Simulations of Ground-Water Flow and Residence Time near Woodbury, Connecticut

By J. Jeffrey Starn and Craig J. Brown

National Water-Quality Assessment Program
Transport of Anthropogenic and Natural Contaminants (TANC) to Public-Supply Wells

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Foreword

The U.S. Geological Survey (USGS) is committed to providing the Nation with credible scientific information that helps to enhance and protect the overall quality of life and that facilitates effective management of water, biological, energy, and mineral resources (*http://www.usgs.gov/*). Information on the Nation's water resources is critical to ensuring long-term availability of water that is safe for drinking and recreation and is suitable for industry, irrigation, and fish and wildlife. Population growth and increasing demands for water make the availability of that water, now measured in terms of quantity and quality, even more essential to the long-term sustainability of our communities and ecosystems.

The USGS implemented the National Water-Quality Assessment (NAWQA) Program in 1991 to support national, regional, State, and local information needs and decisions related to water-quality management and policy (*http://water.usgs.gov/nawqa*). The NAWQA Program is designed to answer: What is the condition of our Nation's streams and ground water? How are conditions changing over time? How do natural features and human activities affect the quality of streams and ground water, and where are those effects most pronounced? By combining information on water chemistry, physical characteristics, stream habitat, and aquatic life, the NAWQA Program aims to provide science-based insights for current and emerging water issues and priorities. From 1991–2001, the NAWQA Program completed interdisciplinary assessments and established a baseline understanding of water-quality conditions in 51 of the Nation's river basins and aquifers, referred to as Study Units (*http://water.usgs.gov/nawqa/studyu.html*).

Multiple national and regional assessments are ongoing in the second decade (2001–2012) of the NAWQA Program as 42 of the 51 Study Units are reassessed. These assessments extend the findings in the Study Units by determining status and trends at sites that have been consistently monitored for more than a decade, and filling critical gaps in characterizing the quality of surface water and ground water. For example, increased emphasis has been placed on assessing the quality of source water and finished water associated with many of the Nation's largest community water systems. During the second decade, NAWQA is addressing five national priority topics that build an understanding of how natural features and human activities affect water quality, and establish links between *sources* of contaminants, the *transport* of those contaminants through the hydrologic system, and the potential *effects* of contaminants on humans and aquatic ecosystems. Included are topics on the fate of agricultural chemicals, effects of urbanization on stream ecosystems, bioaccumulation of mercury in stream ecosystems, effects of nutrient enrichment on aquatic ecosystems, and transport of contaminants to public-supply wells. These topical studies are conducted in those Study Units most affected by these issues; they comprise a set of multi-Study-Unit designs for systematic national assessment. In addition, national syntheses of information on pesticides, volatile organic compounds (VOCs), nutrients, selected trace elements, and aquatic ecology are continuing.

The USGS aims to disseminate credible, timely, and relevant science information to address practical and effective water-resource management and strategies that protect and restore water quality. We hope this NAWQA publication will provide you with insights and information to meet your needs, and will foster increased citizen awareness and involvement in the protection and restoration of our Nation's waters.

The USGS recognizes that a national assessment by a single program cannot address all water-resource issues of interest. External coordination at all levels is critical for cost-effective management, regulation, and conservation of our Nation's water resources. The NAWQA Program, therefore, depends on advice and information from other agencies—Federal, State, regional, interstate, Tribal, and local—as well as nongovernmental organizations, industry, academia, and other stakeholder groups. Your assistance and suggestions are greatly appreciated.

