic conditions in the stream-aquifer system for the 10-year period 1989–98. The USGS modular three-dimensional finite-difference ground-water flow modeling code, MODFLOW-2000 (McDonald and Harbaugh, 1988; Harbaugh and others, 2000), was used to simulate the stream-aquifer system. Results of several hypothetical scenarios that incorporate changes in the stream-aquifer system, including increased pumping rates at existing withdrawals, new supply wells, and increased recharge, are described. The report also describes the application of simulation-optimization modeling in the area of Populatic Pond in Franklin, Medway, and Norfolk. The simulation-optimization model was used to investigate pumping strategies for supply wells in this area that would minimize streamflow depletion during critical low-flow periods.

Description of the Study Area

The upper Charles River Basin encompasses an area of 105 mi^2 in the Charles River Basin in eastern Massachusetts (fig. 1). The Charles River flows about 80 mi from headwaters in Hopkinton to Boston Harbor and meanders extensively in its middle and lower reaches. The lower basin includes communities of metropolitan Boston and contains some of the most highly urbanized areas in Massachusetts. The upper basin, which is the focus of this study, is primarily suburban and semirural (Commonwealth of Massachusetts, 2001). As defined for this study, the upper basin extends from the Charles River headwaters to approximately 1.5 river-miles downstream from the outlet of Populatic Pond; it also includes areas drained by Bogastow Brook, which joins the Charles River outside of the study area, and tributaries to Bogastow Brook upstream of Millis center (fig. 1). Most or substantial parts of Bellingham, Franklin, Holliston, Medway, Milford, Millis, Norfolk, and Wrentham are included in the study area. The boundary of the study area was chosen to include the areas of interest for water management and to coincide as much as possible with natural boundaries of aquifers, as determined from available data (Walker and others, 1975).

Topography in the upper Charles River Basin is gently rolling to hilly. Extensive low-lying and wetland areas are in the eastern parts and along stream valleys in Franklin and Holliston. The land surface generally slopes eastward, with elevations ranging from 550 ft above sea level in Milford and Hopkinton to about 120 and 135 ft above sea level at the Charles River and Bogastow Brook outlets from the study area, respectively. The Charles River drops about 220 ft over about 20 mi and is impounded by nine dams in the study area (Myette and Simcox, 1992); dams also are common on the major tributaries. In addition to ponds and impoundments along streams, the basin contains several kettle ponds, including Kingsbury Pond in Norfolk (fig. 1). Mean annual precipitation is about 46 in/yr, on the basis of data from six nearby climate stations (including West Medway where the average is 46.6 in/yr, 1957–2000; National Oceanic and Atmospheric Administration, 2001) and is distributed uniformly throughout the year.

Land use in the upper basin based on the most recent basin-wide data (land use, 1990; wetlands, 1998; MassGIS, 1997), was primarily forested (50 percent) and low- and medium-density residential (24 percent). Wetlands (8 percent), agriculture (5 percent), commercial and industrial uses (3.5 percent), and high-density residential use (3 percent) represented small fractions of land use in the upper basin in 1990. High-density residential, commercial, and industrial land uses in the study area are centered in Milford and Franklin and in the town centers of Medway, Millis, Holliston, and Bellingham. Milford is the most densely populated town, followed by Franklin and Medway. Since 1990, forested land use has decreased in area as residential, commercial, and industrial land-use areas have increased (for example, see SEA Consultants, 1999). Population has increased most rapidly between 1990 and 2000 in Franklin (36 percent increase) and Medway (32 percent increase), and large population increases also have occurred in Wrentham (15 percent) and Holliston (14 percent) (U.S. Census Bureau data, from Commonwealth of Massachusetts, March 20, 2001, and May 20, 2002). Total population in the upper Charles River Basin was about 100,000 in 2000.

Previous Studies

Information on the hydrogeology and water resources of the upper Charles River Basin is available from many sources. Surficial and bedrock geology of northern parts of the study area were mapped by Volckmann (1975a, 1975b, 1977). Basic hydrologic data, including well and boring logs, water levels, and water quality, and the locations of high transmissivity zones, are available in Walker and others (1975, 1977). An analysis of aquifer yields on the basis of streamflow data was completed by Myette and Simcox (1992). Continuous-record streamflow data for the Charles River are available at a USGS stream-gaging station in West Medway (USGS station number 01103280, fig. 1), in operation since 1997, and at a stream-gaging station operated by ENSR Corporation for American National Power, in Milford, since 1994 (ENSR Corporation, 2000, and references therein). Streamflow data also are available for partial-record stations on the Charles River and its tributaries that were used for low-flow studies (Ries, 1993, 1994, and 1999; Ries and Friesz, 2000). A recent assessment of water resources in the basin focused on water quality and habitat (Fiorintino and others, 2000).

In addition to these basin-wide assessments and data-collection programs, numerous smaller-scale studies have been conducted in the upper Charles River Basin, primarily by consultants to the towns. These include hydrogeologic investigations to locate water-supply sources, to determine wellhead-protection areas for public-supply wells, and to support specific projects such as wastewater discharges or large construction projects. Information available from these reports includes well and boring logs, hydrogeologic maps and sections, and results of aquifer tests and numerical simulations. Reports that were used in this study include Amory Engineers (1989), Anderson-Nichols (1994, 1998), Coffin and Richardson (1977), D.L. Mahar (1986, 1989, 1993, 1995a, 1995b, 1996, 1997), Dufresne-Henry (1996, 1997), Earth Tech (1999), Geophysical Applications (1995), Groundwater Associates (1991), Haley and Ward (1994, 1995), Layne New England (1974), Metcalf & Eddy (1983a, 1983b, 1997), SEA Consultants (1999), and Whitman and Howard (1991, 1992, 1996).

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UPPER CHARLES STREAM-AQUIFER SYSTEM

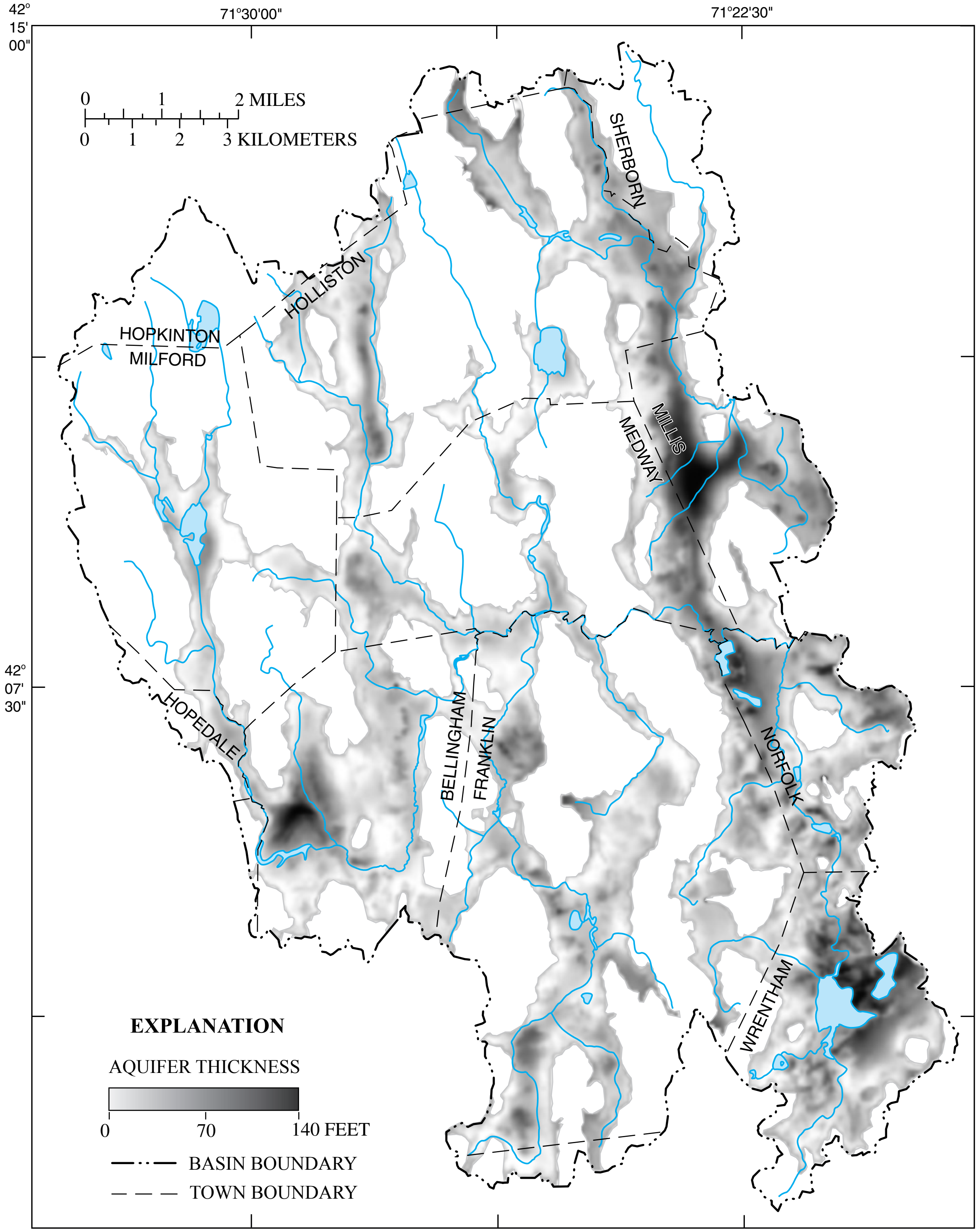
Hydrogeology

Hydrogeologic units in the upper Charles River Basin include stratified glacial deposits, till, and bedrock (fig. 1). Stratified glacial deposits consist of sorted and layered sand, gravel, silt, and clay that was deposited in valleys and lowlands during the last glacial period (Volckmann, 1975a, 1975b). The major aquifers in the basin are contained in the stratified glacial deposits. Till is a poorly sorted, unstratified mixture of clay, silt, sand, gravel, cobbles, and boulders, deposited directly by the glacial ice. Till covers uplands in the study area and typically varies in thickness from 0 to about 15 ft (Volckmann, 1975a, 1975b); thicker till deposits underlie till hills (drumlins). Thin till layers also underlie stratified drift in some areas. The permeability of till is low but variable (0.6 to 2.7 ft/d for tills in New England; Melvin and others, 1992), and till generally is not used for water supply in the study area. Stratified glacial deposits and till are underlain by bedrock that is mostly granite, diorite, and gneiss of Proterozoic and Lower Paleozoic age (Volckmann, 1977). Bedrock outcrops are common in upland areas (Volckmann, 1975a, 1975b). Bedrock also is less permeable than stratified glacial deposits and yields water generally at low rates (for example, 10 gal/min or less); bedrock wells are used only for private supplies in the study area (Walker and others, 1975).

In addition to stratified glacial deposits, till, and bedrock, recent deposits in the study area include alluvium, swamp deposits, and streambed deposits (Volckmann, 1975a, 1975b). Alluvium, mostly fine sand and silt, is along the Charles River in the eastern part of the study area. Swamp deposits of sand, silt, clay, and organic material overlie stratified glacial deposits and are areally extensive in low-lying areas, but generally form thin surficial layers. For ground-water-flow modeling purposes, alluvium and swamp deposits overlying stratified glacial deposits (fig. 1) were combined with the stratified glacial deposits and considered to have the same hydraulic properties.

Stratified glacial deposits occur along the Charles River and its major tributaries and cover about 50 percent of the study area (fig. 1). The areal extent of stratified glacial deposits in the study area was determined from published and unpublished surficial geologic maps (Volckmann, 1975a, 1975b; B.D. Stone and J.R. Stone, U.S. Geological Survey, written commun., 2000). The thickness of the stratified glacial deposits was mapped by contouring the elevation of the underlying bedrock or till surface (J.R. Stone, U.S. Geological Survey, written commun., 2000). Data on depth to bedrock, till, or drilling refusal were obtained from about 800 well logs or borings from USGS files, from reports by private consultants cited previously, and from wells installed during this study. Thicknesses are greatest along a north-south trending preglacial bedrock valley in the eastern part of the study area (fig. 2). Thicknesses of up to 140 ft are reached in this area near Great Black Swamp in Millis, along the Mill River in Norfolk and Franklin, and near Lake Pearl in Wrentham. Thick stratified glacial deposits also are along the Charles River near Box Pond in Bellingham. Aquifer thicknesses elsewhere are generally 70 ft or less (fig. 2). Stratified glacial aquifers in the western and eastern parts of the study area are physically discontinuous, but are connected by the Charles River, which flows across thin till and bedrock in Medway and Franklin (fig. 1).

The distribution, thickness, and character of the stratified glacial deposits reflects their depositional history as well as the configuration of the preglacial bedrock surface. The stratified glacial deposits in the upper Charles River Basin were deposited as a series of sedimentary sequences during temporary standstills of the retreating ice sheet (Stone and others, 1998; J.R. Stone, U.S. Geological Survey, oral commun., 2001). These depositional sequences were formed in valleys and proglacial lakes controlled by local topography, ice distribution, and previously deposited glacial sediments (Stone and others, 1998). The sequences contain ice-contact, fluvial, deltaic, and(or) lacustrine sediments. In each sequence, grain size decreases and sorting increases with increasing distance from the former ice margin. Vertical variations in texture resulted from prograding delta sequences (coarse over fine) or from the deposition of fine-grained or lacustrine sediments over collapsed coarse-grained sediments of an earlier depositional sequence. In the upper Charles River Basin, where the ice-sheet retreated northward, generally east-west trending ice-margin positions resulted in depositional sequences that grade southwards from coarse to fine sediments (J.R. Stone, U.S. Geological Survey, written commun., 2001). In the southern part of the study area, where valleys were shallow and narrow, the stratified glacial deposits are predominantly collapsed coarse-grained deposits. Further northward, the fine-grained deposits are better developed (J.R. Stone, U.S. Geological Survey, oral commun., 2001).



From USGS and MassGIS data sources, Geographic Projection, Spheroid GRS 1980, Datum NAD 83

Figure 2. Thickness of the stratified glacial aquifers in the upper Charles River Basin, eastern Massachusetts.

Information on hydraulic properties of the stratified glacial deposits is available from aquifer tests at public-supply wells and geologic logs from wells and boreholes. Results of aquifer tests at supply-well sites in Bellingham, Franklin, Norfolk, Holliston, and Milford indicated horizontal hydraulic conductivities that ranged from 68 to 247 ft/d (table 1). This range is typical of coarse-grained facies of stratified glacial deposits, valley-fill deposits in New England (Kontis and others, in press; Barlow and Dickerman, 2001; Stone and Dickerman, 2002).

Ground water in the stratified glacial deposits generally is unconfined in the study area. Confined conditions can exist locally, however, where fine-grained facies of the glacial sequences described above are well developed. In the eastern part of the study area, silt and clay sediments underlie Populatic Pond and extend northwestward from the pond; silt and clay sediments also are along the Mill River near Kingsbury Pond in Norfolk (Groundwater Associates, Inc., 1991; D.L. Maher Co., 1996; Dufresne-Henry, Inc., 1996). These fine-grained sediments result in locally confined conditions in the underlying sand and gravel aquifer sediments. In the west model area, silt, clay, and peat sediments near land surface result in semi-confined conditions in the sand and gravel aquifer in wetland areas along the Charles River in northeast Bellingham.

Ground water in the stratified glacial deposits (henceforth, “stratified glacial aquifers” or “aquifers”) generally flows from boundaries with till and bedrock uplands through the aquifers in the downstream direction. Locally, flow is directed towards streams; streams in the aquifer areas are predominantly gaining when unaffected by pumping and are in close hydraulic connection to the aquifers (Randall and others, 1988; Rosenshein, 1988). Stream reaches may be losing under natural conditions near upland boundaries, where abrupt changes occur in the permeability of the material underlying the streams and water-table elevations may change rapidly over short distances. Infiltration of streamflow also may be induced by withdrawals from pumping wells adjacent to streams and ponds.

Water Withdrawals and Return Flow

Water is withdrawn for public supply in the upper Charles River Basin from 33 wells or wellfields in the stratified glacial aquifers and from two locations on the Charles River in Milford (table 2 and fig. 3). In addition, there are several private ground- and surface-water withdrawals in the study area that exceed 0.1 Mgal/d and are registered or permitted under the Massachusetts Water-Management Act (WMA); these include three golf courses and a power plant (table 2 and fig. 3). Data on locations and monthly pumped volumes from 1989 to 1998 were obtained from town officials for municipal sources and from MADEP and MADEM for WMA sources. Average water withdrawals, discharges, and transfers during 1989 to 1998 are summarized in figure 4. Several proposed new municipal ground-water wells were being considered for permits by the MADEP at the time of this study; the proposed wells are located in Franklin, Holliston, Medway, Norfolk, and Wrentham (table 2).

Total withdrawals for public supply from the stratified glacial aquifers and surface-water sources in the study area averaged 9.4 Mgal/d from 1989 to 1998 (table 2). Public-supply withdrawals increased about 15 percent during this time (fig. 5*A*). On average, sources were pumped at less than half their MADEP-approved maximum permitted withdrawal rates on an annual basis (table 2). Withdrawals were greatest during summer months of June, July, and August; these withdrawals were about 30 percent of annual volumes (fig. 5*B*). Summer withdrawals, which increased by about 35 percent from 1989 to 1998, accounted for about half of the increased volume pumped during this period. A small amount of the water withdrawn for public supply is used consumptively. Consumptive use averages about 10 percent annually (Solley and others, 1993). Consumptive use varies seasonally from nearly zero in winter months to about 30 percent in summer months (on the basis of an analysis of monthly water withdrawals in the study area; N.B. Pickering, Charles River Watershed Association, written commun., 2001), a typical pattern resulting from outside water use (for example, lawn watering). Withdrawals varied by town, reflecting differences in size and population density.

Table 1. Hydraulic properties of stratified glacial deposits in the upper Charles River Basin, eastern Massachusetts, as determined by analysis of aquifer tests at public-supply wells

[**Aquifer test well site**: See table 2 for additional identification information; site locations shown on figure 3. ft, feet; ft/d, feet per day; ft^2/d, square feet per day; gal/min, gallons per minute; --, not available]

Aquifer test well site	Predominant grain size of tested interval	Length of test (days)	Well discharge (gal/min)	Transmissivity (ft^2/d)			Saturated thickness (ft)	Mean hydraulic conductivity (ft/d)	Specific yield	Reference
				Mean	Minimum	Maximum				
BL-12G	Fine to coarse sand and gravel	5	384	6,020	4,950	16,400	26	231	[1]0.001–0.5	Groundwater Associates, 1991
FR-01G,-02G	Fine to coarse sand and gravel	2	--	8,710	7,480	9,940	60	145	--	D.L. Maher Co., 1995a
FR-07G	Fine to coarse sand and gravel	2	--	7,860	--	--	61	129	--	D.L. Maher Co., 1995a
FR-02P	Sand and gravel	--	--	10,200	8,010	11,900	--	231	[1]0.0003–0.1	D.L. Maher Co., 1996
HL-01G	Medium to coarse sand and gravel, silt	2	190	7,550	7,032	7,990	60	126	0.07–0.14	Whitman and Howard, Inc., 1996
HL-02G	Medium to coarse sand and gravel	2	240	4,850	4,520	5,070	30	162	0.13–0.15	Whitman and Howard, Inc., 1996
HL-05G	Medium to coarse sand and gravel	2	440	12,800	11,900	13,800	52	247	0.19	Whitman and Howard, Inc., 1996
HL-06G	--	--	--	11,230	--	--	47	239	--	Whitman and Howard, Inc., 1996
MF-03G,-04G,-05G	Fine to coarse sand and gravel	2	50	2,450	2,260	2,650	36	68	[1]0.01	D.L. Maher Co., 1995b
NF-01G	Fine to coarse sand and gravel	17	250	5,050	2,760	9,800	53	95	[1]0.01–0.28	Dufresne-Henry, Inc., 1997
NF-01P	Fine to coarse sand and gravel	23	--	7,220	4,200	9,800	60	120	0.04	Dufresne-Henry, Inc., 1996

[1]Confining unit present at site.

Table 2. Water withdrawals for existing and proposed municipal public-supply and large non-municipal sources and existing wastewater discharges in the upper Charles River Basin, eastern Massachusetts

[**Source type:** GW, ground water; SW, surface water. **Well depth**: All wells completed in the stratified glacial deposits unless otherwise noted. **Maximum per-mitted withdrawal rate**: Data from B.R. Bouck, Massachusetts Department of Environmental Protection, written commun., 2001 (municipal sources) and D. LeVangie, Massachusetts Department of Environmental Management, written commun., 1999 (non-municipal sources). Proposed withdrawals include sources currently submitted for approval by municipalities to the Massachusetts Department of Environmental Protection; Mgal/d, million gallons per day; --, not applicable or not known]

Well identifier	Town	Source name	Source type	Well depth (feet)	Mean annual withdrawal or discharge rate, 1989–98 (Mgal/d)	Maximum permitted withdrawal rate (Mgal/d)
Municipal Public-Supply Withdrawals						
BL-05G	Bellingham	Well No. 5	GW	32	0.20	0.29
BL-07G	Bellingham	Well No. 7	GW	50	.10	.42
BL-08G	Bellingham	Well No. 8	GW	48	.27	.68
BL-12G	Bellingham	Well No. 12	GW	62	.02	.55
FR-01G	Franklin	Well No. 1	GW	35	[1].11	.47
FR-02G	Franklin	Well No. 2	GW	45	[1].11	.72
FR-03G	Franklin	Well No. 3	GW	56	.30	.32
FR-04G	Franklin	Well No. 4	GW	73	.67	.92
FR-05G	Franklin	Well No. 5	GW	60	.17	.50
FR-06G	Franklin	Well No. 6	GW	60	.34	.53
FR-07G	Franklin	Well No. 7	GW	45	.28	.58
FR-08G	Franklin	Well No. 8	GW	51	.30	.26
FR-09G	Franklin	Well No. 9	GW	30	.26	.50
FR-10G	Franklin	Well No. 10	GW	30	.22	.50
FR-01P	Franklin	Proposed well No. 11	GW	--	--	.72
FR-02P	Franklin	Proposed Populatic Pond well	GW	--	--	.47
HL-01G	Holliston	Well No. 1 Lake Winthrop	GW	50	.05	.32
HL-02G	Holliston	Well No. 2 Maple St.	GW	40	.06	.31
HL-04G	Holliston	Well No. 4 Washington St.	GW	50	.14	.48
HL-05G	Holliston	Well No. 5 Central St.	GW	60	.45	.71
HL-06G	Holliston	Well No. 6 Brook St.	GW	60	.45	.86
HL-01P	Holliston	Proposed well No. 7	GW	--	--	.86
MD-01G	Medway	Well No. 1 Populatic St.	GW	60	.41	.38
MD-02G	Medway	Well No. 2 Oakland St.	GW	69	.14	.59
MD-03G	Medway	Well No. 3 Village St.	GW	58	.33	.60
MD-01P	Medway	Medway Proposed well	GW	--	--	.43
MF-01G	Milford	Dilla St. wells No. 1 and 2	GW	32	.02	.68
MF-02G	Milford	Clark Island wellfield	GW	34	.43	.80
MF-03G,-04G,-05G	Milford	Godfrey Brook wells	GW	52	.36	.79
MF-01S	Milford	Charles River	SW	--	.41	[2]1.57
MF-02S	Milford	Echo Lake	SW	--	1.77	[2]1.57
NF-01G	Norfolk	Well No. 1 Gold St.	GW	49	.24	.43
NF-01P	Norfolk	Proposed Mill River well	GW	--	--	1.08
WR-02G	Wrentham	Well No. 2	GW	51	.32	.72
WR-03G	Wrentham	Well No. 3	GW	61	.42	.95
WR-01P	Wrentham	Proposed well	GW	--	--	1.01

Table 2. Water withdrawals for existing and proposed municipal public-supply and large non-municipal sources and existing wastewater discharges in the upper Charles River Basin, eastern Massachusetts—*Continued*

Well identifier	Town	Source name	Source type	Well depth (feet)	Mean annual withdrawal or discharge rate, 1989–98 (Mgal/d)	Maximum permitted withdrawal rate (Mgal/d)
Large Non-Municipal Withdrawals						
NEA-01G,-02G, -03G	Bellingham	Northeast Energy Association wells No. 1, 2 and 3	GW	--	0.53	[3]0.66
NEA-04G, -05G	Bellingham	Northeast Energy Association wells No. 4 and 5	GW	--	.01	[3].66
GECC-01S	Holliston	Glen Ellen Country Club Bogastow Brook	SW	--	.04	.16
FCC-01G	Franklin	Franklin Country Club well	GW	--	.06	[4].20
FCC-01S	Franklin	Franklin Country Club reservoir	SW	--	.04	[4].20
MGCC-01G	Holliston	Maplegate Country Club well	GW	--	.06	.15
Wastewater Discharges						
MTF	Milford	Milford Treatment Facility	--	--	3.56	--
CRPCD	Medway	Charles River Pollution Control District Treatment Facility	--	--	3.92	--

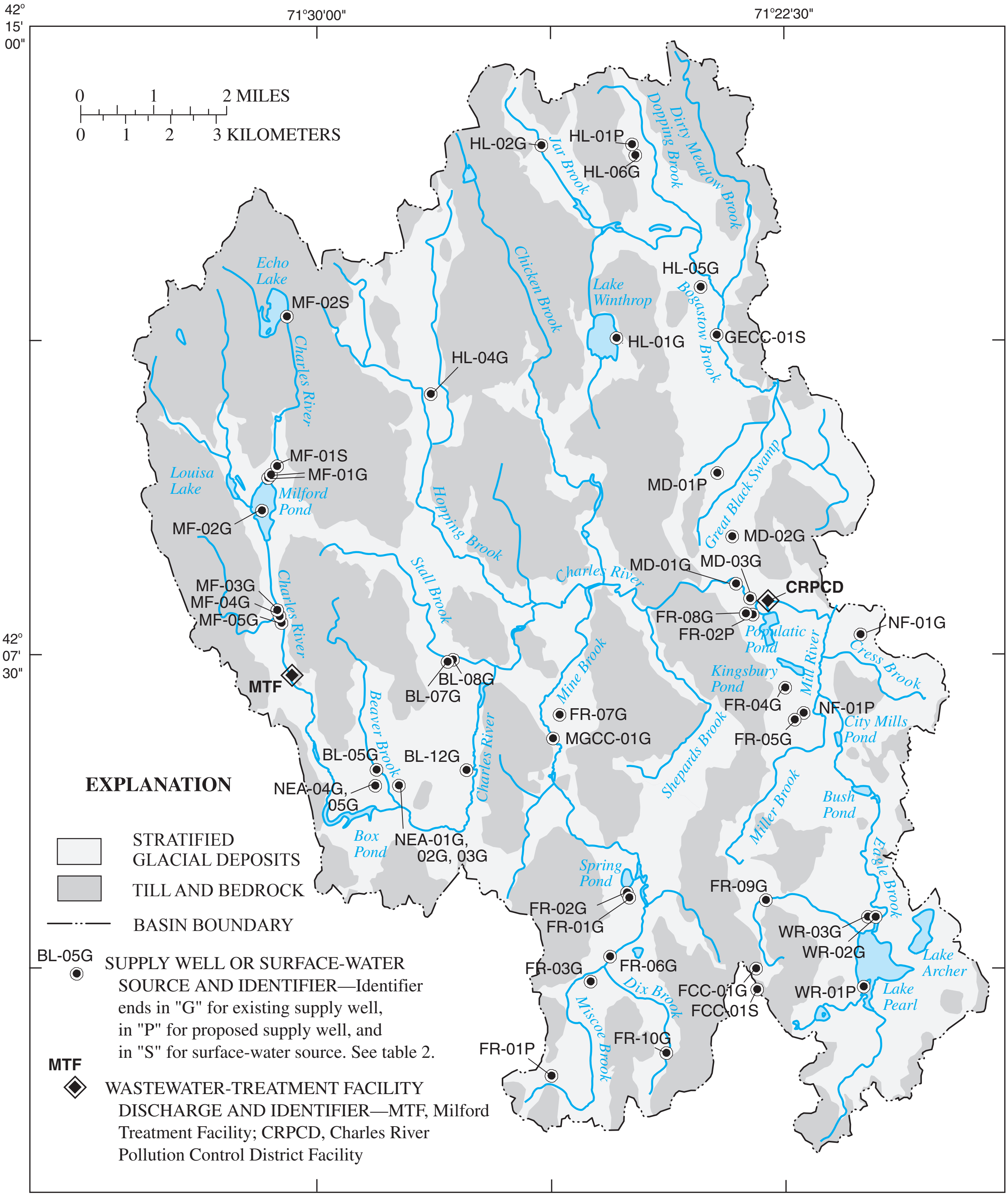
[1]Mean annual withdrawal rate determined as one-half of reported volumes for Franklin wells No. 1 and 2.
[2]Maximum permitted withdrawal rate is combined rate for Charles River and Echo Lake sources.
[3]Maximum permitted withdrawal rate includes Northeast Energy Association wells No. 1 through 5.
[4]Maximum permitted withdrawal rate includes Franklin Country Club well and reservoir.

Franklin and Milford each accounted for 30 percent of total annual withdrawals in the study area, and the six remaining towns each accounted for about 10 percent or less. Three new sources came on-line during this period, Bellingham well No. 12 (1997) and Franklin wells No. 9 and No. 10 (1990).

Withdrawals by large nonmunicipal sources that are registered or permitted under the WMA were variable as determined from the limited available data. Water was withdrawn for golf-course irrigation ("Country Club" sources, table 2) from April to October. Some of the water withdrawn for golf-course irrigation likely was returned to the aquifer as recharge. Withdrawals for Northeast Energy Association, a power-generating facility, were 100 percent consumptive (N.B. Pickering, Charles River Watershed Association, written commun., 2001) and were evenly distributed throughout the year. Several other WMA-registered or permitted users that are not listed in table 2 are in the study area, but location or pumping data were limited or unavailable for these sources; several may not be active. Total withdrawals from 1989 to 1998 for the non-municipal sources estimated with available data averaged 0.74 Mgal/d (table 2).

Water also is withdrawn from the stratified glacial and bedrock aquifers in the study area by domestic users and by small private suppliers. Water withdrawn from low-yielding wells for domestic supply generally is recharged to the aquifer through on-site septic systems, minus consumptive use. Consumptive use results in a small net (local) loss of water to the aquifer in the areas served by private supply. Comparison of water-distribution and sewer lines also indicates that there are minimal areas of private supply that are sewered, especially in the areas of stratified deposits (fig. 6).

Water is transferred in and out of the study area by five municipal supply systems (fig. 4). Milford Water Company exported on average 0.13 Mgal/d from 1989 to 1998, or about 4 percent of its total withdrawals, out of the Charles River Basin to Hopedale and Mendon. Some of this water is returned to the Charles River Basin as return flow to the Milford Water Treatment Facility (MTF). Bellingham, whose area is in both the Blackstone and Charles River Basins, distributes water throughout the town from supply wells in both basins. A net transfer of water into the Charles River Basin results, which equalled about 0.26 Mgal/d, or 17 percent of total withdrawals for Bellingham sources, in 1996 (D.F. DiMartino, Town of Bellingham, oral commun., 1999).



From USGS and MassGIS data sources, Geographic Projection, Spheroid GRS 1980, Datum NAD 83

Figure 3. Supply wells, surface-water sources, and wastewater discharges in the upper Charles River Basin, eastern Massachusetts.

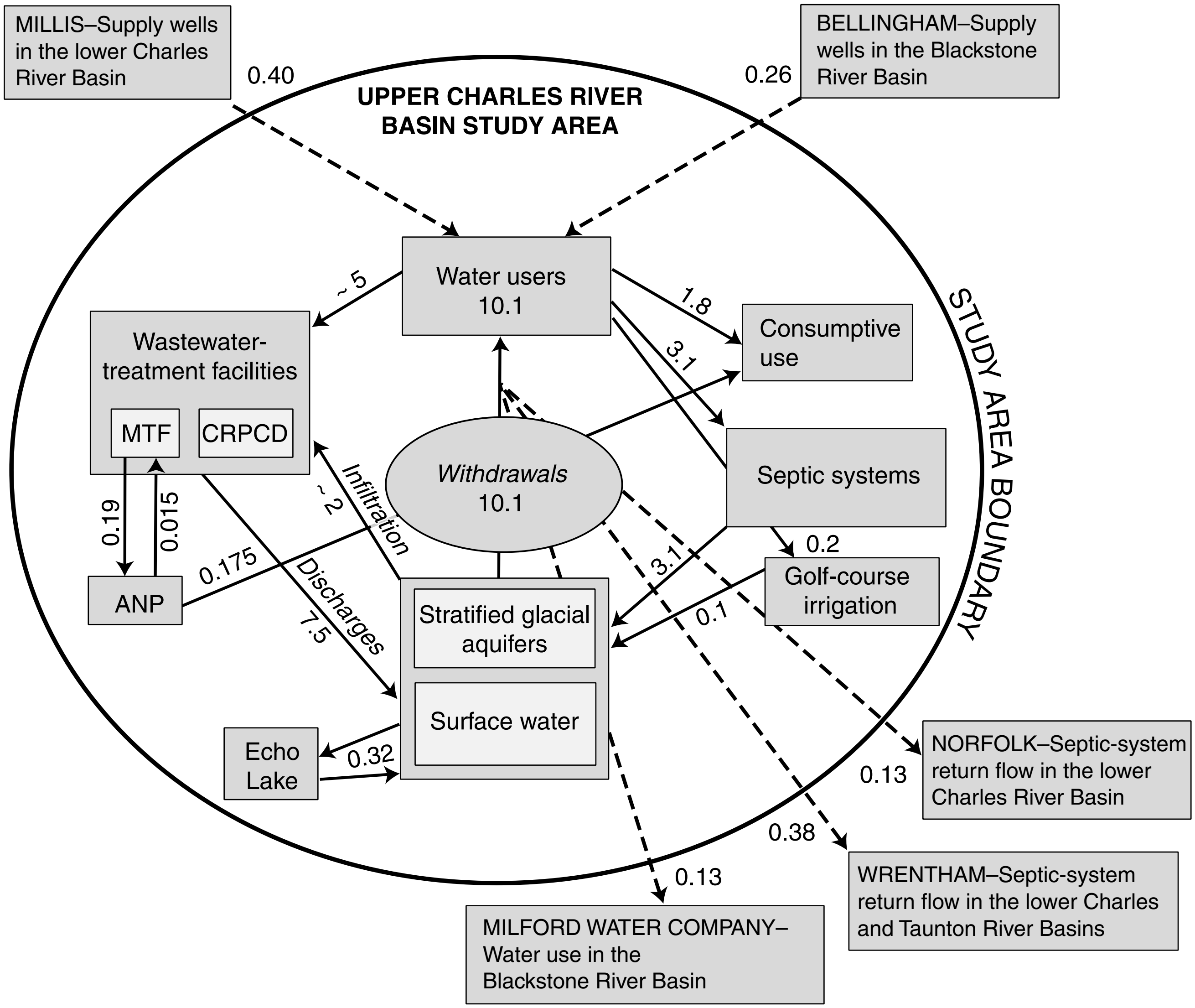


Figure 4. Summary of water withdrawals, discharges, and transfers in the upper Charles River Basin, eastern Massachusetts, 1989–98, in millions of gallons per day. Dashed lines indicate transfers across the study-area boundary. (ANP, American National Power; CRPCD, Charles River Pollution Control District Treatment Facility; MTF, Milford Treatment Facility.)

Similarly, pumping from supply wells in Millis, Norfolk, and Wrentham results in net transfers of water across the study-area boundary. These transfers were estimated from the proportions of total land area and pumped volumes within and outside of the study area for each town. The fact that all three supply wells for Millis are outside of the study area results in an estimated transfer of 0.40 Mgal/d to the study area. Pumping in Norfolk and Wrentham results in estimated net transfers of 0.13 and 0.38 Mgal/d, respectively, out of the study area. These small transfers into and out of the study area balance one another, such that water use (consumptive plus non-consumptive) by municipal and large non-municipal users in the study area (10.1 Mgal/d) is about equal to total municipal and large non-municipal withdrawals (fig. 4 and table 2).

Within the study area, water is redistributed within towns by public-water systems. Populations served by municipal water supplies ranged by town from about 50 to more than 90 percent (table 3). In Milford, Franklin, Medway and Millis, sewer systems are extensive and serve about half or more of the towns' populations (table 3 and fig. 6). Much of the water used for municipal supply in these towns is transported to treatment facilities that discharge to the Charles River, minus the small amount that is used

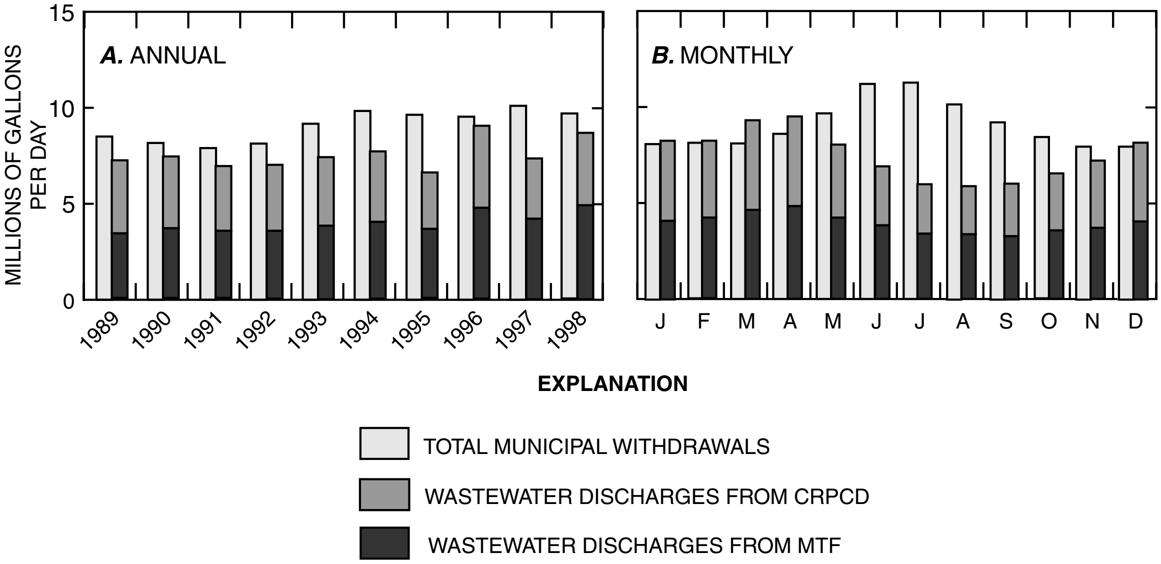
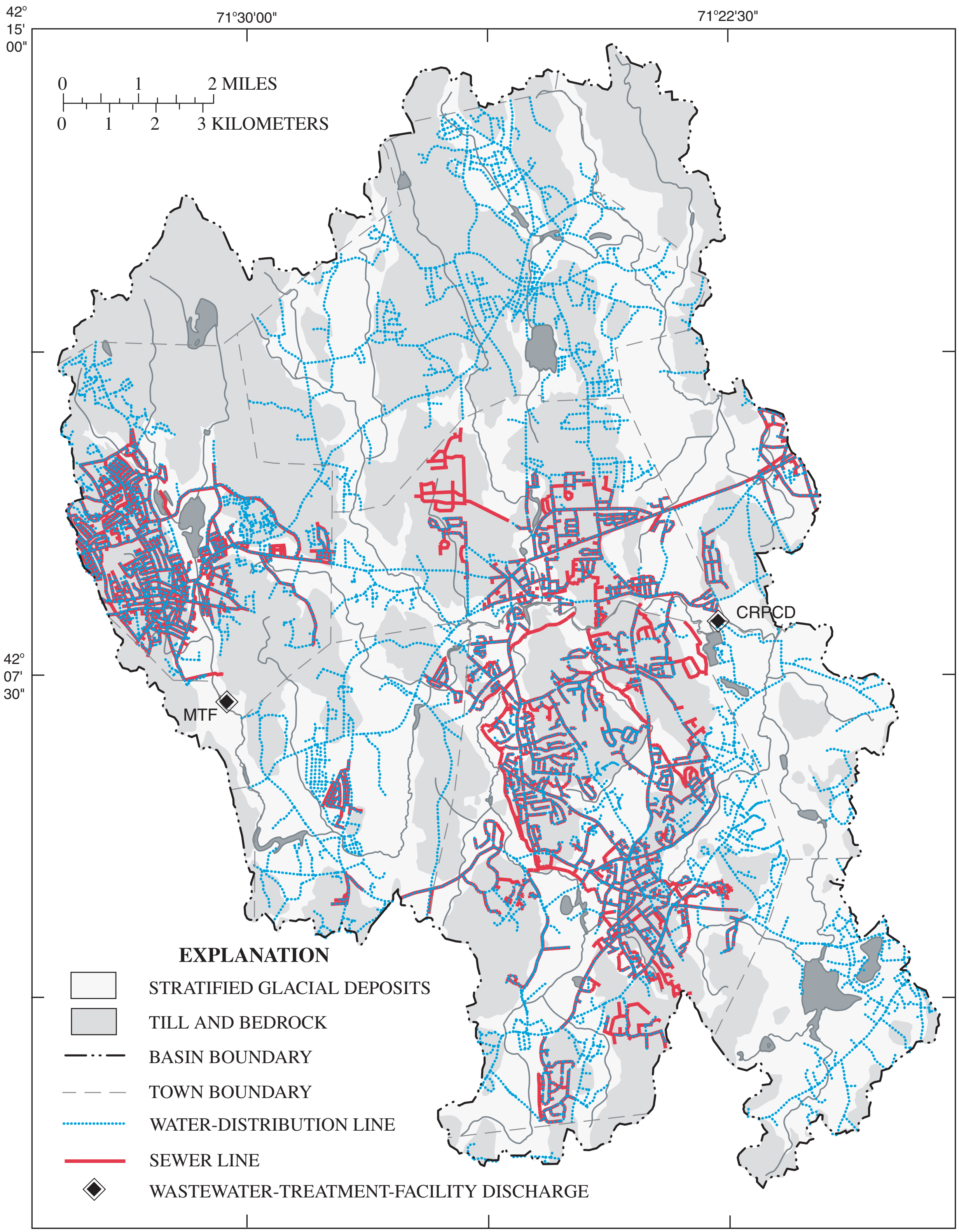


Figure 5. Average annual and monthly municipal withdrawals and wastewater discharges in the upper Charles River Basin, eastern Massachusetts, 1989–98: (*A*) Average annual municipal withdrawals and wastewater discharges, and (*B*) Average monthly municipal withdrawals and wastewater discharges. (CRPCD, Charles River Pollution Control District Treatment Facility; MTF, Milford Treatment Facility.)

consumptively. Wastewater from Milford is treated and discharged at the MTF in Hopedale, Mass., and wastewater from Medway, Franklin, Millis, and Bellingham is treated and discharged at the Charles River Pollution Control District facility (CRPCD) in Medway (fig. 3). About half of the water transferred into the study area for use in Millis also is discharged from the CRPCD; this estimate is based on town-wide percentages of sewered populations. Bellingham, Holliston, Norfolk, and Wrentham currently have little or no sewering (table 3 and fig. 6). In these towns and in unsewered parts of the other towns, water withdrawn from the sand and gravel aquifers from municipal sources or imported from outside the study area is recharged as return flow through residential and commercial septic systems.