Robert M. Hirsch
Associate Director for Water

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Conversion Factors, Abbreviations, and Datums

SI to Inch/Pound

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
meter (m)	3.281	foot (ft)
	Area	
square kilometer (km^2)	247.1	acre
hectare (ha)	2.471	acre
hectare (ha)	0.003861	square mile (mi^2)
	Volume	
cubic meter (m^3)	264.2	gallon (gal)
	Flow rate	
meter per day (m/d)	3.281	foot per day (ft/d)
centimeter per year (cm/yr)	0.3937	inch per year (in./yr)
cubic meter per second per square kilometer [$(m^3/s)/km^2$]	91.49	cubic foot per second per square mile [$(ft^3/s)/mi^2$]
cubic meter per day (m^3/d)	35.31	cubic foot per day (ft^3/d)
cubic meter per day (m^3/d)	264.2	gallon per day (gal/d)
gallon per minute (gal/min)	0.06309	liter per second (L/s)
	Load	
gram per day (g/d)	0.002204	pounds per day (lbs/d)
	Hydraulic conductivity	
meter per day (m/d)	3.281	foot per day (ft/d)

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

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

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

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

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Abstract

Water withdrawn for public use from glacial stratified deposits in Woodbury, Connecticut, is a mixture of water from different source areas, each having a characteristic water-quality signature. The physical processes leading to this mixture were explored using a numerical model to simulate steady-state ground-water source areas and residence times for a public water-supply well (PSW-1) in Woodbury. Upland areas contribute water to the well that is primarily from undeveloped and agricultural land. Valley bottoms contribute water to the well that is primarily from developed land. From 1985 to 2002, 6 percent of the contributing recharge area to the well changed from agricultural and undeveloped to developed land. The pattern of recharge areas and land use causes stratification of ground water by residence time and by characteristic water quality, which is related to land use. As land use changes with time, the water-quality signature of developed land moves deeper into the aquifer. Predicted nitrate concentrations decreased from 1985 to 1995 because of the conversion from agricultural land to developed land, but then began to increase after 1995 because of the conversion of undeveloped land to developed land. Total dissolved solids concentrations, on the other hand, increased from 1985 to 2002 because agriculture is associated with lower total dissolved solids concentrations than is developed land.

About 40 percent of the water withdrawn from PSW-1 originated as upland recharge before flowing through glacial deposits in the valley. About 44 percent of the water originated as recharge in either fluvial deposits (mean residence time 7 years) or deltaic deposits (mean residence time 4 years). About 16 percent of the water originated as recharge through storm drains with ground-water discharge (often known as "dry wells"). The residence time for water that originated as recharge in dry wells is 2 to 4 years, and the mean residence time is 3 years. Dry wells are a fast pathway for water to enter the aquifer and provide a significant amount of water to PSW-1; therefore, PSW-1 is more susceptible to contamination in runoff from the commercial area, which enters the dry wells, than to recharge elsewhere in the area.

Water withdrawn from a well is a mixture of waters with different residence times, and a single residence time does not fully characterize the susceptibility of the well to recent contamination. The mean simulated flow-weighted residence time in PSW-1 is 6 years, which compares reasonably well with the apparent residence time measured using tritium/helium data of 6 and 7 years (samples for age dating were collected twice from this well). There are at least two modes to the distribution of ages, one mode with residence times less than 5 years and one mode with residence times greater than 5 years. About 34 percent of the ground-water in PSW-1 is younger than 5 years and 56 percent of the water is from 5 to 9 years.

The estimated nitrate loading rate from a single-family septic system is 18 grams per day. If each household in the contributing recharge area contributes nitrate at that loading rate to the well PSW-1, each additional septic system in the contributing recharge area is responsible for a 0.045-milligram-per-liter increase in nitrate at PSW-1 at the current pumping rate.

Uncertainty in the predicted contributing recharge area can be propagated through the analysis using a Monte Carlo technique. There is a greater degree of certainty in the delineation of the recharge area near the well, and as one moves from the well toward the recharge areas, the uncertainty in the model increases. The area that possibly contributes water to the well using the Monte Carlo model is much larger than the recharge area delineated using the optimal parameter estimates. Within the probabilistic recharge area, the number of septic systems could be twice the number initially estimated.

Introduction

Aquifers in Connecticut are a valuable natural resource and a major source of public drinking water. Reliance on ground water is expected to increase because opportunities to develop new surface-water supplies are diminishing due to the rising cost of land and increasing development (Connecticut General Assembly, 2005). In Connecticut, about 11 percent of the public water supply is from ground water; this, combined with the fact that almost all self-supplied water is from residential wells, means that a total of 32 percent of the state's drinking-water supply is from ground water. In addition, many schools, small businesses, churches, restaurants, and camps rely on ground water (about 165,000 people; sum of transient and non-transient non-community water systems; Connecticut Department of Public Health, 2007a).

Although population growth in Connecticut has been slow in the past decade, the changing distribution of people has implications for the quality of ground water. Connecticut's population grew from 3.3 million in 1990 to 3.4 million in 2000 (Connecticut State Comptroller, 2007). This 3.6-percent population gain was the fourth lowest among all states; however, the modest gain was not uniformly distributed. The current trend in Connecticut is toward decreasing population in cities and increasing population in more recently developed areas. For example, the projected growth rate for 2005 to 2015 for suburban areas is twice that for urban areas (University of Connecticut, 2007a). The state's five largest cities had an overall population decline of 5.5 percent in 2002, whereas 58 towns with an average population of 11,000 grew by more than 10 percent during the last decade (Connecticut State Comptroller, 2007). Rural and suburban areas are less likely to be served by large public-water suppliers and are more likely to rely on small water suppliers or on self-supplied water, both of which are more likely to use ground water as the source of drinking water.

As a result of the redistributed population, the land cover of the state is changing. From 1985 to 2002, high-density development, such as building, parking lots, and roads, increased by 4.9 ha per day while forest cover decreased by 7.3 ha per day (University of Connecticut, 2007b). Developed land expanded by 308 km^2 and forest land decreased by 440 km^2 during that time period. Developed land increased by 15 percent from 1985 to 2002, roughly twice the rate of population increase. The density of new development is lower than in the past so more land per person is required (University of Connecticut, 2007b). In 2002, 56 percent of the state was forested and 19 percent was developed.

The link between land use and the quality of shallow ground water has been well documented (Grady, 1994; Grady and Mullaney, 1998). In particular, there are differences in water quality in shallow ground water beneath undeveloped areas, tilled and untilled agricultural areas, sewered and unsewered residential areas, and commercial and industrial areas (Grady, 1994). Contaminants have been detected in 52 percent of all public drinking-water sources in Connecticut, but the Maximum Contaminant Levels (MCLs) for drinking water established by the U.S. Environmental Protection Agency are rarely exceeded (with the exception of microbiological contaminants). Although less attention has been paid to the relation between microbiological contamination and land cover, a total of 454 MCL violations were issued to public-water systems during calendar year 2005, of which 393 were for microbiological contamination. Nitrate was the prevalent non-microbiological contaminant detected in public-supply wells. The MCL for nitrate is 10 mg/L (reported as nitrogen concentration). Samples from 36 percent of all community water systems contained nitrate concentrations from 1 to 10 mg/L in 2005. Eight percent of systems had detections of volatile organic compounds (VOCs) in 2005. The most commonly detected VOCs in 2005 were methyl *tert*-butyl ether (MTBE) and trichloroethylene (TCE) (Connecticut Department of Public Health, 2007b). Other contaminants that were found above their MCL in 2005 include net gross-alpha radiation, radium 226, radium 228, uranium, chloride, total haloacetic acids, and total trihalomethanes (Connecticut Department of Public Health, 2007c). Some of these same contaminants (nitrate, VOCs, and trihalomethanes) in shallow ground water are related to land use (Grady, 1994).

There is a cost of treatment and remediation associated with high levels of organic contaminants in ground water. For example, removal of tetrachloroethylene from ground water to levels below the MCL can cost from $11 to $73 per household per year for medium-sized water systems (Connecticut Department of Public Health, 2007c), and the cost per household would be higher for smaller systems. In 2005, 293 systems in 149 towns had public-supply wells in which organic contaminants were detected that were either below the MCL or for which there was no MCL. Sixty-seven public drinking-water systems in the state in 2005 were treating their water to remove organic contaminants. After this treatment, all 67 suppliers provided drinking water that meets all legal standards (Connecticut Department of Public Health, 2007c). Of the organic chemicals detected in Connecticut's public drinking-water supplies, benzene, bromodichloromethane, carbon tetrachloride, chloroethane, chloroform, 1,2-dichloroethane, dichloromethane, 1,2-dichloropropane, naphthalene, and TCE are considered known or probable carcinogens, and 1,1-dichloroethane, p-dichlorobenzene, MTBE, 1,1,2-trichloroethane are considered possible human carcinogens (Connecticut Department of Public Health, 2007c).