Water also is redistributed in Milford by specific water-supply and wastewater-management practices. Water is withdrawn in winter months (January, February, and March) from the Charles River above Milford Pond in excess of that used for immediate demand and is pumped into Echo Lake Reservoir. This transfer averaged 0.32 Mgal/d in 1998–2000 (H.C. Papuga, Milford Water Company, written commun., 2000). Wastewater from the MTF is diverted to a power-generation facility, American National Power (ANP), for use as cooling water (ENSR Corporation, 2000). This diversion is allocated by regulatory permit in terms of flow in the Charles River, such that flows of 3 ft^3/s or greater must be maintained at the streamgaging station immediately downstream of the MTF discharge. Water use at the ANP power-generating facility is about 80 percent consumptive; the unused portion is returned to the MTF. The diversion began in 1994 and has averaged 0.43 Mgal/d annually from 1995 to 1998 (0.19 from 1989 to 1998); monthly averages range from less than 0.1 to 0.9 Mgal/d (Mark Gerath, ENSR Corporation, written commun., 1998).

Wastewater return flow from the CRPCD and MTF averaged 3.92 and 3.56 Mgal/d, respectively, from 1989 to 1998 (fig. 5*C*). Discharges at the MTF were variable but essentially unchanged during that period, whereas discharges at the CRPCD increased by about 40 percent. Increased discharge from the CRPCD likely resulted from the sewer extensions in the towns served by the facility (table 3). At both treatment facilities, flows were higher in winter and spring months, particularly March, April, and May, than at other times of the year (fig. 5*D*). These high flows likely result because of an increased infiltration of ground water into the sewer lines during periods of seasonally high water levels. Discharges from the treatment facilities represent substantial fractions of flow in the Charles River at both discharge locations.



From USGS and MassGIS data sources, Geographic Projection, Spheroid GRS 1980, Datum NAD 83

Figure 6. Water and sewer systems in the upper Charles River Basin, eastern Massachusetts.

Table 3. Population on public water and sewer in eight towns in the upper Charles River Basin, eastern Massachusetts, 1990 and 2000

[Total population from U.S. Census Bureau data, obtained from Commonwealth of Massachusetts, 2001, 2002. Public-water and -sewer population data from N.B. Pickering, Charles River Watershed Association, written commun., 2000]

Town	Proportion of town in study area, in percent	Total population		Population on public water, in percent		Population on public sewer, in percent	
		1990	2000	1990	2000	1990	2000
Bellingham	52	14,877	15,314	96	96	5	29
Franklin	91	22,095	29,560	76	80	51	61
Holliston	100	12,926	13,801	97	98	0	0
Medway	100	9,931	12,448	79	73	37	45
Milford	86	25,355	26,799	99	94	75	80
Millis	46	7,613	7,902	85	85	40	60
Norfolk	31	9,270	10,460	40	58	0	0
Wrentham	18	9,006	10,554	80	80	0	0

At the MTF, the annual average discharge volume corresponds to 5.5 ft^3/s, which is about 20 percent of the average annual flow at the ANP stream-gaging station for water years[1] 1996 through 1998. The annual discharge at the CRPCD corresponds to 6.1 ft^3/s, about 6 percent of average annual flow at the Medway stream-gaging station for water years 1997 and 1998. During low-flow periods, the relative magnitudes of the wastewater discharges compared to total streamflow are considerably higher than annual averages.

Rates of return flow from septic systems were determined by CWRA as part of a recharge analysis for this study and for a CRWA watershed model of the Charles River Basin (N.B. Pickering, Charles River Watershed Association, written commun., 2001). Water and sewer lines (fig. 6) were used to identify areas receiving public water and served by on-site septic systems (PWSS). Return-flow rates were assigned to the PWSS areas, which corresponded to residential, commercial, and other urban land uses. Rates were based on estimates of population density (5.4 people per acre for high-density residential and 2.2 people per acre for other land uses) and per capita water use (80 gallons per person per day). This approach is similar to that used in a USGS watershed model of the Ipswich River Basin in eastern Massachusetts (Zarriello and Ries, 2000). Septic-system return-flow rates thus calculated were from 2.1 to 3.2 in/yr for commercial, from 2.2 to 3.2 in/yr for low-density residential, and from 5.8 to 7.8 in/yr for high-density residential land uses. Summed across the entire study area, septic-system return flow was 3.1 Mgal/d. This volume includes some septic-system return flow of water that originated from outside the study area in Bellingham, Millis, Norfolk, and Wrentham. The return-flow volumes calculated in this way from population density and water uses were checked against measured withdrawals and wastewater discharges, and found to be consistent.

Ground-water infiltration to sewers is another likely transfer of water within the study area. However, it is difficult to quantify. Ground-water infiltration to sewers is generally considered to account for as much as 10 to 60 percent of a town's wastewater flow to a treatment facility. As described previously, ground-water infiltration to sewers is greatest during times of seasonally high water levels; it also varies spatially and is affected by physical characteristics of the sewer infrastructure. One way of estimating ground-water infiltration to sewers in the study area is to compare wastewater discharge rates in winter and spring, when water levels are high, with discharge rates in summer and fall. Average monthly discharges from December to May averaged 2.3 Mgal/d greater than average monthly discharges from June to November for the treatment facilities in the study area. This difference is about 30 percent of total wastewater discharges and is probably a reasonable estimate for ground-water infiltration to sewers in the study area.

[1]A water year begins on October 1 of the year noted and ends on September 30 of the following year.

Streamflow and Water Levels

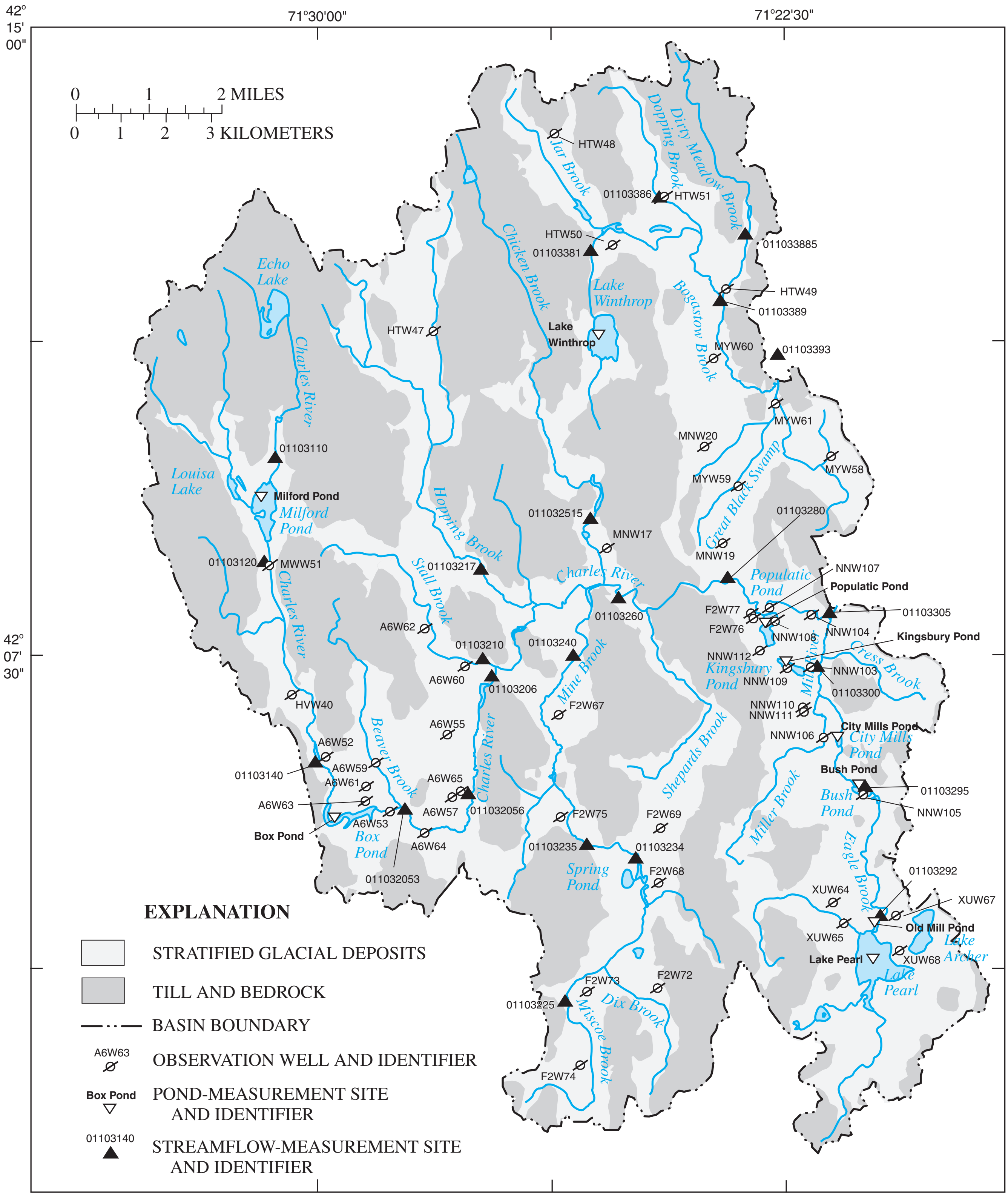
Streamflow and water levels were measured during this study at 24 streamflow-measurement sites, 47 observation wells, and 9 ponds (fig. 7). Measurement sites were distributed throughout the area of stratified glacial aquifers; some were clustered in areas of particular interest for purposes of water-resource management (near Populatic and Kingsbury Ponds) or located at study-area boundaries. Streamflow and water-level measurement-site characteristics are listed in tables 4 and 5. Many of the streamflow sites were partial-record stations that had been measured in previous low-flow investigations (Ries, 1993, 1994, 1997, 1999), and historical data were available for these sites (table 4). Observation wells were installed in 1998 and 1999 for this study, except for two wells in Franklin (F2W76 and 78) and several wells in Norfolk (NNW108-112), which were existing wells at aquifer test sites. Measurements were made monthly at most sites from late summer 1999 through September 2000. At sites in the Dopping Brook aquifer area in east Holliston, measurements were made from December 1999 to November 2000. Streamflow measurements generally were made after several days of little or no precipitation in order to represent base-flow conditions. Streamflow and water-level data collected during this study are stored in the USGS NWIS database.

Continuous streamflow data were available for two sites on the Charles River in the study area, but the data were limited. The USGS maintains a stream-gaging station in Medway (station number 01103280) and a stream-gaging station is operated by ENSR Corporation for American National Power (ANP) at the MTF in Hopedale (fig. 1). The ANP stream-gaging station is operated primarily for low-flow data, and its stage-discharge relation is not well defined at moderate and high flows (ENSR Corporation, 2000; R.S. Socolow, U.S. Geological Survey, oral commun., 2001). Periods of record for both stream-gaging stations are short, extending from October 1997 to the present (2002) for the USGS stream-gaging station and from July 1994 to the present for the ANP stream-gaging station.

Water year 2000, during which most of the streamflow and water-level measurements were made, was about average in terms of total annual precipitation, based on records from five NOAA weather stations in and near the study area (National Oceanic and Atmospheric Administration, 2001). Average streamflows during water year 2000 at several nearby stream-gaging stations were within 20 percent of long-term average flows and were neither consistently higher nor lower than average (table 6). Ground-water levels at nearby observation wells were slightly higher than long-term averages (table 7). Thus, hydrologic conditions during the measurement period of this study were not exceptionally wet or dry. Streamflow in the Charles River at the Medway stream-gaging station (based on continuous stream-gaging station data; fig. 8) averaged 103 ft^3/s during water year 2000, with mean daily streamflow in August 2000 equal to 24 ft^3/s. The summer of 1999, preceding water year 2000, was exceptionally dry; however, this period of extended low base-flow conditions and low ground-water levels was ended by two unusually large storms in September (including Tropical Storm Floyd), which increased stream base flow and ground-water levels in the study area to average conditions.

Ground-water levels in the stratified glacial aquifers generally follow topography and decrease from west to east and from upstream to downstream locations in the study area. Measured mean annual ground-water levels ranged from more than 250 ft above sea level in headwater basins in the western part of the study area to less than 140 ft above sea level in low-lying areas draining to Bogastow Brook and along the Charles River near the eastern boundaries of the study area (table 5). Ground-water levels typically reached maximum levels in late spring, and water levels declined through summer months; this pattern reflects seasonal patterns in recharge and evapotranspiration (fig. 9). Measured annual fluctuations in ground-water level ranged from more than 3 ft, in wells located on hillsides and near boundaries of the stratified glacial aquifers with till and bedrock uplands (for example, F2W72 and HTW48; fig. 9), to about 1 ft, in wells in low-lying areas near streams and ponds (for example, MWW51 and HTW 51; figs. 7 and 9).

Measured streamflows and water levels during the study period were used to estimate average flows for the period 1989–98 ("current average conditions"). Base flow is the component of streamflow resulting from ground-water discharge and excludes direct runoff; this flow is the component of streamflow that is calculated in the ground-water-flow model.



From USGS and MassGIS data sources, Geographic Projection, Spheroid GRS 1980, Datum NAD 83

Figure 7. Observation wells, pond-measurement sites, and streamflow-measurement sites in the upper Charles River Basin, eastern Massachusetts.

Table 4. Drainage-area characteristics of streamflow-measurement sites in the upper Charles River Basin, eastern Massachusetts

[Site locations shown in figure 7. mi^2, square miles; --, not determined]

Station No.	Station name	Years of historical data	Total number of measure-ments	Drainage-area characteristics: Area (mi^2)	Area of sand and gravel (percent)	Mean slope (percent)
Mainstem Charles River						
01103110	Charles River above Milford Pond near Milford	none	15	3.3	3	3.4
01103120	Charles River at Milford	1967	14	8.3	17	3.2
01103140	Charles River at Factory Pond at South Milford	none	14	12.8	20	3.1
011032053	Charles River at Bellingham	none	15	17.9	28	2.8
011032056	Charles River near Bellingham	none	14	19.3	31	2.7
01103206	Charles River at North Bellingham	1982–83	22	21.0	35	2.6
01103260	Charles River at West Medway	1968–71, 1982–83	31	60.0	38	2.4
01103280	Charles River at Medway	1997–2000	[1]1,141	65.2	38	2.4
01103305	Charles River near Millis	1968, 1982–83, 1989–90	31	83.8	45	2.4
Tributaries to Charles River						
01103210	Stall Brook at North Bellingham	1968–70, 1983	30	3.9	43	1.9
01103217	Hopping Brook near West Medway	1968–71, 1983, 1989–90	30	9.9	40	2.4
01103225	Miscoe Brook at Washington Street near Franklin	1968–70, 1983	26	2.2	64	2.3
01103234	Mine Brook at Franklin	1983	11	8.6	50	2.5
01103235	Mine Brook near Franklin	1967, 1983, 1989	23	2.9	52	2.5
01103240	Mine Brook near Franklin	1968–70, 1983, 1989–91	35	14.2	50	2.5
011032515	Chicken Brook below Milk Pond near West Medway	none	15	6.1	13	2.3
01103292	Eagle Brook near Wrentham	1967, 1983	23	7.5	68	2.7
01103295	Mill River below Bush Pond near City Mills	none	15	9.4	72	2.8
01103300	Mill River near Norfolk	1968–71, 1983, 1989–90	29	13.8	--	--
Bogastow Brook and Tributaries						
01103381	Winthrop Canal at Lindon Street, Holliston	none	11	2.7	11	1.9
01103386	Dopping Brook at Whitney Street, Holliston	none	10	1.4	49	4.5
011033885	Dirty Meadow Brook at Hollis Street, Holliston	none	11	2.1	12	1.8
01103389	Bogastow Brook at Central Street, Holliston	none	11	12.5	38	2.7
01103393	Bogastow Brook below Great Black Swamp near Millis	1983	15	21.0	46	2.1

[1]Mean daily values at streamflow-gaging station for the period indicated.

Table 5. Characteristics and water levels at observation wells and ponds in the upper Charles River Basin, eastern Massachusetts

[Site locations shown in figure 7. All wells for which well depth is given are screened at bottom; screened interval equal to 5 feet. **Latitude and longitude**: In degrees, minutes, and seconds. -- not applicable or not known]

Well identifier or pond name	Latitude ° ′ ″	Longitude ° ′ ″	Well depth (feet below and surface)	Mean depth to water (feet below land surface)	Mean water-level elevation (feet above sea level)	
					Water year 2000	Estimated, 1989–98
Bellingham						
A6W52	42 06 15	071 29 50	17.0	7.88	229.76	229.70
A6W53	42 05 35	071 28 48	57.8	40.15	212.70	212.69
A6W55	42 06 31	071 27 52	17.2	6.55	208.49	208.47
A6W57	42 05 46	071 27 48	15.9	11.14	220.94	220.82
A6W59	42 06 09	071 28 59	28.9	8.62	215.56	215.51
A6W60	42 07 20	071 27 36	30.5	13.13	205.78	205.69
A6W61	42 05 54	071 29 11	51.2	35.79	219.05	218.99
A6W62	42 07 47	071 28 14	24.4	12.73	210.97	210.96
A6W63	42 05 36	071 28 48	29.2	10.30	220.26	220.27
A6W64	42 05 20	071 28 14	26.4	9.88	224.13	224.09
A6W65	42 05 50	071 27 40	22.4	6.84	203.28	203.16
Box Pond	42 05 41	071 29 05	--	--	221.34	221.32
Franklin						
F2W67	42 06 44	071 26 06	55.4	20.17	178.90	179.01
F2W69	42 05 23	071 24 24	32.4	21.51	257.99	258.08
F2W72	42 03 28	071 24 30	27.3	20.33	286.57	286.51
F2W73	42 03 26	071 25 38	45.2	29.99	254.31	254.38
F2W74	42 02 34	071 25 45	37.5	11.02	265.48	265.50
F2W75	42 05 32	071 26 03	29.0	19.96	188.51	188.51
F2W76	42 07 54	071 22 58	--	6.07	127.17	127.09
F2W77	42 07 54	071 22 58	--	5.21	126.96	126.84
Holliston						
HTW47	42 11 22	071 28 09	--	4.05	238.71	238.56
HTW48	42 13 42	071 26 09	20.0	8.88	205.61	205.24
HTW49	42 11 50	071 23 44	28.2	12.30	143.97	143.84
HTW50	42 12 22	071 25 13	12.0	6.71	167.16	167.07
HTW51	42 12 56	071 24 23	37.2	5.66	149.02	149.00
Lake Winthrop	42 11 35	071 25 37	--	--	[1]174.41	174.60
Hopedale						
HVW40	42 06 59	071 30 22	21.8	12.42	238.32	238.36

Table 5. Characteristics and water levels at observation wells and ponds in the upper Charles River Basin, eastern Massachusetts—*Continued*

Well identifier or pond name	Latitude ° ′ ″	Longitude ° ′ ″	Well depth (feet below and surface)	Mean depth to water (feet below land surface)	Mean water-level elevation (feet above sea level)	
					Water year 2000	Estimated, 1989–98
Medway						
MNW17	42 08 45	071 25 19	26.8	9.49	183.07	183.02
MNW19	42 08 48	071 23 27	33.5	28.33	137.88	137.75
MNW20	42 09 57	071 23 44	32.8	11.57	140.86	140.71
Milford						
MWW51	42 08 32	071 30 45	31.6	13.16	248.70	248.71
Milford Pond	42 08 39	071 30 47	--	--	265.71	265.69
Millis						
MYW58	42 09 50	071 21 42	33.5	12.62	144.75	144.71
MYW59	42 09 29	071 23 12	33.3	5.03	135.69	135.57
MYW60	42 10 59	071 23 35	21.9	7.01	138.51	138.48
MYW61	42 10 28	071 22 36	26.7	5.26	131.82	131.64
Norfolk						
NNW103	42 07 19	071 22 03	29.9	5.79	130.24	130.25
NNW104	42 07 57	071 22 20	28.3	7.57	128.79	128.74
NNW105	42 05 48	071 21 12	24.1	9.63	173.17	173.31
NNW106	42 06 29	071 21 50	35.5	15.74	140.39	140.39
NNW107	42 08 02	071 22 41	--	5.98	126.75	126.50
NNW108	42 07 52	071 22 36	--	2.94	127.01	126.85
NNW109	42 07 18	071 22 25	--	15.67	129.37	129.31
NNW110	42 06 48	071 22 10	--	6.66	132.55	132.55
NNW111	42 06 46	071 22 06	--	10.41	132.11	132.11
NNW112	42 07 31	071 22 52	--	17.13	130.95	130.81
Bush Pond	42 05 51	071 21 04	--	--	173.77	173.84
City Mills Pond	42 06 34	071 21 43	--	--	148.87	148.83
Kingsbury Pond	42 07 17	071 22 13	--	--	129.39	129.74
Populatic Pond	42 07 44	071 22 42	--	--	[2]126.98	127.27
Wrentham						
XUW64	42 04 30	071 21 41	14.2	16.37	231.47	231.39
XUW65	42 04 15	071 21 31	43.4	25.60	205.73	205.65
XUW67	42 04 21	071 20 40	62.1	56.28	195.41	195.35
XUW68	42 03 56	071 20 37	40.2	22.20	204.20	204.20
Lake Pearl	42 04 11	071 21 08	--	--	[2]199.55	199.60
Old Mill Pond	42 04 21	071 20 59	--	--	194.43	194.33

[1]Data for July, August, September, and November 2000 only.
[2]No data for January, April, and May 2000.

Table 6. Drainage-area characteristics, streamflow, and estimated base flow at stream-gaging stations near the upper Charles River Basin, eastern Massachusetts

[See Socolow and others (2001) for location information; ft^3/s, cubic feet per second; in/yr, inches per year; mi^2, square miles; --, not determined]

Station No.	Station name	Period of record	Drainage area characteristics: Area (mi^2)	Area of sand and gravel (percent)	Mean slope (percent)
01096000	Squannacook River near West Groton, Mass.	1950–present	63.7	--	--
01097300	Nashoba Brook near Acton, Mass.	1964–present	12.2	61	2.3
01105730	Indian Head River at Hanover, Mass.	1967–present	30.3	68	2.3
01105600	Old Swamp River near South Weymouth, Mass.	1966–present	4.5	27	.5
01109000	Wading River near Norton, Mass.	1926–present	43.3	59	--
01111300	Nipmuc River near Harrisville, R.I.	1965–present	15.9	28	3.1
01175670	Sevenmile River near Spencer, Mass.	1961–present	8.7	13	5.4

Station No.	Mean annual streamflow (ft^3/s): Period of record (years)	1989–98	Water year 2000	Estimated mean annual base flow: Period of record (in/yr)	Period of record (ft^3/s)	1989–98 (in/yr)	1989–98 (ft^3/s)	Water year 2000 (in/yr)	Water year 2000 (ft^3/s)
01096000	113	120	119	17.5	82.1	18.7	87.5	19.5	91.2
01097300	20.5	20.5	19.6	17.7	15.9	17.1	15.4	--	--
01105730	66.2	66.4	58.7	20.5	45.7	21.4	47.7	--	--
01105600	9.2	9.2	9.1	16.9	5.6	16.3	5.4	17.8	5.9
01109000	73.7	76.9	61.1	19.3	61.6	20.4	65.0	18.9	60.2
01111300	30.4	32.5	31.4	19.1	22.4	20.0	23.5	19.4	22.8
01175670	15.1	16.7	17.0	18.9	12.1	20.7	13.3	19.5	12.5

Table 7. Characteristics and water levels at long-term observation wells near the upper Charles River Basin, eastern Massachusetts

[**Well location**: See Socolow and others (2001) for additional location information]

Well identifier	Well location	Period of record	Well-screen interval (feet below land surface)	Mean depth to water (feet below land surface)	Mean water-level elevation (feet above sea level): Period of record	1989–98	Water year 2000
DVW10	Dover, Mass.	1964–present	52–54	33.35	126.65	126.72	126.74
FXW3	Foxborough, Mass.	1964–present	30–32	19.09	270.91	270.93	270.94
NNW27	Norfolk, Mass.	1965–present	16–18	6.10	153.90	154.39	154.29
NXW54	Northbridge, Mass.	1984–present	10–12	4.24	365.76	365.90	365.91
NSW21	North Smithfield, R.I.	1947–present	[1]16	8.04	230.64	230.87	231.24

[1]Well open at bottom.

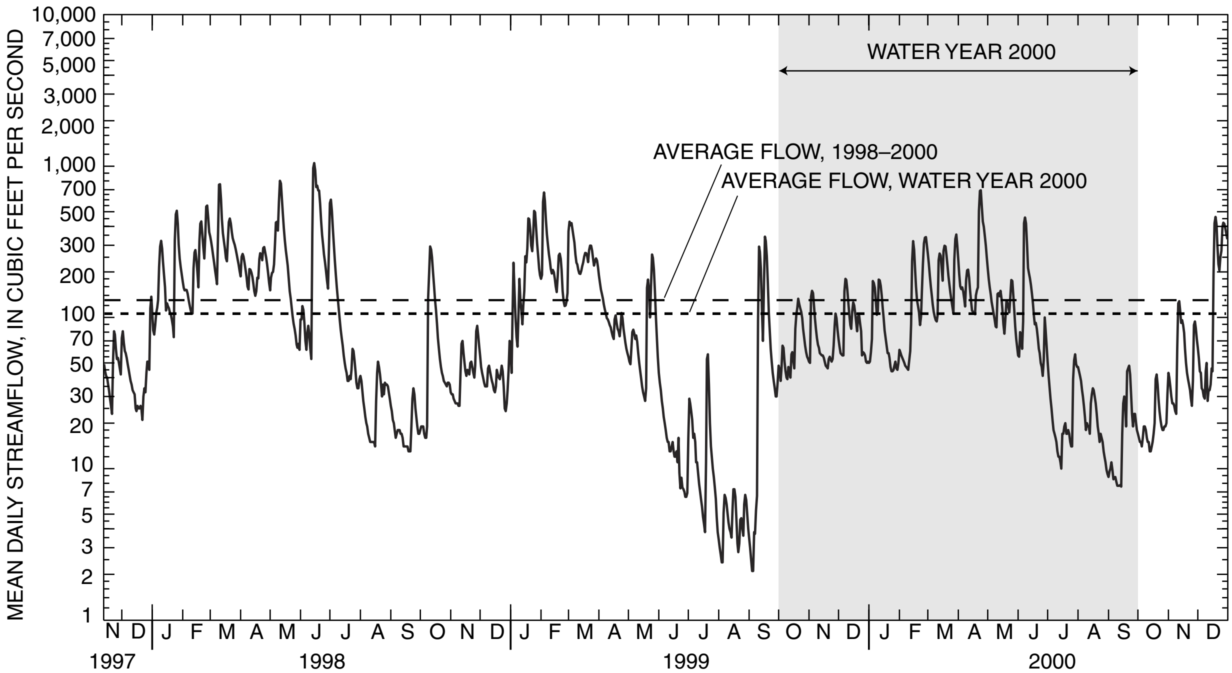


Figure 8. Streamflow in the Charles River at the U.S. Geological Survey gaging station at Medway, Massachusetts (01103280) for the period November 1997 through December 2000.

Average base flows and water levels were estimated by (1) determining a functional relation between (a) flows and water levels at study measurement sites and (b) same-day flows (mean daily) and water levels at nearby long-term continuous stream-gaging stations and observation wells; then, (2) determining current average conditions at the long-term sites; and, finally (3) using the functional relation to estimate current average conditions at the study measurement sites. Stream-gaging stations and observation wells used in the analysis are listed in tables 6 and 7. Selected long-term stream-gaging stations were near the study area and largely unregulated. Their drainage areas also spanned ranges of area and percent area covered by stratified glacial deposits similar to those for flow-measurement sites (table 4). Selected long-term observation wells also were nearby, screened in sand and gravel aquifers, and included a range of depths to water that was representative of the study area.

Scatterplots of flows and water levels were used to determine if the relations of data at study measurement sites and at long-term sites were linear and to identify long-term sites that best correlated with each measurement site. For each measurement site, two to five long-term sites were identified. These included sites for which relations were log-linear and R^2 values of correlations (on log-transformed data for flows) were greater than 0.8 (for a few sites with very good correlations overall, an R^2 value of greater than about 0.9 was used as the selection criterion).

Equations relating flows and water levels at study and long-term sites were developed with the Maintenance of Variance Extension, Type 1 (MOVE.1) method (Hirsch, 1982). This method is applicable where relations are log-linear between same-day discharges at partial-record sites and nearby stream-gag-ing stations; it is commonly used to estimate low-flow statistics at partial-record sites (Bent, 1995, 1999; Ries and Friesz, 2000). The MOVE.1 equation is:

$$y_i = y_m + \frac{s_y}{s_x} \cdot (x_i - x_m) \qquad (1)$$

where

y_i = streamflow or water level statistic of interest at the measurement site,

y_m = mean instantaneous streamflow or water level at the measurement site,

s_y = standard deviation of instantaneous streamflow or water level at the measurement site,

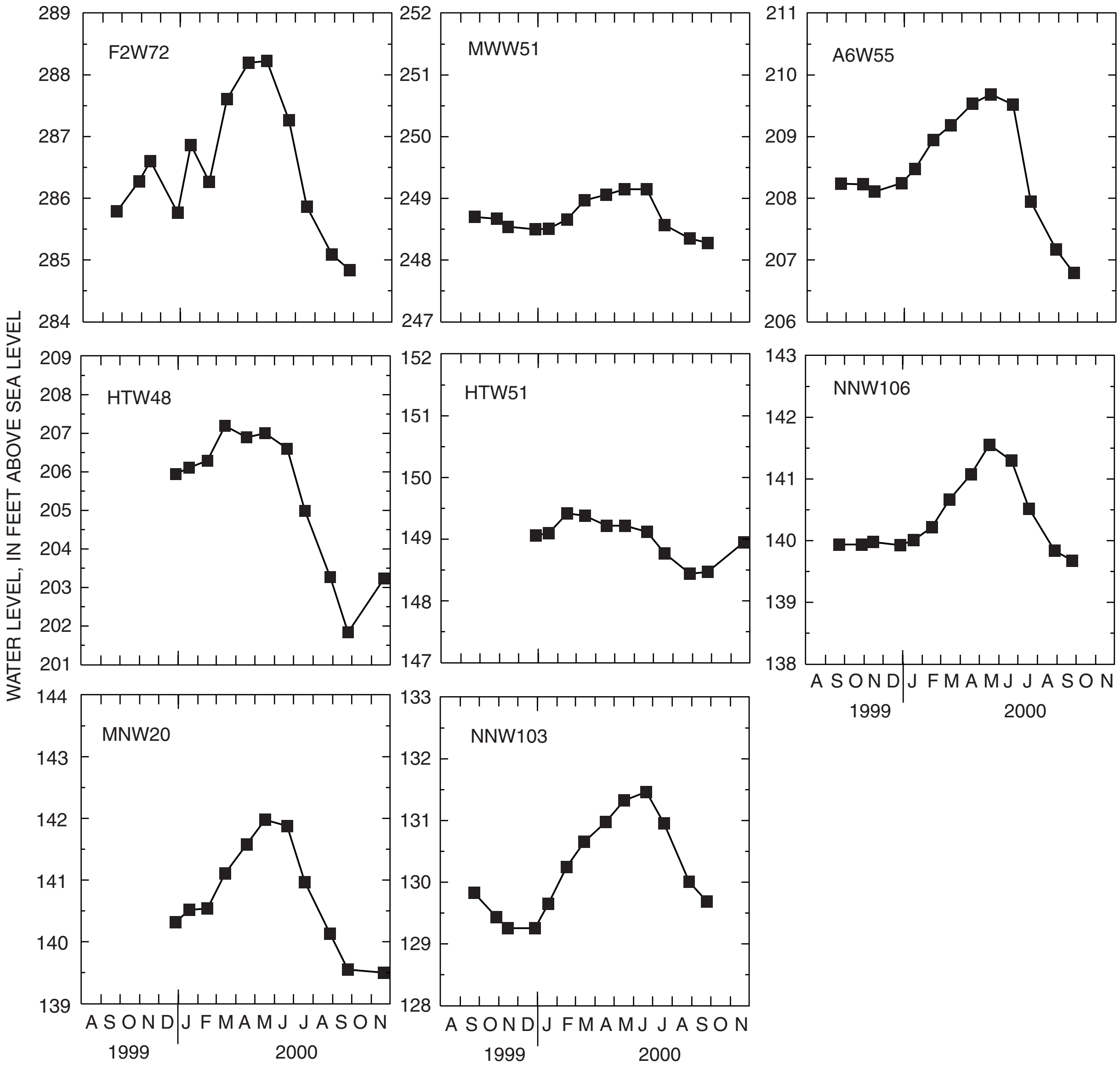


Figure 9. Ground-water levels measured in selected observation wells in the upper Charles River Basin, eastern Massachusetts, 1999–2000.

x_i = streamflow or water level statistic of interest at the long-term site,

x_m = mean daily streamflow or water level at the long-term site, and

s_x = standard deviation of mean daily streamflow or water level at the long-term site.

For the continuous stream-gaging station on the Charles River at Medway (01103280), mean daily flows were used rather than mean instantaneous streamflow (y_m in equation 1). In this study, the statistics of interest were mean annual or monthly base flow for streamflow sites and mean annual water level for observation wells. Base flow was chosen to represent streamflow because it is the base flow, or ground-water-discharge, component of streamflow that is simulated in a ground-water-flow model.

Mean annual and monthly base flow at continuous stream-gaging stations was estimated with an automated hydrograph separation technique, PART (Rutledge, 1993, 1998). The computer program PART uses streamflow partitioning and is applied to a record of daily streamflows. Base flow is equated to streamflows on days when antecedent and subsequent recession requirements are satisfied; these requirements describe the duration of substantial runoff and interflow after a streamflow peak. Linear interpolation is used to estimate base flow on days when flows do not meet these requirements. PART is applicable to basins of moderate size (about 1 to 500 mi^2) with little or no flow regulation and where ground-water underflow at the stream measurement site is negligible. Mean annual base flow for continuous stream-gaging stations estimated with PART are listed in table 6.

Mean annual and monthly base flow for 1989 to 1998 at continuous stream-gaging stations (table 6) was substituted into the MOVE.1 equations (equation 1), along with means and standard deviations of daily and instantaneous streamflows for stream-gaging stations and measurement sites, respectively, to obtain estimated 1989–98 mean annual and monthly base flow at streamflow sites. For ground-water levels, mean annual water levels at long-term observation wells for 1989 to 1998 (table 7) were used to estimate mean annual water levels at study observation wells (table 5). Two to five estimates were determined for each streamflow-measurement site and observation well; the number of estimates depended on the number of stream-gaging stations or observation wells that were identified as correlating well with each site or well, as described previously. Individual estimates from long-term sites (Q_{BF_i}, in log units) were combined to generate a weighted-average base flow (Q_{BF}, in log units) or water level as (Ries and Friesz, 2000; K.G. Ries, U.S. Geological Survey, written commun., 2001):

$$Q_{BF} = \frac{\sum_{i=1}^{n} (Q_{BF_i} / MSE_{BF_i})}{\sum_{i=1}^{n} (1 / MSE_{BF_i})} \qquad (2)$$

where MSE_{BF_i} is the mean-square error (Helsel and Hirsch, 1992, p. 227) of the individual MOVE.1 relations. Upper (CI_{high}) and lower (CI_{low}) 90-percent confidence intervals for the weighted-average base flow estimates were determined as (Tasker and Driver, 1988; K.G. Ries, U.S. Geological Survey, written commun., 2001):

$$CI_{low} = Q_{BF} / \left(10^{(1.645 \cdot \sqrt{MSE_{BF}})}\right) \qquad (3a)$$

and

$$CI_{high} = Q_{BF} \cdot \left(10^{(1.645 \cdot \sqrt{MSE_{BF}})}\right) \qquad (3b)$$

where MSE_{BF} is the weighted-average MSE, in log units, and Q_{BF}, CI_{high}, and CI_{low} are as defined previously but in arithmetic units (ft^3/s). The weighted-average MSE (MSE_{BF}) is calculated from the mean-square error of the individual MOVE.1 relations (MSE_{BF_i}) as (Ries and Friesz, 2000; K.G. Ries, U.S. Geological Survey, written commun., 2001):

$$MSE_{BF} = \frac{1}{\sum_{i=1}^{n} (1 / MSE_{BF_i})} \qquad (4)$$

Weighted-average base flow was retransformed back into arithmetic units after verification that the log retransformation bias was negligible (about 1 to 2 percent as determined by the smearing estimator, Helsel and Hirsch, 1992).

It should be noted that average base flow and water levels determined through the procedures described above do not take into account measurement errors at long-term or study-area sites, errors in the daily mean discharge records at the stream-gaging stations, or errors in interpolated water levels at long-term observation wells. In addition, the average base flow and water-level estimates obtained for study measurement sites with MOVE.1 relation with data from multiple long-term sites are not truly independent, because streamflow and water-level records at multiple long-term sites are correlated. Thus, the weighted average base flow and water-level estimates obtained with equations 2 and 4 may not be the best possible, and the actual confidence intervals are larger than those calculated with equations 3a and 3b (Ries and Friesz, 2000).

Estimated mean annual base flow and water levels are given in tables 5 and 8. Estimated mean annual base flow at the continuous stream-gaging station on the Charles River at Medway (01103280, 91.0 ft^3/s) was 88 percent of mean annual streamflow during water year 2000 (103 ft^3/s). This flow value is consistent with estimates of the proportion of streamflow that is base flow at sites of stream-gaging stations in Massachusetts and Rhode Island that drain basins with both till and bedrock uplands and stratified glacial deposits (Bent, 1995, 1999). Estimated mean annual

Table 8. Measured and estimated base flow at streamflow-measurement sites in the upper Charles River Basin, eastern Massachusetts

[**Measured base flow:** Streamflow measurements generally made after several days of little or no precipitation. August flows measured on August 28–30, 2000, unless otherwise indicated. --, not determined]

Station No.	Measured base flow, water year 2000 (ft^3/s)		Estimated base flow, 1989–98 (ft^3/s)					
			Mean annual			August		
	Mean annual, water year 2000	August, water year 2000	Flow	90-percent confidence limits		Flow	90-percent confidence limits	
				Lower	Upper		Lower	Upper
Mainstem Charles River								
01103110	2.4	0.1	2.0	1.2	3.2	0.3	0.2	0.5
01103120	4.8	.0	5.6	3.0	10.5	1.3	.7	2.5
01103140	18.7	1.2	15.8	10.6	23.7	5.6	3.7	8.3
011032053	21.4	3.4	20.1	14.7	27.5	6.4	4.6	8.7
011032056	23.3	5.5	24.2	17.2	34.0	9.2	6.5	12.9
01103206	33.0	5.6	28.8	21.4	38.9	10.4	7.7	14.1
01103260	[1]69.8	15.4	86.9	62.4	121	25.8	18.5	35.9
01103280	--	--	91.0	59.7	139	24.2	15.9	36.9
01103305	113	26.6	131	103	168	51.7	40.4	66.0
Tributaries to Charles River								
01103210	5.5	0.8	7.6	0.8	71.7	1.1	0.1	10.1
01103217	14.1	.4	14.8	5.8	37.8	1.5	.6	3.8
01103225	3.3	.3	2.5	1.5	4.2	.5	.3	.9
01103234	--	--	13.6	10.2	18.0	3.3	2.5	4.3
01103235	17.4	4.1	15.0	7.9	28.6	5.8	3.1	11.1
01103240	--	--	24.4	14.9	40.0	7.7	4.7	12.6
011032515	10.2	.7	8.1	5.5	12.1	1.7	1.1	2.5
01103292	11.7	4.1	11.3	8.0	16.0	4.7	3.3	6.6
01103295	15.5	5.9	18.4	11.9	28.4	8.9	5.8	13.8
01103300	22.8	6.7	26.0	18.3	37.0	12.4	8.7	17.5
Bogastow Brook and Tributaries								
01103381	--	0.3	3.3	2.2	4.9	0.7	0.5	1.1
01103386	--	.1	1.0	.5	2.0	.1	.1	.3
011033885	--	.2	1.6	1.1	2.2	.3	.3	.4
01103389	--	1.6	12.8	8.0	20.5	2.9	1.8	4.6
01103393	--	1.6	30.2	20.9	43.6	6.2	4.3	9.0

[1]No measurements in February or March 2000, due to ice conditions.

base flow at other study flow-measurement sites ranged from 73 to 122 percent of measured mean annual base flow, based on monthly measurements during base-flow conditions in water year 2000; measured values all were within 90-percent confidence limits of estimated values. Comparison of estimated mean monthly base flow with individual streamflow measurements indicates that the monthly estimates match seasonal variations and the magnitudes of measured flows; exceptions are due to precipitation events or snowmelt, such as the February 2000 measurements at Charles River sites (fig. 10). Mean annual water levels estimated with MOVE.1 and data from long-term observation wells for 1989–98 (table 5) were nearly identical with measured water levels during water year 2000 (the median difference was less than 0.05 ft).