Connecticut has a comprehensive and coordinated system of land-use regulation that includes provisions to protect public-drinking water in wells that tap glacial stratified deposits. The provisions, administered by the Connecticut Departments of Public Health (DPH) and Environmental Protection (DEP), identify vulnerable ground water and protect ground-water supplies. A source-water assessment, conducted by the Connecticut Department of Public Health in 2003, revealed that 20 percent of public-supply wells that serve more than 1,000 people ranked as highly susceptible to contamination. These wells were considered susceptible in part because they are in an area of moderate to high density of potential contaminant sources, are in an area of high-density development, and have a previous history of the detection of contaminants (Connecticut Department of Public Health, 2007b). Sixty percent of the potential contaminant sources that were identified in the assessment involve fuel storage or automotive-related activity.

One way to mitigate the effects of contamination is to protect the source of the water. In Connecticut, the DEP manages the Aquifer Protection Program. Preliminary aquifer-protection areas have been delineated at all 122 active community well fields. Final aquifer-protection mapping, conducted using techniques similar to those used in this study, is complete at 27 well fields, with an additional 64 delineations underway as of March 2007 (Connecticut Department of Environmental Protection, 2007). Land use will be regulated within the aquifer-protection areas. Regulated activities include businesses that use hazardous materials, pesticides, and petroleum products. These businesses include some manufacturing industries, chemical wholesale storage facilities, gasoline stations, automobile service stations, dry cleaners, and furniture strippers (Connecticut Department of Environmental Protection, 2007).

In 2001, the U.S. Geological Survey's National Water-Quality Assessment (NAWQA) Program began an intensive study to assess the vulnerability of public-supply wells to contamination from a variety of compounds (Eberts and others, 2005). This study, referred to as the TANC study—for Transport of Anthropogenic and Natural Contaminants—builds on previous NAWQA studies from 1991 to 2001 that found low levels of multiple contaminants in about 90 percent of samples from shallow ground-water-monitoring wells in urban areas across the Nation. One goal of the TANC study is to synthesize and compare data and simulation results among TANC study areas in Woodbury, Connecticut; Modesto, California; Tampa, Florida; and York, Nebraska. To perform this analysis across multiple scales, the TANC study design called for development of nested simulation models of ground-water flow. A Large-Area Simulation (LAS) model of ground-water flow in the aquifer in the Pomperaug River drainage basin (Lyford and others, 2007) was developed to estimate areas contributing recharge and ground-water residence times to many public water-supply wells over a 128-km^2 area in the Pomperaug River Basin, Connecticut (fig. 1). A Small-Area Simulation (SAS) model within the LAS boundaries is a more detailed model (fig. 1).

Purpose and Scope

This report documents the SAS model in the TANC study at Woodbury, Connecticut. The results presented here will be the basis for comparing this study area with other TANC study areas; those areas and study results will be reported elsewhere. This report describes (1) the approaches used to refine the LAS model developed by Lyford and others (2007), (2) apparent ground-water residence times using the tritium/helium method, (3) construction and calibration of the SAS model used to simulate ground-water residence time, and (4) results from the SAS model and a comparison of the measured and simulated ground-water residence times. Details of the water quality in the study area will be discussed in a separate report (C.J. Brown, U.S. Geological Survey, written commun., 2007).

The study area discussed in this report is the estimated contributing recharge area to one public water-supply well (referred to herein as well PSW-1). In order to minimize the effects of the model boundaries on the conclusions drawn from the SAS model, ground-water flow was simulated in an area larger than the contributing recharge area. This area is referred to herein as the SAS area. Data collection was focused on a smaller area within the SAS area in and around the contributing recharge area to well PSW-1.