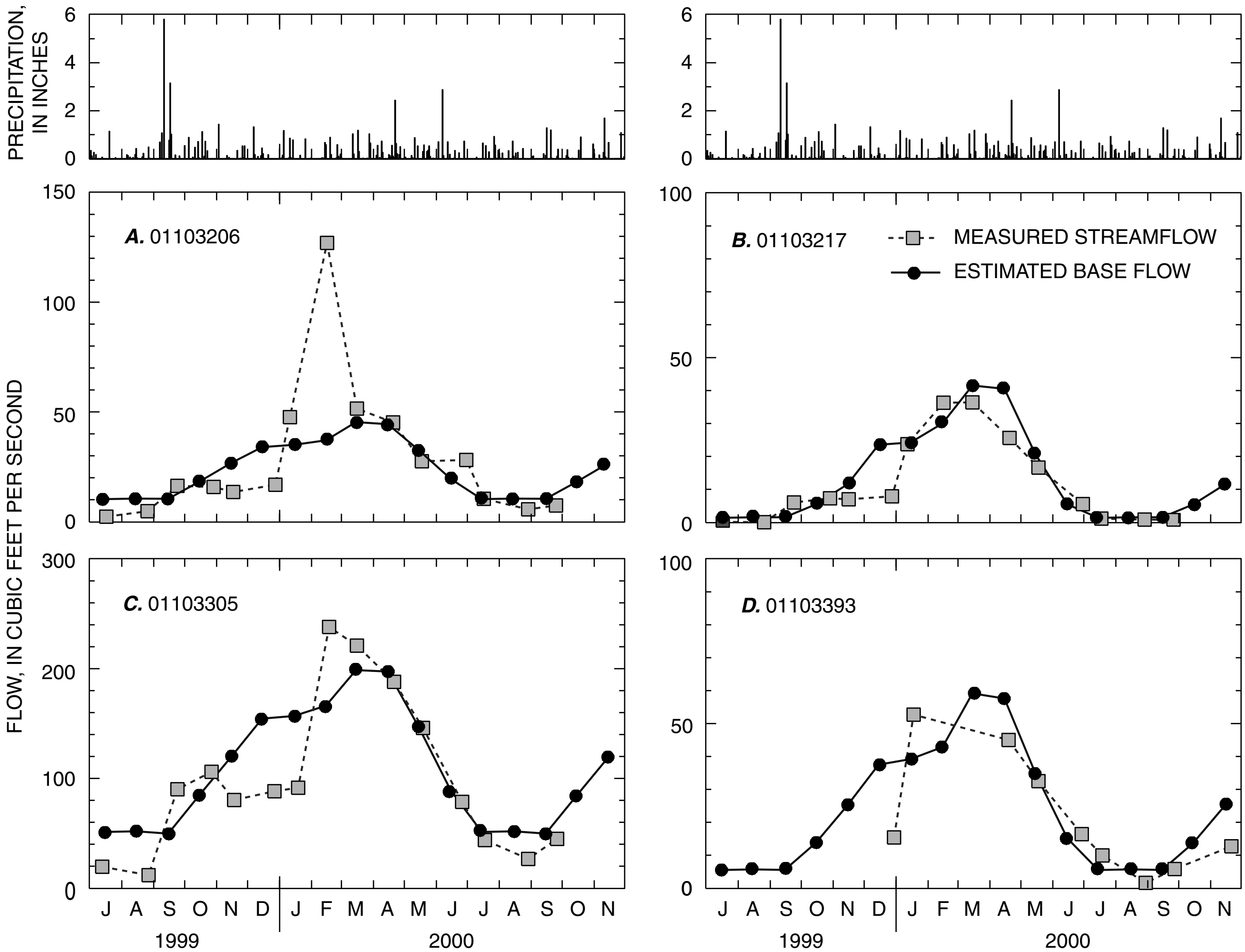


Figure 10. Daily precipitation and measured streamflow during base-flow conditions, 1999–2000, and estimated monthly average base flow at selected streamflow-measurement sites in the upper Charles River Basin, eastern Massachusetts: (*A*) Charles River at North Bellingham (01103206), (*B*) Hopping Brook near West Medway (01103217), (*C*) Charles River near Millis (01103305), and (*D*) Bogastow Brook below Great Black Swamp near Millis (01103393). Precipitation data from the National Oceanic and Atmospheric Administration (2001).

Water Balance

A steady-state water balance for the upper Charles River Basin describes the stream-aquifer system and provides a conceptual framework for the ground-water flow models (table 9). In the steady-state water balance, inflows to and outflows from the combined stream-aquifer system are identified and quantified on an average annual basis. Stratified glacial aquifers and their associated streams as well as upland till and bedrock areas were included. The water balance described here includes the components simulated in a ground-water-flow model; thus, the direct runoff component of streamflow is not included.

Inflows include ground-water recharge from precipitation, ground-water recharge from septic-system return flow, and wastewater discharge to streams from treatment facilities (fig. 11). Outflows include stream outflows from the study area, withdrawals from supply wells and surface-water intakes, ground-water evapotranspiration (ET), and infiltration into sewer lines (fig. 11). Water moves from till and bedrock areas in uplands to areas of stratified glacial aquifers in valleys (fig. 1) through lateral ground-water inflow and streamflow from uplands (fig. 11).

Ground-water recharge from precipitation is the primary source of water. Precipitation recharge (fig. 11) is the component of precipitation that

Table 9. Estimated average annual water balance for the upper Charles River Basin, eastern Massachusetts, 1989–98

[Numbers in parentheses correspond to 90-percent confidence intervals for base-flow estimates]

Water-balance component	Rate of flow	
	Million gallons per day	Cubic feet per second
Inflow		
Recharge from precipitation	[1]100–130	[1]155–201
Septic-tank return flow	3.1	4.8
Wastewater discharge to streams	7.5	11.6
Total inflow	111–141	171–217
Outflow		
Ground-water discharge to streams (stream base flow)	104 (80–137)	161 (124–212)
Water-supply withdrawals	10.1	15.6
Ground-water evapotranspiration in wetlands	5.0	7.7
Total outflow	119 [1](95–152)	184 [1](147–235)

[1]Values correspond to recharge rates of 20–26 inches per year.

infiltrates the land surface and reaches the water table; it is equal to total precipitation minus direct surface-water runoff and ET at or near land surface. Precipitation recharge rates vary temporally and spatially with climatic conditions, land-surface permeability and slope, vegetation, and soil saturation. Precipitation recharge rates, which can be estimated with a number of methods, were compiled from literature sources for the water balance. Basin-wide precipitation-recharge rates have been estimated from long-term streamflow records to range from 23.8 to 25.5 in/yr for basins in eastern Massachusetts and Rhode Island that contain about equal areas of stratified glacial deposits and till and bedrock uplands (Bent, 1995; Barlow, 1997; Barlow and Dickerman, 2001); these values were about 52 to 54 percent of average annual precipitation. Rates from streamflow records in basins in southern New England with varying percentages of stratified glacial deposits and basin slopes ranged from 17.5 to 28.1 in/yr (Bent, 1995; 1999). A range of precipitation-recharge rates, 14 to 28 in/yr (reflecting a range of specific-yield values) was estimated from ground-water levels at a long-term observation well in the Charles River Basin downstream of the study area (Myette and Simcox, 1992). Recent estimates of precipitation recharge to aquifers on western Cape Cod, where surface runoff is negligible, are about 26 in/yr, or 60 percent of average annual precipitation (Masterson and others, 1998). Precipitation-recharge rates to stratified glacial deposits, which may be higher than basin-wide rates in basins with a mix of till and stratified deposits, ranged from 19.3 to 29.4 in/yr at locations throughout New England and New York (Barlow, 1997). For the upper Charles River Basin, recharge estimates of 20 to 26 in/yr (43 to 56 percent of average annual precipitation at West Medway), which are typical rates based on literature sources, correspond to inflow volumes of 100 to 130 Mgal/d to the study area (table 9).

Other inflows to the study area (fig. 11) are small in comparison with precipitation recharge (table 9). Septic-system return flow was estimated for 1989–98, at 3.1 Mgal/d, as described previously, or about 2 to 3 percent of precipitation recharge. Wastewater return flow to streams from 1989–98 was 7.5 Mgal/d, or about 5 to 6 percent of precipitation recharge. Total inflows to the study area, thus, were estimated at 111 to 141 Mgal/d (table 9).

The primary outflow from the study area is streamflow at the study-area boundary. Streamflow leaves the study area through the Charles River and Bogastow Brook in Norfolk and Millis (stations 01103305 and 01103393, fig. 7). Streamflow at these locations includes direct runoff (stormflow) and base flow (ground-water discharge to streams, fig. 11); however, stormflow runoff, which is not simulated in ground-water-flow models, is not included in the water balance. Average annual base flow in the Charles River and Bogastow Brook at the study-area boundary was estimated at 104 Mgal/d (161 ft^3/s, table 9). Confidence limits (90 percent) around these estimates yielded a range from 80 to 137 Mgal/d. Withdrawals for water supply are another outflow from the stream-aquifer system (fig. 11). Water-supply withdrawals were equal to 10.1 Mgal/d in 1989–98 (table 9).

Ground-water ET, ground-water underflow, and infiltration to sewer lines are additional outflows from the stream-aquifer system that are difficult to quantify accurately. Ground-water ET, or ET directly from the water table (fig. 11), may occur in discharge areas such as along streams, in wetlands, or in other areas where the water table is close to land surface. On a basin-wide basis, ground-water ET is likely to be small relative to the total water balance because the areas where it can occur are small. Ground-water ET was estimated to be 1 to 2 in/yr on the basis of streamflow records for about

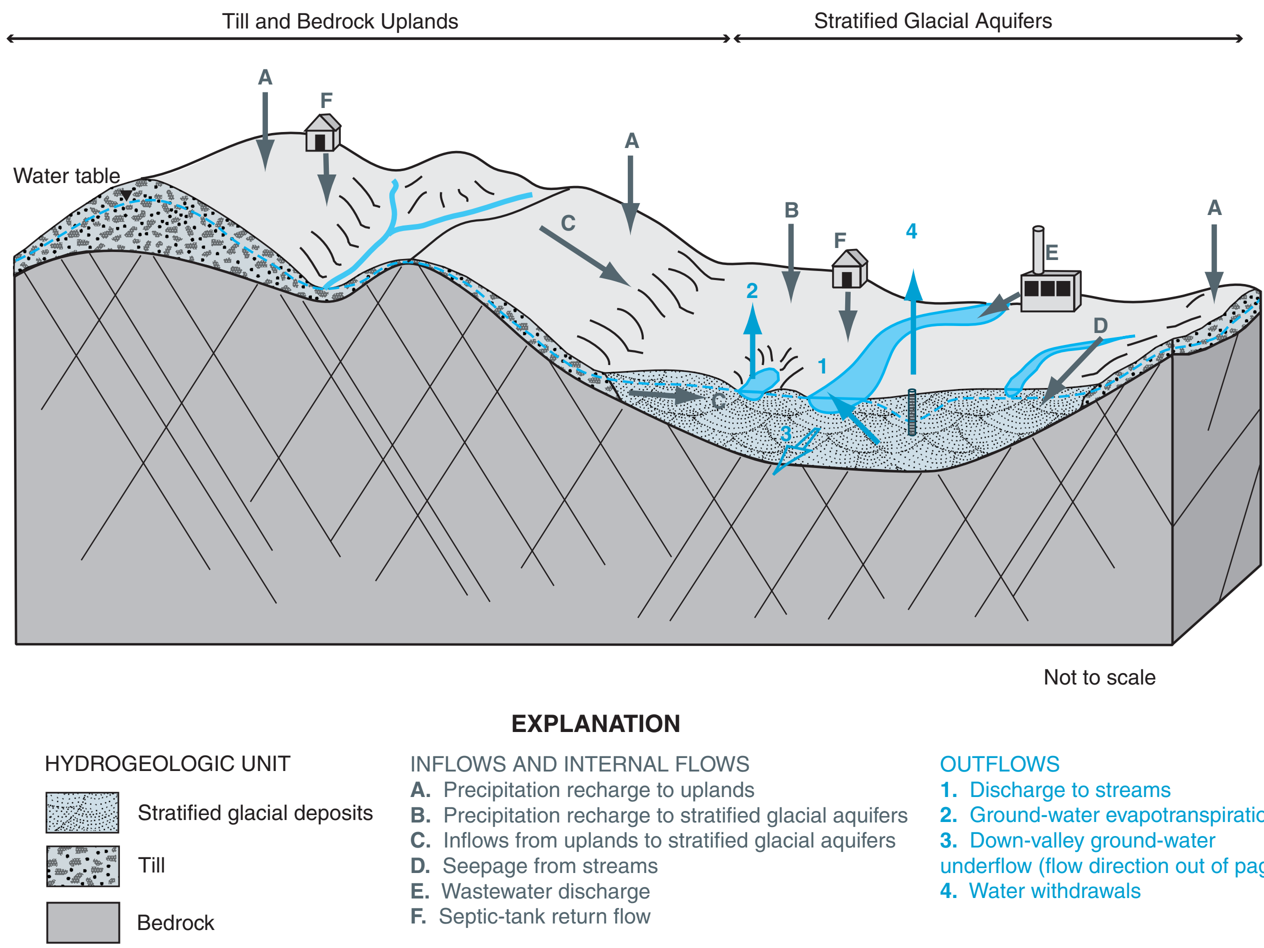


Figure 11. Schematic diagram showing hydrogeologic units and flow components of the water balance for the upper Charles River Basin, eastern Massachusetts. (Modified from Randall and others, 1988, fig. 1.)

150 basins in the east-central United States; these estimates were calculated as the difference between average annual recharge and base-flow rates (Rutledge, 1998). By means of a similar approach, ground-water ET was determined to be 2 to 3 in/yr for two basins in southern Rhode Island. (Barlow and Dickerman, 2001). One way of estimating ground-water ET from available data for the upper Charles River Basin is to use wetland areas, as mapped with the land-use data, and regional evaporation rates. Wetland areas represent areas where the water table is near land surface, and, thus, where ground-water ET may occur. Free-water-surface evaporation rates can be considered potential evaporation rates from a saturated surface and may be used to approximate actual ET rates, which are unknown (Farnsworth and others, 1982). Application of a growing-season free-water-surface evaporation rate of 21 in/yr (determined at a regional scale for east-central Massachusetts, Farnsworth and others, 1982) to wetland areas in the basin yields a basin-wide ground-water ET estimate of about 1 in/yr. This estimate corresponds to an outflow rate of about 5 Mgal/d (table 9). This estimate, which is about 5 percent of streamflow and withdrawal outflows, indicates that ground-water ET is likely to be a small but substantial component of the water balance. Total estimated outflow from the stream-aquifer system, including the estimate of ground-water ET, was 119 Mgal/d, and 95 to 152 Mgal/d represented the 90-percent confidence limits on the base-flow estimates (table 9).

Ground-water underflow (fig. 11) may occur as flow through the stratified glacial aquifers across study-area boundaries. Ground-water underflow is unlikely to occur where Bogastow Brook exits the study area, because the stratified glacial deposits pinch out at the stream in this area. Near the Charles River in Norfolk, however, stratified glacial deposits are thin but do not entirely pinch out at the study-area boundary, and ground-water underflow out of the study area may occur. Ground water also may flow in or out of the study area where the boundary is a surface-water divide that does not coincide with a ground-water divide. Generally, the assumption that ground-water and surface-water divides coincide is valid in the upper Charles River Basin because of the rolling topography, shallow aquifers, and close hydraulic connection between surface and ground water; the assumption also is supported by surface-water and land-surface elevations near the study-area boundary. Pumping wells may alter these conditions, however. In Millis, supply wells in stratified glacial deposits adjacent to the study area, along with the flat topography in this area, may result in some ground-water outflow across the study area boundary. In Norfolk, a supply well may draw water into the study area from stratified glacial deposits that extend across a surface-water divide east of the Charles River. The magnitude of these flows could not be quantified. However, they are likely to be small relative to the total water balance, given the magnitude of supply-wells withdrawals relative to total outflows.

Infiltration of ground-water into sewer lines is a potential outflow in the water balance that is hard to quantify, but probably is small compared to the total water balance. Ground-water infiltration changes seasonally with water-table fluctuations and varies greatly with the age and integrity of sewer-system infrastructure. Ground-water infiltration represents some fraction of the total wastewater flows from sewage-treatment facilities, with estimates ranging from 10 to 60 percent. Few data were available for towns in the upper Charles River Basin, but ground-water infiltration and inflow of stormwater runoff into sewer lines averaged 26 percent of total wastewater flows for six towns in an adjacent basin (Earth Tech, 2001). Similarly, comparison of winter and summer wastewater-discharge rates in the basin suggested an infiltration rate of about 30 percent of total wastewater flows, as described previously. At this rate, ground-water infiltration in the upper Charles River Basin would be about 2 Mgal/d, or about 2 percent of total study-area outflows (table 9).

Within the study area, lateral ground-water inflow and stream base flow from uplands (upland inflows; fig. 11) accounts for a large fraction of water inflows to the stratified glacial aquifers. Upland flows originate as precipitation recharge in till and bedrock areas. Recharge rates in these areas are poorly known, but may be as little as one-half of the rates in nearby stratified glacial aquifers (Kontis and others, in press). If a range of upland recharge rates from 50 to 100 percent of rates in stratified glacial aquifers is assumed, upland flows account for about 35 to 50 percent of total recharge inflows to stratified glacial aquifers.

Overall, estimated water inflows—precipitation recharge and wastewater return flow—and outflows—stream base flow at study-area boundaries, water withdrawals, ground-water ET, and ground-water infiltration to sewers—agree reasonably well (within 20 percent; table 9), given the uncertainties in individual estimates. Ground-water outflows that were not directly quantified in the water balance, such as ground-water underflow, may account for some of the discrepancy between inflows and outflows.

SIMULATION OF GROUND-WATER FLOW

Steady-State Numerical Models

Ground-water flow and head distribution across the upper Charles River Basin were simulated with the three-dimensional, finite-difference ground-water flow modeling code, MODFLOW-2000 (Harbaugh and others, 2000). Ground-water flow and heads were only simulated in the stratified glacial aquifers. Upland till and bedrock areas, which constitute about 50 percent of the study area, were not simulated, although stream base flow and recharge from these areas were included. Active model areas are shown in figure 12. Physically discontinuous stratified glacial aquifers in west and east parts of the basin were simulated with separate models. The two models use the same discretization (cell size and orientation). Model-calculated flow in the Charles River at the exit from the west model is specified as inflow in the east model. Both steady-state and transient conditions were simulated. The steady-state models simulated average flow conditions during the 10-year period, 1989–98. Areal recharge rates and hydraulic conductivity were estimated with inverse modeling (based on nonlinear regression) to obtain the best possible fit between observed and simulated head or flow values (Hill and others, 2000) and to calibrate the steady-state models.

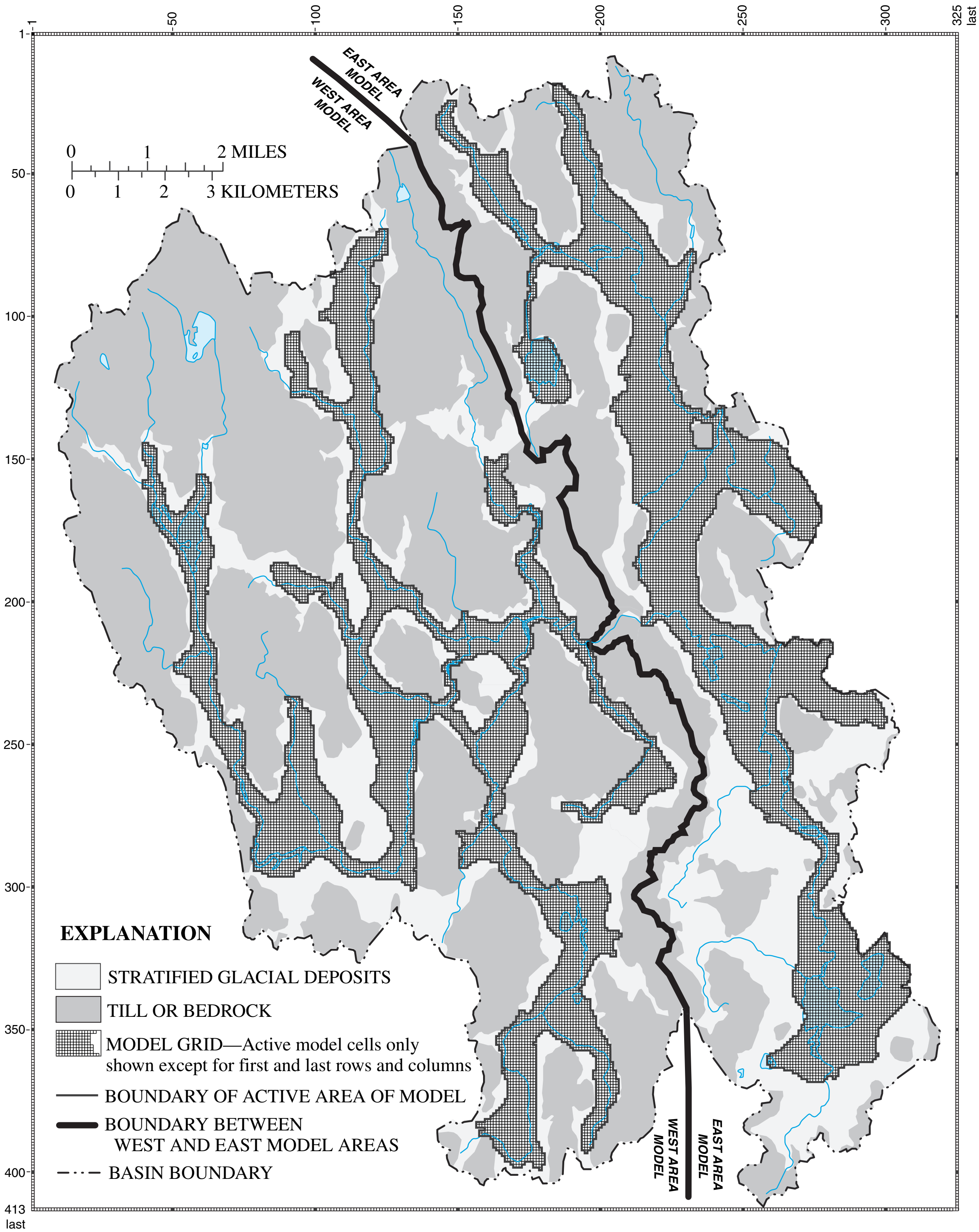


Figure 12. West and east model areas, grid, and boundary of active model areas for the simulation models of the upper Charles River Basin, eastern Massachusetts.

Spatial Discretization

The model domain was discretized into a grid of 420 rows and 325 columns of cells with uniform horizontal dimensions of 200 ft (fig. 12). West and east aquifer models are discretized with one model grid that contains both areas; in the west model, cells representing stratified glacial deposits of the east aquifer area are inactive, and opposite in the east model. The vertical discretization consists of two sloping layers of variable thickness (fig. 13). The bottom of the lower layer corresponds to the top of the bedrock and till surface, and was determined by intersecting the model grid with an elevation surface that was linearly interpolated from elevation contours of the bedrock and till surface. The bottom of the upper layer was set at 40 ft below a simulated water-table surface that was determined by means of uncalibrated, two-dimensional versions of the models. Where saturated thickness, based on the preliminary simulated water table, was less than 40 ft, only the upper layer is active, extending vertically to the bedrock surface. There were 11,197 active cells in the west model, (9,868 cells in layer 1 and 1,329 cells in layer 2) and 13,824 active cells in the east model (9,724 cells in layer 1 and 4,100 cells in layer 2).

Horizontal boundaries of the active model were defined so as to exclude areas where the saturated thickness of the stratified glacial aquifers was thin (fig. 12). Thinly saturated areas along the margins of the active model area (at the boundary with till and bedrock uplands) lead to numerical instabilities and the inability of the model to converge to a solution. Thinly saturated areas were identified using available data on aquifer geometry and preliminary model testing. Preliminary model testing also was used to identify and solve numerical problems with the models. Various two- and three-dimensional versions of an initial model, in which west and east areas were both active, were numerically unstable and did not yield steady-state solutions. A transient version was tested to investigate a quasi-steady-state solution (defined in terms of negligible change in storage) as an alternative to a steady-state model, but also was unstable. Consequently, the initial model was simplified by simulating the study area with two, two-dimensional models for west and east aquifer areas, for which robust solutions were obtained.

Boundary Conditions

Physically, the active model areas are bounded laterally by till and bedrock uplands or by areas of thin stratified glacial deposits; they are bounded below by bedrock or till (figs. 12 and 13). In the numerical models, the lower boundaries of the active model areas are no-flow boundaries (fig. 13). Flow from adjacent uplands is simulated as specified flow along the lateral boundaries. No-flow boundaries are specified along ground-water drainage divides near downgradient boundaries of the model domain and in small areas near the headwaters of some tributary streams. In these areas, ground-water divides were determined from surface topography. The assumption that ground-water divides coincide with surface-water divides likely is valid in the upper Charles River Basin under natural conditions, because of the shallow aquifer depths and close connection between surface and ground water; however, there may be local deviation between surface- and ground-water divides where pumping wells are close to divides. The study area boundary was chosen to minimize the use of surface-water divides as model boundaries, but this use could not be entirely avoided because of the interfingering geometry of stratified glacial aquifers in the basin.

Streams are hydrologic boundaries within the models. Streams were simulated as head-dependent flow boundaries with a version of the Stream Routing Package (Prudic, 1989) that is compatible with MODFLOW-2000. This package simulates hydraulic interaction between the aquifer and adjoining streams and tracks the amount of water in each simulated stream. Simulated streams are divided into reaches, corresponding to individual model cells, and segments, which are groups of reaches that are connected in downstream order (Prudic, 1989). Water may flow in either direction at the boundary between the aquifer and streams. Flow, or leakage, is calculated by multiplying the head difference between the stream stage and the underlying aquifer by the specified streambed conductance. Stage in the stream may be either specified or calculated. In the upper Charles River Basin models, stream stage was specified. If the model-calculated head in the aquifer is higher than the stream-stage elevation, ground-water discharges to the stream reach; when the model-calculated head in the aquifer falls below the

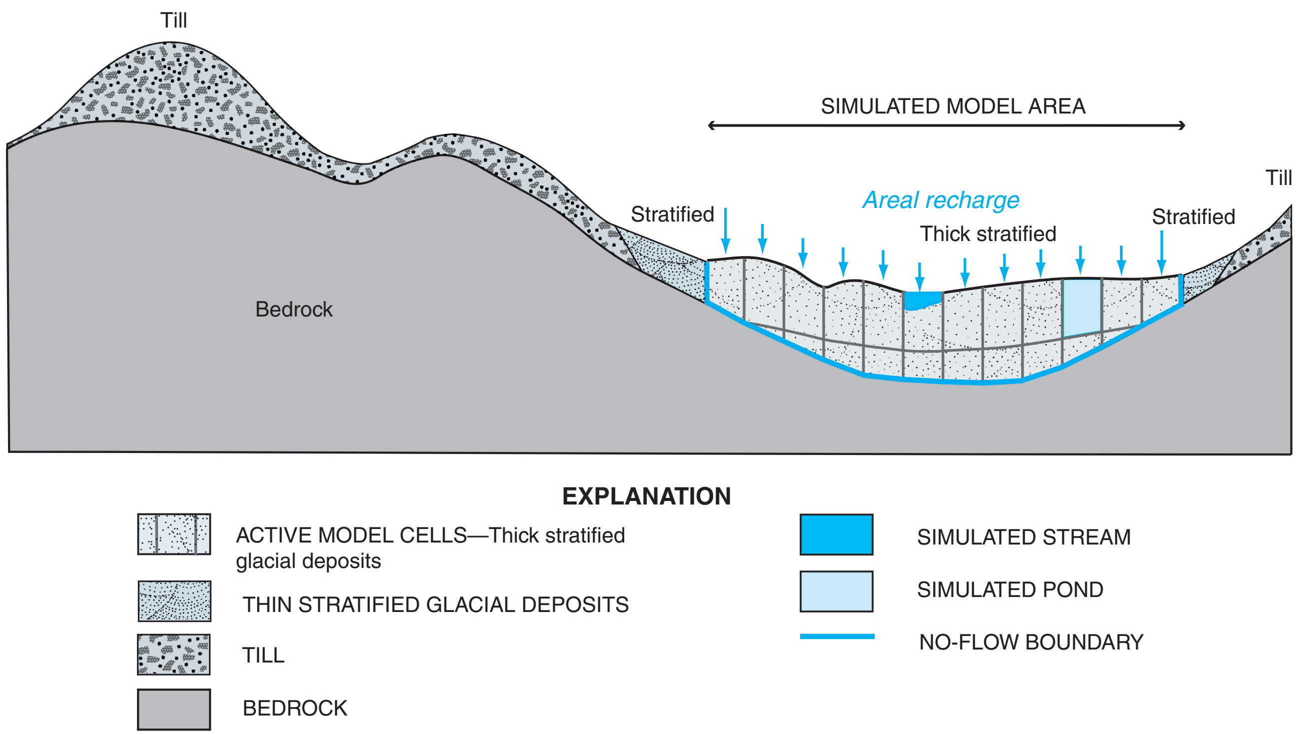


Figure 13. Schematic section showing active model cells and model layers for the simulation models of upper Charles River Basin, Massachusetts.

stream-stage elevation, stream water leaks to the aquifer. The quantity of water in each stream reach is tracked and routed to downstream reaches. Simulated streams may go dry when stream leakage to the aquifer exceeds inflows from upstream reaches.

Simulated streams in the models included the Charles River and Bogastow Brook and their major tributaries: Beaver Brook, Stall Brook, Hopping Brook, Miscoe Brook, Dix Brook, Mine Brook, Chicken Brook, Shepards Brook, Eagle Brook, Mill River, Dirty Meadow Brook, Dopping Brook, and Jar Brook (figs. 1, 12*A* and *B*). Small headwater streams that drain substantial areas of upland also were included.

Stream-stage elevations used as input to the model were determined from DEM data (Elassal and Caruso, 1983) and measured elevations. The DEM data provided elevations (based on 3-meter contour intervals) of stream surfaces at a grid resolution of 30 meters. These data were linearly interpolated to generate surface elevations at a finer scale to more accurately represent elevations along stream channels. The resulting elevations for stream channels were manually edited to ensure that stream elevations decreased monotonically downstream throughout the active model areas and to ensure agreement with measured elevations at outlets from major impoundments along streams. A uniform water depth of 1 ft and streambed thickness of 5 ft were assumed to calculate streambed top and bottom elevations from stream-surface elevations.

Streambed conductances were determined initially from literature sources and assumed stream geometries. Conceptually, streambed conductance is calculated from streambed hydraulic properties and geometry as (Prudic, 1989):

$$C_{SB} = \frac{K_{SB} \cdot L_S \cdot W_S}{T_{SB}} \tag{5}$$

where

C_{SB} = streambed conductance, in ft^2/d,

K_{SB} = vertical hydraulic conductivity of the streambed, in ft/d,

L_S = length of the stream, equal to the model cell dimension in the direction of streamflow, in ft,

W_S = width of the stream, in ft, and

T_{SB} = thickness of the streambed, in ft.

In practice, the values used to compute streambed conductance, particularly the hydraulic conductivity and thickness of the streambed, are poorly known. Vertical hydraulic conductivities of streambed sediments have been reported to range from 0.1 to 15 ft/d, with typical values of 1 to 2 ft/d, for streams draining stratified glacial deposits at locations in central Massachusetts, southern Rhode Island, Long Island, New York, and northern New Jersey (Rosenshein, 1968; DeLima, 1991; Prince and others, 1988; Dysart and Rheaume, 1999). In the upper Charles River Basin models, initial streambed conductances were calculated from assumed uniform stream widths and streambed thicknesses of 10 ft and 5 ft, respectively, stream lengths equal to the model cell size of 200 ft, and a streambed hydraulic conductivity of 5 ft/d. Streambed conductances were varied during model calibration, as described below.

Streamflow entering the active model area was specified at two locations where the Charles River enters the west and east models. Flow in the first reach of the segment representing the Charles River in the west model was specified at 2.3 ft^3/s. Flow in the first reach of the segment representing the Charles River in the east model was specified equal to the simulated flow in the Charles River at the exit from the west model. Flow from the headwaters to the Charles River was specified to facilitate simulation of water withdrawals from Echo Lake Reservoir, which is located in this upland subbasin.

Stresses

Recharge

Recharge to the models consisted of infiltration from precipitation and wastewater return flow from septic systems. Recharge was applied to the active model area as a spatially varying, specified flux to the uppermost active layer (fig. 13). In general, precipitation recharge varies spatially with land-surface permeability, which is a function of soil characteristics and land use. Recharge to the aquifer from septic-system return flow occurs in areas where wastewater disposed of through septic systems originated as water withdrawn from the aquifer at a distant location, such as a public-supply well. Septic-system return flow in areas of co-located, private water supplies is assumed to result in no net inflow to the aquifer and was not explicitly represented in the models.

The spatial distribution of recharge rates in the upper Charles River Basin was determined by CRWA from 1991 land-use data (1:25,000 scale; MassGIS, 1997), the areal extent of stratified glacial deposits (Volckmann, 1975a, 1975b; B.D. Stone and J.R. Stone, U.S. Geological Survey, written commun., 2000), and digital data layers of water and sewer systems (fig. 6). Digital water- and sewer-system data was based on maps provided by each town. Twenty recharge categories were defined with seven types of land use (forest, open space, wetland, water, high-density residential, low-density residential, and commercial), two surficial-geology types to represent soil conditions (stratified glacial deposits and till or bedrock), and the presence or absence of septic-system return flow to develop a range of recharge rates for the study area. The recharge categories and associated rates were based on those used in a watershed model of the Ipswich River Basin in northeastern Massachusetts (Zarriello and Ries, 2000). Recharge rates (excluding rates to wetlands, which were zero) ranged from 7.8 in/yr for commercial land-use areas to 28.9 in/yr for high-density residential land use in stratified glacial aquifers with return flow.

Both the area-weighted average rate for the study area, 16.4 in/yr, and the rates corresponding to natural precipitation recharge (forested land cover), 10 or 17.5 in/yr for areas of till and bedrock or stratified glacial deposits (fig. 1), respectively, were low relative to recharge rates determined from literature sources and those used in the water balance. Thus, it was expected that recharge rates based on the CRWA analysis and Ipswich model would be increased during model calibration. To facilitate this process, recharge rates in model cells of the active model area were divided by the natural precipitation recharge rate for stratified glacial aquifers (17.5 in/yr); thus, a multiplier array was used to incorporate the spatial variability in recharge rate into the models.

Flow from upland areas typically enters areas of stratified glacial aquifers through lateral ground-water inflow (unchanneled flow) or as flow in streams draining upland subbasins (channeled flow) (Randall and others, 1988). Inflows from upland areas through both these pathways were simulated as specified flow to the uppermost model layer at active model boundaries (except for flow from the headwaters of the Charles River, which was included as input the stream package). Upland areas were divided into subbasins to account for these inflows (fig. 14*A* and *B*). Subbasins were defined for each of the simulated streams on the basis of topography from points where the streams met active model boundaries. Subbasins were defined for upland areas between simulated streams on the basis of previously defined subbasin boundaries (MassGIS, 1997) and aquifer geometry. Using the digital data layer of recharge polygons, an area-weighted recharge rate for each upland subbasin was calculated. This rate, applied to the entire subbasin area, was summed to apply as a specified flow to one or more model cells along the active model boundary (one cell for subbasins drained by streams; multiple cells for subbasins drained by lateral ground-water flow). This flow rate was added to the precipitation recharge rate for each boundary model cell, and converted to a recharge multiplier as described previously. The recharge data layer, adjusted to model-calibrated recharge rates (described below), is shown in figure 15.

The approach of using recharge multipliers to simulate boundary flows from the till and bedrock uplands to the active model area had advantages and disadvantages. It was used so that rates of inflow to the active model area from uplands, consisting of either lateral ground-water inflow or base flow in upland streams, could be varied, along with recharge rates to active model areas, during parameter estimation for model calibration. These inflows, which conceptually originate as precipitation recharge in uplands, were expected to be large components of the total hydrologic budgets for the stream-aquifer systems. Thus, it was important to vary the inflows from uplands when calibrating the model to obtain the best possible match between observed and model-calculated stream base flows. The approach required delineation of upland basins drained by channeled and unchanneled flow in uplands, as do other approaches for simulating recharge in uplands (for example, Kontis, 2001). Addition of base flow in streams draining uplands as a flow directly to the aquifer was based on the assumption that all base flow in these streams is lost to the aquifer immediately after the streams cross the upland/aquifer boundary.

Ponds drained by streams and wetlands in the upper Charles models were conceptualized as areas of net ground-water discharge, where, on average, water levels were equal to or less than the surrounding water table. Saturated soils during most of the year and low-permeability sediments likely result in no net recharge of water to aquifers from precipitation in most wetlands, under natural conditions. Similarly, precipitation onto surfaces of ponds drained by streams becomes direct stream (stormwater) runoff rather than contributing to ground-water recharge or constituting a component of the stream base flow that is simulated in the models. Water leaves these areas through ET, however. Thus, ponds drained by streams and wetlands were simulated as areas of net water loss from the stream-aquifer system. Water loss from these areas in the active model area was simulated as a loss equal to the rate of free-water-surface evaporation in the growing season in east-central Massachusetts, or 21 in/yr (Farnsworth and others, 1982). Although vegetation in wetlands may result in actual ET rates that differ from free-water-surface evaporation rates, the direction and magnitude of these differences is uncertain; thus, free-water-surface evaporation rates are likely to be a good approximation of actual ET rates in wetlands (Mitsch and Gosselink, 1993). In uplands, ET from ponds drained by streams and wetlands was not simulated. However, wetland and pond areas in uplands are small relative to non-wet areas and ET from these areas was considered negligible.

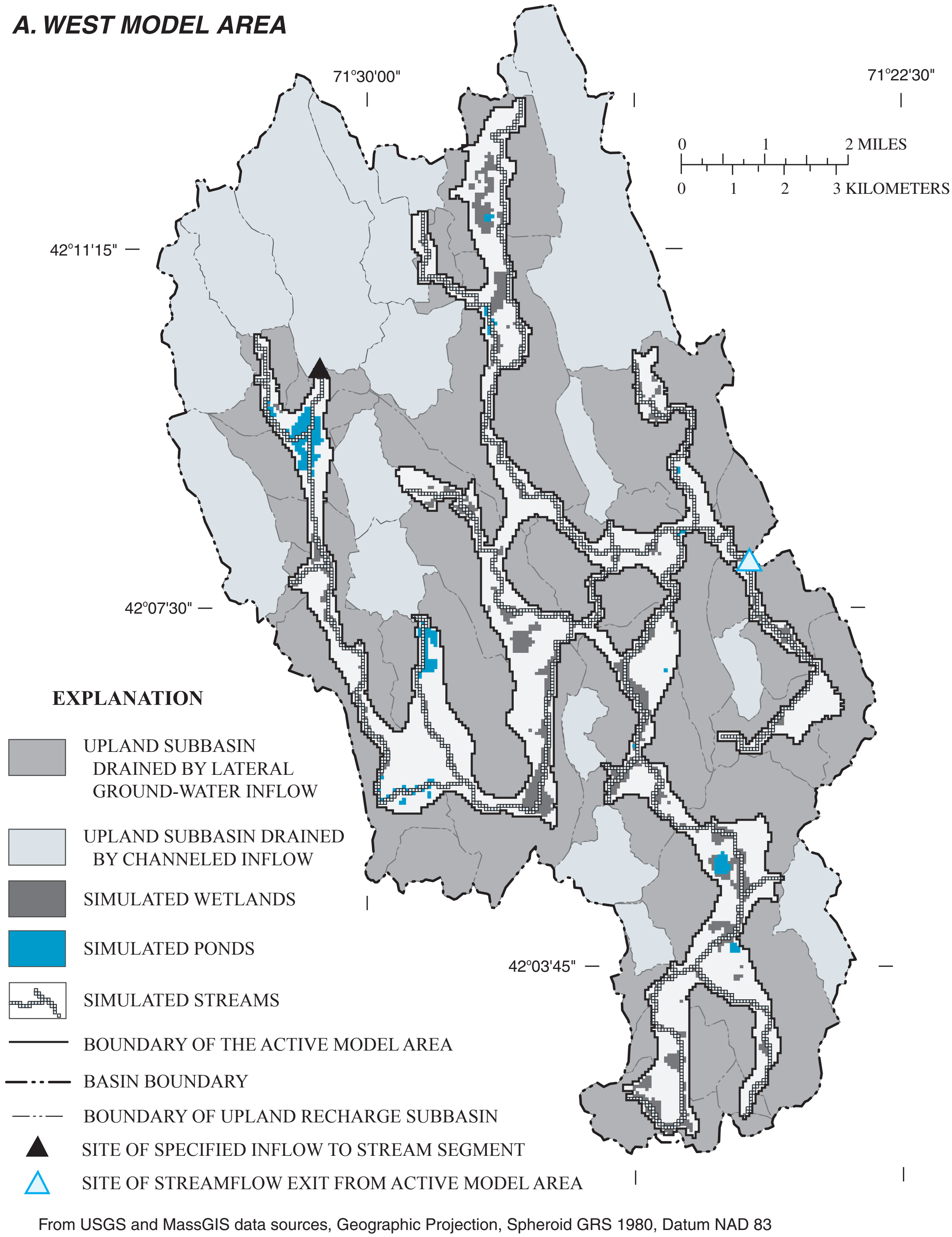


Figure 14. Upland recharge subbasins and simulated surface-water features for the simulation models of the upper Charles River Basin, eastern Massachusetts: (*A*) West model area, and (*B*) East model area.

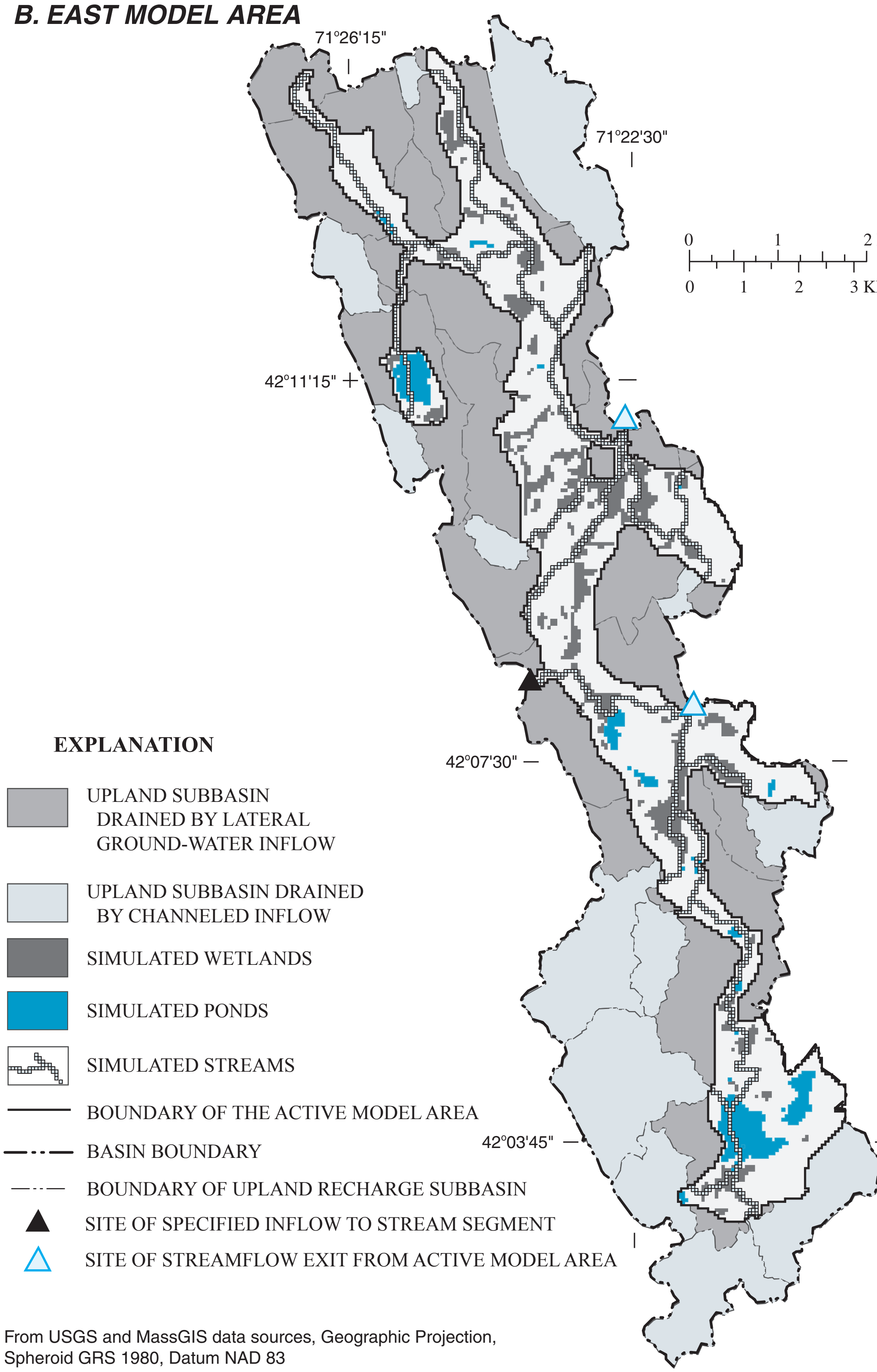
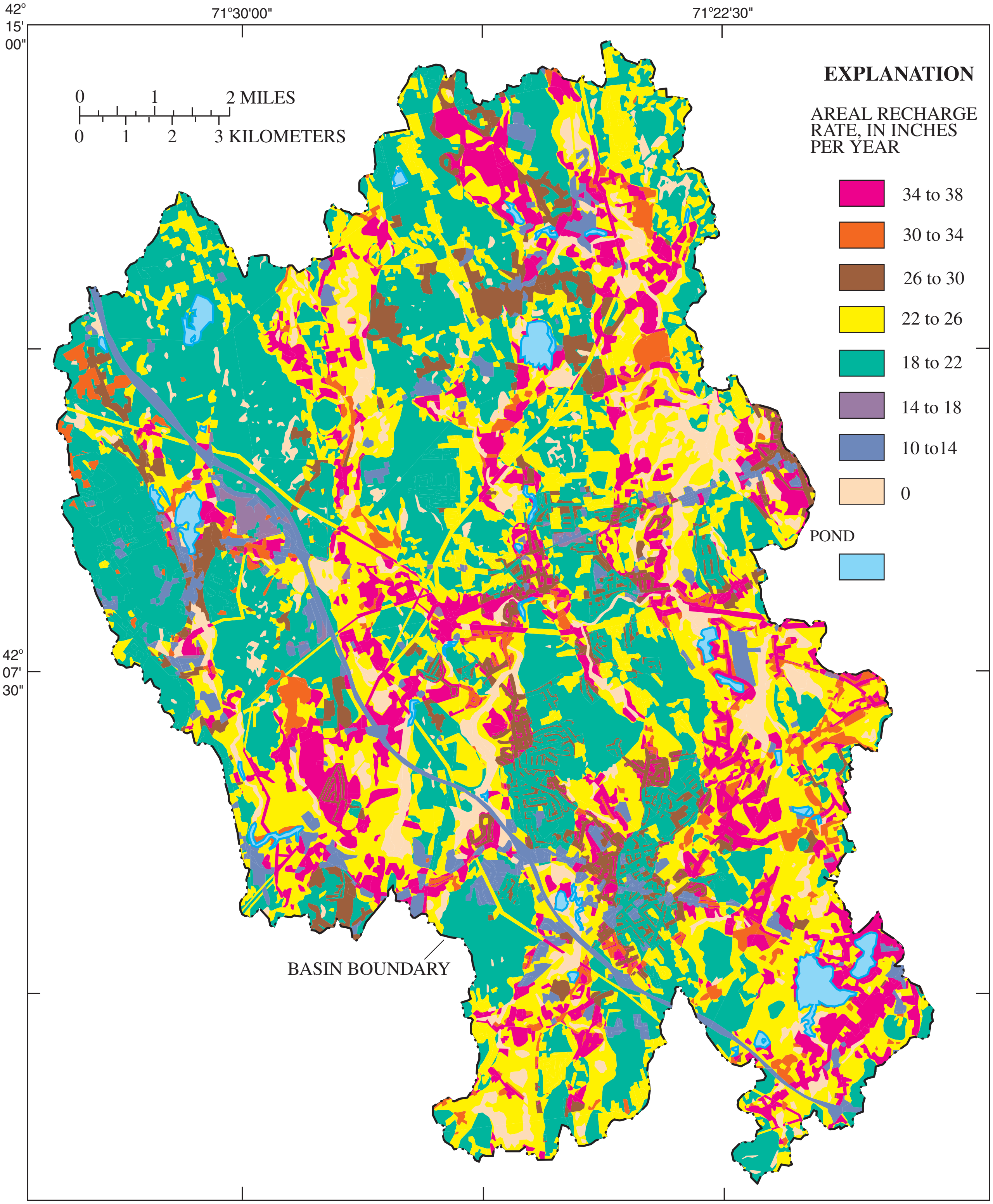


Figure 14. Upland recharge subbasins and simulated surface-water features for the simulation models of the upper Charles River Basin, eastern Massachusetts: (*A*) West model area, and (*B*) East model area—*Continued*.



From USGS and MassGIS data sources, Geographic Projection, Spheroid GRS 1980, Datum NAD 83

Figure 15. Areal recharge rate from precipitation recharge and septic-system return flow for the simulation models of the upper Charles River Basin, eastern Massachusetts. Rates are based on the calibrated steady-state models.

Ponds that are not drained by streams, typically kettle-hole ponds, have no outlet for surface-water runoff of precipitation. These areas were simulated as ground-water recharge areas and received simulated recharge of 22 in/yr in the active model area. This value is a representative recharge rate from literature sources cited previously and represents the estimated difference between precipitation and evaporation in the basin. Recharge rates for kettle ponds, and recharge rates representing ET from in-stream ponds and wetlands, were not varied in the calibration process.

Water Withdrawals and Discharges

Pumping wells, water withdrawals from surface water, and wastewater discharges to streams were simulated with the Well Package for MODFLOW-2000. Withdrawals from pumping wells were simulated as specified flows from the aquifer; surface-water withdrawals and discharges were simulated as specified flows from the model cells hydraulically connected to streams. Flow rates in the steady-state model (table 10) were equal to average annual withdrawal and discharge rates for 1989–98 for most sources (table 2). For irrigation (country club) withdrawals, flow rates were specified as 50 percent of the total withdrawal rates, to account for return flow (infiltration) of irrigation water. All simulated wells were in layer 1 for both models, in accordance with available data about well-screen intervals (table 2).

Several water withdrawals were outside the active model area. These were the town of Franklin supply well No. 9 and Franklin Country Club well and reservoir in the east model area and the Echo Reservoir in the west model area. The Franklin withdrawals were incorporated into the east model by reducing the inflow of water from the upland subbasin in which they were located by a volume equal to their combined annual average withdrawals (table 10). In the west model, inflow from the upland subbasin that includes the Echo Lake Reservoir was altered to exclude all flow that originated upstream of the reservoir outlet. This approach is based on the assumption that the reservoir, which stores water pumped to it in winter from a location further downstream on the Charles River, does not have substantial stream releases during base-flow conditions. Water use from Echo Lake Reservoir is represented by the winter withdrawals for storage from the downstream Charles River location, which averaged 0.34 Mgal/d annually, based on available data (1998–2000).

Hydraulic Properties

Aquifer hydraulic properties required for the steady-state simulations were horizontal and vertical hydraulic conductivity. Aquifer-test data at public-supply wells (table 1) and geologic logs from wells and boreholes were used to characterize hydraulic conductivities. The conceptual model of aquifer-sediment deposition also provided a framework for estimating hydraulic conductivity throughout the model area. A generalized approach was used to estimate the spatial distribution of hydraulic conductivity and values were adjusted during model calibration to match observed data.

Hydraulic conductivity was assigned in zones that were delineated based on aquifer and stream geometry and streamflow-measurement sites and were modified during model calibration (fig. 14*A* and *B*). Horizontal hydraulic conductivity (K_h) ranged from 150 to 290 ft/d in the west model, and from 70 to 220 ft/d in the east model. Vertical hydraulic conductivity (K_v) was based on published ratios of vertical to horizontal conductivities, which generally range from 1:3-1:5 for coarse-grained stratified glacial deposits to 1:30-1:100 for fine-grained deposits (Dickerman and others, 1990; Masterson and Barlow, 1997, and references therein; Masterson and others, 1998; Stone and Dickerman, 2002). Vertical hydraulic conductivities ranged from 10 to 70 ft/d (K_v:K_h equal to 1:4 to 1:17) in the west model and from 5 to 30 ft/d (K_v:K_h equal to 1:6 to 1:20) in the east model. Horizontal and vertical hydraulic conductivities were distributed in the same way in layers 1 and 2, except for cells in layer 1 that simulated ponds. Model areas used to simulate ponds were assigned horizontal and vertical hydraulic conductivity values of 10,000 ft/d. Simulating ponds as active model cells allowed pond levels to change with changing stresses in the aquifer. The large value of hydraulic conductivity effectively simulates the lack of resistance to flow through the ponds and results in little or no head drop across adjacent pond cells and in realistic flow patterns in the aquifer surrounding the ponds.

Table 10. Simulated water withdrawals and discharges in calibrated models and in scenarios 1 and 2 for existing and proposed municipal public-supply sources, large non-municipal sources, and wastewater-treatment facilities in the upper Charles River Basin, eastern Massachusetts

[**Identifier:** See table 2 for additional identification information; site locations shown in figure 3; identifiers ending in “G,” “S,” and “P” denote ground-water, surface-water, and proposed sources, respectively. **Simulated withdrawal or discharge rate**: Parentheses denote discharges. **Average withdrawal or discharge rate for 1989–98:** Rates used in calibrated steady-state and transient models. **Average summer withdrawal or discharge rate**: Average of monthly average June, July, and August rates. Mgal/d, million gallons per day; --, not applicable]

Identifier	Model location			Simulated withdrawal or discharge rate (Mgal/d)					
	Layer	Row	Column	1989–98		Scenario 1		Scenario 2	
				Annual	Summer	Annual	Summer	Annual	Summer
West Model									
Charles River									
MF-01S[1]	1	163	61	0.84	0.12	0.84	0.12	0.84	0.12
MF-01G	1	166	58	.008	.009	.34	.34	.34	.34
	1	166	59	.008	.009	.34	.34	.34	.34
MF-02G	1	179	56	.43	.46	.80	.80	.80	.80
MF-03G	1	215	61	.12	.11	.26	.26	.26	.26
MF-04G	1	217	62	.12	.11	.26	.26	.26	.26
MF-05G	1	218	63	.12	.11	.26	.26	.26	.26
MTF	1	240	66	(3.56)	(2.72)	(4.87)	(3.75)	(4.87)	(3.75)
BL-12G	1	278	130	.02	.04	.41	.35	.41	.38
Beaver Brook Subbasin									
BL-05G	1	278	98	.20	.20	.29	.29	.29	.29
NEA-01G, -02G, -03G	1	279	105	.53	.50	.66	.61	.66	.61
NEA-04G, -05G	1	279	97	.010	.010	.014	.013	.014	.013
Stall Brook Subbasin									
BL-07G	1	234	123	.10	.15	.21	.37	.21	.37
BL-08G	1	233	125	.27	.46	.44	.68	.44	.68
Hopping Brook Subbasin									
HL-04G	1	137	117	.14	.24	.14	.24	.14	.24
HL-05G	1	98	215	.45	.47	.45	.47	.45	.47
Mine Brook Subbasin									
FR-01G	1	317	188	.11	.20	.24	.40	.19	.34
FR-02G	1	319	189	.11	.20	.24	.40	.19	.34
FR-03G	1	350	175	.30	.35	.32	.32	.32	.32
FR-06G	1	341	182	.34	.39	.50	.53	.46	.53
FR-07G	1	253	163	.28	.31	.48	.55	.41	.49
FR-10G	1	376	202	.22	.25	.39	.48	.32	.41
FR-01P	1	384	161	.00	.00	.00	.00	.47	.47
MGCC-01G	1	261	161	.030	.060	.075	.154	.075	.15
East Model									
Charles River									
FR-08G	1	216	231	0.30	0.322	0.26	0.26	0.26	0.26
FR-02P	1	217	234	.00	.00	.00	.00	.72	.72
MD-01G	1	206	228	.41	.46	.41	.46	.37	.38
MD-03G	1	211	233	.33	.38	.34	.38	.37	.47
CRPCD	1	212	239	(3.92)	(3.52)	(5.80)	(5.85)	(6.78)	(6.33)

Table 10. Simulated water withdrawals and discharges in calibrated models and in scenarios 1 and 2 for existing and proposed municipal public-supply sources, large non-municipal sources, and wastewater-treatment facilities in the upper Charles River Basin, eastern Massachusetts—*Continued*

Identifier	Model location			Simulated withdrawal or discharge rate (Mgal/d)					
				1989–98		Scenario 1		Scenario 2	
	Layer	Row	Column	Annual	Summer	Annual	Summer	Annual	Summer
East Model—*Continued*									
Bogastow Brook Subbasin									
HL-01G	1	116	184	0.05	0.13	0.05	0.13	0.049	0.13
HL-02G	1	46	157	.06	.16	.056	.15	.057	.15
HL-06G	1	50	191	.45	.52	.45	.52	.45	.52
HL-01P	1	46	190	.00	.00	.00	.00	.35	.46
GECC-01S	1	118	221	.02	.07	.08	.23	.08	.23
MD-02G	1	188	226	.14	.19	.14	.19	.16	.26
MD-01P	1	165	221	.00	.00	.00	.00	.41	.43
Mill River Subbasin									
FR-04G	1	243	245	.67	.74	.92	.92	.85	.92
FR-05G	1	255	249	.17	.20	.32	.40	.26	.34
FR-09G[2]	--	--	--	.26	--	.44	--	.37	--
NF-01G	1	224	273	.24	.30	.38	.42	.24	.30
NF-01P	1	252	252	.00	.00	.00	.00	.26	.31
WR-02G	1	326	275	.32	.41	.40	.51	.32	.41
WR-03G	1	326	278	.42	.49	.52	.62	.42	.49
WR-01P	1	351	274	.00	.00	.00	.00	.19	.23
FCC-01G, -01S[2]	--	--	--	.05	--	.10	--	.10	--

[1]Includes water transfers to Echo Lake Reservoir.
[2]Withdrawals simulated as reduced inflow from upland basin.

Model Calibration

The steady-state models were calibrated by varying model input parameters—recharge, horizontal and vertical hydraulic conductivity, and streambed conductance—to obtain as close a match as possible between simulated and observed water levels and stream base flows. Calibration data consisted of stream base flow at 23 measurement sites and water levels at 35 observation wells and 7 ponds. The models were calibrated to average annual conditions during 1989–98. This 10-year period was chosen so that the calibrated models would reflect both current stresses and average hydrologic conditions. The observed values used in calibration consisted of estimated average annual stream base flow and water levels for the 1989–98 period.

A combination of inverse modeling and trial-and-error methods was used to calibrate the steady-state models. In inverse modeling, model input parameters are solved for as unknowns, with automated mathematical methods (Hill, 1998) to determine parameter values that yield the best match between observed and simulated values. In trial-and-error methods, parameter values are adjusted manually until the match between observed and simulated values is considered reasonable. For the upper Charles models, inverse modeling was used to determine areal recharge rates, on the basis of observed stream base flows. This approach was based on the conceptual model of the study area and was supported by the water balance, in which all water enters the model area as recharge and exits as streamflow (except for minor outflows such as underflow, assumed to be negligible). The approach was considered appropriate for recharge because of the available streamflow data and the lack of long-term data from which recharge rates could be independently calculated. Trial-and-error methods were used to calibrate the models with respect to aquifer hydraulic properties.

Recharge

Calibrated recharge rates were determined through parameter estimation with a version of UCODE, an inverse modeling code, that is incorporated into MODFLOW-2000 (Hill, 1998; Hill and others, 2000). In UCODE, nonlinear regression is used to determine parameter values that minimize values of a weighted least-squares (WLS) objective function. The WLS objective function quantifies the difference between observed and simulated values. The parameter-estimation process incorporates an analysis in which model sensitivities for the estimated parameter are determined at all specified observation points. The primary estimated parameter was the natural precipitation-recharge rate for stratified glacial aquifers with forested land use. Cell-by-cell recharge rates used in the model were the product of this rate and a multiplier that incorporated the variations resulting from differences in land use, surficial geology, and septic-system return flow. Thus, adjustments to the natural precipitation-recharge rate made during the parameter-estimation process also proportionately adjusted all spatially varying recharge rates specified in the model. Inflows to the active model area from uplands also were adjusted similarly, because these inflows were simulated through the recharge-multiplier values at active model boundaries. This approach allowed the automated adjustments during parameter estimation while preserving the spatial variations in recharge that reflected local land-surface permeability and return flow throughout the model area (fig. 15).

Recharge parameters were defined to correspond to natural precipitation-recharge rates in two aquifer zones of the east model. The northern zone included the aquifer associated with Dopping Brook, Jar Brook, and Bogastow Brook in Holliston, Millis, and Medway. The southern zone included the aquifer associated with the Mill River and the Charles River in Wrentham, Franklin, Norfolk, and Medway. The two zones are separated by a ground-water divide and are drained by streams to different model exits. Observed values were stream base flow at four measurement sites in the northern zone and five sites in the southern zone (sites shown in fig. 7). Uniform values of horizontal and vertical hydraulic conductivity of 250 ft/d and 50 ft/d, respectively, were used in the model during the estimation of the recharge parameter. For streambed conductance, a uniform value of 2,000 ft^2/d was used, which corresponds to a streambed vertical hydraulic conductivity of 5 ft/d, as previously described.

The rate of natural precipitation recharge in undeveloped sand and gravel areas used in the calibrated west and east models was 24.1 in/yr. Rates of natural precipitation recharge determined by parameter estimation were 23.9 in/yr and 24.3 in/yr, for northern and southern zones, respectively, of the east model. These rates (weighted by area) were averaged to yield 24.1 in/yr for natural recharge precipitation for the east model. This rate yielded simulated stream base flows that closely matched observed flows in the east model (see below). A value of 24.1 in/yr also yielded a close match between simulated and observed stream base flow in the west model. Because of this close agreement and because natural precipitation recharge rates would not be expected to vary from east to west model areas, the calibrated value from the east model also was used in the west model. The natural precipitation recharge rate of 24.1 in/yr corresponded to spatially varying rates ranging from 10.7 in/yr for commercial land use areas to 38 in/yr for high-density residential land use in stratified glacial aquifers with return flow. Area-weighted rates across the model domain were 21.5 and 23.0 in/yr for active areas of the west and east models, (excluding wetland areas and ponds drained by streams), respectively, and 21.9 and 24.0 in/yr for flows to the active model area from till and bedrock uplands of west and east model areas, respectively. The basin-wide area-weighted rate was 22.5 in/yr. These rates are consistent with recharge rates determined for stratified glacial deposits and basins with a mix of stratified glacial aquifers and till and bedrock uplands in previous studies (Bent, 1995, 1999; Barlow, 1997; Barlow and Dickerman, 2001; Masterson and others, 1998; Myette and Simcox, 1992).

Hydraulic Properties

Hydraulic properties were adjusted manually to obtain a close match between observed and simulated ground-water and pond levels. The degree of fit between observed and simulated values was measured with normal and weighted residual statistics, which are calculated by MODFLOW-2000. Twenty well- and pond-observation points were used in the west model and 30 points were used in the east model. Hydraulic properties adjusted during model calibration were horizontal and vertical hydraulic conductivity and streambed conductance.

Horizontal and vertical hydraulic conductivity values were adjusted in zones (fig. 16*A* and *B*), from initial values of 250 ft/d and 50 ft/d, respectively. Initial zones were modified to better match observed heads by adding separate zones in the east model for the Jar Brook aquifer in Holliston and for the eastern part of Millis. Calibrated horizontal hydraulic conductivity values (70 ft/d) were lower in these areas than in adjacent zones. Available lithologic data were sparse in these areas, but did indicate the presence of fine-grained sediments, particularly in the Millis zone. As described previously, final calibrated values of horizontal hydraulic conductivity ranged from 150 to 290 ft/d in the west model and from 70 to 220 ft/d in the east model; calibrated values of vertical hydraulic conductivity ranged from 10 to 70 ft/d in the west model and from 5 to 30 ft/d in the east model (fig. 16*A* and *B*).

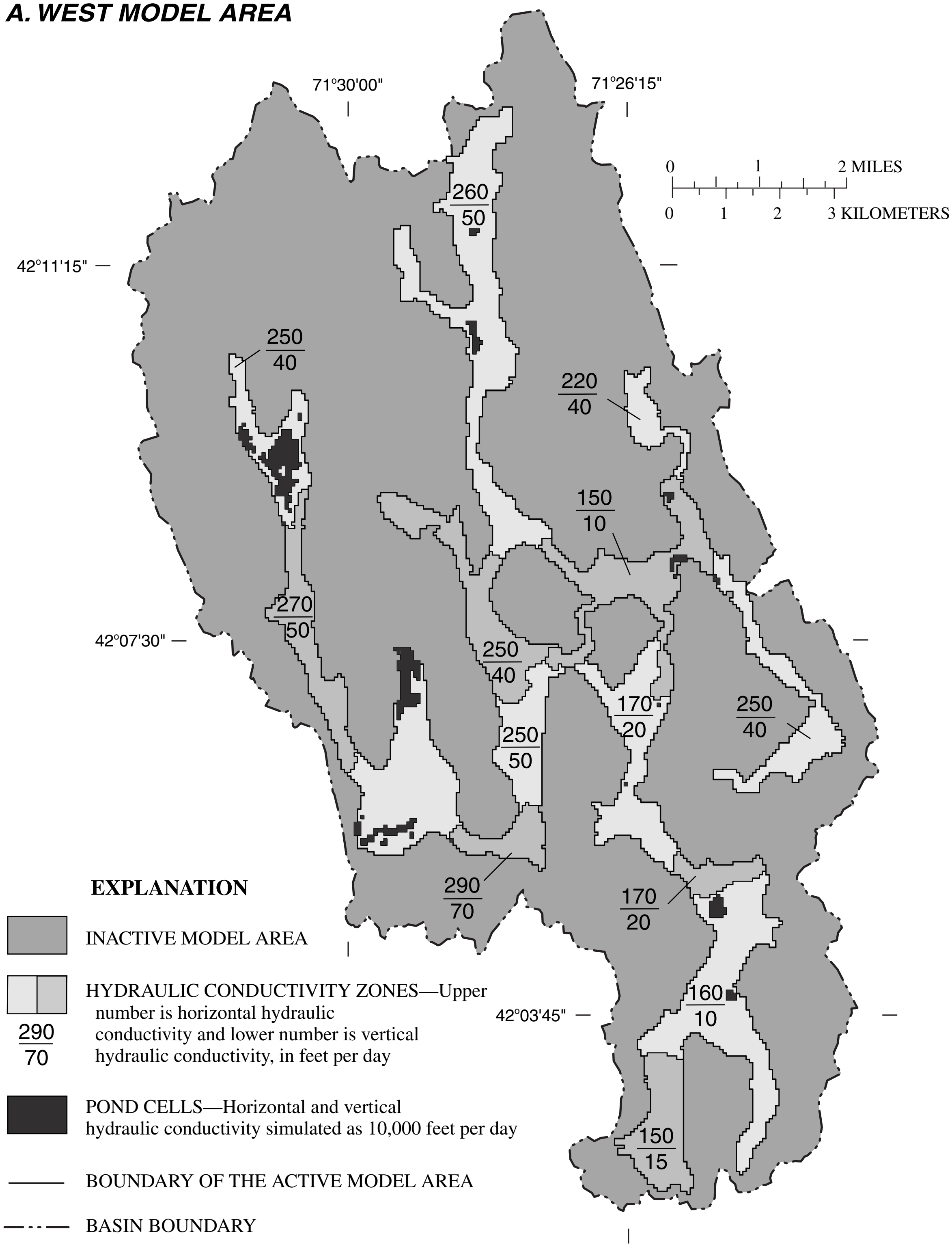
Streambed conductances also were adjusted to improve the match between observed and simulated heads and flows. Streambed conductances were adjusted from initial values of 2,000 ft^2/d in groups that corresponded to hydraulic conductivity zones and areas simulated as wetlands and ponds. In areas simulated as wetlands, calibrated streambed conductances generally were lower than in other areas, ranging from about 500 to 2,000 ft^2/d in both models; these values correspond to K_{SB} values of about 1 to 5 ft/d, using reasonable assumptions about streambed thickness and stream width. In areas simulated as ponds, streambed conductances generally were higher than in other areas, ranging from about 2,000 ft^2/d to high values that simulated no resistance to vertical flow. In areas not simulated as wetlands or ponds, streambed conductances generally ranged from about 1,000 to 7,000 ft^2/d (K_{SB} about 3 to 15 ft/d) in the west model and from about 5,000 to 15,000 ft^2/d (K_{SB} about 10 to 30 ft/d) in the east model. Additional changes were made to streambed conductances to resolve numerical instabilities during development of transient versions of the model. Most changes were slight, affecting a few model cells only, except for changes made in several headwater streams in the southern part of the east model. In these areas, streambed conductances were lowered substantially, to about 30 ft^2/d (K_{SB} about 0.1 ft/d), which is still, however, within the range of typical values.

Calibration Results

Calculated water levels for observation wells and ponds for the calibrated steady-state model are shown in table 11 and figure 17*A*. The mean absolute difference between observed and model-calculated water levels (mean absolute water-level residual) was 2.6 ft for the west model and 2.0 ft for the east model; both these values are about 1.4 percent of the total head (water-level) change across the simulated water table for the west (180 ft) and east (142 ft) models. In some cases, larger differences between observed and model-calculated water levels may have resulted from the position of observation wells near the model boundaries (for example, well MNW17 in the west model; table 11), where model discretization or simulation of boundary stresses may have led to less accurate simulated water levels. The simulated water table for east and west model areas (fig. 18) is consistent with the conceptual model of flow in the stratified glacial aquifers. Water-table contours decrease in the downstream direction in tributary valleys and along the Charles River, and bend at large streams reflecting the effect of ground-water discharge.

Calculated base flow at streamflow-measurement sites for the calibrated steady-state models is shown in table 12 and figure 17*B*. Observed and model-calculated stream base flow at or near model exits (stations 01103260 for the west model and stations 01103305 and 01003393 for the east model) are in good agreement, differing by less than 4 percent. The mean difference between model calculated and observed base flow (mean absolute base-flow residual) was 1.6 ft^3/s for the west model and 3.4 ft^3/s for the east model, or 2.3 and 3.4 percent of the total range of observed flows in west and east model areas, respectively. At individual measurement sites, mean absolute base-flow residuals ranged from about 3 to 60 percent of observed values; large differences (greater than 30 percent) generally were associated with sites on small streams with low flows (table 12).

A. WEST MODEL AREA



From USGS and MassGIS data sources, Geographic Projection, Spheroid GRS 1980, Datum NAD 83

Figure 16. Hydraulic conductivity zones for the simulation models of the upper Charles River Basin, eastern Massachusetts: (*A*) West model area, and (*B*) East model area.

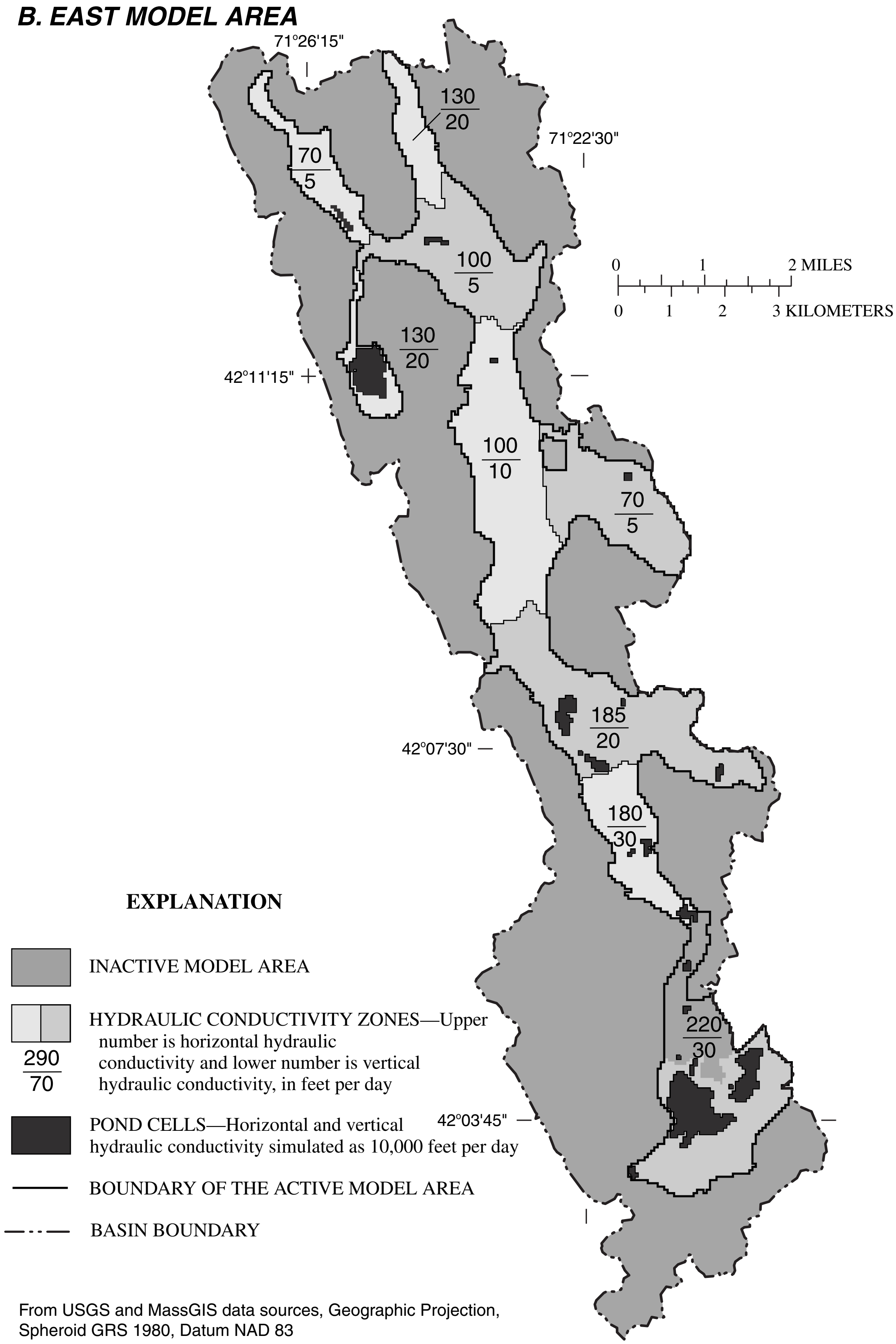


Figure 16. Hydraulic conductivity zones for the simulation models of the upper Charles River Basin, eastern Massachusetts: (*A*) West model area, and (*B*) East model area—*Continued*.

Table 11. Model-calculated steady-state water levels and observed water levels at observation wells and ponds in the upper Charles River Basin, eastern Massachusetts

[Site locations shown in figure 7. **Observed water level**: Estimated for 1989–98 from measurements made during 1999–2000, as described in text]

Well identifier or pond name	Model location			Average annual water level		
	Layer	Row	Column	Model calculated (feet above sea level)	Observed (feet above sea level)	Difference (model calculated minus observed, in feet)
West Model						
A6W52	1	269	80	224.99	229.70	-4.71
A6W53	1	289	103	214.62	212.69	+1.93
A6W55	1	261	124	208.79	208.47	+.32
A6W59	1	272	98	216.25	215.51	+.74
A6W60	1	236	130	199.65	205.69	-6.04
A6W61	1	280	94	218.85	218.99	-.14
A6W62	1	223	116	214.81	210.96	+3.85
A6W63	1	285	94	220.17	220.27	-.10
A6W65	1	282	129	205.85	203.16	+2.69
F2W67	1	254	164	177.27	179.01	-1.74
F2W72	1	354	200	287.07	286.51	+.56
F2W73	1	355	175	248.55	254.38	-5.83
F2W74	1	381	172	260.60	265.50	-4.90
F2W75	1	291	165	189.99	188.51	+1.48
HTW47	1	114	118	240.04	238.56	+1.48
HVW40	1	247	67	234.81	238.36	-3.55
MWW51	1	200	59	250.38	248.71	+1.67
MNW17	1	193	182	173.44	183.02	-9.58
Box Pond	1	290	88	221.38	221.32	+.06
Milford Pond	1	176	57	265.44	265.69	-.25
East Model						
F2W76	1	219	235	129.96	127.09	+2.87
HTW48	1	44	163	204.34	205.24	-.90
HTW49	1	99	225	141.62	143.84	-2.22
HTW50	1	82	184	165.31	167.07	-1.76
HTW51	1	66	202	149.84	149.00	+.84
MNW19	1	192	224	141.15	137.75	+3.40
MNW20	1	157	217	137.76	140.71	-2.95
MYW58	1	160	263	140.28	144.71	-4.43
MYW59	1	171	229	138.09	135.57	+2.49
MYW60	1	125	221	138.46	138.48	-.02
MYW61	1	141	143	126.09	131.64	-5.55
NNW103	1	237	256	129.46	130.25	-.79
NNW104	1	218	256	125.10	128.74	-3.64
NNW105	1	283	275	173.68	173.31	+.37
NNW106	1	262	260	148.96	140.39	+8.57

Table 11. Model-calculated steady-state water levels and observed water levels at observation wells and ponds in the upper Charles River Basin, eastern Massachusetts—*Continued*

Well identifier or pond name	Model location			Average annual water level		
	Layer	Row	Column	Model calculated (feet above sea level)	Observed (feet above sea level)	Difference (model calculated minus observed, in feet)
				East model—*Continued*		
NNW107	1	215	241	130.49	126.50	+3.99
NNW108	1	220	243	128.77	126.85	+1.92
NNW109	1	237	247	130.20	129.31	+.89
NNW110	1	252	253	134.50	132.55	-1.95
NNW112	1	231	237	131.27	130.81	+.46
XUW65	1	330	268	205.02	205.65	-.63
XUW67	1	237	250	196.93	195.35	+1.58
XUW68	1	339	288	200.58	204.20	-3.62
Bush Pond	1	282	276	173.72	173.84	-.12
City Mills Pond	1	263	264	149.06	148.83	+.23
Kingsbury Pond	1	237	250	130.18	129.74	+.44
Lake Pearl	1	340	278	200.06	199.60	-.46
Lake Winthrop	1	116	180	176.14	174.60	+1.54
Old Mill Pond	1	330	278	194.59	194.33	+.26
Populatic Pond	1	219	239	128.68	127.27	+1.41

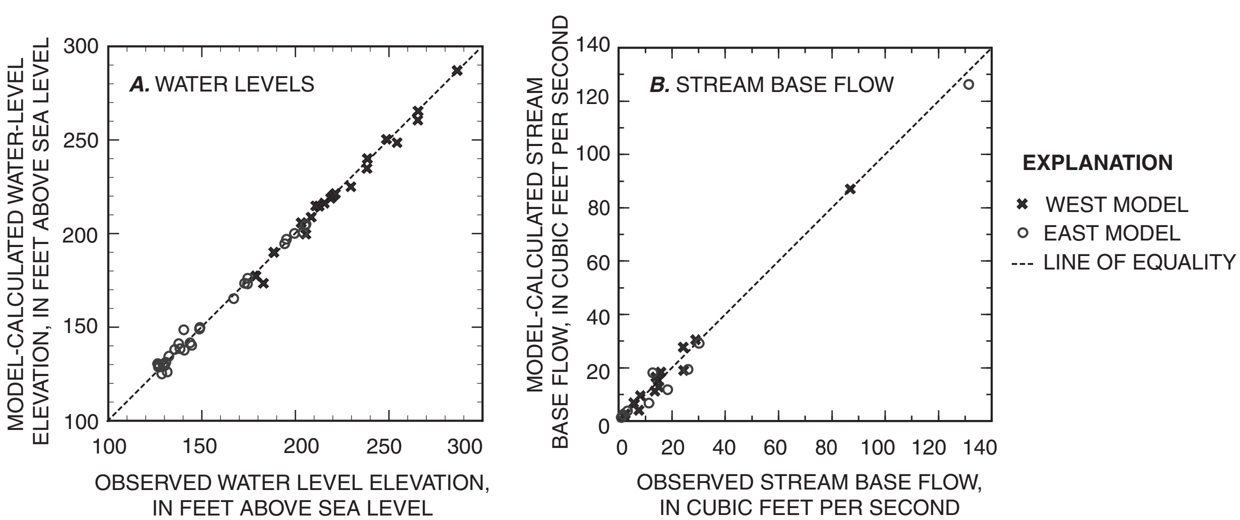
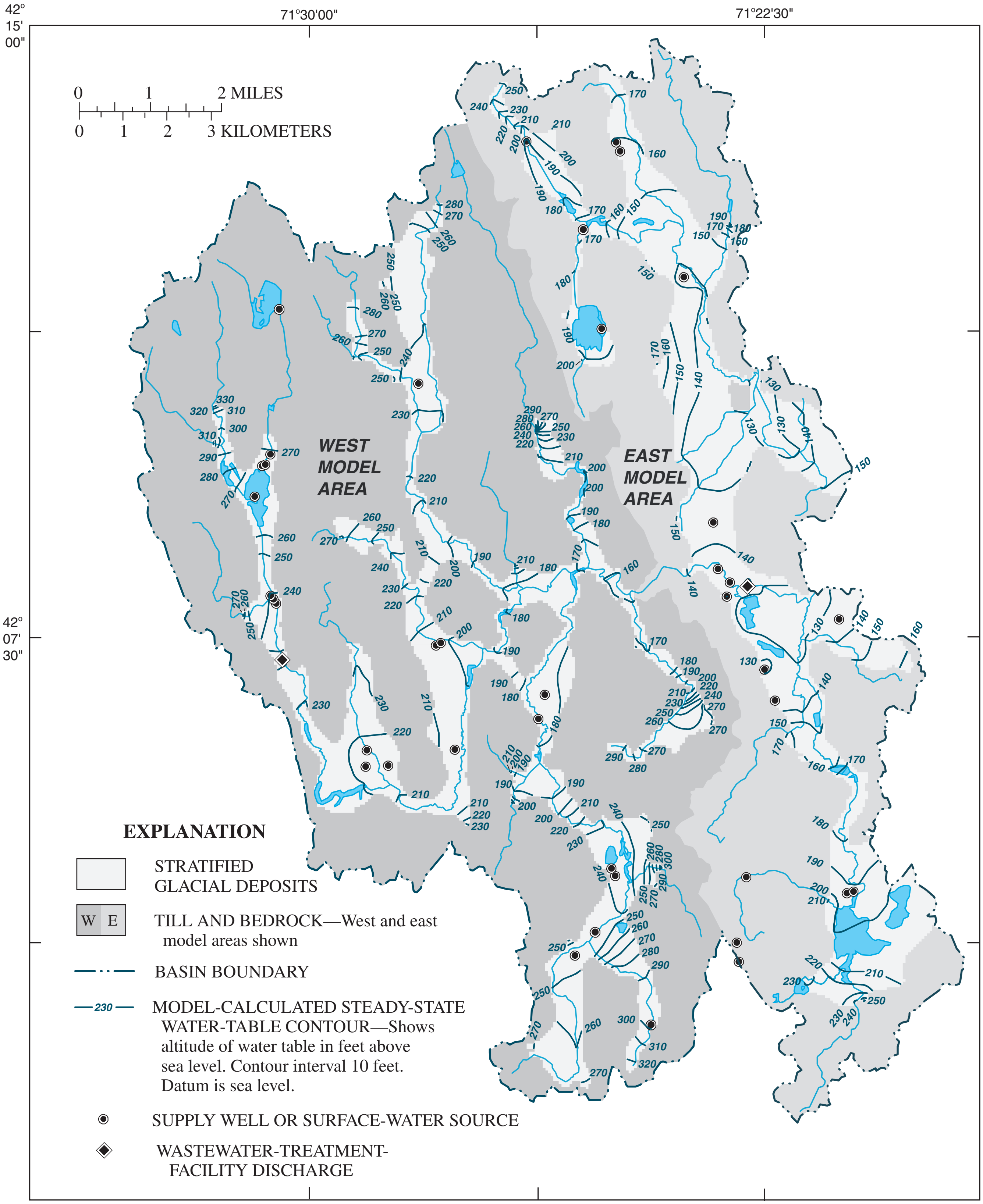


Figure 17. Relation between observed and model-calculated water levels and stream base flow for average conditions, 1989–98, for steady-state simulation models of the upper Charles River Basin, eastern Massachusetts: (*A*) Water levels, and (*B*) Stream base flow. Observed values are estimates for 1989–98 from measurements made in 1999–2000 as described in the text. Line is line of equality between measured and simulated water level.