Description of Study Area

The SAS area is in the Connecticut, Housatonic, and Thames River Basins NAWQA Study Unit (fig. 1). Characteristics of the aquifer system selected for this study are similar to those of many other valley-fill-aquifer systems in the Eastern Hills and Valley Fills region, defined and described by Randall (2001), which encompasses much of southern New England, northern New Jersey, and eastern New York.

The SAS area covers about 15 km^2 of the Pomperaug River Basin in the west-central Connecticut towns of Woodbury and Southbury. The major river valley trends north-south and is bounded on the east and west by till-covered bedrock uplands drained by numerous perennial streams. Streams in upland areas are oriented mostly from east to west on the eastern side and northwest to southeast on the northern and western sides. Ponds have been constructed on several tributary streams. Altitudes range from about 30 m near the confluence of the Pomperaug River with the Housatonic River to about 280 m at places on the basin divide.

Precipitation in the Pomperaug River Watershed averages about 117 cm/yr (Randall, 1996). Basin runoff measured in the Pomperaug River at Southbury, Connecticut, has averaged 60 cm/yr from 1933 to 2001 (Morrison and others, 2006). The balance of about 57 cm/yr is lost mainly to evapotranspiration or is transferred out of the basin for use as drinking water in other basins.

Changes in land use alter the quality and availability of water, so it is useful to have some understanding of how land use and population have changed in the past. This is especially true for ground water, which can have a long residence time in the environment, thus carrying the imprint of past land use. The population of Woodbury was 2,150 people in 1850 and 2,564 in 1950, an increase of 41 people per decade (U.S. Census Bureau, 2007a). From 1950 to 1990, the population grew steadily to 8,131 people, an average increase of 1,390 people per decade. From 1990 to 2000, the rate of population growth decreased to about 1,065 people per decade. The rate of growth for Woodbury is projected to decrease slightly from 2000 to 2020 (Connecticut Office of Policy and Management, 2007).

Land use in the Pomperaug River Watershed has changed over the past 50 years from primarily undeveloped or agricultural lands to residential, commercial, and light-industrial areas. An 1822 map of the study area shows that the SAS area was then predominantly agricultural or undeveloped, with two churches, one saw mill, one fulling mill (for processing cloth), a grist mill, and a small number of houses. A 1934 aerial photograph shows that the valley bottom in the SAS area was predominantly agricultural. Uplands were a mixture of