From USGS and MassGIS data sources, Geographic Projection, Spheroid GRS 1980, Datum NAD 83

Figure 18. Model-calculated steady-state water table in the upper Charles River Basin, eastern Massachusetts.

Table 12. Model-calculated steady-state stream base flow and observed stream base flow at measurement sites in the upper Charles River Basin, eastern Massachusetts

[Site locations shown in figure 7. **Observed base flow:** Estimated for 1989–98 from measurements made during 1999–2000, as described in text]

Station No.	Model location			Average annual stream base flow (cubic feet per second)				
	Layer	Row	Column	Model calculated	Observed			Difference (model calculated minus observed)
					Flow	90-percent confidence limits		
						Lower	Upper	
West Model								
Charles River								
01103110	1	161	61	1.8	2.0	1.2	3.2	-0.2
01103120	1	197	58	6.9	5.6	3.0	10.5	+1.3
01103140	1	271	76	18.4	15.8	10.6	23.7	+2.6
011032056	1	282	131	27.7	24.2	14.7	27.5	+3.5
01103206	1	239	140	30.5	28.8	17.2	34.0	+1.7
01103260	1	210	186	87.1	86.9	21.4	38.9	+.2
Tributaries to Charles River								
01103210	1	233	137	4.1	7.6	0.8	71.7	-3.5
01103217	1	200	136	15.3	14.8	5.8	37.8	+.5
01103225	1	375	167	2.7	2.5	1.5	4.2	+.2
01103234	1	305	192	11.3	13.6	10.2	18.0	-2.3
01103235	1	302	175	12.8	15.0	7.9	28.6	-2.2
01103240	1	232	170	19.0	24.4	14.9	40.0	-5.4
011032515	1	182	176	9.5	8.1	5.5	12.1	+1.4
East Model								
Charles River								
01103280	1	204	226	95.8	91.0	59.7	139	+4.5
01103305	1	216	263	126	131	103	168	-5.0
Tributaries to Charles River								
01103292	1	326	281	6.8	11.3	8.0	16.0	-4.8
01103295	1	279	275	11.9	18.4	11.9	28.4	-6.5
01103300	1	236	258	19.4	26.0	18.3	37.0	-6.6
Bogastow Brook and Tributaries								
01103381	1	85	176	3.8	3.3	2.2	4.9	+.5
01103386	1	66	201	1.3	1.0	.5	2.0	+.3
011033885	1	79	232	2.4	1.6	1.1	2.2	+.8
01103389	1	103	223	18.2	12.8	8.0	20.5	+5.4
01103393	1	132	243	29.1	30.2	20.9	43.6	-1.1

The steady-state, average annual hydrologic budget calculated with the calibrated model is shown in table 13. Total inflows were 70 Mgal/d to the west model area and 49 Mgal/d to the east model area. In both models, inflows from uplands were large components, representing about 70 percent of inflows to the west model and about 50 percent of inflows to the east model. These proportions are larger than estimates (30 to 50 percent) made in the water balance for the study area, because upland inflows in the model budget include precipitation and return-flow recharge to thin areas of stratified glacial deposits that are excluded from the active model area. Leakage to the aquifer from streams was a small component (less than 10 percent) of both model budgets. Stream leakage resulted near the model boundary downstream of the specified streamflow in the west model; stream leakage occurred elsewhere in the models near pumping wells (induced infiltration) or downstream of constrictions in aquifer geometry. Inflows from wastewater discharge to

Table 13. Model-calculated steady-state hydrologic budget for the upper Charles River Basin, eastern Massachusetts

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

Hydrologic budget component	Rate of flow					
	West model		East model		Entire upper Charles Basin	
	Cubic feet per second	Million gallons per day	Cubic feet per second	Millions gallons per day	Cubic feet per second	Million gallons per day
Inflow						
Recharge from precipitation and septic-tank return flow	19.7	12.7	25.4	16.4	45.1	29.1
Lateral ground-water inflow[1]	56.1	36.3	22.9	14.8	79.0	51.1
Streamflow from uplands[1, 2]	22.3	14.4	17.8	11.5	40.1	25.9
Stream leakage to aquifer	4.7	3.0	3.7	2.4	8.4	5.4
Wastewater discharge to streams	5.5	3.6	6.1	3.9	11.6	7.5
Total inflow[2]	108.3	70.0	75.8	49.0	184.1	119.0
Outflow						
Ground-water discharge to streams	97.1	62.8	64.5	41.7	161.6	104.5
Evapotranspiration from wetlands and ponds	4.4	3.3	5.1	3.3	9.5	6.1
Water-supply withdrawal	6.7	4.3	6.2	4.0	12.9	8.3
Total outflow[2]	108.2	69.9	75.8	49.0	184.0	118.9
Budget error (inflow-outflow)	0.1	0.0	0.0	0.0	0.1	0.1

[1]Simulated as enhanced recharge at model boundaries.

[2]Does not include specified stream inflows to the Charles River, equal to 2.3 ft^3/s for the west model, 97.1 ft^3/s for the east model, and 2.3 ft^3/s for the entire basin. These inflows are not part of the model budget calculations.

streams also were small components of the model budgets. Total net inflow to west and east models, 119 Mgal/d, was consistent with the range of values determined for the study area with the water balance (111 to 141 Mgal/d).

Calculated outflows from the models consisted primarily of discharge to simulated streams; these outflows were 90 and 85 percent of total outflows from west and east models, respectively. Although small relative to total outflows, model-calculated ET from wetlands and ponds was similar in magnitude (though much less well quantified) to withdrawals for water supply, indicating the potential importance of this flux. Stream outflows that are calculated in the model budget represent ground-water discharge to streams within the active area. The calculated stream outflows do not include inflows specified at model boundaries in the stream package; specified inflows are routed through the model unaltered unless water in stream cells leaks into the aquifer, in which case the inflows are available for aquifer recharge. Thus, the simulated stream base flow at the exit of the west model was equal to the model-calculated flow from ground-water discharge to streams (97.1 ft^3/s) minus stream leakage to the aquifer (4.7 ft^3/s) plus the specified inflow for the Charles River in Milford (2.3 ft^3/s), or 94.7 ft^3/s. This value was specified as the inflow to the east model for the Charles River in Medway. Similarly, the simulated stream base flow at the exits of the study area (Charles River and Bogastow Brook in Norfolk and Millis) was equal to the specified inflow to the east model, plus the model-calculated flow from the ground-water discharge to streams for the east model (64.5 ft^3/s), minus the model-calculated stream leakage to the aquifer in the east model (3.7 ft^3/s), or 155.5 ft^3/s (100.5 Mgal/d). The simulated stream base flow at model exits plus other model outflows of evapotranspiration and water-supply-withdrawals yields total net outflows that are consistent with the values determined for the study-area water balance (119 Mgal/d with a 90-percent confidence-interval range of 95 to 152 Mgal/d on the basis of base-flow estimates).

Withdrawals for water supply account for about 7 percent of model-calculated average annual inflows or outflows on a basin-wide basis (table 13). Within individual tributary basins, however, water-supply withdrawals constituted more or less than this basin-wide average. For example, water-supply withdrawals accounted for 13 percent of average annual inflows in the headwaters Charles River area in Milford (upstream of station 01103140), about 10 percent of inflows in the Stall Brook, Mine Brook, and Mill River Subbasins (areas upstream of stations 01103210, 01103240, and 01103300, respectively), about 8 percent of inflows in the Bogastow and Dopping Brook Subbasins (upstream of station 01103989), and about 1.5 percent of inflows in the Hopping Brook Subbasin (upstream of station 01103217).

Transient Numerical Models

Transient models were developed to simulate the variations in hydrologic conditions within an average annual cycle. The transient models are based on the steady-state models but incorporate time-varying hydraulic stresses and boundary conditions. The spatial discretization of the model grid, boundary conditions other than specified flows, and spatial variations in stresses and hydraulic conductivities are the same in transient and steady-state models. The transient models are designed to simulate dynamic equilibrium, or the condition in which there is no net change in storage over the annual cycle (Barlow and Dickerman, 2001). The transient models were calibrated by comparing simulated stream base flow and water levels to average monthly flows and levels estimated for the 1989–98 period.

With the transient models, the low-flow periods of the annual cycle can be simulated. Low-flow periods, typically the late summer months, often are of particular concern in the evaluation of the effects of water-management alternatives. During these periods, the effects of water withdrawals and other management practices on aquatic life and stream-water quality often are greatest, because their effects are combined with naturally low flows and ground-water levels. Water demands also typically are highest during summer months. Thus, seasonally varying stresses and fluxes within the annual cycle often must be considered when the effects of water-management alternatives are tested.

Temporal Discretization and Initial Conditions

The annual hydrologic cycle was divided into 12 monthly time periods that varied in length from 28 to 31 days. Within each month, hydraulic stresses and boundary flows were uniform. Multiple model stress periods (with the same hydraulic stresses and boundary flows) were used to simulate some months to improve model stability, particularly during months of low recharge. Fifteen stress periods per year were used in the west model and 19 to 23 stress periods per year were used in the east model. Stress periods were divided into 30 time steps, regardless of length. The time steps increased in length by a factor of 1.3 during each stress period, for example, from 4.6 x10^{-4} to 0.9 day for stress periods of 4 days and from 3.6 x10^{-3} to 7.2 days for stress periods of 31 days.

Water-level elevations from the calibrated steady-state models were specified as the initial conditions for the transient simulations. The calibrated steady-state water levels represent average annual conditions. The transient simulations also began with stresses and boundary flows representing conditions in the month of January. Because January conditions differ from average annual conditions, changes in calculated water levels and flows in early parts of the transient simulations resulted from this initial discrepancy as well as from time-varying stresses. Transient simulations were run for repeated 1-year cycles until the effects of the initial conditions were eliminated and there was no change in storage over a 1-year cycle. After five annual cycles for both west and east models, the difference between flow into and out of storage was 0.2 and 0.1 percent for west and east models, respectively. These difference indicated that net changes in storage were negligible, compared to the total water budget. Five annual cycles consisted of 75 stress periods for the west model and 107 stress periods for the east model.

Boundary Conditions and Stresses

Boundary conditions in the transient models were similar to those used in the steady-state models. As in the steady-state models, stream base flow was specified in the stream package at locations where the Charles River entered the active area of west and east models. For the Charles River at the upstream boundary of the west model, monthly average base flow was varied on the basis of the monthly distribution of upland inflows, described below, and the base flow estimated from measurements at streamflow site 01103110 (fig. 7). Monthly average base flow in the Charles River at the upstream boundary of the east model was equal to the model-calculated monthly average base flow in the Charles River at the west model exit. Stresses in the transient models were of the same type as in the steady-state models, but varied monthly.

Average monthly recharge rates were based on the average annual recharge rate from the calibrated steady-state models, analyses of the monthly distribution of annual recharge in the Hunt River Basin in southern Rhode Island (Barlow and Dickerman, 2001), and observed streamflows at two sites near active model boundaries. Temporal recharge patterns in the Hunt River Basin were considered representative for the upper Charles River Basin because climatic conditions, average annual recharge rates, and the areal percentage of stratified glacial deposits in the two basins are similar. The monthly distribution of annual recharge in the Hunt River Basin was determined by Barlow and Dickerman (2001) from the analysis of long-term streamflow records with a base-flow recession-curve displacement method developed by Rorabaugh (1964) and automated by Rutledge (1993). This approach to estimating average monthly varying recharge rates has the advantage over water-balance methods applied at the monthly scale. Because it is implemented with a daily time step, the Rutledge (1993) method can yield greater-than-zero recharge rates in summer months, whereas comparison of monthly average potential ET and monthly average precipitation would yield monthly average recharge estimates equal to zero. Although the recession-curve displacement method may under- or over-estimate recharge rates in months when ground-water ET, underflow, or water withdrawals (assumed negligible in this approach) are substantial outflows from the stream-aquifer system (Nicholson and Watt, 1997; R.S. Nicholson, U.S. Geological Survey, written commun., 2001), the method was applicable to the Hunt River Basin because these withdrawals were small relative to streamflow (Barlow and Dickerman, 2001).

Average monthly recharge rates were calculated by varying the calibrated rate for natural precipitation recharge for stratified glacial aquifers proportionately through the year on the basis of the distribution of annual recharge among months in the Hunt River Basin. This approach made rates for all recharge categories uniformly proportional, because the cell-by-cell recharge rates applied in the model were the product of the natural precipitation rate to stratified glacial deposits and a multiplier array. Initially, the same distribution of annual recharge among months was used for recharge to interior cells of the active model area, which represents only precipitation and return-flow recharge to stratified glacial aquifers, and for enhanced recharge to cells along the active model boundary, which represents inflows from uplands as well as precipitation and return-flow recharge. This initial uniform monthly distribution of recharge was modified during model calibration. Final monthly recharge rates to interior cells (those representing stratified glacial deposits without inflow from uplands) and flow rates to cells receiving upland inflows are shown in figure 19*A*. The monthly rates for upland inflows and for recharge to stratified glacial deposits both correspond to the same average annual rate as used in the steady-state models. Flow rates to cells receiving upland inflows are more variable from month to month than recharge rates to interior aquifer cells. The variability in upland flow rates is comparable to the variability observed in estimated monthly base flow for streams draining upland basins in the upper Charles River Basin (stations 01103110 and 011033885; fig. 19*B*).

Wetlands and ponds drained by streams were simulated in the steady-state models as areas of net specified water loss from the stream-aquifer system. The rate of loss was equal to the growing-season, free-water-surface evaporation rate, or 21 in/yr (Farnsworth and others, 1982). In the transient simulations, this loss was assumed to occur uniformly throughout the months of May to October, for an average monthly rate of 3.5 in/month. Ponds not drained by streams were simulated as net recharge areas (22 in/yr) in the steady-state models. This recharge was assumed to be distributed between summer and winter months by removing free-water-surface evaporation rates from precipitation during these periods (Farnsworth and others, 1982). Recharge to ponds not drained by streams was specified as 2.7 in/month from November to April and 0.9 in/month from May to October in the transient models.

Monthly withdrawals for water supply and discharges from wastewater-treatment facilities were set equal to average monthly volumes for 1989–98. Withdrawals at water supplies outside the active area of the east model, the town of Franklin supply well No. 9 and the Franklin Country Club well and reservoir (fig. 3),

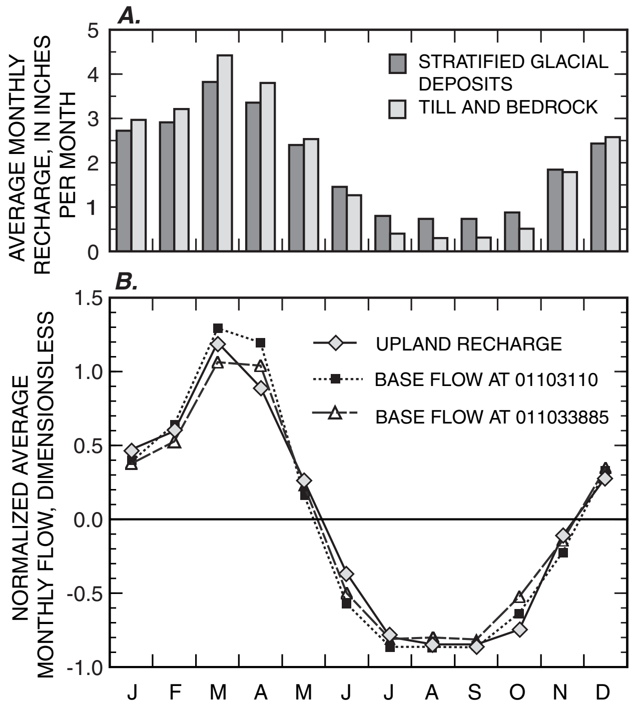


Figure 19. Monthly average recharge rates for transient simulation models and comparison of monthly average rates for flows to the active model area from upland till and bedrock areas with estimated monthly base flow in two streams draining upland watersheds in the upper Charles River Basin, eastern Massachusetts: (*A*) Monthly average recharge rates for stratified glacial deposits and rates for flows to the active model area from upland till and bedrock areas. Monthly average recharge rates for stratified glacial deposits are from Barlow and Dickerman (2001), and (*B*) Monthly average rates for flows to the active model area from upland till and bedrock areas and estimated monthly average base flow in the Charles River above Cedar Swamp Pond near Milford (01103110) and Dirty Meadow Brook at Hollis Street near Holliston (011033885). Rates shown are normalized to average annual rates.

were not varied monthly. The average annual withdrawals were used for these sources. In steady-state and transient models, withdrawals at these sources were represented by reductions in the upland inflows, which were input to the models by use of a recharge multiplier. Echo Lake Reservoir (fig. 3), in the west model, also was outside of the active model area. Withdrawals at Echo Lake Reservoir were represented, as in the steady-state model, by excluding the drainage area of Echo Lake Reservoir from the upland area contributing water to the active model area and by including winter withdrawals of water for transfer to Echo Lake Reservoir from the Charles River surface-water source (fig. 3). Water withdrawals for transfer from the Charles River source averaged 0.57 Mgal/d in January, 1.30 Mgal/d in February, 1.78 Mgal/d in March, 1.56 Mgal/d in April, and 0 Mgal/d in other months, based on available data.

Hydraulic Properties

A uniform value of 0.28 for specific yield was used for stratified glacial deposits in most active model areas. This value is similar to values of specific yield determined for stratified glacial deposits on Cape Cod and in southern Rhode Island (Allen and others, 1963; Moench and others, 2000) and to values determined from laboratory studies (Morris and Johnson, 1967; Johnson, 1967; Nwankwor and others, 1984). For areas along the Mill River in the east model, where available data indicated locally confined conditions, specific yield values of 0.2 and 0.15 were used. These values were used to represent spatially averaged effects of the local confining units, which otherwise were not simulated. A specific yield of 1.0 (100 percent porosity) was specified for all simulated ponds. A uniform value for specific storage of 1.0 x 10^{-3} per foot was specified for stratified glacial deposits in most areas. This value corresponds to a storage coefficient of 1.0 x 10^{-2} for a 10-ft saturated thickness of aquifer, which is within the range of values determined for stratified glacial deposits in New England (Kontis and others, in press).

Model Calibration

The transient models were calibrated to water-level fluctuations measured at observation wells and ponds during the study (August 1999 through November 2000) and estimated average (1989–98) monthly base flow at streamflow-measurement sites (“observed values”). Trial-and-error adjustment of storage parameters and, to some extent, the monthly distribution of annual recharge, was used to obtain reasonable agreement between simulated and observed values for both water levels and flows. Calibration of west and east models proceeded concurrently. Particular weight was given to the distribution of monthly streamflow at measurement sites near model exits on the Charles River (stations 01103260 and 01103305) and Bogastow Brook (01103393; fig. 7), because these sites integrate conditions throughout the model areas. Greater consideration also was given to simulated water-level fluctuations at observation wells near the horizontal center of aquifers than to simulated fluctuations at wells near model boundaries with inactive upland areas.

Simulated and observed water-level fluctuations at selected observation wells and ponds are shown in figure 20. Fluctuations are shown relative to the average of model-calculated monthly water-level elevations for model-calculated values and relative to the average of water levels measured in water year 2000 for the observed values (table 5). In general, seasonal patterns of measured water-level fluctuations were reasonably well matched by simulated water levels. The timing of model-calculated and measured maximum and minimum water levels within the annual cycle was similar at most sites, although simulated water levels peaked 1–2 months before measured water levels at some sites. Model-calculated water levels also tended to fluctuate more during the annual cycle than water levels measured during the study. The average difference between model-calculated and observed annual water-level fluctuations was 0.9 ft (median 0.6 ft) for the 48 measurement sites (table 5) in the active model area. Large differences (greater than about 2 ft) generally occurred at observation wells adjacent to model boundaries (for example, F2W72, fig. 20*A*). Large differences also occurred at some ponds on the stream network (for example, Box Pond, fig. 20*B*) and may have resulted from the inability of the model to simulate streamflow hydraulics, such as pond storage. Differences between observed and simulated fluctuations also may have resulted from deviations of observed conditions during the measurement period from long-term average conditions.

Simulated and observed average monthly base flow at selected measurement sites on the Charles River, Bogastow Brook, and tributaries are shown in figure 19. Simulated and observed average monthly base flow agreed reasonably well at most sites. Most model-calculated monthly base flow was within the range of values represented by 90-percent confidence intervals for observed base-flow estimates. Temporal patterns in base flow were simulated and the mean absolute base-flow residuals averaged 41 percent of observed flows overall. At sites on the Charles River, observed monthly flows were better matched, with the mean absolute base-flow residuals averaging 26 percent of observed values overall at eight measurement sites (table 12 and fig. 21*A*–*D*). The mean absolute base-flow residuals averaged 22 percent of observed values at each of the model exit sites on the Charles River (01103260 and 01103305) and 50 percent of observed values at the model exit site on Bogastow Brook (01103393), respectively. At most sites, including sites on the Charles River and some tributaries (for example, Chicken and Dopping Brook sites, fig. 21*H* and *J*), model-calculated spring high base flow was larger than observed flows (fig. 22*A*; mean base-flow residual for March flows averaged 11 percent of observed values; mean absolute base-flow residuals averaged 29 percent of observed values). Model-calculated late-summer low flows were less biased than high flows (mean base-flow residuals for September flows averaged 5 percent of observed values), but there was more scatter in observed and model-calculated values at the low flows (fig. 22*B*). Late-summer low flows occurred later in the annual cycle for simulated flows than for observed flows at most of these sites (fig. 21). At a number of sites on tributaries draining proportionately large upland areas (for example, Hopping Brook, 01103217, fig. 21*F*) or on tributaries in the Mine Brook aquifer area, spring flows were underestimated and summer flows overestimated. The flashiness of streams in these areas may have been more strongly affected by lateral and channeled inflow from uplands than streams in other aquifer areas. Finally, discrepancies between model-calculated and observed base flow from the calibrated steady-state model resulted in large differences between model-calculated and observed monthly average base flow for some sites in the transient models, including those in the Mill River aquifer area (for example, 01103295, fig. 21*I*). Generally, seasonal patterns in flows are reasonably well matched at these sites, however.

During model calibration, aquifer storage properties and the distribution of annual recharge among months were varied to minimize the differences between simulated and measured water-level fluctuations and stream base flow. Specific yield was varied between values of 0.26 and 0.30 (except in zones where locally confined conditions were present). Although a specific yield of 0.30 improved the east model fit slightly with respect to water-level fluctuations, a value of 0.28 was retained for both models because it was considered more representative for aquifer sediments in the study area, on the basis of literature sources, and because of greater numerical instability in the west model with the higher value of specific yield. Specific storage also was varied, but had little effect on water-level fluctuations or stream base flow. The distribution of inflows from upland areas, simulated as enhanced recharge to cells along the active model boundary, was varied as described previously to increase the annual range of simulated base flow and better match the observed annual values.

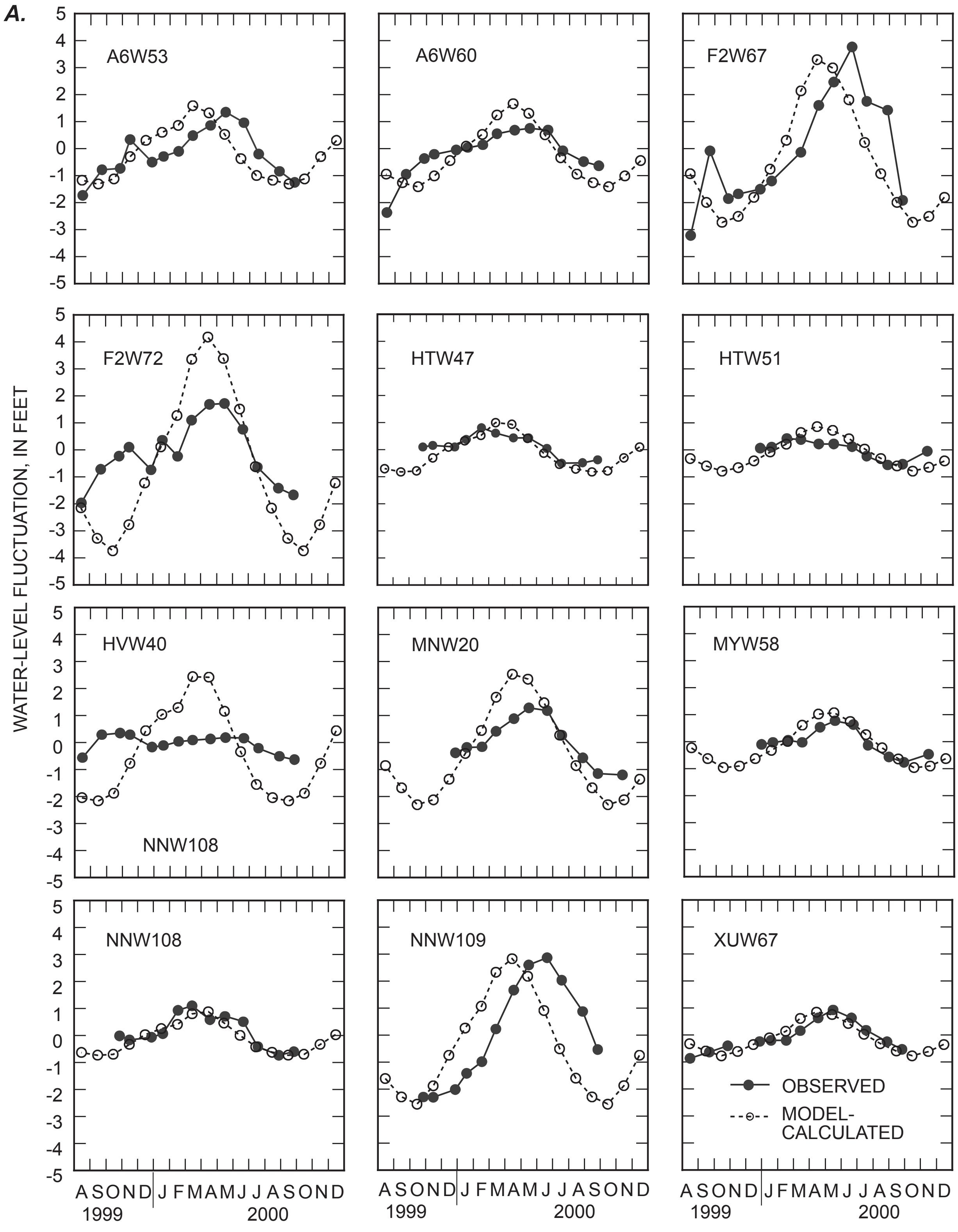


Figure 20. Model-calculated and observed water-level fluctuations during the annual average cycle for selected observation wells and ponds in the upper Charles River Basin, eastern Massachusetts: (*A*) Observation wells, and (*B*) Ponds. Fluctuations are shown relative to the average of water levels measured in water year 2000 for the measured values (see table 5) and relative to the annual average of calculated monthly water levels for model-calculated values (see table 11).

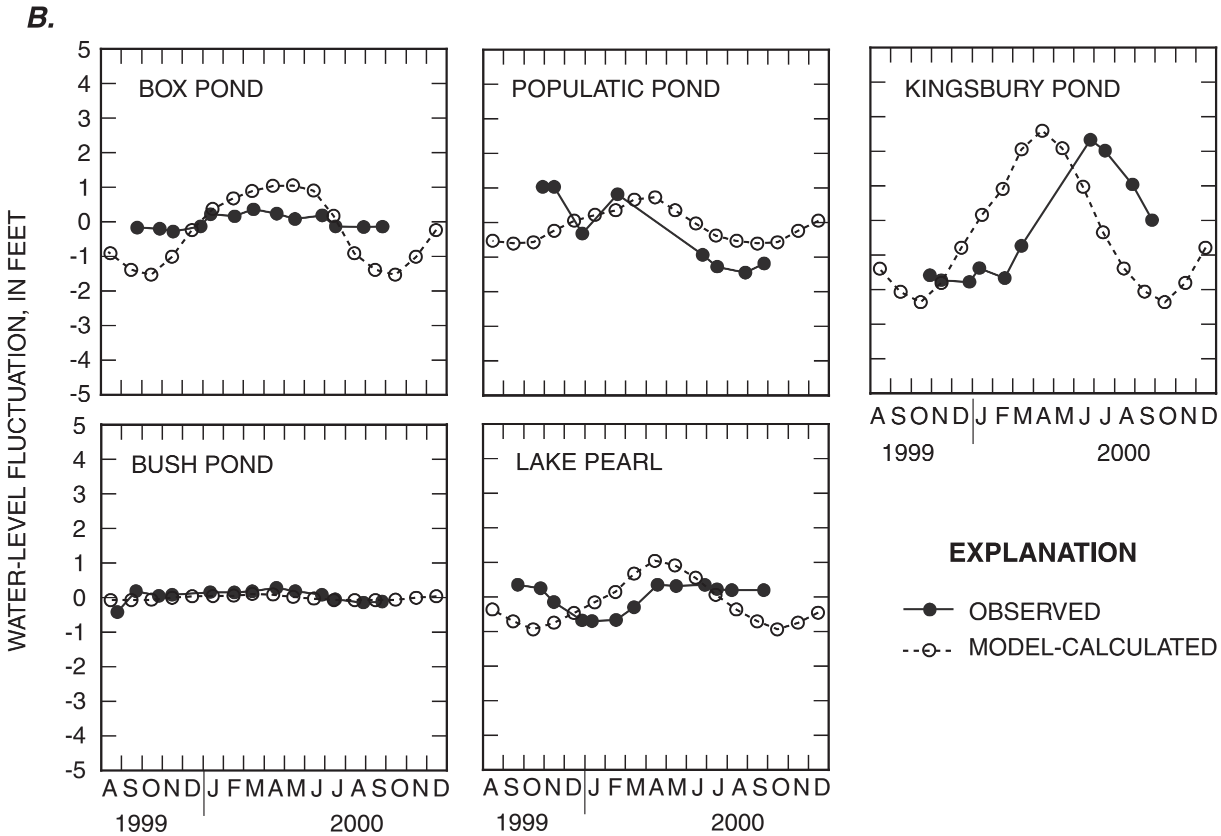


Figure 20. Model-calculated and observed water-level fluctuations during the annual average cycle for selected observation wells and ponds in the upper Charles River Basin, eastern Massachusetts: (*A*) Observation wells, and (*B*) Ponds. Fluctuations are shown relative to the average of water levels measured in water year 2000 for the measured values (see table 5) and relative to the annual average of calculated monthly water levels for model-calculated values (see table 11)—*Continued*.

Differences between simulated and observed water-level fluctuations and stream base flow may have resulted from various causes, including model calibration error, discretization effects, or inadequate simulation of aquifer geometry, storage properties, recharge or other hydrologic processes. Deviations between the annual cycle of water levels, as represented by model-calculated, long-term monthly averages and as represented by the instantaneous measurements made during the study period also may have contributed to the differences between simulated and measured water-level fluctuations. The observed long-term monthly average streamflows, which were estimates from partial streamflow records, also may have contained multiple sources of error. Further modifications to aquifer-storage properties, to specified rates of precipitation and return-flow recharge, or to specified rates of enhanced recharge representing inflows from uplands might have been made to improve the match between simulated and observed water-level fluctuations and streamflows. However, these modifications were judged to be inappropriate given the limited availability of data for these variables and the expected sources of error inherent in the observations against which simulated water-level fluctuations and stream base flow were compared.

The average annual hydrologic budget for the entire upper Charles River Basin calculated with the transient models agreed well with the steady-state budget. Inflow and outflow components, except for stream leakage to the aquifer, differed by 1 percent or less between transient and steady-state models; thus, the flows specified in the steady-state models were accurately represented in the transient models. Larger percentage differences between the steady-state and transient models in stream leakage, which constitutes about 5 percent of total flow, were probably insignificant.

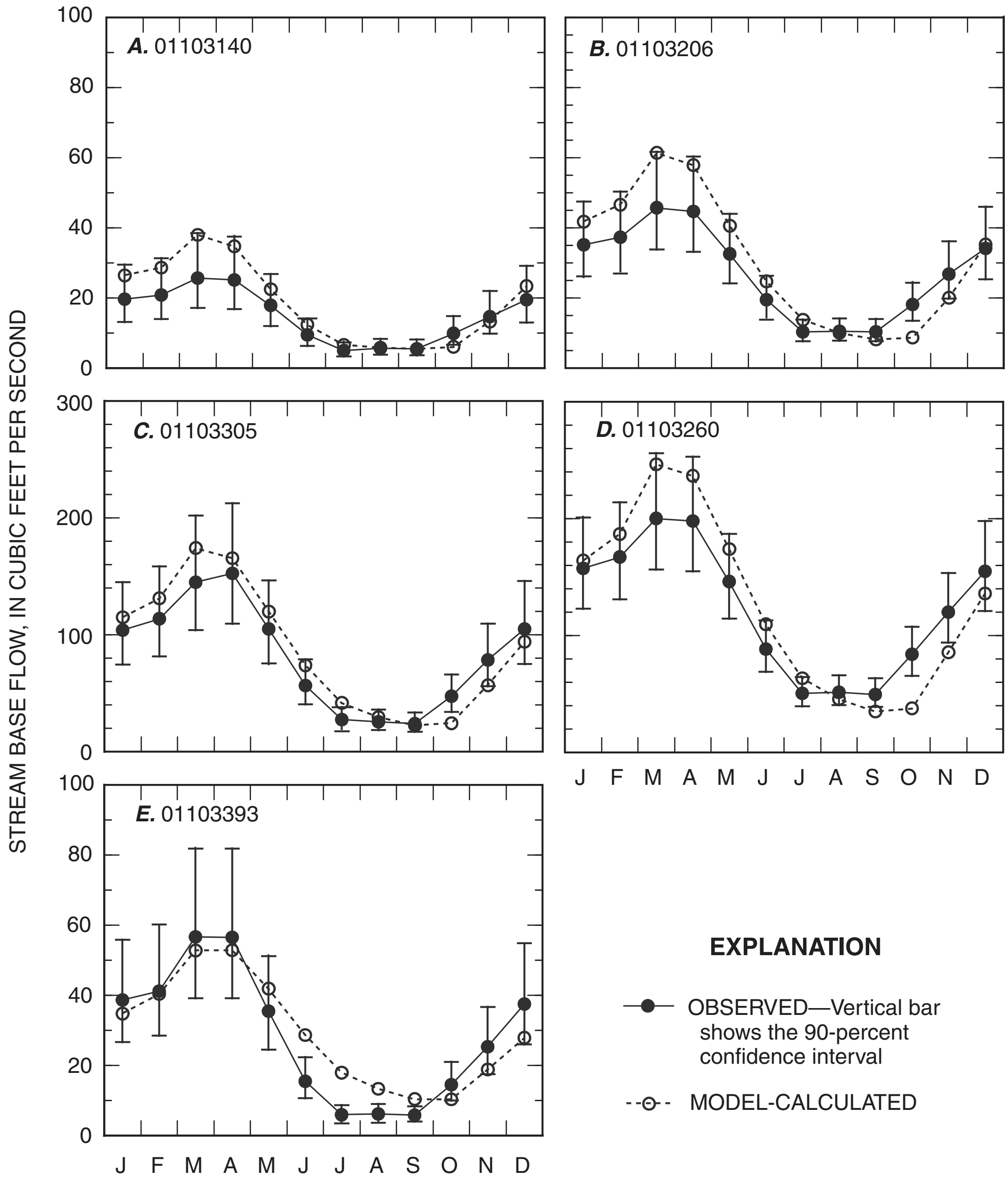


Figure 21. Model-calculated and observed average monthly base flow in the Charles River, Bogastow Brook, and tributaries, upper Charles River Basin, eastern Massachusetts: (*A*) Charles River at Factory Pond at South Milford (01103140), (*B*) Charles River at North Bellingham (01103206), (*C*) Charles River at West Medway (01103260), (*D*) Charles River near Millis (01103305), (*E*) Bogastow Brook below Great Black Swamp near Millis (01103393), (*F*) Hopping Brook near West Medway (01103217), (*G*) Mine Brook near Franklin (01103235), (*H*) Chicken Brook below Milk Pond near West Medway (011032515), (*I*) Mill River below Bush Pond near City Mills (01103295), and (*J*) Dopping Brook at Whitney Street, Holliston (01103386). Observed values are estimates for 1989–98 based on measured flows in 1999–2000, as described in text.

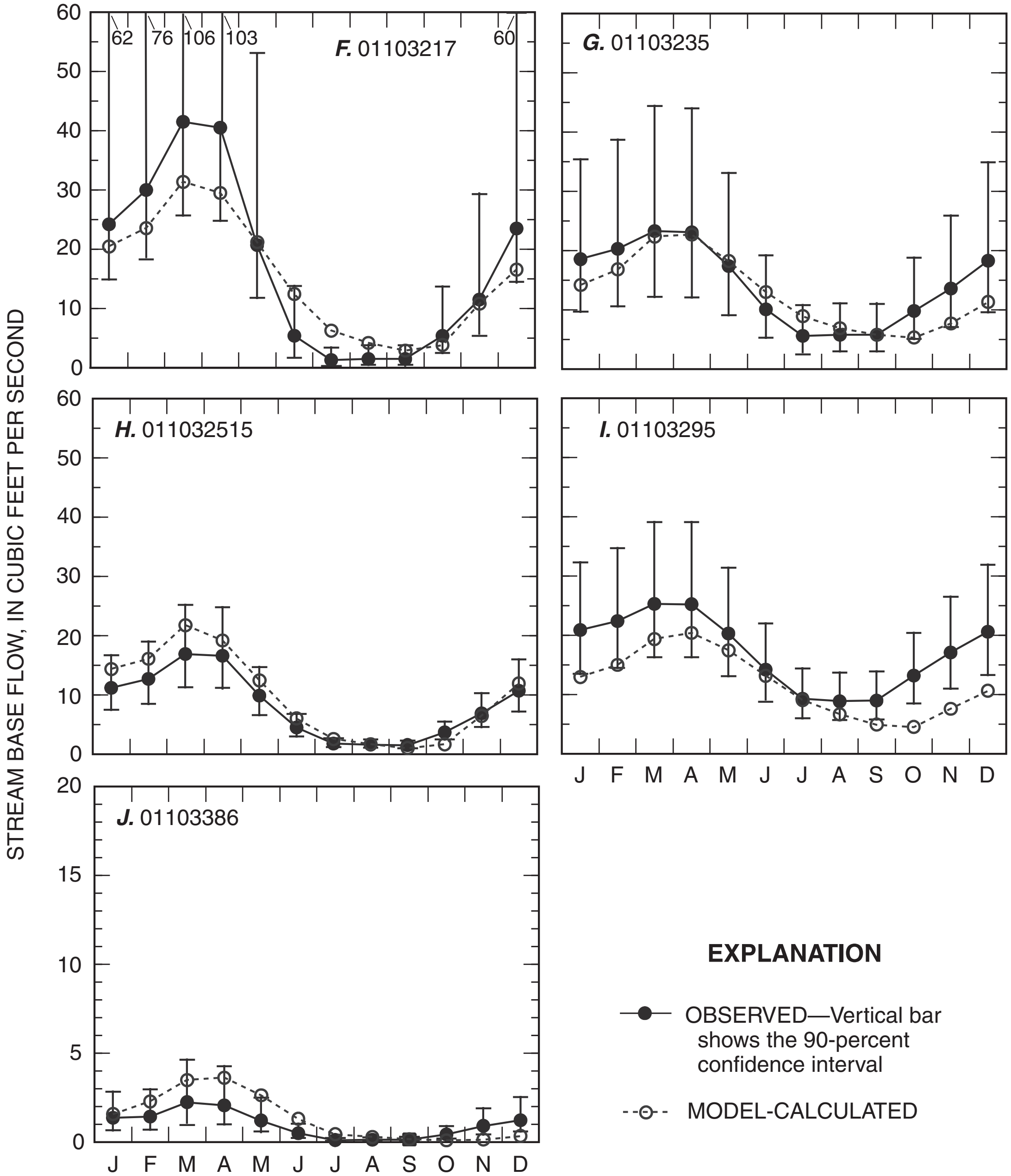


Figure 21. Model-calculated and observed average monthly base flow in the Charles River, Bogastow Brook, and tributaries, upper Charles River Basin, eastern Massachusetts: (*A*) Charles River at Factory Pond at South Milford (01103140), (*B*) Charles River at North Bellingham (01103206), (*C*) Charles River at West Medway (01103260), (*D*) Charles River near Millis (01103305), (*E*) Bogastow Brook below Great Black Swamp near Millis (01103393), (*F*) Hopping Brook near West Medway (01103217), (*G*) Mine Brook near Franklin (01103235), (*H*) Chicken Brook below Milk Pond near West Medway (011032515), (*I*) Mill River below Bush Pond near City Mills (01103295), and (*J*) Dopping Brook at Whitney Street, Holliston (01103386). Observed values are estimates for 1989–98 based on measured flows in 1999–2000, as described in text—*Continued*.

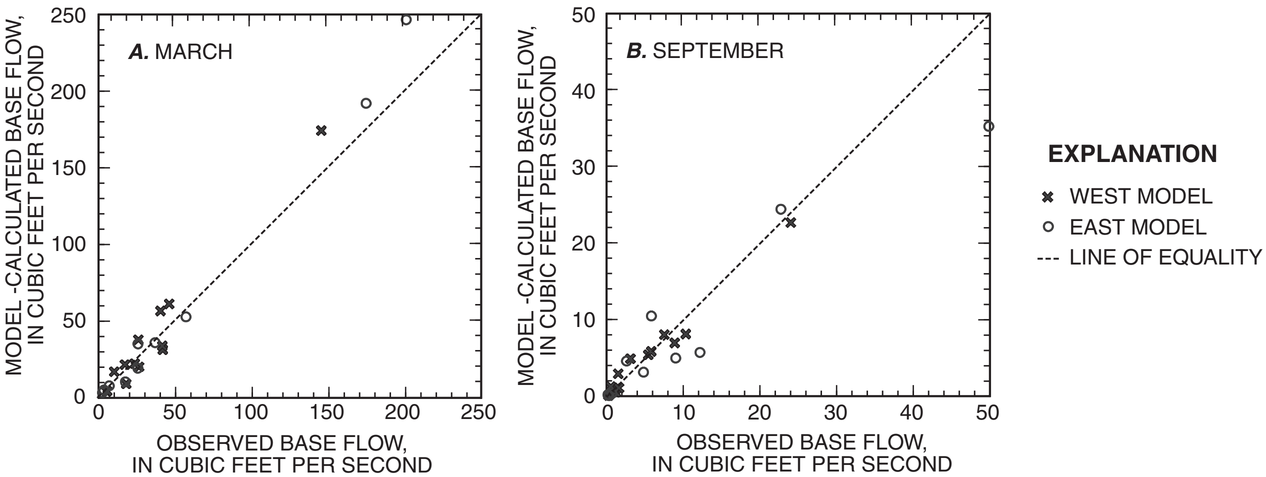


Figure 22. Relation between observed and model-calculated average March and September base flow for measurement sites in the upper Charles River Basin, eastern Massachusetts: (*A*) March, and (*B*) September. Observed values are estimates for 1989–98 from measurements made in 1999–2000 as described in the text. Line is line of equality between observed (estimated) and model-calculated values.

The average annual rate of inflow to and outflow from aquifer storage calculated with the transient models was 21 Mgal/d, or about 18 percent of total flow through the stream-aquifer system. Water is added to aquifer storage from November through April and removed from storage from May through October. Hydrologic budgets calculated with the transient models also indicated that monthly water withdrawals, which averaged 7 percent of total flow on an annual average basis, range from about 4 to 5 percent of model-calculated total monthly flow in winter and spring months (November to May) to about 10 to 12 percent of model-calculated total monthly flow in summer and early fall months (June to October).

Model Limitations

The steady-state and transient flow models of the upper Charles River Basin stream-aquifer system provide a regional-scale simulation of ground-water flow in the stratified glacial aquifers in the study area. As with all mathematical models of natural systems, the simplifications and assumptions incorporated into the models result in limitations to their appropriate uses and to the interpretations that may be made of simulation results.

The upper Charles River Basin flow models simulate ground-water flow, water levels, and the interaction with surface-water features at the regional scale. Hydrologic processes and spatial variability in hydraulic properties and stresses are simplified and approximated to a degree consistent with this scale. The model calibration also represents the best fit to estimates and observations made throughout the upper Charles River study area. Thus, the agreement between simulated water levels or stream base flow in specific areas of the flow system may not be adequate to support local-scale model applications. The model discretization also may not be applicable for local-scale problems. In fact, model calibration generally should be taken into consideration whenever the effects of simulated changes in water-management practices or hydrologic conditions at specific observation points are evaluated. Because of the error associated with regional-scale calibration, simulation results for many applications may best be viewed as evidence of relative changes in water levels or base flow rather than as absolute changes at specific locations.

The effects of temporal and spatial discretization also impose limitations on model use. Hydrologic processes and hydraulic stresses were represented in the transient models as monthly averages. Simulation results are monthly average ground-water levels and flows. The models were not intended to be used to simulate changes that occur at finer time scales, for example, daily values, which may substantially exceed or fall below monthly average values. The spatial resolution of the simulation results was limited by the area of the 200x200-ft grid cell. Water withdrawals, discharges, and streamflow and water-level observations were averaged within grid cells and their exact locations were approximated by the centers of the cells in which they occur.

The upper Charles River Basin models described in this report simulate ground-water flow and water levels. Flows and water levels in surface-water features are simulated to the extent that they represent discharge areas or boundaries for the ground-water system. Thus, flows in simulated streams are base flow (ground-water discharges) and do not include the direct runoff component of streamflow. Moreover, the hydraulics of the surface-water system, such as storage provided by impoundments and wetlands, were not simulated. The effects of these controls on surface-water flow were not included in the simulated stream base flows. Finally, the approach of representing stream stage by a fixed value representing average conditions may lead to some small inaccuracies in flow rates between aquifers and streams, particularly during periods of high flow.

Ground-water flow through till and bedrock are not directly simulated in the models. Ground water in fractured bedrock can have a widely variable area of recharge and natural discharge. Thus, although water withdrawals from bedrock aquifers may be simulated indirectly (as reductions in recharge), the effects of these withdrawals on the stream-aquifer system may not be appropriately addressed with the models. Assumptions about ground-water flow and recharge from till and bedrock to stratified glacial aquifers and streams also may place limitations on interpretation of simulation results. In the absence of any information about flow rates or pathways through bedrock, recharge in till and bedrock upland areas was routed directly to the lateral boundaries of the stratified glacial deposits, and no flow was assumed to occur between stratified glacial deposits and the underlying bedrock. Where flow through bedrock is substantial, the spatial or temporal distribution of simulated flows may not be adequately represented.

Finally, the models were calibrated to average stream base flow and water levels that were estimated from short-term monitoring data and correlations with long-term monitoring records from stations outside the basin. Errors in estimated averages may have contributed to the discrepancies between model-calculated water levels and stream base flow and actual conditions.

EVALUATION OF GROUND-WATER-MANAGEMENT ALTERNATIVES

The ground-water-flow models were developed as a tool to evaluate the response of the upper Charles River stream-aquifer system to changes in water-management practices or hydrologic conditions. Increased water withdrawals, alternative pumping schedules for existing withdrawals, additional surface-water discharges, land disposal of treated wastewater, sewering, or stormwater recharge are examples of water-management practices that may be tested with the models. Altered hydrologic conditions that could be simulated include drought conditions or conditions of altered recharge that may result, for example, from land-use changes.

Two approaches were used to investigate alternative ground-water management practices and altered hydrologic conditions in the upper Charles River Basin. First, the flow models alone were used to determine the effects of increased withdrawals and altered recharge in several hypothetical scenarios. The scenarios were selected to represent possible future changes in water use in the basin, or to investigate the effects of water-management practices that could mitigate potential adverse effects of increased water withdrawals. Second, the flow models were used in conjunction with optimization techniques. This approach was used to quantify possible increases in withdrawals or streamflow that could be obtained by managing water withdrawals. The simulation-optimization approach is advantageous for many applications, because it determines the combination of practices to best (optimally) meet management goals for a particular set of constraints. It was applied in a sub-area of the model to demonstrate its use in addressing complex water-management questions in the valley-fill aquifer hydrogeologic setting. Results of both approaches are described in terms of changes in average monthly stream base flow and pond levels from the base flow and pond levels calculated with the calibrated transient models.

Simulation of Increased Water Withdrawals and Altered Recharge

Hypothetical scenarios of increased water withdrawals and altered recharge in the upper Charles River Basin were identified for testing through consultation with the Technical Advisory Committee, MADEM, MADEP, CRWA, and town officials. The scenarios were (1) increasing water withdrawals from existing sources in the study area to levels currently permitted under the WMA; (2) increasing water withdrawals from increasing and proposed sources in the study area to levels currently permitted or proposed for permitting under the WMA; and (3) altering recharge in the Mine Brook aquifer area in Franklin to represent hypothetical changes in sewering or management of residential rooftop runoff.

Simulation of Increased Withdrawals

Two scenarios of increased withdrawals were tested with the flow models. In the first scenario (S1), water withdrawals at existing sources in the study area (table 2) were increased such that the total pumping rates for each town or non-municipal source were equal to the rates currently permitted under the WMA. In the second scenario (S2), water withdrawals in the study area were increased to the currently permitted rates under the WMA or, in cases where increases have been requested, to proposed levels. Withdrawals are made from both existing and proposed sources in S2. Proposed sources were identified as those in the process of being permitted by MADEP at the time of this study. In both S1 and S2, inflows from precipitation recharge remained the same and corresponded to average conditions during the model calibration period (1989–98).

WMA-permitted withdrawal rates for water systems in the upper Charles River Basin are shown in figure 23 (“WMA-permitted” in this report refers to withdrawals that are both registered and permitted under the WMA). Combined withdrawals from all sources in a water system within a river basin are permitted under the WMA as system-wide average annual rates (Duane LeVangie, Massachusetts Department of Environmental Protection, oral commun., 2000). Withdrawals from individual sources (“Zone II” approved rates) are separately permitted by MADEP (B.R. Bouck, Massachusetts Department of Environmental Protection, oral commun., 2000). For most towns and all non-municipal systems, system-wide average annual rates during 1989–98 were less than WMA-permitted rates (fig. 23). System-wide average annual rates during 1989–98 met or slightly exceeded currently permitted WMA rates in Holliston and Medway; simulated withdrawals for sources in these towns were set to 1989–98 rates in S1 and to proposed rates in S2. Simulated withdrawals for sources in all other water systems were increased so that system-wide average annual rates equalled currently permitted rates for S1 and equalled all currently permitted plus proposed rates for S2. For Norfolk, which is only partly in the study

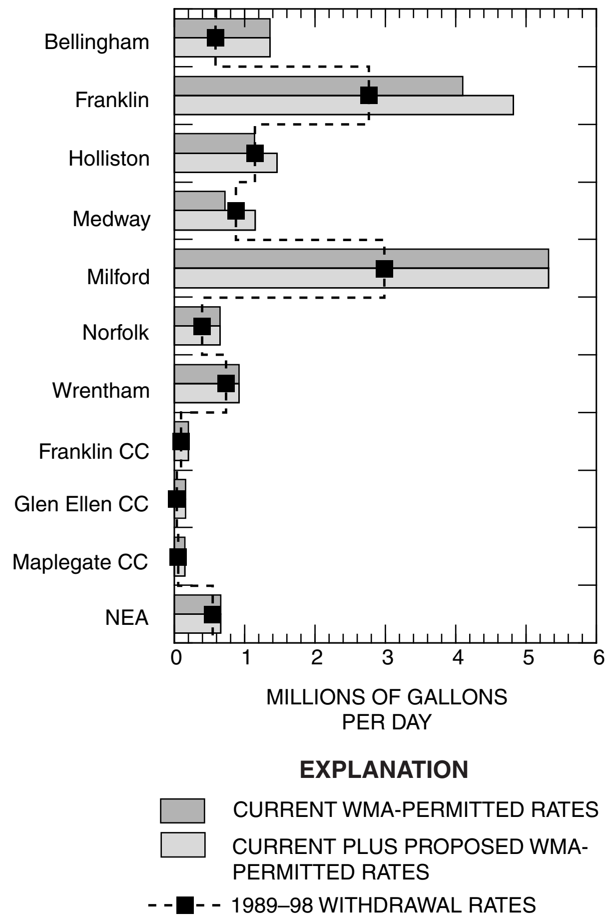


Figure 23. Permitted and registered withdrawal rates under the Massachusetts Water Management (WMA) and 1989–98 average withdrawal rates for municipal and large non-municipal water systems in the upper Charles River Basin, eastern Massachusetts. Rates are system-wide annual averages. (CC, country club; NEA, Northeast Energy Association.)

area but entirely within the Charles River Basin, allowable increases under the WMA for S1 and S2 were prorated. S1 and S2 withdrawals for Norfolk were increased to levels reflecting the contribution of sources in the study area to total Norfolk withdrawals during 1989–98. For Millis, with all its water-supply sources outside of the study area, neither water withdrawals nor return flows were increased.

Simulated system-wide increases in average annual total withdrawal rates were distributed among months of the annual cycle on the basis of the monthly distribution of 1989–98 withdrawals for each system. For S1, increases were distributed among individual sources in each system in proportion to the historical (1989–98) contribution of each source to monthly total withdrawals in the system. Because withdrawals from a new source in Bellingham (well No. 12) began in mid-1997, increased withdrawals were distributed among sources in that town on the basis of 1998 data. For S2, increased withdrawals were preferentially assigned to new sources. Average monthly withdrawals at municipal sources were not allowed to exceed their Zone II approved rates in either scenario. If distribution of system-wide increases according to the procedures described would have resulted in its Zone II permitted rate being exceeded at a source during any month, withdrawals at the source were set to the Zone II rate during the month and the excess withdrawal was distributed among the remaining sources. For Milford, the currently WMA-permitted withdrawals could not be met with existing sources without increasing surface-water withdrawals above historical levels or exceeding Zone II permitted rates at ground-water sources. Consequently, withdrawals for Milford for S1 and S2 were set to Zone II permitted rates in all months for ground-water sources and to 1989–98 withdrawals for surface-water withdrawals. The monthly distribution of simulated municipal withdrawals and discharges, totaled by town or facility, for S1 and S2 are shown in figure 24; average withdrawal rates by source are listed in table 10. Overall, total simulated withdrawals increased in S1 by 3.2 Mgal/d (4.9 ft^3/s) to 7.7 Mgal/d (11.9 ft^3/s) in the west model and by 0.92 Mgal/d (1.42 ft^3/s) to 5.1 Mgal/d in the east model. In S2, total simulated withdrawals increased by 2.9 Mgal/d (4.5 ft^3/s) to 7.4 Mgal/d in the west model and 2.4 Mgal/d (3.7 ft^3/s) to 6.6 Mgal/d in the east model. Withdrawals in the west model were slightly less in S2 than in S1 because, for some towns, pumping at proposed wells located in the east model area in S2 resulted in slightly lower pumping rates at existing wells in the west model area. Thus, total withdrawals simulated in S1 and S2 increased by 42 and 54 percent, respectively, over 1989–98 withdrawals.

Return flow from increased withdrawals for municipal water systems was routed to existing wastewater-treatment plants or directly to the aquifer as septic-system return flow, according to 1989–98 wastewater-management practices in towns in the study area. Simulated additional return flow equaled the additional average monthly withdrawals per town, reduced by a factor representing consumptive use. Consumptive use, which averages about 10 percent annually, was varied from 0 percent in winter months to about 30 percent in June and July. For towns with sewers, additional return flow was added to simulated monthly discharges from the MTF (Milford) and the CRCPD treatment facility (Bellingham, Franklin, and Medway). For towns without sewers (Holliston, Norfolk, and Wrentham), additional septic-system return flow was added, as areal recharge. Recharge from additional septic-system return flow was applied in the same month as the additional withdrawal. For each town, the additional recharge was distributed between upland and active model areas in proportion to land areas. Additional return flow was assumed to be distributed throughout the total town areas for Norfolk and Wrentham. Additional monthly return-flow rates averaged 0.49 in/yr for Holliston (S2), 0.47 in/yr (S1) and 0.86 in/yr (S2) for Norfolk, and 0.14 in/yr (S1 and S2) for Wrentham. Return flow was not simulated for non-municipal sources. As in the calibrated models of current conditions, withdrawals for golf courses were reduced by 50 percent to account for return flow from irrigation use, and withdrawals for the Northeast Energy Association sources were considered to be 100 percent consumptive.

Model-calculated stream base flow under scenarios of increased withdrawals (S1 and S2) is compared with model-calculated base flow for average 1989–98 conditions in table 14 and figures 25–27. Comparisons are shown of average monthly flows for March and September. The magnitude of reductions in base flow relative to 1989–98 flows was variable throughout the annual cycle, and was strongly affected by the timing of increased withdrawals at upstream sources.

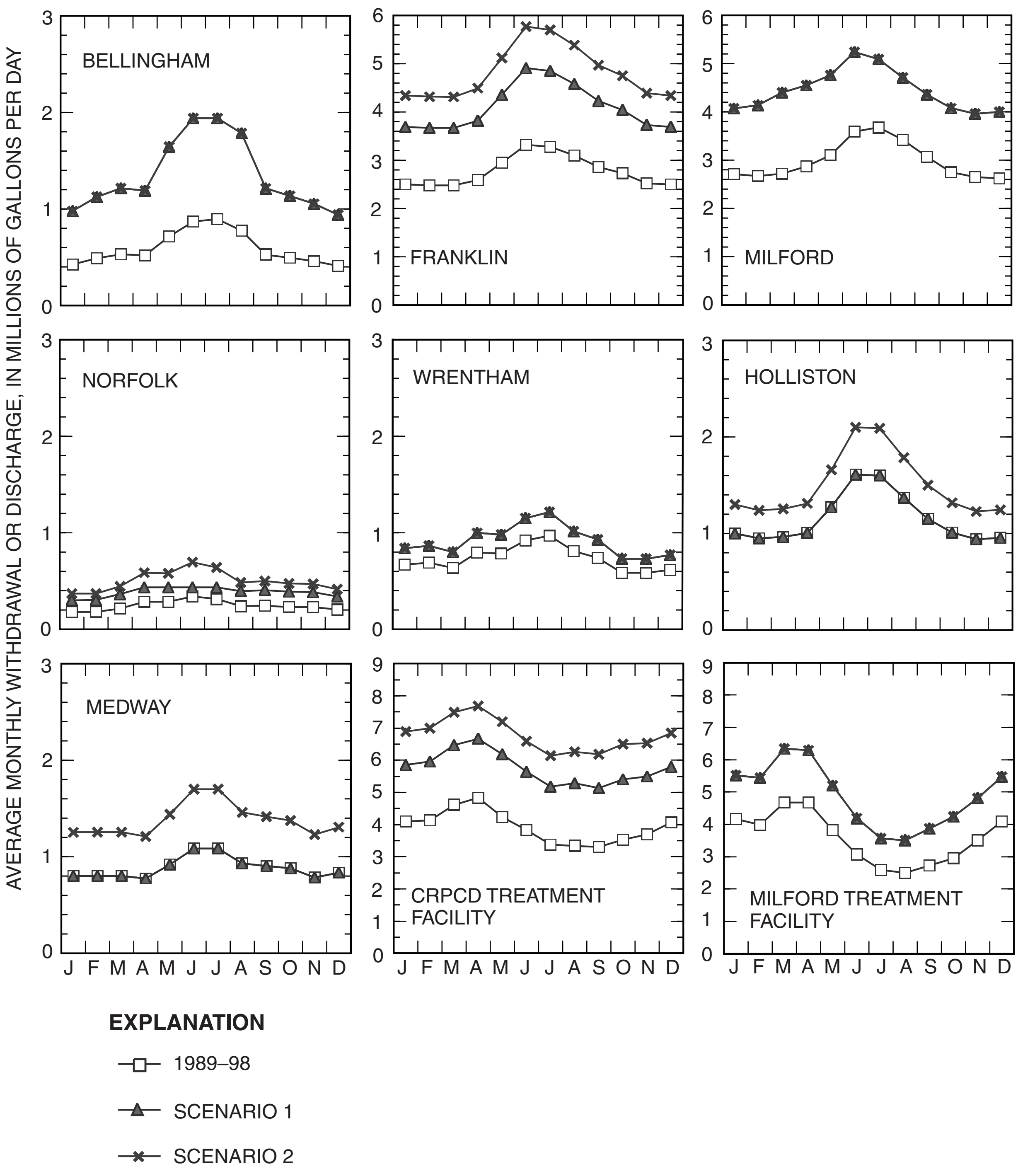


Figure 24. Average monthly withdrawals and discharges by town or treatment facility for 1989–98 and two hypothetical scenarios of increased withdrawals in the upper Charles River Basin, eastern Massachusetts. (CRPCD, Charles River Pollution Control District.)

Table 14. Changes in model-calculated average annual, March, and September stream base flow from 1989-98 base flow for hypothetical scenarios of increased withdrawals and altered recharge in the upper Charles River Basin, eastern Massachusetts

[**Streamflow site:** See figure 7 for location of measurement sites; locations of sites along tributaries are immediately upstream of confluences with the Charles River unless otherwise indicated. **Model-calculated 1989–98 flow**: Average of model-calculated monthly flows from transient simulations. **Change in model-calculated base flow**: Number in parentheses is percent of model-calculated 1989–98 average annual or monthly flow; ft^3/s, cubic feet per second]

Streamflow site	Model-calculated 1989–98 flow (ft^3/s)	Scenario	Change in model-calculated stream base flow		
			Average annual flow (ft^3/s)	Average March flow (ft^3/s)	Average September flow (ft^3/s)
West Model					
Charles River					
Charles River above the Milford Treatment Facility near the Milford/Hopedale town line	10.5	1	-2.1 (20)	-2.3 (9)	-0.7 (69)
		2	-2.1 (20)	-2.3 (9)	-0.7 (69)
Charles River at 01103140	18.6	1	-0.16 (0.9)	+0.27 (0.7)	+0.97 (18)
		2	-0.16 (0.2)	+0.27 (0.7)	+0.97 (18)
Charles River at 011032056	27.9	1	-0.55 (2)	-0.11 (0.2)	+0.60 (9)
		2	-0.55 (2)	-0.11 (0.2)	+0.59 (9)
Charles River at 01103260	87.5	1	-2.6 (3)	-2.4 (1)	-0.19 (5)
		2	-2.7 (3)	-2.7 (2)	-0.19 (5)
Tributaries to Charles River					
Beaver Brook	4.7	1	-0.21 (5)	-0.21 (3)	-0.19 (10)
		2	-0.21 (5)	-0.21 (3)	-0.19 (10)
Stall Brook at 01103120	4.1	1	-0.36 (9)	-0.31 (3)	-0.34 (68)
		2	-0.36 (9)	-0.31 (3)	-0.34 (68)
Hopping Brook	15.8	1	+0.01 (0)	0.00 (0)	+0.02 (0.8)
		2	+0.16 (1)	+0.16 (0.5)	+0.11 (4)
Miscoe Brook at South Street above Mine Brook	1.2	1	0.00 (0)	0.00 (0)	0.00 (0)
		2	-0.45 (37)	-0.73 (34)	-0.17 (36)
		3a	+0.11 (9)	+0.15 (7)	+0.08 (18)
		3b	+0.02 (1)	+0.02 (1)	+0.01 (3)
		3c	+0.03 (2)	+0.04 (2)	+0.02 (5)
Mine Brook at 01103225	2.7	1	-0.01 (0.3)	-0.02 (0.4)	0.00 (0.2)
		2	-0.66 (25)	-1.02 (23)	-0.34 (27)
		3a	+0.23 (9)	+0.30 (7)	+0.19 (15)
		3b	+0.04 (1)	+0.05 (1)	+0.03 (3)
		3c	+0.06 (2)	+0.08 (2)	+0.05 (4)
Mine Brook at 01103235	12.8	1	-0.74 (6)	-0.92 (4)	-0.43 (7)
		2	-1.15 (9)	-1.6 (7)	-0.64 (11)
		3a	+0.99 (8)	+1.2 (6)	+0.58 (10)
		3b	+0.16 (1)	+0.20 (0.9)	+0.10 (2)
		3c	+0.27 (2)	+0.34 (2)	+0.17 (3)
Mine Brook at 01103240	19.1	1	-1.1 (6)	-1.3 (4)	-0.81 (10)
		2	-1.4 (7)	-1.8 (5)	-0.92 (11)
		3a	+1.4 (7)	+1.7 (5)	+0.96 (12)
		3b	+0.23 (1)	+0.28 (0.8)	+0.16 (2)
		3c	+0.39 (2)	+0.47 (1)	+0.27 (3)

Table 14. Changes in model-calculated average annual, March, and September stream base flow from 1989-98 base flow for hypothetical scenarios of increased withdrawals and altered recharge in the upper Charles River Basin, eastern Massachusetts—*Continued*

Streamflow site	Model-calculated 1989–98 flow (ft^3/s)	Scenario	Change in model-calculated stream base flow		
			Average annual flow (ft^3/s)	Average March flow (ft^3/s)	Average September flow (ft^3/s)
East Model					
Charles River					
Charles River at 01103280	96.3	1	-2.5 (3)	-2.3 (1)	-1.0 (4)
		2	-2.6 (3)	-2.5 (1)	-0.96 (4)
Charles River above Populatic Pond	97.2	1	-1.9 (2)	-1.7 (0.9)	-0.32 (1)
		2	-2.0 (2)	-1.9 (1)	-0.35 (1)
Charles River above Mill River	104	1	+0.17 (1)	+0.03 (.04)	+1.8 (6)
		2	+0.53 (2)	+0.60 (0.3)	+2.2 (8)
Charles River at 01103305	127	1	-0.97 (0.2)	-1.38 (0.6)	+0.97 (3)
		2	-0.46 (0.5)	-0.50 (0.2)	+1.42 (4)
Tributaries to Charles River					
Mill River between 01103292 and 01103295	11.6	1	-0.60 (5)	-0.87 (5)	-0.35 (6)
		2	-0.60 (5)	-0.87 (5)	-0.35 (6)
Mill River at 01103300	19.4	1	-0.98 (5)	-1.3 (4)	-0.69 (12)
		2	-1.0 (5)	-1.2 (3)	-0.79 (14)
Mill River above Charles River	22.2	1	-1.1 (5)	-1.4 (3)	-0.77 (12)
		2	-0.99 (5)	-1.1 (3)	-0.79 (13)
Cress Brook above Mill River	1.5	1	-0.07 (4)	-0.07 (2)	[1]0
		2	+0.07 (4)	+0.07 (2)	[1]0
Miller Brook above Mill River	3.3	1	-0.23 (7)	-0.22 (3)	-0.22 (85)
		2	-0.37 (11)	-0.33 (4)	-0.26 (100)
Bogastow Brook and Tributaries					
Dopping Brook at 01103386	1.4	1	0.00 (0)	0.00 (0)	0.00 (0.1)
		2	-0.49 (36)	-0.93 (27)	-0.04 (26)
Great Black Swamp tributaries above Bogastow Brook	7.7	1	+0.03 (0.4)	+0.04 (0.3)	+0.01 (0.2)
		2	-0.59 (8)	-0.65 (5)	-0.4 (10)
Bogastow Brook above Great Black Swamp tributaries	21.4	1	-0.10 (0.4)	0.00 (0)	-0.18 (3)
		2	-0.43 (2)	-0.8 (2)	-0.06 (0.9)
Bogastow Brook at 01103393	29.1	1	-0.07 (0.2)	+0.03 (0.1)	-0.18 (2)
		2	-1.0 (4)	-1.4 (3)	-0.5 (4)

[1]Not determined because simulated stream is dry in September under 1989–98 conditions.

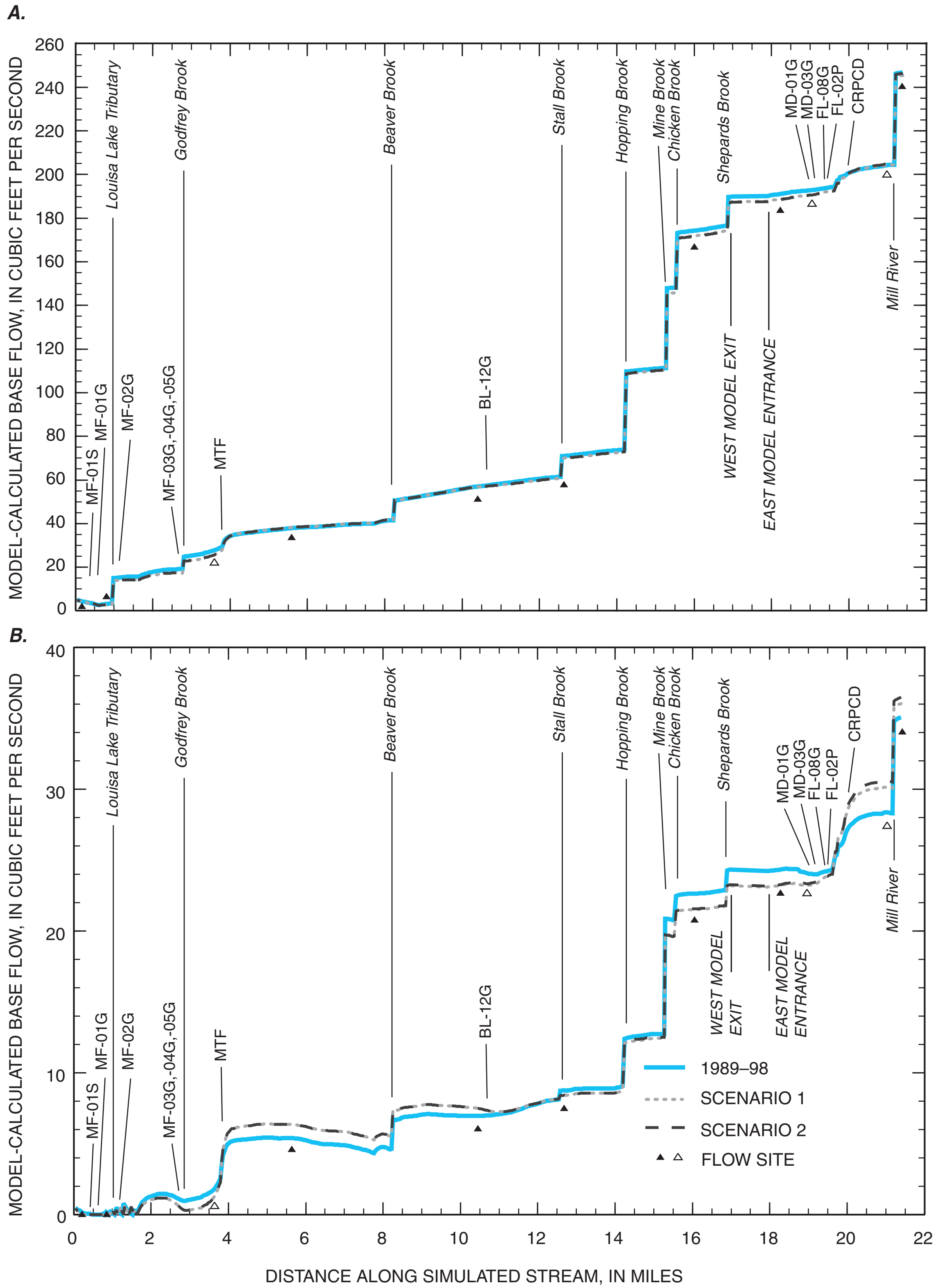


Figure 25. Model-calculated monthly average base flow for March and September along the Charles River for 1989–98 average conditions and two scenarios of increased withdrawals in the upper Charles River Basin, eastern Massachusetts: (*A*) March, and (*B*) September. Flow sites shown are streamflow-measurement sites (solid symbols; see figure 7) or streamflow-observation sites described in table 14 (open symbols).

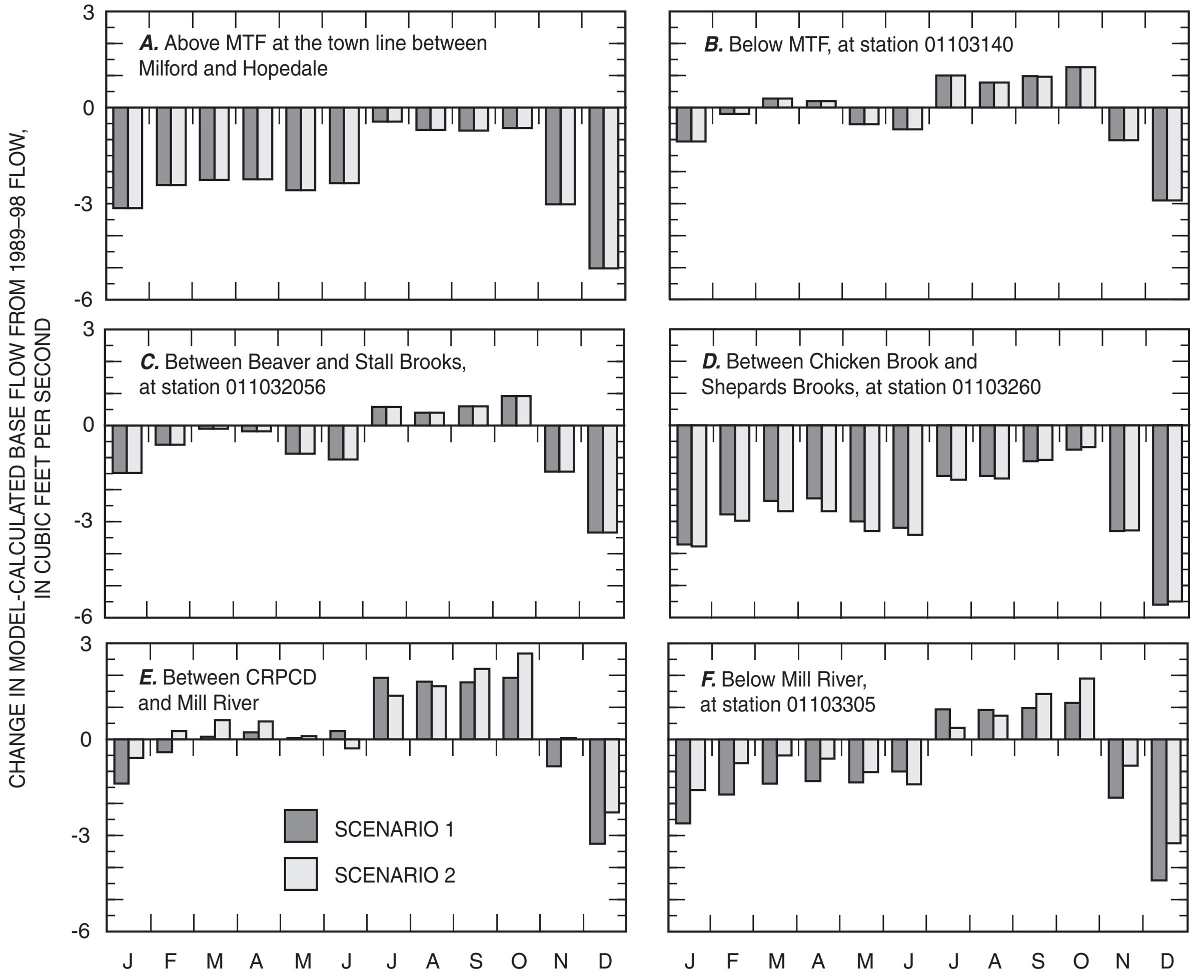


Figure 26. Change in model-calculated base flow from 1989–98 flows at selected sites along the Charles River for two scenarios of increased withdrawals in the upper Charles River Basin, eastern Massachusetts: (*A*) Charles River above the Milford Treatment Facility (MTF) at the town line between Milford and Hopedale, (*B*) Charles River below the MTF at station 01103140, (*C*) Charles River between Beaver and Stall Brooks at station 011032056, (*D*) Charles River between Chicken and Shepards Brooks at station 01103260, (*E*) Charles River between the Charles River Pollution Control District Treatment Facility (CRPCD) and Mill River, and (*F*) Charles River below Mill River at station 01103305.

However, March and September typically were months of maximum and minimum monthly average flows, respectively, at most measurement sites. Thus, changes in base flow typically were proportionally largest or smallest relative to 1989–98 flows in these months.

Along the Charles River, stream base-flow reductions from increased withdrawals were nearly balanced by flow augmentations from increased waste-water discharges in both scenarios overall. Results of S1 and S2 simulations were very similar (fig. 25). Along specific reaches of the river or during various months of the annual cycle, however, reductions or augmentations dominated (figs. 23 and 24). Generally, model-calculated base flows were reduced relative to model-calculated 1989–98 flows upstream of waste-water treatment facilities in most months and were augmented relative to 1989–98 flows in low-flow months (July to October) downstream of treatment facilities (fig. 26). This results because withdrawals at upstream sources affect stream baseflow over an extended period of time, rather than instantaneously. Relative to total average monthly flow for 1989–98, reductions in model-calculated base flows were greatest in September (table 14) or October. Base-flow reductions were associated with increased withdrawals (a) from sources along the Charles River, such as wells in Milford and Bellingham, and (b) from decreased inflows from tributaries, such as from Mine Brook.

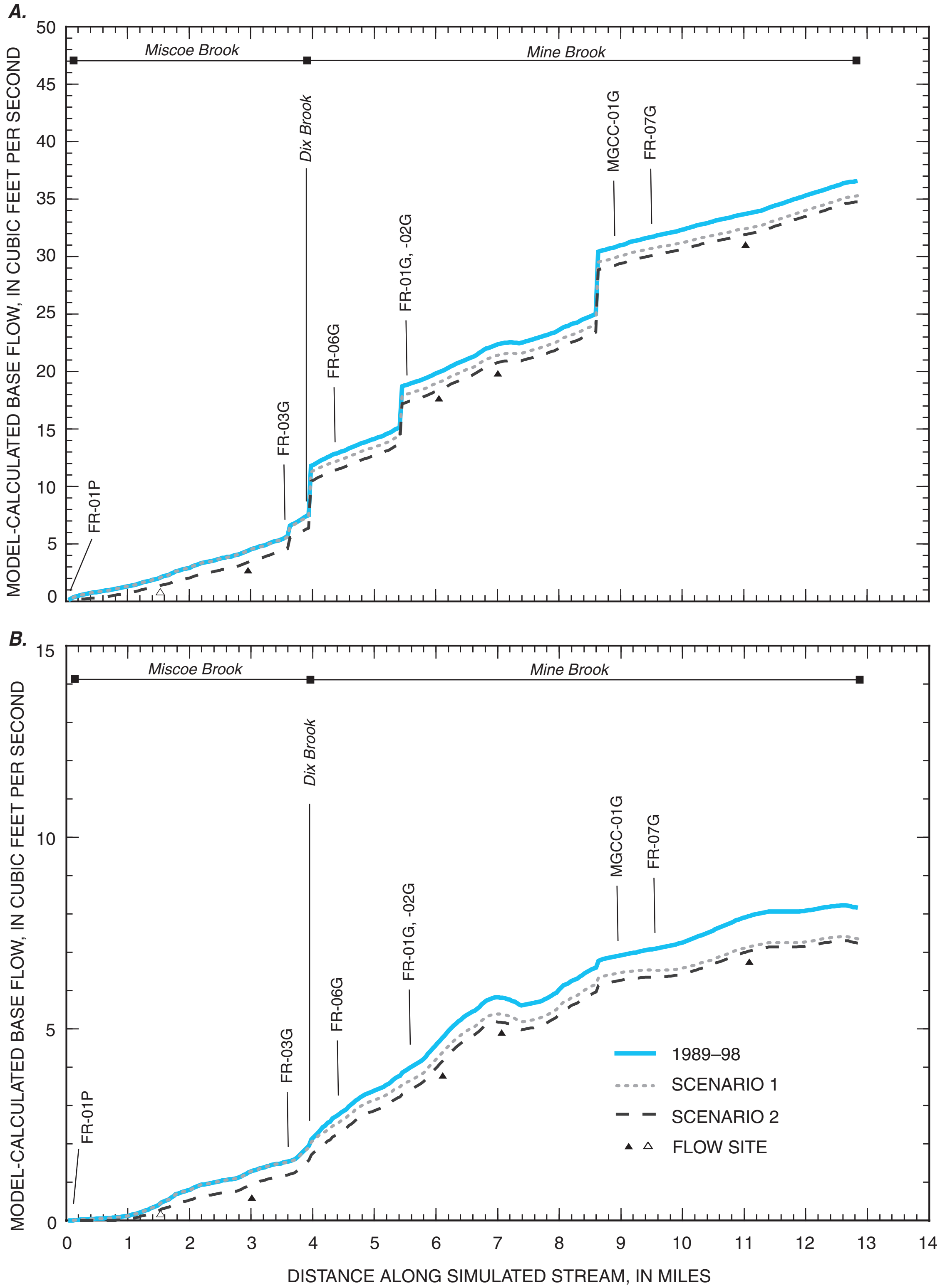


Figure 27. Model-calculated monthly average base flow for March and September along Mine Brook for 1989–98 average conditions and two scenarios of increased withdrawals in the upper Charles River Basin, eastern Massachusetts: (*A*) March, and (*B*) September. Flow sites shown are streamflow-measurement sites (solid symbols; see fig. 7) or streamflow-observation sites described in table 14 (open symbols).

Model-calculated September base flow was reduced relative to 1989–98 flows along three river reaches in S1 and S2: by about 70 percent (0.7 ft^3/s) downstream of Milford sources, by about 2 to 4 percent (0.2 to 0.3 ft^3/s) between Stall and Mine Brooks, and by about 5 percent (1.1 ft^3/s) downstream of Mine Brook (fig. 25*B* and table 14). In March, reductions in model-calculated base flow along these three reaches were larger in magnitude than in September but were smaller fractions (9, 1.5, and 1.5 percent, respectively) of total flow (fig. 25*A* and table 14). Model-calculated augmentations from wastewater discharges exceeded upstream depletions in September; as a result, flow in September increased downstream of treatment facilities. In March, model-calculated augmentations (less than 1 percent of total flow) were less than upstream reductions. The increased discharges from the treatment facilities resulted in wastewater constituting greater proportions of stream base flow during low-flow months. Treated wastewater constituted about 90 percent of simulated September flows immediately downstream from the MTF (S1 and S2) and about 27 (S1) and 32 (S2) percent of simulated September flows downstream from the CRPCD treatment facility; corresponding values for simulated 1989–98 conditions were 80 percent for the MTF and 18 percent for the CRPCD treatment facility.

Along Mine Brook, simulated streamflows were reduced relative to 1989–98 flows from increased withdrawals at several Franklin municipal-supply wells and a golf course located along the stream and its tributaries (fig. 27). Return flow was routed out of the subbasin to the CRPCD treatment facility from these withdrawals. Thus, reductions in model-calculated stream base flow in Mine Brook were comparable to upstream increases in withdrawals from the aquifers (0.83 and 1.02 Mgal/day, or 1.29 and 1.58 ft^3/s, in S1 and S2, respectively) and averaging 1.1 ft^3/s (S1) and 1.4 ft^3/s (S2) over the annual cycle (table 14). Base-flow reductions along specific reaches of Mine Brook varied depending on the proximity of the reach to individual supply wells and varied as proportions of total flow. Model-calculated 1989–98 September base flow in the aquifer along Mine Brook was reduced in S2 by more than 50 percent in its headwaters (Miscoe Brook), and in S1 and S2 by about 10 percent along downstream reaches (fig. 27). Results of S1 and S2 are dissimilar in upstream reaches of Mine Brook because of the increased withdrawals at the single proposed well at the headwaters of Miscoe Brook, which is included in S2 but not in S1.

Similar results to those described for Mine Brook were obtained for other tributaries in areas where return flow was routed to wastewater-treatment facilities. Reductions in model-calculated stream base flow along Stall Brook and Beaver Brook in Bellingham and along a Great Black Swamp tributary in Medway were about equal to upstream withdrawals from the aquifers (tables 9 and 14). Along Stall Brook, reductions in simulated stream base flow were substantial and represented more than 60 percent of model-calculated 1989–98 flows during low-flow months for S1 and S2 (table 14).

Along tributaries in areas where return flow was wholly or partly returned to the aquifer as added recharge, reductions in model-calculated base flow generally were less than increases in upstream withdrawals from the aquifers. Reductions in model-calculated base flow in tributaries were not eliminated even in areas where return flow was routed to the aquifer, primarily because of differences in the contributing areas to the streams and areas where return-flow recharge was applied (that is, intrabasin and interbasin transfers). For example, in S2, average annual withdrawals of 0.54 ft^3/s from a new source in the aquifer along Dopping Brook (0.35 Mgal/d at HL-01P; table 10) were distributed throughout the town of Holliston. Thus, simulated average annual base flow in Dopping Brook was reduced by 0.49 ft^3/s (measurement site 01103386; this reduction includes effects of an increased average annual withdrawal of 0.06 ft^3/s at an upstream golf course), whereas simulated base flow was slightly increased in Hopping Brook (average annual increase of 0.16 ft^3/s at the confluence with the Charles River; table 14) and in Chicken Brook, where withdrawals were unchanged but returnflow increased.

Along the Mill River, reductions in model-calculated stream base flow relative to 1989–98 flows averaged 1.1 ft^3/s in S1 and 0.99 ft^3/s in S2 above the Charles River (table 14). Results of S1 and S2 are similar, because increased withdrawals at the two proposed wells included in S2 largely are balanced by smaller additional withdrawals at existing wells in S2 than in S1 (fig. 28). Return flow from sources in the Mill River aquifer area either was routed to the CRPCD treatment facility (Franklin withdrawals) or to the aquifer as added return flow (Norfolk and Wrentham withdrawals), and average reductions in base flow over the annual cycle are about 75 percent of average increased withdrawals at the confluence with the Charles River.

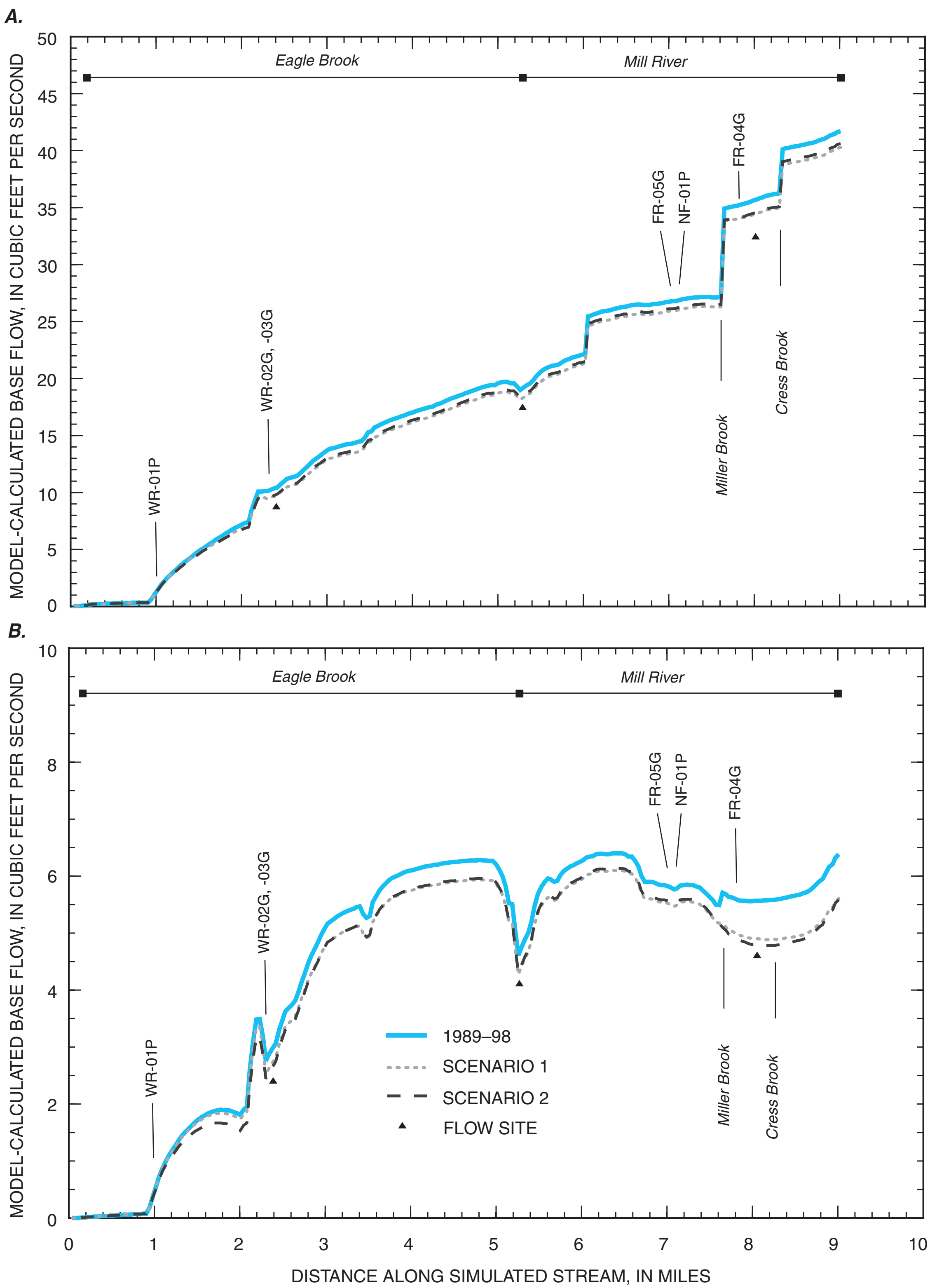


Figure 28. Model-calculated monthly average base flow for March and September along the Mill River for 1989–98 average conditions and two scenarios of increased withdrawals in the upper Charles River Basin, eastern Massachusetts: (*A*) March, and (*B*) September. Flow sites shown are streamflow-measurement sites (solid symbols; see fig. 7).

Reductions in model-calculated base flow in the September low-flow period for S1 and S2 are about 6 percent of model-calculated 1989–98 flows downstream of Wrentham withdrawals and about 12 to 14 percent of 1989–98 flows downstream of Norfolk and Franklin withdrawals (table 14 and fig. 28*B*). It should be noted that simulated base flow in the Mill River may be affected by aquifer geometry. Constrictions in the areal extent of the aquifer (fig. 13*B*) result in enhanced ground-water discharge to streams, some of which is returned to the aquifer as stream leakage downstream of the constrictions. This effect is indicated by the rapid increases in model-calculated September base flow from about stream miles 2.5 to 4.0, upstream of an aquifer constriction, and abrupt decrease in model-calculated base flow where the aquifer broadens near flow-measurement site 01103295 (fig. 28*B*). The effect may be larger in model-calculated base flow than in actual flows because the constriction is more abruptly defined, relative to actual aquifer boundaries, in the active model area.

Increased withdrawals in the Mill River aquifer area also resulted in changes in model-calculated base flow in small tributaries in the Populatic Pond area (fig. 3 and table 14) and in water levels in Kingsbury Pond. Model-calculated September base flow in Miller Brook, near Franklin wells No. 4 and No. 5 (FR-04G and FR-05G) and the Norfolk proposed well (NF-01P), is reduced by 85 percent relative to 1989–98 flows in S1 and is entirely depleted in S2. In Cress Brook, model-calculated average annual flows are reduced by about 4 percent in S1 when withdrawals at the nearby existing Norfolk well are increased. In S2, when increased withdrawals for Norfolk are obtained from the proposed well along the Mill River, flows in Cress Brook are augmented by 4 percent because of increased septic-system return flow in this area.

Water levels in Kingsbury Pond, which is near several existing wells and between two proposed wells in the Populatic Pond area, are reduced by about 0.8 ft on average over the annual cycle in both S1 and S2 (fig. 29). Kingsbury Pond is a kettle pond without surface water inflow or outflow. Water levels in the pond are representative of water levels in the aquifer and indicate a general lowering of the water table in response to increased withdrawals in this area. The response of Kingsbury Pond was similar in S1 and S2 even though increases in withdrawals from wells in the aquifer near the pond (northern part of Mill River aquifer and Charles River aquifer, table 10) in S1 were

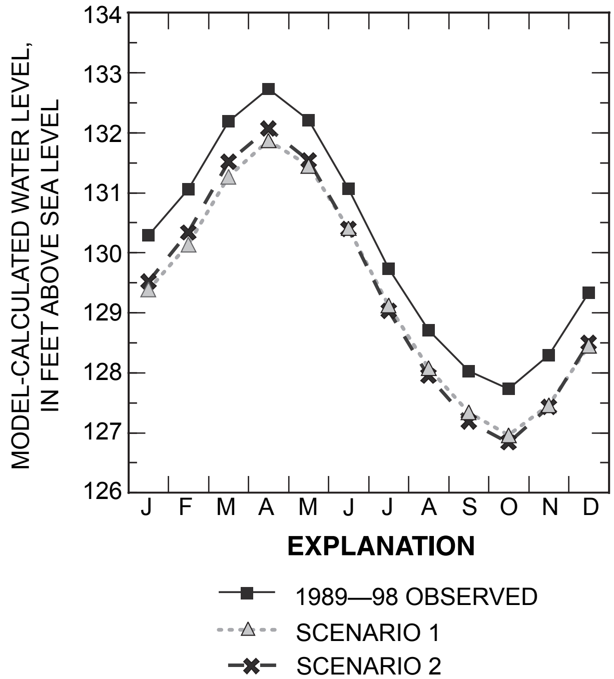


Figure 29. Model-calculated average monthly water levels in Kingsbury Pond for 1989–98 conditions and in two hypothetical scenarios of increased water withdrawals in the upper Charles River Basin, eastern Massachusetts.

nearly twice the increases in withdrawals from these wells in S2. This resulted because withdrawals from wells closest to the pond (particularly FR-04G and FR-05G) were increased less in S2 than in S1.

In summary, the increased withdrawals in the upper Charles River Basin in scenarios 1 and 2 resulted in localized reductions in model-calculated stream base flow relative to model-calculated 1989–98 flows. Reductions ranged from less than 5 percent to more than 60 percent of simulated 1989–98 base flow during low-flow periods in the Charles River and its major tributaries. The largest reductions occurred in headwater areas or in small subbasins where withdrawals were large relative to base flow and where the withdrawn water was transferred out of the subbasin. Additional recharge from septic-system return flow from increased withdrawals, as simulated in S1 and S2, augmented model-calculated base flow in a few tributaries where withdrawals were not increased, and to some extent decreased the effects on model-calculated base flow in areas of increased withdrawals. However, the effects were not large. Overall, increased water withdrawals were transferred among subbasins but were returned nearly entirely to the stream and aquifer system within the model area. Thus, overall changes in simulated outflows from the study area were small, about 1 to 1.5 ft^3/s on average over the annual cycle in S1 and S2, and resulted primarily (80 percent) from consumptive use. In S2, water (about 0.5 ft^3/s on average) was transferred from the Bogastow Brook drainage area to the Charles River drainage area through withdrawals at a proposed well in Medway and routing of the water to the CRPCD treatment facility. Finally, increased withdrawals and transfer to treatment facilities potentially increased the effects of these discharges on water quality in the river by increasing the wastewater component of streamflow during low-flow periods.

Simulation of Altered Recharge

A third set of scenarios, in which recharge to the aquifer is increased through water-management alternatives (S3), was simulated for the aquifer along Mine Brook in Franklin. This area was selected to represent a typical area of the model domain where streamflow depletion currently occurs and is likely to increase with additional withdrawals. In the Mine Brook Subbasin, water is transferred out of the subbasin by sewering and discharge of treated wastewater to the Charles River at the CRPCD facility in Medway. First, recharge was increased by an amount equal to the out-of-subbasin transfer currently resulting from sewering (S3a). About 50 percent of the Mine Brook Subbasin is sewered (fig. 6). Although it is unlikely that existing sewers would be removed, this scenario serves to quantify the effects of sewering on streamflow depletion in a typical aquifer area. Second, recharge was increased to simulate artificial recharge of residential rooftop runoff (S3b and c). Recharge of rooftop runoff is a water-management practice that often is proposed in densely populated areas to reduce stormwater flows. This scenario illustrates the relative magnitude of potential effects of this practice on water quantity in a less densely populated, suburban area. Recharge volumes associated with the management practices simulated in S3a and S3b were estimated by CRWA, as described below (N.B. Pickering, Charles River Watershed Association, written commun., 2002). In all three scenarios, water withdrawals, inflows from precipitation recharge, and 1989–98 septic-system return flow remained the same as those used to simulate 1989–89 conditions in the calibrated models.

Added recharge equivalent to water currently transferred out of the Mine Brook Subbasin through sewering was estimated at 1.47 in/yr. This value was based on an analysis of land-use data (1990), the areal extent of sewers, and population densities per land use category similar to that described previously for 1989–98 septic-system return flow (N.B. Pickering, Charles River Watershed Association, written commun., 2002; table 15). Population values determined from 1990 data were adjusted using 2000 data to represent average 1989–98 populations. Sewered land uses were grouped into categories of low-density residential (lots greater than one-half acre in area), medium-density residential (lot sizes from one-quarter to one-half acre), multi-family residential land use, and other developed land uses (table 15). Wastewater volumes of 140 gal/day per person were estimated from 1989–98 flows to CRPCD from Franklin (N.B. Pickering, Charles River Watershed Association, written commun., 2002); this value includes ground-water infiltration into sewer lines. The additional recharge was added uniformly in all months of the annual cycle.

Table 15. Land-use characteristics, population density, and rooftop-drainage characteristics used to calculate rates of additional recharge in the Mine Brook aquifer area, upper Charles River Basin, eastern Massachusetts

[Data from N.B. Pickering, Charles River Watershed Association, written commun., 2002; --, not applicable]

Land use category	Area (acres)			Population density (people per acre)	Number of rooftops	Effectiveness of rooftop impervious surface (percent of total rooftop area)	
	Total	Sewered	Unsewered			Low	High
Low-density residential	760	240	520	2.2	832	7.5	12.5
Medium-density residential	1,023	692	331	4.4	2,238	34	56
Multi-family residential	134	87	47	15.2	340	45	75
Other developed[1]	1,186	260	926	2.2	--	--	--
Undeveloped	5,038	0	5,038	0	--	--	--

[1]Includes commercial, industrial, transportation, urban open and public, and spectator-recreation land uses.

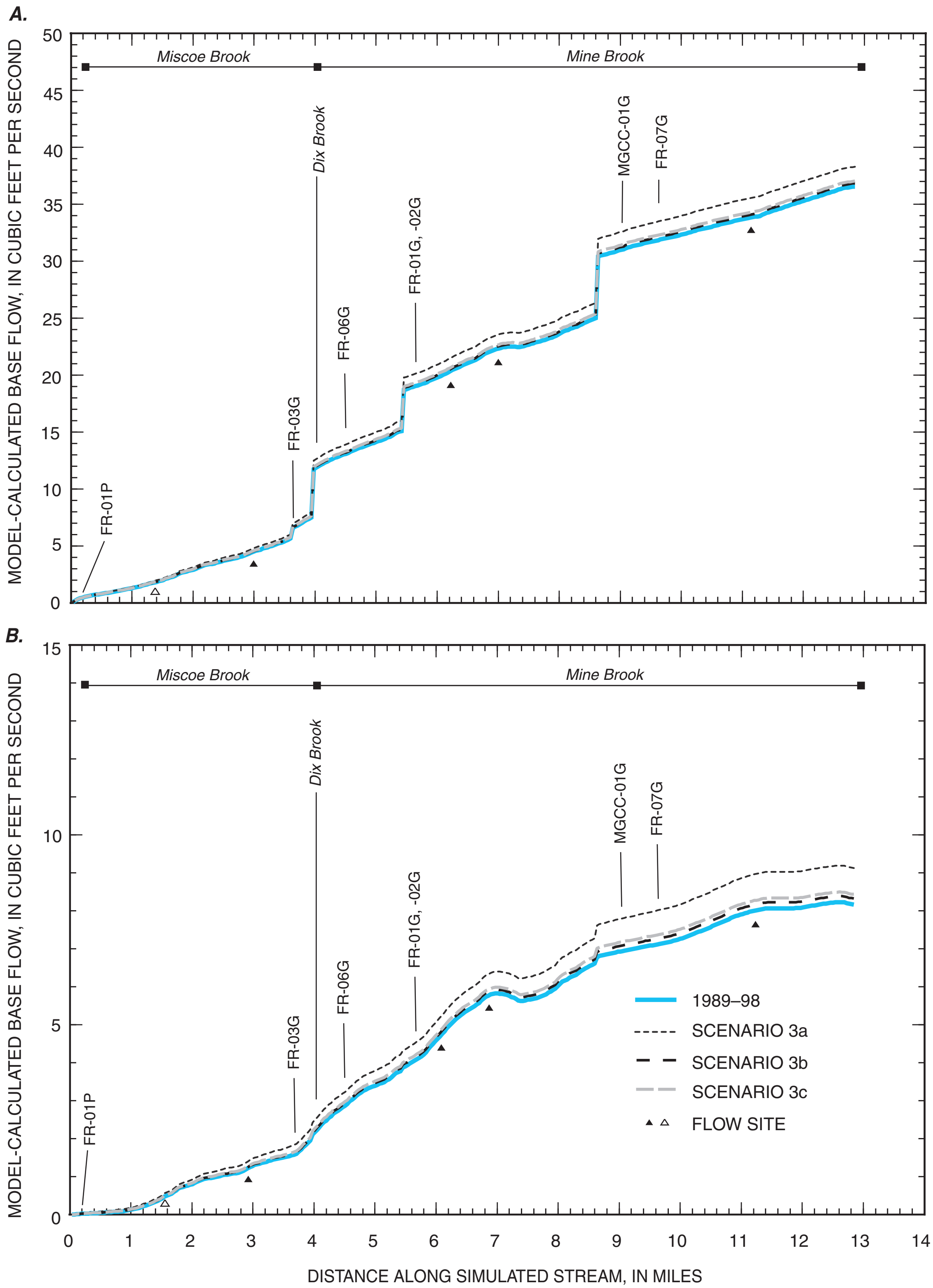


Figure 30. Model-calculated monthly average base flow for March and September along Mine Brook for 1989–98 average conditions and three scenarios of altered recharge in the upper Charles River Basin, eastern Massachusetts: (*A*) March, and (*B*) September. Flow sites shown are streamflow-measurement sites (solid symbols; see fig. 7) or streamflow-observation sites described in table 14 (open symbols).

Added recharge resulting from artificial recharge of rooftop runoff was estimated as low and high values of 0.24 in/yr (3b) to 0.40 (3c) in/yr. These values were based on several assumptions (N.B. Pickering, Charles River Watershed Association, written commun., 2002). First, the number of residential rooftops in the Mine Brook aquifer area was estimated from land-use data, population densities per land-use category, and an average number of people per residential household (table 15). The average number of people per household, 2.9, was determined from population data for Franklin and Millis. Multifamily residential structures were assumed to contain three households. Second, an average rooftop area of 2,000 ft^2 was assumed, based on an analysis of orthophotographs (MassGIS, 1997; N.B. Pickering, Charles River Watershed Association, written commun., August 2002). Third, the fraction of water currently drained from residential rooftops such that it becomes surface-water runoff (effectiveness of rooftop impervious area) was assumed to range from 7.5 percent, a low estimate for low-density residential land use, to 75 percent, a high estimate for multifamily residential land use. Finally, it was assumed that all of the water currently drained from residential rooftops could be captured and recharged to the aquifer. The added recharge was added uniformly in all months of the annual cycle.

Model-calculated stream base flow in Mine Brook increased under the three scenarios of increased recharge (S3a, b, and c; table 14). Average annual increases in simulated base flow at measurement site 01103240, the downstream end of the subbasin area where recharge was added, were 1.4 ft^3/s for S3a (effects of sewers), 0.23 ft^3/s for S3b (rooftop runoff recharge—low estimate), and 0.39 ft^3/s for S3c (rooftop runoff recharge—high estimate). The increases were nearly equivalent to the volumes added in the drainage basin for the three scenarios: 1.54 ft^3/s for S3a, 0.25 ft^3/s for S3b, and 0.42 ft^3/s for S3c, as expected under conditions of dynamic equilibrium. The streamflow augmentations increased in absolute magnitude downstream as drainage area and total flow increased (fig. 30). Model-calculated streamflow augmentations were about 12 percent of model-calculated 1989–98 September base flow for the scenario of sewer effects and about 2 to 3 percent of September base flow for the scenarios of rooftop-runoff recharge (table 14). The effect of added recharge on model-calculated base flow was linear, but varied throughout the annual cycle (fig. 31). Recharge added in low-flow months went partly to replenish aquifer storage, such that increases in model-calculated base flow in May through October were less than (80 percent of) the added recharge rate. In high-flow months, from November through April, added water was released from storage and streamflow increased by amounts greater than (120 percent of) the added recharge.

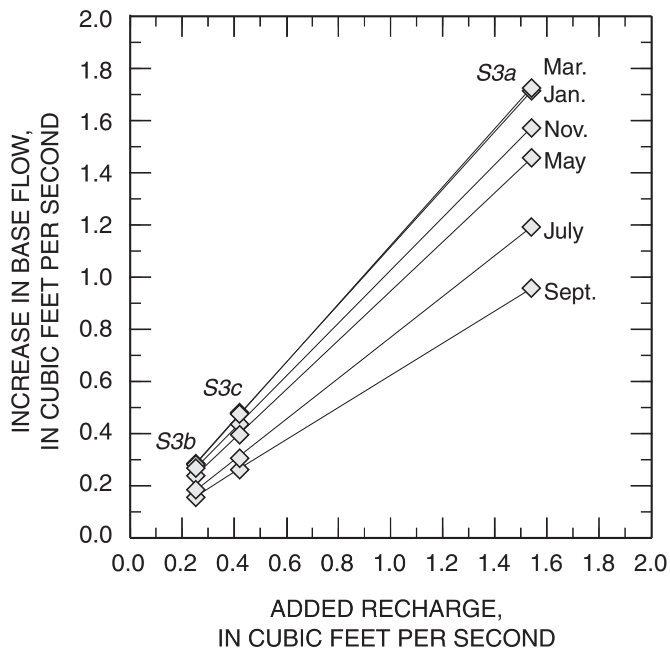


Figure 31. Relation of increases in model-calculated monthly average base flow in Mine Brook (measurement site 01103240) and added recharge in the Mine Brook Subbasin, upper Charles River Basin, eastern Massachusetts. Data points are for three model runs in which areal recharge is increased by 0.25, (S3b) 0.42 (S3c), and 1.54 (S3a) cubic feet per second (0.24, 0.40, and 1.47 inches per year) to simulate recharge of rooftop runoff (0.25 and 0.42 cubic feet per second) and the effects of out-of-basin transfers by sewering (1.54 cubic feet per second).

Simulation-Optimization of Water Withdrawals and Stream Base Flow in the Populatic Pond Area

Simulation-optimization methods were applied in a subarea of the study area near Populatic Pond and the confluence of the Charles and Mill Rivers (fig. 32). In the Populatic Pond area, water is withdrawn from the stratified glacial aquifer for public supply for three towns—Franklin, Medway, and Norfolk.

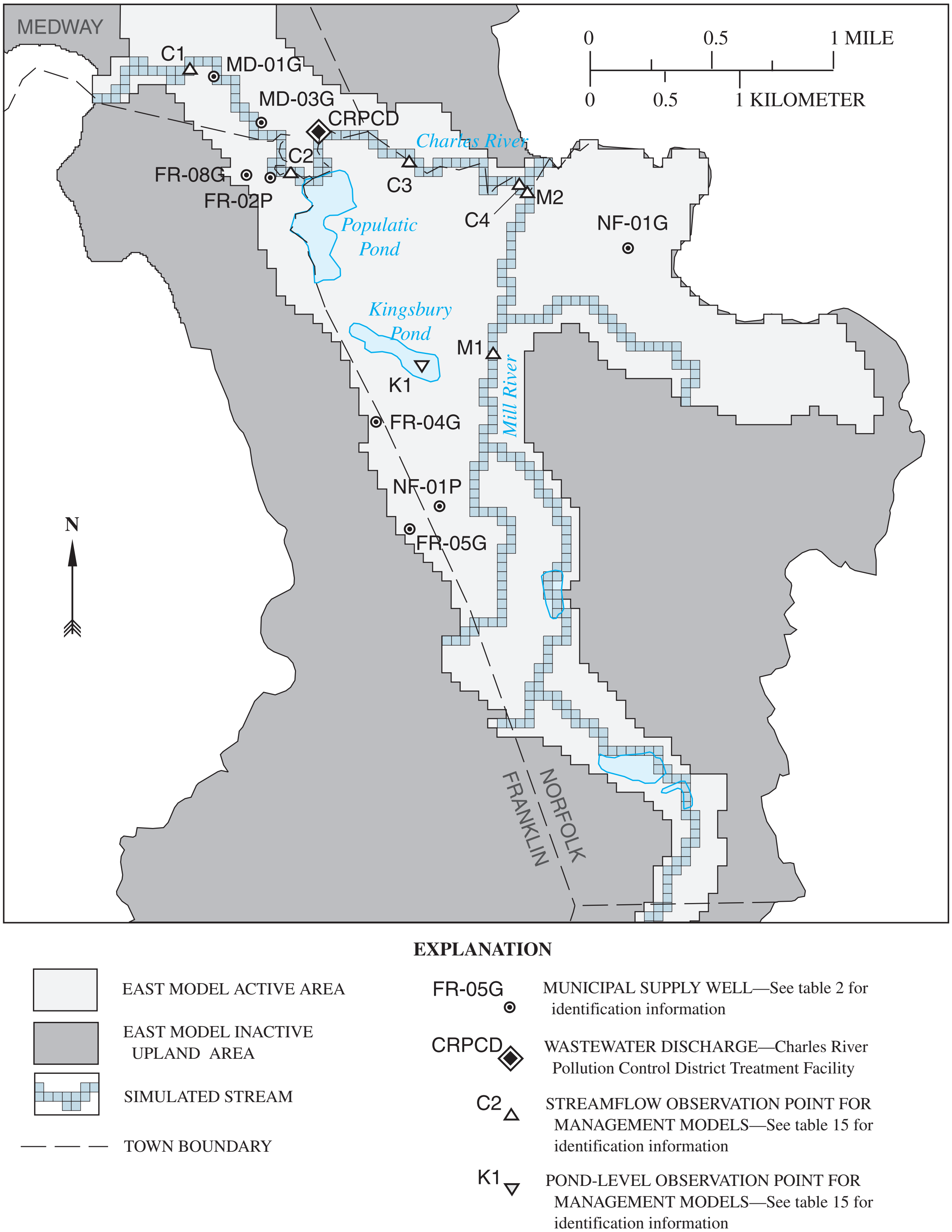


Figure 32. Aquifer area near Populatic Pond where optimization methods were applied in the upper Charles River Basin, eastern Massachusetts.

Water levels in ponds as well as streamflows are affected by these withdrawals. Simulation-optimization methods can provide information about how to minimize the effects of water withdrawals on the stream-aquifer system, or, conversely, on how to maximize water withdrawals from an area within specific limitations (constraints) on hydrologic responses. The simulation-optimization approach was applied in the Populatic Pond area to test its efficacy for water management in a typical area of the upper Charles River Basin where multiple water-supply and water-resource needs coexist. Three management questions, identified through consultation with the Technical Advisory Committee, MADEM, MADEP, and CRWA, were posed:

1. Can withdrawals at existing and proposed wells be increased, and by how much, while maintaining flow in the Charles and Mill Rivers above specified minimum-flow requirements?
2. Can existing withdrawal volumes be increased with no additional streamflow depletion?
3. Can streamflow depletion in the Charles River be reduced relative to current conditions while maintaining current water withdrawals?

Theory and Methods of the Simulation-Optimization Approach

The simulation-optimization approach relies on the numerical ground-water-flow model to simulate the hydrologic response of the stream-aquifer system to applied stresses such as water withdrawals. Optimization techniques are used to formulate and answer specific management questions about the applied stresses and hydrologic responses. The mathematical representation of these questions can be termed a management model that is formulated and solved separately from the flow model. Further information about the use of the simulation-optimization methods for ground-water resource management can be found in Ahlfeld and Mulligan (2000), and in a recent application (Barlow, 1997; Barlow and Dickerman, 2001).

A management model has three components: objective function, decision variables, and constraints (Ahlfeld and Mulligan, 2000). (1) The objective function represents the goal of the management process. It typically is represented by a mathematical formula in which some quantity (typically a stress on the aquifer) is maximized or minimized. For example, it may be desired to maximize total pumping (Qw_T) at three supply wells (Maximize $Qw_T = Qw_1 + Qw_2 + Qw_3$). (2) The decision variables are the quantities that are to be determined in the solution of the management question. In the example given, pumping rates at the individual supply wells (Qw_i) are the decision variables. (3) The constraints set limits on the values that decision variables may take in the solution to the question. Constraints for the example given could be that total withdrawals from all supply wells must exceed a specified value ($Qw_1 + Qw_2 + Qw_3 > 1$ Mgal/d) or that pumping rates at individual wells cannot exceed established upper limits ($Qw_1 \leq 0.5$ Mgal/d; $Qw_2 \leq 0.3$ Mgal/d; and $Qw_3 \leq 0.2$ Mgal/d). Constraints also may consist of limitations on the hydrologic response of the stream-aquifer system, such as maximum allowed quantities of streamflow depletion or minimum ground-water levels at specified observation points.

The management model is linked to the ground-water flow model through a matrix of response coefficients (Gorelick and others, 1993; Ahlfeld and Mulligan, 2000). Response coefficients quantify the relation between a stress applied to the stream-aquifer system and the response of the system to the stress, in terms of streamflow depletion or altered ground-water levels. Response coefficients are calculated for streamflow depletion and changes in water levels at specific observation points and times as:

$$R_{i,j,tp,tr} = \frac{D_{i,j,tp,tr}}{Qw_{i,tp}} \qquad (6)$$

where

$R_{i,j,tp,tr}$ = response coefficient describing change at observation point j during month tr caused by a change in pumping at well i during month tp, dimensionless for streamflow depletion and with units of ft/(ft^3/d) for changes in water levels,

$D_{i,j,tp,tr}$ = streamflow depletion or water-level change at observation point j during month tr caused by a change in pumping at well i during month tp, in ft^3/d for streamflow depletion and in ft for water-level change, and

$Qw_{i,tp}$ = specified change in pumping at well i during month tp, in ft^3/d.

Response coefficients are determined for specific locations of interest (observation points) in the active model area. A separate response coefficient represents the hydraulic response at each observation point to changes in the pumping rate at each pumping well during each stress period.

The response-matrix approach assumes a linear relationship between aquifer stresses and hydrologic responses. The assumption of linearity allows multiple responses to be added through superposition. Linearity is assumed both with the magnitude of the stress (streamflow and water-level changes are proportional to changes in pumping) and with the time of the stress (streamflow and water levels change equally in response to equal changes in pumping at all times in the annual cycle). The assumption of linearity may not be valid under some conditions; for example, when the proximity of a stressed well to a flow boundary causes non-linear head responses. In the upper Charles River Basin, aquifer geometry and the distribution of permeable materials is such that supply wells generally are located near streamflow boundaries. Thus, the assumption of linearity of aquifer response to imposed stresses was tested for management questions investigated in this study, as described below.

A linear programming computer software package, LINDO (Shrage, 1997), was used to solve management questions posed for the Populatic Pond area in this study. Solutions consist of optimal values for all the decision variables, which, for the questions posed, were average monthly pumping rates at each well included in the management models.

Application in the Populatic Pond area

In the area near Populatic Pond (fig. 32), the aquifer is narrow, and supply wells for the three municipalities in many cases affect the same ground- and surface-water resources. Issues of concern for water management in this area are (1) the multiple water-supply needs, and (2) the effects of the withdrawals on streamflows in the Charles and Mill Rivers and on water levels in Kingsbury Pond. Water-supply needs and hydrologic effects of withdrawals are of particular concern in summer months, when water use typically is greatest and recharge, streamflows, and water levels are low. Management models that formalize three management questions identified for testing were developed. The models were formulated similarly in terms of hydrologic stresses (decision variables) and responses, but differed in their objective functions and in the constraints set on management model solutions.

Hydrologic Stresses and Responses

Hydrologic stresses included in the management models of the Populatic Pond area are water withdrawals at existing and proposed supply wells for the towns of Franklin and Norfolk (table 16). Monthly average pumping rates at these six wells constitute the decision variables in the models. Pumping rates for the Medway supply wells (MD-01G and MD-03G) were not included as decision variables. Changes in pumping rates at these two wells, which are immediately adjacent to the Charles River (fig. 32), would have little effect on the Mill River or Kingsbury Pond. For this reason, and to minimize the complexity of the management models, pumping rates at Medway wells were maintained at average 1989–98 levels. Similarly, wastewater discharge from the CRPCD treatment facility also was maintained at average 1989–98 levels. These simplifications were adopted for the hypothetical questions posed in this study and likely would be modified in a more comprehensive application of the simulation-optimization approach in this area.

Streamflows in the management models were represented by flows at six representative observation points along the Charles and Mill Rivers (table 16 and fig. 32). Observation points are model-cell locations where response coefficients were evaluated and where constraints were set in the management models. Observation points were selected near the confluences of the rivers and upstream and downstream of managed water withdrawals. Four points were selected along the Charles River (C1–C4) and two were selected along the Mill River (M1 and M2; fig. 32). Kingsbury Pond, a kettle pond, also was represented by an observation point. Water levels in Populatic Pond, which is an extension of the Charles River, are controlled by stream stage, and, thus, an observation point in this pond was not included in the management models. Observation points C1 and M1 approximately coincided with measurement sites 01103280 and 01103300, respectively.

Response coefficients for streamflow depletion and changes in head were calculated for each observation point (table 16) by means of flow simulations with the east model and equation 6. A base simulation was run in which withdrawals were set to zero for all wells included in the management models. Then, four

Table 16. Pumping wells, stream locations, and ponds in management models of the Populatic Pond area, upper Charles River Basin, eastern Massachusetts

[**Identifier**: See figure 32 for locations. **Minimum flow**: Based on a minimum-flow requirement of 0.21 cubic feet per second per square mile; --, not applicable or not used]

Identifier	Name	Model location			Role in management model		Drainage area (square miles)	Minimum flow (cubic feet per second)
		Layer	Row	Column	Decision variable	Observation and (or) constraint location		
Pumping wells								
FR-04G	Franklin well No. 4	1	243	245	X	--	--	--
FR-05G	Franklin well No. 5	1	255	249	X	--	--	--
FR-08G	Franklin well No. 8	1	216	231	X	--	--	--
FR-02P	Franklin proposed well	1	217	234	X	--	--	--
NF-01G	Norfolk well No. 1	1	224	273	X	--	--	--
NF-01P	Norfolk proposed well	1	252	252	X	--	--	--
Stream locations								
C1	Charles River at Medway[1]	1	205	225	--	X	65.2	13.7
C2	Charles River above Populatic Pond	1	216	236	--	X	66.2	13.9
C3	Charles River between Populatic Pond and Mill River	1	216	250	--	X	66.9	14.0
C4	Charles River above Mill River	1	218	261	--	X	67.6	14.2
M1	Mill River near Kingsbury Pond[2]	1	236	258	--	X	13.8	2.90
M2	Mill River above Charles River	1	220	262	--	X	16.0	3.36
Ponds								
K1	Kingsbury Pond	1	237	250	--	X	--	--

[1]Approximate location of streamflow measurement site 01103280.
[2]Location of streamflow measurement site 01103300.

simulations were run for each well in which pumping was simulated at the well for a 1-month period. Response coefficients were calculated by dividing streamflow depletion ($D_{i,j,tp,tr}$) by the pumping rate ($Qw_{i,tp}$), as in equation 6.

To test the assumption of linearity used in the response coefficient methodology, a high pumping rate (1.0 Mgal/d) and a lower pumping rate (0.5 Mgal/d) were simulated for each well during both January and July (a total of four simulations per well). For the assumption of linearity to be valid, response coefficients at an observation point generated from the four simulations for the same pumping well (R values with common i,j values; equation 6) should be equal or nearly so. The lower pumping rate of 0.5 Mgal/d equals the average State-permitted maximum pumping rate for the wells considered, and the high pumping rate of 1.0 Mgal/d exceeds the State-permitted maximum pumping rate for the wells, except for NF-01P which has a proposed limit of 1.1 Mgal/d. For the wells, times, and withdrawal rates tested, the response coefficients with common (i,j) values typically varied by less than 10 percent. The variation among response coefficients was the highest for observation points and times when the effect of the increased pumping was very small. This is likely due at least in part to numerical model errors because the variation in coefficients did not show any consistent correlation with either pumping rate or time of pumping. For coefficients that were greater than 0.20, the differences among coefficients were in all cases less than 20 percent. These typically slight disagreements among response coefficients indicate that the stream-aquifer system in the Populatic Pond area may be weakly nonlinear. Although a sequential linearization approach could be used to address the nonlinearities, such an approach was considered unnecessary based on previous studies (Barlow and Dickerman, 2001).

A mass-balance test of the stream-aquifer system response in the Populatic Pond area also was performed by analysis of response coefficients. Stream cells are the only boundaries, other than pumping wells, where water can exit the model in the management-model area. Thus, streamflow depletions resulting from a pumping stress should equal the total pumping withdrawal, under steady-state conditions, at a stream location sufficiently downstream from the pumping well. Under transient conditions, the timing of pumping withdrawals and streamflow depletions at observation points may be offset, but mass balance should hold when total flows are summed over the annual cycle. This mass-balance relation means that the sum of response coefficients for each month of the annual cycle at a downstream observation point should equal 1.0 (Barlow, 1997). This test was verified for all wells upgradient of streamflow observation points (fig. 32). The response coefficients for streamflow depletion in the Charles and Mill Rivers for these wells satisfied the mass-balance requirement to within 3 percent (table 17).

Final response coefficients for the supply wells and observation points in the management models were determined by selecting values from one of the four simulation runs for each $R_{i,j}$ pair. The selected values were close to middle of the range of values and best met the mass-balance relation. Final response coefficients are given in table 18.

The variability in hydrologic responses to pumping stresses in the model area, in terms of streamflow depletion, is illustrated by the distribution of response coefficients among months of the annual cycle from pumping at each supply well for 1 month at 1 ft^3/s (0.65 Mgal/d) (table 18). For example, pumping at FR02P, which is immediately adjacent to the Charles River, results in rapid streamflow depletion in the Charles River but only small depletions in the Mill River. More than 90 percent of streamflow depletion from the Charles River from 1 month of pumping at FR-02P has occurred by the end of the first month after the pumping month (table 18). In contrast, pumping at FR-04G for 1 month results in streamflow depletions that extend for 5 subsequent months, primarily in the Mill River. This variability is summarized in figure 33, which shows the time required for 90 percent of the depletion that occurs in the Charles and Mill Rivers from pumping at each well for 1 month. The time required for depletions to occur increases with increasing distance of the pumping well from a simulated stream. This variability is advantageous for optimization analysis, in which pumping schedules at individual wells are managed to minimize streamflow depletion during critical time periods.

Response coefficients for the Kingsbury Pond observation point also indicate the variable effects of pumping at supply wells in the management-model area on pond-water levels. The largest effect resulted from pumping at FR-04G, as would be expected from its close proximity to the pond. This relation of pumping at FR-04G and water levels in Kingsbury Pond is consistent with the previous understanding of the ground-water flow system near the pond (Williams, 1967; Bouck, 1998). Water levels in Kingsbury Pond also were affected, to a lesser degree than by pumping at FR-04G, by pumping at FR-05G and NF-01P. Response coefficients indicate that pumping at FR-04G at rate of 1 ft^3/s (0.65 Mgal/d) for 1 month results in a decline in water levels at the pond of about 0.5 ft in each of the first 2 months following the beginning of pumping, with additional declines in subsequent months. Continuous pumping at FR-04G at this rate would result in a total quasi-steady-state decline in water levels of about 2 ft (monthly response coefficients are additive). These results are consistent with the results of the two scenarios of increased pumping described previously. It should be noted, also, that determination of the effects of pumping at supply wells on pond levels (as well as on streamflow) by analysis of response coefficients is subject to at least the same limitations and assumptions as the numerical flow models, as wells as assumptions incorporated in the optimization approach.

Table 17. Hydrologic response coefficients for streamflow depletion at downstream observation points from pumping wells used to verify mass balance in the management models of the Populatic Pond area, upper Charles River Basin, eastern Massachusetts

[See figure 32 and table 16 for observation-point locations]

Pumping well identifier	Sum of monthly response cofficients at downstream observation points		
	Mill River above Charles River (C4)	Charles River above Mill River (M2)	Sum of Mill and Charles Rivers
FR-04G	0.79	0.24	1.03
FR-05G	.93	.05	.98
FR-08G	.03	.95	.98
FR-02P	.02	.98	.99
NF-01P	.96	.06	1.01

Table 18. Hydrologic response coefficients for the Populatic Pond area of the upper Charles River Basin, eastern Massachusetts

[Response coefficents represent change in streamflow or water level per unit withdrawal (1 cubic foot per second) for 1 month of pumping at supply wells, dimensionless for changes in streamflow and in units of feet per cubic foot per second for changes in water level (1 cubic foot per second equals 86,400 cubic feet per day). **Observation sites:** C1-C4 and M1-M2 are streamflow sites, and K1 is a pond site; sites are described in table 16. **Month**s: Month 1 is the month in which pumping occurs; --, no hydrologic response at observation point from pumping at supply well]

Supply well	Obser-vation point	Months											
		1	2	3	4	5	6	7	8	9	10	11	12
FR-04G	C1	--	--	--	--	--	--	--	--	--	--	--	--
	C2	--	--	--	--	--	--	--	--	--	--	--	--
	C3	0.007	0.030	0.045	0.037	0.022	0.016	0.010	0.006	--	--	--	--
	C4	.007	.041	.052	.043	.031	.020	.015	.007	0.007	0.007	0.007	--
	M1	.145	.192	.121	.073	.042	.025	.015	.010	.006	--	--	--
	M2	.151	.220	.150	.098	.067	.040	.027	.018	.010	.007	--	--
	K1	.504	.543	.362	.233	.155	.103	.065	.039	.026	.013	.013	--
FR-05G	C1	--	--	--	--	--	--	--	--	--	--	--	--
	C2	--	--	--	--	--	--	--	--	--	--	--	--
	C3	--	--	.007	.015	.007	.008	--	--	--	--	--	--
	C4	--	--	.007	.015	.015	.009	.006	--	--	--	--	--
	M1	.310	.307	.150	.073	.036	.019	.011	.007	--	--	--	--
	M2	.295	.307	.159	.082	.042	.023	.013	.008	--	--	--	--
	K1	.026	.103	.142	.116	.078	.065	--	--	--	--	--	--
FR-08G	C1	.015	.015	.015	--	--	--	--	--	--	--	--	--
	C2	.401	.085	.023	.008	--	--	--	--	--	--	--	--
	C3	.675	.192	.052	.017	.007	--	--	--	--	--	--	--
	C4	.675	.195	.054	.018	.008	--	--	--	--	--	--	--
	M1	--	.010	.006	--	--	--	--	--	--	--	--	--
	M2	--	.012	.008	.006	.006	--	--	--	--	--	--	--
	K1	--	.013	.013	--	--	--	--	--	--	--	--	--
FR-02P	C1	.009	.009	.009	--	--	--	--	--	--	--	--	--
	C2	.319	.047	.013	.005	--	--	--	--	--	--	--	--
	C3	.765	.147	.038	.012	.005	--	--	--	--	--	--	--
	C4	.767	.151	.040	.014	.007	--	--	--	--	--	--	--
	M1	--	--	--	--	--	--	--	--	--	--	--	--
	M2	--	.005	.005	.005	--	--	--	--	--	--	--	--
	K1	--	.015	.015	--	--	--	--	--	--	--	--	--
NF-01G	C1	--	--	--	--	--	--	--	--	--	--	--	--
	C2	--	--	--	--	--	--	--	--	--	--	--	--
	C3	--	--	--	--	--	--	--	--	--	--	--	--
	C4	--	.015	.015	.015	.007	.005	--	--	--	--	--	--
	M1	--	.006	--	--	--	--	--	--	--	--	--	--
	M2	.111	.164	.112	.074	.050	.035	.025	.012	.010	.010	.013	0.007
	K1	.006	.013	.019	.019	.019	.019	.013	.006	.006	--	--	--
NF-01P	C1	--	--	--	--	--	--	--	--	--	--	--	--
	C2	--	--	--	--	--	--	--	--	--	--	--	--
	C3	--	--	.015	.015	.007	.007	--	--	--	--	--	--
	C4	--	--	.015	.015	.015	.007	.006	--	--	--	--	--
	M1	.439	.214	.113	.062	.037	.021	.012	.009	.006	--	--	--
	M2	.439	.221	.121	.071	.045	.025	.016	.010	.007	--	--	--
	K1	.038	.115	.128	.103	.064	.051	.038	.013	.013	--	--	--

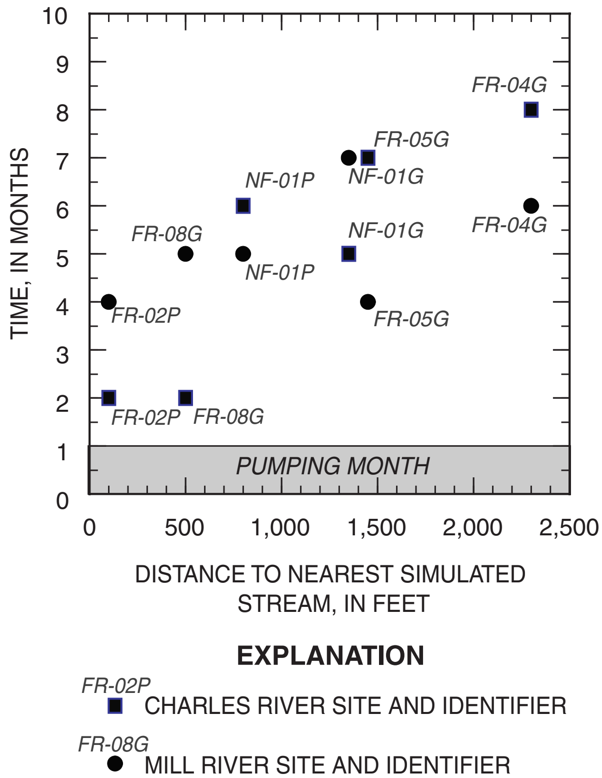


Figure 33. Relation of the time required for streamflow depletion to occur in response to pumping for one month at supply wells and the distance of the supply well from the nearest simulated stream for six supply wells in the Populatic Pond area, upper Charles River Basin, eastern Massachusetts. Times shown are months, from the end of the pumping month, within which more than 90 percent of the total streamflow depletion that results from the 1-month pumping stress in the nearest stream.

Management Model Applications

Management models were developed and applied to address the three management questions described previously. For the first two questions, the goals were to increase water withdrawals in the Populatic Pond area within specified constraints on streamflow depletion. For the third question, the goal was to reduce current streamflow depletion while maintaining current total withdrawals.

Increase Withdrawals while Maintaining Stream Base Flow above Specified Minimum-Flow Requirements

The first set of applications examined whether it would be possible to increase water withdrawals from existing and proposed sources in the Populatic Pond area, while maintaining streamflows in the Charles and Mill Rivers above specified minimum-flow requirements. Because water demand is greatest in June, July, and August, the objective function was formulated to maximize pumping during these months. The objective function for these applications is:

$$maximize \sum_{i=1}^{NW} \sum_{t=6}^{8} ND_t Qw_{i,t} \tag{7}$$

where NW is the total number of wells, ND_t is total number of days in month t, and $Qw_{i,t}$ is the average pumping rate at well i in month t, as described previously. The value that is maximized is the sum of pumped volumes in June, July, and August from all wells in the management models (table 16).

The value of the objective function was limited by constraints on streamflow depletion, pumping rates at wells, and, initially, the water level in Kingsbury Pond. Constraints were placed on streamflow depletion in the Charles River and Mill River, such that model-calculated streamflows were not allowed to fall below the requirement of 0.21 $(ft^3/s)/mi^2$ at each of the streamflow observation sites listed in table 16. Imposition of the streamflow constraint at all streamflow observation points was designed to ensure that flows in the rivers were above the specified limit throughout the area of interest. A minimum-streamflow requirement of 0.21 $(ft^3/s)/mi^2$ currently is in use by MADEM for permitting activities that affect flows in the Charles River (V. Gartland, Massachusetts Department of Environmental Management, written commun., 2002). It should be noted that minimum-flow requirements such as this one typically refer to total streamflow, rather than to the base-flow component of streamflow that is simulated by the ground-water flow models. However, during low-flow period such as summer months, most streamflow is base flow, so that use of minimum-flow requirements as constraints in the management models was considered appropriate.

Pumping rates at individual wells were constrained so as not to exceed MADEP-approved Zone II rates (table 2). The streamflow and Zone II constraints on pumping were imposed equally in all months of the annual cycle. Pumping rates also were constrained so that combined pumping from all existing Franklin and Norfolk sources in each month of the annual cycle equaled or exceeded 1989–98 monthly average withdrawals, plus an added volume for each proposed well.

The added volumes for proposed wells were equal to one-half the estimated Zone II rate (table 2; B.R. Bouck, Massachusetts Department of Environmental Protection, written commun., 2002). These constraints were applied separately to Franklin and Norfolk sources, and ensured that total withdrawals for each town were at least as great as current withdrawals (as represented by 1989–98 averages).

Initially, a constraint also was imposed on water levels in Kingsbury Pond. Water levels in Kingsbury Pond have declined in response to pumping since the 1960s (Williams, 1967; Bouck, 1998). Current average water levels in Kingsbury Pond (table 5) were estimated at about 9 ft below pre-1960s levels of about 139 ft above mean sea level. Some septic-system leachfields installed near the pond since these water-level declines have been below the 5-ft minimum depth to the water table required by State regulations (Title V) for septic systems (B.R. Bouck, Massachusetts Department of Environmental Management, oral commun., 2002). Thus, a constraint was imposed to maintain water levels in Kingsbury Pond below 129.9 ft above sea level. This level was below the average 1989–98 level. Inclusion of this constraint, however, resulted in no feasible solution to the management model. This outcome resulted because the high pumping rates required to keep water levels in Kingsbury Pond below the specified limit would have been greater than the maximum allowed Zone II pumping rates at wells. When constraints for maximum head in Kingsbury Pond were removed, a feasible solution was possible.

Total withdrawals for June, July, and August in the solution to this optimization problem averaged 3.98 Mgal/d, more than twice the average withdrawals during 1989–98 in these months (1.55 Mgal/d). Average annual withdrawals were about doubled, from 1.38 Mgal/d to 2.7 Mgal/d. Most of the increases resulted from pumping at the proposed wells. However, pumping rates were increased at most wells in the management model in June, July, and August, such that all wells pumped at their maximum approved rates in these months (fig. 34*A*). Pumping rates were lower in months other than June, July, and August (although the total pumping for each town was still at or above 1989–98 rates) because the other months were not part of the objective function. Rates also were lower in other months because lower rates in winter months help to maintain higher streamflow during the summer months when streamflow approaches the minimum flow rates.

These substantial increases in withdrawals were accompanied by additional streamflow depletion relative to model-calculated 1989–98 flows. Model-calculated September streamflows were reduced by about 40 percent (2.4 to 3.0 ft^3/s) at observation points in the Mill River, by about 5 percent (1.7 to 1.8 ft^3/s) at observation points in the Charles River downstream of Populatic Pond, and by about 2 percent (0.6 ft^3/s) at observation points in the Charles River upstream of Populatic Pond. Streamflow depletions in the Charles River downstream of Populatic Pond probably would be partly offset by increased discharges at the CRPCD treatment facility, which were not simulated. Model-calculated stream base flows were not reduced to the constraint limit of 0.21 $(ft^3/s)/mi^2$ because the constraints on maximum pumping rates at individual wells were reached first. The increased withdrawals also were accompanied by a decline in model-calculated average annual water level in Kingsbury Pond of about 0.3 ft.

Minimum streamflow requirements to maintain aquatic habitat are the subject of active research in Massachusetts (for example, Armstrong and others, 2001). Alternative values could be considered applicable for streams in the upper Charles River Basin. For example, the U.S. Fish and Wildlife Service (USFWS) recommends a minimum-flow requirement of 0.5 $(ft^3/s)/mi^2$ in summer months for New England streams (aquatic base flow or ABF; Armstrong and others, 2001). Thus, alternative values to the 0.21 $(ft^3/s)/mi^2$ minimum-flow requirement may be considered in future watershed planning or permitting in the upper Charles River Basin. For this reason, additional management models were formulated in which the effects of various minimum-streamflow requirements were investigated. Streamflow-depletion constraints were reformulated to impose minimum-flow requirements of 0.35, 0.40, 0.43, and 0.50 $(ft^3/s)/mi^2$. Other aspects of the management model, including the objective function, constraints on pumping, and the locations of streamflow constraints (observation points), remained the same. Each of the alternate minimum-flow requirements was implemented in a separate optimization formulation and solution.

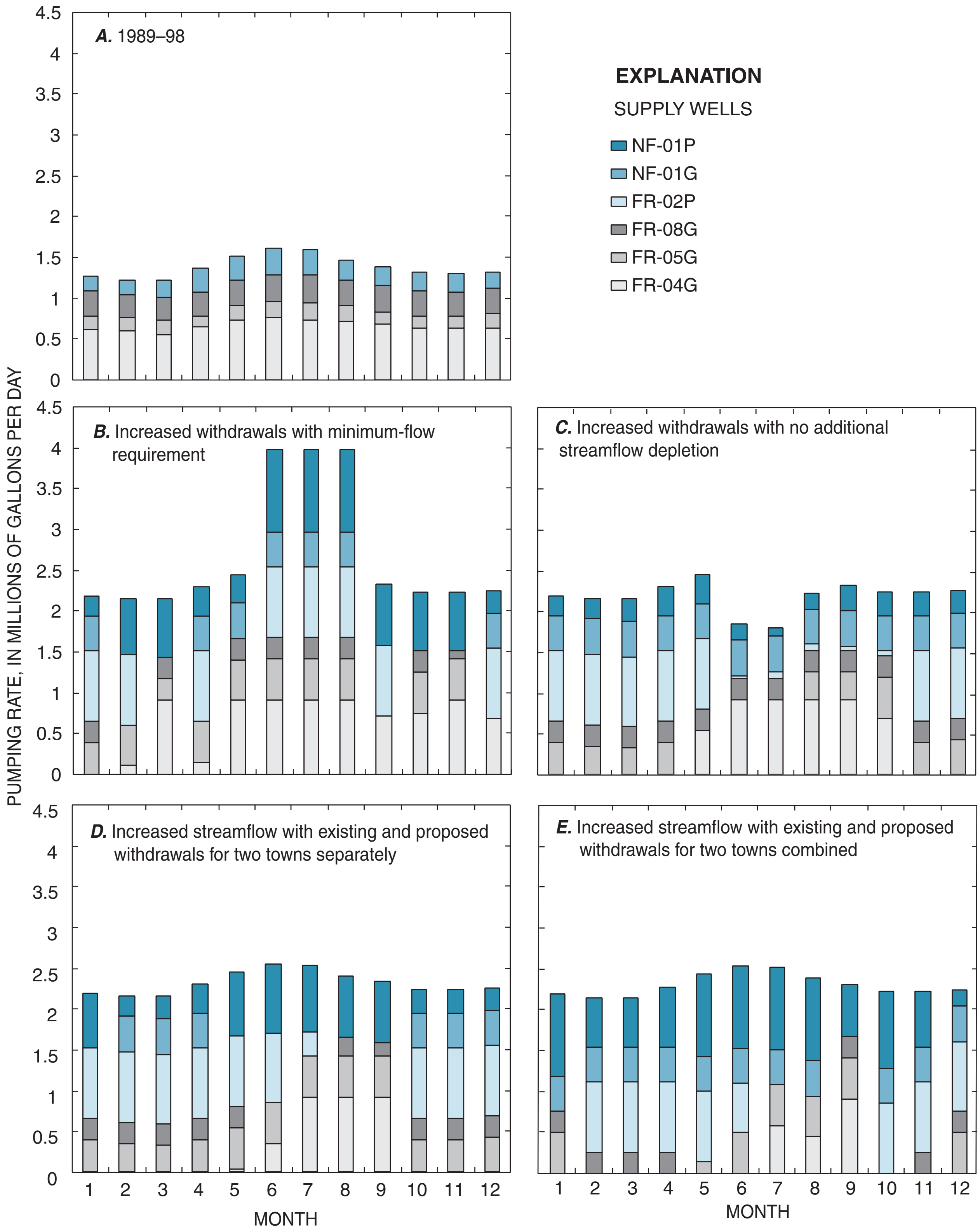


Figure 34. Monthly pumping rates for 1989–98 and for management-model applications for increased withdrawals or increased streamflow in the Charles River during the low-flow period in the Populatic Pond area of the upper Charles River Basin, eastern Massachusetts: (*A*) 1989–98, (*B*) Increased withdrawals with a minimum-streamflow requirement (0.21 cubic feet per second per square mile), (*C*) Increased withdrawals with no additional streamflow depletion, (*D*) Increased streamflow with existing and proposed withdrawals for two towns separately, and (*E*) Increased streamflow with existing and proposed withdrawals for two towns combined.

The relation between total withdrawals in June, July, and August and minimum-streamflow requirements in the Populatic Pond area, as determined by the management models, is shown in figure 35. No reductions in total allowable summer withdrawals result until the minimum-flow requirement is greater than 0.39 $(ft^3/s)/mi^2$. For minimum-flow requirements of 0.43 $(ft^3/s)/mi^2$ or greater, no solution could be found that allowed 1989–98 withdrawals in the area to be maintained.

Increase Water Withdrawals while Maintaining Stream Base Flow at Current Levels

A second set of model applications examined whether it would be possible to increase water withdrawals in the Populatic Pond area while maintaining current streamflows in the Charles and Mill Rivers. Current streamflows were represented by model-calculated 1989–98 average monthly flows. Streamflow-depletion constraints were reformulated in these applications to allow no decrease in monthly average stream flows. The objective function was to maximize pumping in June, July, and August at existing and proposed sources. Constraints on pumping also were imposed initially such that combined total pumping for each town (Franklin and Norfolk) was required to meet or exceed 1989–98 rates for each month with the addition of proposed sources as described previously. However, no feasible solution was found with these constraints imposed. When these constraints were removed for the months of June, July, and August, an optimal solution was found (fig. 34*B*). Total withdrawals for June, July, and August in the optimal solution averaged 1.95 Mgal/d, exceeding 1989–98 withdrawals in these months by about 25 percent. Total average annual withdrawals were 2.19 Mgal/d, about 60 percent greater than 1989–98 withdrawals. In this solution, combined withdrawals from Franklin sources were less than 1989–98 withdrawals in June and July, but greater in August (table 19).

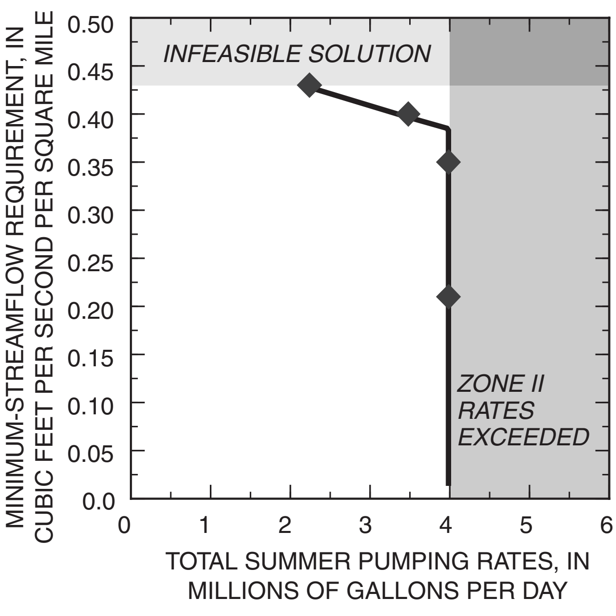


Figure 35. Relation of minimum streamflow requirements and total summer pumping rates from managed wells, as determined by optimal solutions to management models in the Populatic Pond area of the upper Charles River Basin, eastern Massachusetts. Rates shown are sums of average daily pumping rates in June, July, and August for supply wells FR-04G, FR-05G, FR-08G, FR-02P, NF-01G, and NF-01P (see tables 4 and 16).

Most of the additional volume of water withdrawn in the optimal solution for this management model, relative to 1989–98 withdrawals, occurred at the proposed wells. However, pumping schedules at existing wells also differed from 1989–98 conditions to take advantage of the variability in hydrologic response to pumping stress at the wells. For example, FR-04G, which is farther from simulated streams than other wells in the management model (fig 30), was pumped at higher rates than in 1989–98 in June through September and was not pumped at all from November through April. In contrast, pumping at FR-05G did not occur in May through July, but occurred at higher rates than in 1989–98 in August through December.

Increase Stream Base Flow while Maintaining Existing and Proposed Withdrawals

A third set of model applications was made to determine whether it would be possible to increase streamflows in the Populatic Pond area while maintaining withdrawals at 1989–98 levels, with added increments for proposed wells. This management model required that the objective function be formulated in terms of streamflows. The objective function for these applications is:

$$minimize Qsd_{C4,\,8} \qquad (8)$$

where $Qsd_{C4,8}$ is streamflow depletion in the Charles River at observation point C4 (Charles River above the Mill River) in August (eighth month). A more complex objective function could be formulated in which streamflow depletion is minimized at multiple observation points and at multiple times; however, that level of complexity was beyond the scope of this study.

Constraints on pumping and streamflow-depletion rates were the same as in the applications in which June, July, and August withdrawals were maximized and the minimum-flow requirement was set at 0.21 $(ft^3/s)/mi^2$.

August stream base flow in the Charles River above Mill River in the optimal solution of the objective function was 40.2 ft^3/s. This value was about 1 ft^3/s greater than model-calculated August base flow (39.3 ft^3/s) for 1989–98 withdrawals at existing sources and withdrawals from proposed wells equal to one-half their estimated MADEP-approved rate as described previously (fig. 34*C*). In this solution, Franklin and Norfolk sources separately withdrew the same water volumes in all months of the year as in 1989–98, with added increments for proposed wells. Thus, Franklin and Norfolk wells were managed separately to achieve reductions in streamflow depletion. Franklin wells were managed by eliminating or reducing pumping at wells adjacent to the Charles River (FR-08G and FR-02P) in summer relative to 1989–98 rates, increasing pumping at these wells and at FR-05G in winter, and eliminating pumping at FR-04G in winter (table 20). Norfolk's wells were managed by eliminating pumping in summer at NF-01G, which is the closer of the two wells to the Charles River, and increasing summer pumping at NF-01P, which is farther from the Charles River. The changes in pumping schedules also resulted in a decline in the average annual water level in Kingsbury Pond of about 1 ft.

An alternative formulation for this question was tested to examine how collaborative management of water supplies could result in further reductions in streamflow depletion. In this application, the constraint that Franklin and Norfolk sources separately withdrew the same water volumes as in 1989–98 (with increments for proposed wells) was replaced with a constraint that total withdrawals from all Franklin and Norfolk sources jointly met this requirement. This is equivalent to assuming that water-supply systems for these towns were connected and regulated so that supplies could be shared. The optimal solution for this application resulted in August stream base flow in the Charles River above Mill River of 45.8 ft^3/s, an increase of about 14 percent relative to the optimal solution for separate management of water-supply systems. The additional reductions in streamflow depletion resulted from decreased pumping at Franklin sources adjacent to the Charles River and increased pumping at NF-01G, in summer, and at NF-01P, throughout the year, relative to 1989–98 rates (fig. 34*D* and *E*). Thus, collaborative management of water systems in the Populatic Pond area results in a better solution in terms of the model objective because the optimization analysis can make use of greater variability in hydrologic responses to pumping stresses to better manage individual withdrawals.

Limitations of Management-Model Applications

The management-model applications for the upper Charles River Basin described in this study are subject to limitations that should be considered when their results are interpreted. In all applications of management models in this study, allowable streamflow depletion is quantified relative to model-calculated, rather than observed, stream base flows. Thus, differences between observed and model-calculated flows (model calibration error) should be considered when the results of the management-model applications are interpreted. Theoretically, alternative formulations of the management models could specify streamflow depletion relative to observed rather than model-calculated flows. Observed streamflows, however, are available only for a limited number of locations. In addition, simulation-optimization results would need to be interpreted in terms of differences between observed flow values, represented by estimated monthly averages for 1989-98, and model-calculated values (model-calibration error). Model-calibration error, in general, resulted from multiple, poorly known sources of error and varied spatially and over the annual cycle. Thus, it could be difficult to formulate streamflow constraints that eliminate model calibration error consistently and accurately throughout the management model. This effort was beyond the scope of this study.

The results of the management-model applications that include streamflow constraints are for average monthly base-flow conditions. They do not represent dry years or describe the variability in stream base flow within a month during the average year. In dry years, stream base flow is less than long-term average flows, represented by the 1989–98 time period in this study. In average and dry years, mean daily streamflows on some days in any month are less than the monthly average. Thus, mean daily streamflows occurring under pumping schedules determined by management-model applications that include minimum-flow constraints are likely to fall below the specified minimum flow constraints on some days, because the streamflow constraint is applied in the models as a limit on monthly average values. Finally, minimum-flow requirements typically are set in terms of total streamflow, which is easily measured, rather than base flow, which is calculated by the ground-water-flow models.

SUMMARY

Ground water is the primary source of drinking water for towns in the upper Charles River Basin, an area in eastern Massachusetts that is undergoing rapid growth. Stratified glacial aquifers in the basin are thin and discontinuous and are in close hydraulic connection with streams, ponds, and wetlands. Increased water withdrawals, combined with out-of-basin or downstream transfers of wastewater and decreased natural recharge from changes in land use, have stressed water resources by contributing to streamflow depletion and lowering water levels in ponds and wetlands. The stratified glacial aquifers extend across municipal boundaries, potentially leading to conflicts over water availability and downstream effects. Numerical modeling tools were developed by the U.S. Geological Survey, in cooperation with the Massachusetts Departments of Environmental Management and Environmental Protection, to evaluate the effects of water-management alternatives and to address the regional-scale needs for information on water resources in the basin.

The upper Charles River Basin is an area of 105 mi^2 that encompasses most or substantial parts of Towns of Bellingham, Franklin, Holliston, Medway, Milford, Millis, Norfolk and Wrentham. Land use is primarily forested and residential. Population has increased by about 15 percent in the past decade, to about 125,000 in 2000. Water withdrawals from 33 public-supply wells or wellfields, 2 surface-water withdrawals from the Charles River, and several large non-municipal sources averaged 10.1 Mgal/d in 1989–98 and are likely to increase. Six new public-supply wells were being permitted at the time of this study. About half of the population in the basin is served by public sewers, and the treated wastewater discharges to the Charles River from facilities in Milford and Medway.

Stratified glacial aquifers provide the primary source of water for municipal supply and large non-municipal users. Stratified glacial aquifers occur along the Charles River and major tributaries and are thin and irregular in shape. Areally, they cover about 50 percent of the study area. Stratified glacial aquifers consist of depositional sequences that include sand, gravel, silt, and clay. Maximum thicknesses of 120 to 130 ft are reached in a preglacial bedrock valley in the eastern part of the study area, but elsewhere aquifer thickness generally is 70 ft or less. Stratified glacial deposits in western and eastern parts of the study area are discontinuous, but are hydrologically connected by the Charles River. Ground water generally is unconfined, but local confining conditions are present in areas where silt and clay sediments are well developed.

Streamflow averaged 103 ft^3/s in the Charles River at Medway during water year 2000. Average annual base flow, estimated for the period 1989–98 from correlations of measurements made during the study with records at long-term stream-gaging stations, was 88 percent of average annual streamflow during water year 2000. Estimates were similarly made of average annual and monthly base flow at 24 sites where flow measurements were made during the study on the Charles River and its tributaries. Estimated mean annual base flow at these sites ranged from 73 to 122 percent of measured annual base flow, on the basis of monthly measurements during base-flow conditions in water year 2000; measured values all were within the 90-percent confidence intervals of estimated values.

Measured ground-water levels at 50 observation wells ranged from 250 ft above sea level in headwater areas in the western part of the study area to less than 140 ft above sea level in low-lying areas along streams near the eastern boundary. Measured annual fluctuations in ground-water levels varied with topography and position of the observation well with respect to aquifer boundaries. Annual fluctuations during the study ranged from 1 ft to more than 3 ft, and typically reached maximum elevations in late spring. Mean annual water levels for 1989–98, estimated by correlation with nearby long-term observation wells, were nearly identical with measured water levels during water year 2000.

Inflows to the upper Charles River Basin included ground-water recharge from precipitation, ground-water recharge from septic-system return flow, and wastewater discharge to streams from treatment facilities. These inflows were estimated at 100 to 130 Mgal/d for precipitation recharge, 3.1 Mgal/d for septic-system return flow, and 7.5 Mgal/d for wastewater discharges to streams. Outflows included stream

base flow at downstream boundaries, water withdrawals, ground-water evapotranspiration, possible ground-water underflow at study area boundaries, and infiltration into sewer lines. Outflows through streamflow were estimated at 104 Mgal/d (90-percent confidence intervals for streamflow corresponding to 80 to 137 Mgal/d). Outflows for water withdrawals were 10.1 Mgal/d, and outflows through ground-water evapotranspiration from wetlands were estimated at about 5 Mgal/d. Small transfers for water supply and wastewater discharge in and out of the study area balanced each other. The inflows and outflows considered in the water balance agreed within 20 percent.

Steady-state and transient ground-water-flow models were developed for the upper Charles River Basin with MODFLOW-2000. Separate models were developed for east and west aquifer areas to improve numerical stability of the models. Stratified glacial aquifers only were included as active model areas. The models were calibrated to 1989–98 conditions of water withdrawals, water levels, and stream base flows. Recharge rates were simulated as varying spatially in active model areas and in inactive till and bedrock uplands; the rates varied by land use, surficial geology, and return flow. Recharge rates were calibrated to estimated 1989–98 average annual stream base flow with a parameter-estimation technique. Area-weighted rates were 21.5 and 23.0 in/yr for active areas of the west and east models, respectively, and 21.9 and 24.0 in/yr for flows to the active model area from till and bedrock uplands of the west and east models, respectively. Horizontal hydraulic conductivity ranged from 70 to 290 ft/d and vertical hydraulic conductivity ranged from 5 to 50 ft/d in the calibrated models. In the transient models, specific yield varied from 0.15 to 0.28 and specific storage was set at 1.0×10^{-3} in the calibrated models.

Total inflows to the calibrated steady-state models were 119 Mgal/d. In both west and east models, inflows from uplands, which were simulated as additional recharge at active model boundaries, were large components of total inflows. Inflows from uplands were about 70 percent of inflows to the west model and about 50 percent of inflows to the east model. Calculated outflows consisted primarily of discharge to simulated streams, which were 90 and 85 percent of total outflows from the west and east models, respectively. Although small relative to total outflows, simulated ground-water evapotranspiration from wetlands was of similar magnitude (although much less well quantified) to water withdrawals; this result indicates the importance of this flux. Water withdrawals accounted for 7 percent of simulated total fluxes overall through the stream aquifer, and varied from about 1.5 percent (Hopping Brook subbasin) to 13 percent (area near the headwaters of the Charles River) of total fluxes in subbasins of the study area. In the transient models, water withdrawals ranged from 4 to 5 percent of total monthly flows in winter and spring months to about 10 to 12 percent of total monthly flows in summer and early fall.

Two hypothetical scenarios of increased withdrawals in the upper Charles River Basin were simulated with the numerical flow models. Increasing withdrawals to levels allowed in the basin under the Massachusetts Water Management Act would result in withdrawals of about 15 Mgal/d, or about 50 percent more than current withdrawals. Simulated effects of these increased withdrawals include reductions in stream base flow that are greatest in proportion to total flow in late summer and early fall, but varied through the annual cycle as a result of the timing of increased withdrawals at upstream sources. Reductions in model-calculated stream base flow ranged from less than 5 percent to more than 60 percent of model-calculated 1989–98 base flow during low-flow periods in the Charles River and major tributaries.

Along the Charles River, reductions in stream base flow from increased ground-water withdrawals were nearly balanced by flow augmentations from increased wastewater discharges overall. Along specific reaches of the river, reductions or augmentations dominated. Generally, model-calculated base flow was reduced relative to 1989–98 flows upstream of wastewater-treatment facilities in most months and was augmented downstream of treatment facilities in low-flow months. Model-calculated September base flow in the Charles River was reduced relative to 1989–98 flows along three river reaches: by about 70 percent downstream of Milford sources, by about 2 to 4 percent between Stall and Mine Brooks, and by about 5 percent downstream of Mine Brook. The proportion of wastewater in the Charles River downstream of treatment facilities increased relative to 1989–98 conditions from 80 to 90 percent of model-calculated September base flow in Milford and from 18 to 27 percent in Medway.

Along tributaries such as Mine Brook, where return flow was routed to a wastewater discharge on the Charles River, reductions in model-calculated stream base flow was about equal to upstream withdrawals.

Model-calculated 1989–98 September base flow in Mine Brook were reduced by more than 50 percent in headwaters (Miscoe Brook) and by about 10 percent along downstream reaches. Along tributaries in areas where return flow was wholly or partly returned to the ground-water flow system as added recharge, reductions in model-calculated base flow generally were less than increases in upstream withdrawals. Reductions in model-calculated base flow in tributaries were not eliminated even in areas where return flow was routed to the aquifer, primarily because of differences in the contributing areas to the streams and areas where return-flow recharge was applied. Along the Mill River, where return flow is routed to a treatment facility and to septic systems, average reductions in base flow over the annual cycle are about 75 percent of average increased withdrawals. Reductions in model-calculated base flow for the Mill River during the September low-flow period were from about 12 to 14 percent of 1989–98 flows. Increased withdrawals in the Mill River aquifer area also resulted in changes in water levels in Kingsbury Pond, which were reduced by about 0.8 ft on average over the annual cycle in scenarios of increased withdrawals.

In a third set of hypothetical scenarios, additional recharge equal to the transfer of water out of a typical subbasin by sewers was simulated and was found to increase model-calculated base flows. Flow augmentations from this added recharge were about 12 percent of model-calculated 1989–98 base flows, about equal to base flow reductions simulated in the scenarios of increased withdrawals. Addition of recharge equal to that available through artificial recharge of residential rooftop runoff had smaller effects, augmenting simulated September base flow by about 3 percent. The effect of added recharge on model-calculated base flow was linear, but varied throughout the annual cycle. Recharge added in low-flow months went partly to replenish aquifer storage. Thus, increases in model-calculated base flow in low-flow months were less than (60 percent of) the added recharge rate, whereas in high-flow months, added water was released from storage and streamflow increased by amounts greater than (110 percent of) the added recharge.

Simulation-optimization methods were applied in the aquifer area near Populatic Pond and the confluence of the Mill and Charles Rivers to demonstrate the use of these methods in the basin. Water is withdrawn from six supply wells for the three towns in this area. Management models were developed to maximize water withdrawals within specified constraints on streamflow depletion and to minimize streamflow depletion within minimum requirements for water supply. Application of the simulation-optimization methods indicates that hydrologic responses to pumping from different supply wells vary in time and duration in the Populatic Pond area. This variability suggests that water withdrawals could be managed to minimize the effects of increased withdrawals on streams and ponds.

Solutions of preliminary management models for the Populatic Pond area suggested that water withdrawals could be substantially increased from existing and proposed sources while maintaining stream base flow in the Charles and Mill Rivers above minimum flow requirements with active management of withdrawals. Using a minimum-streamflow requirement of 0.21 $(ft^3/s)/mi^2$, management model solutions indicated that annual average withdrawals from existing and proposed sources in the Populatic Pond area could be about doubled. The minimum streamflow constraints were applied on a monthly average basis, however, so that mean daily flows below the specified levels would likely occur in average years. Management-model solutions also indicated that it was possible to increase or maintain existing withdrawals with higher minimum-streamflow requirements, but that 1989–98 withdrawals could not be maintained for minimum flow requirements above 0.43 $(ft^3/s)/mi^2$. A second set of management models indicated that it might be possible to increase withdrawals by about 60 percent, while allowing no further reductions in stream base flow in the Charles and Mill Rivers, by managing withdrawals jointly between two towns in the Populatic Pond area. Withdrawals could be increased by about 25 percent if sources for the two towns are managed separately in these scenarios. Finally, a third set of management models indicated that base flow during the low-flow period in the Charles River might be increased without reducing withdrawals. The result was obtained by eliminating pumping in wells along the river in summer months, increasing pumping at these wells in the winter, and increasing pumping at wells farther from the river in summer months. This solution again was improved by collaborative management of sources between towns. This analysis indicates that the simulation-optimization approach could be useful for water management in areas of the upper Charles River Basin and elsewhere in Massachusetts where multiple water-supply and water-resource needs coexist.

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