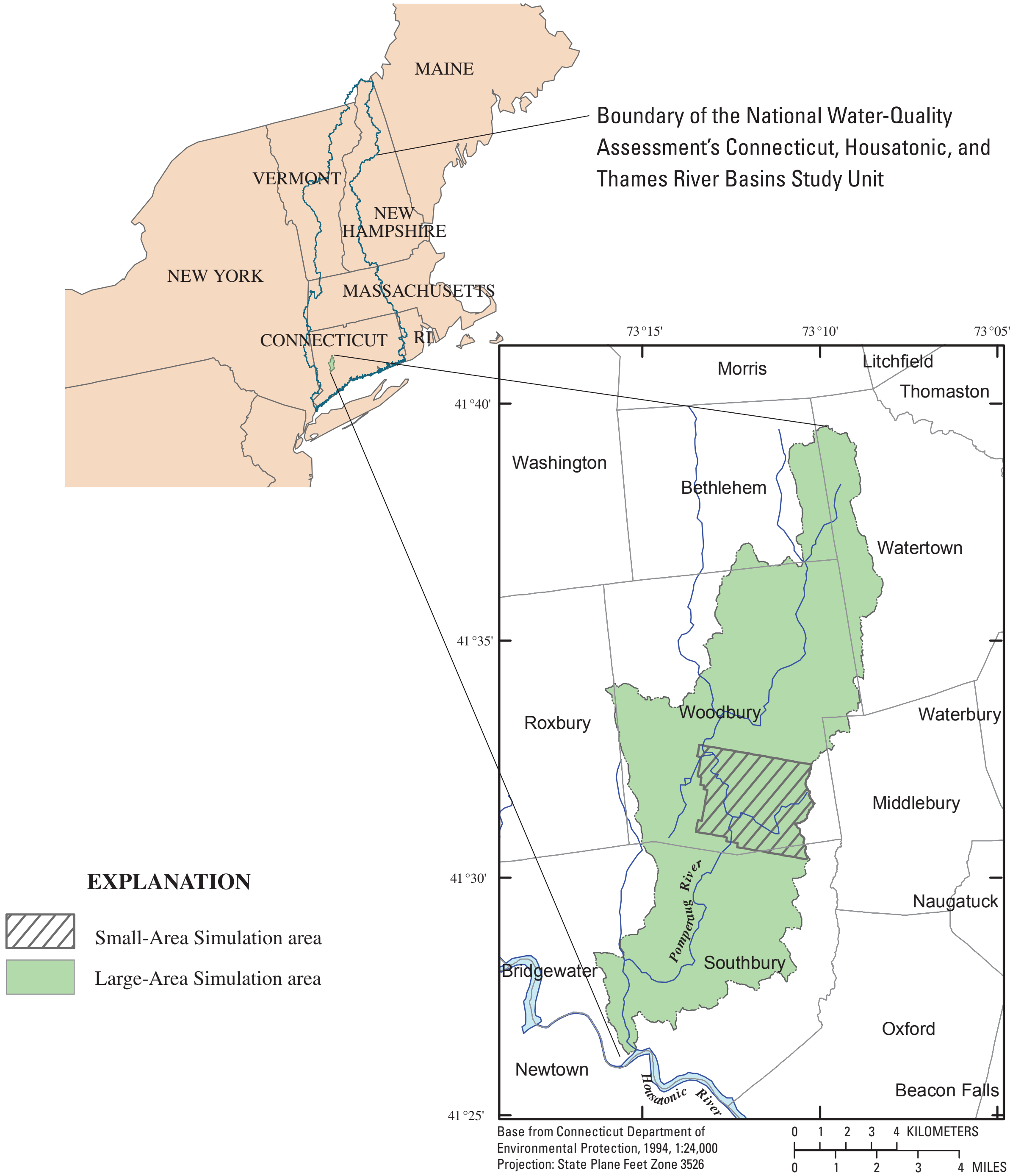


Figure 1. Location of large-area simulation and small-area simulation models, Woodbury, Connecticut.

wooded and partially cleared land that may have been pasture. From 1985 to 2002, Woodbury lost about 196 ha of deciduous forest (out of a town area of 9,510 ha) to development and land clearing (which may be a precursor to development; University of Connecticut, 2003). In 2002, the SAS area was more developed than the rest of the town (16 and 10 percent, respectively; fig. 2), but the patterns of land use were similar (commercial development primarily in the valley, residential primarily in the uplands). Only 12 percent of the SAS area was used by agriculture (fig. 2).

Currently (2006), residential areas are unsewered and are characterized by low- to medium-density housing (Lyford and others, 2007). Agricultural lands are mostly in the upland areas. Industrial uses are limited and include small, modern, high-tech industries. Upland areas are largely forested with scattered residences on 0.4 ha or larger lots. Most water for public supply is obtained from wells completed in valley fill, although four condominium complexes in the uplands obtain water from wells completed in bedrock. Numerous residents in the valley and uplands obtain water from private wells for domestic uses, including lawn irrigation. Most of the water pumped from wells is used within the SAS area. Wastewater is returned to the ground through private septic systems and on-site treatment facilities.

Previous Investigations

Ground-water conditions in the Pomperaug River Watershed have been described by Meinzer and Stearns (1929), Mazzaferro (1986a), Mazzaferro (1986b), Grady and Weaver (1988), Starn and others (2000), and Lyford and others (2007). In addition, the surficial geology has been described by Pessl (1970), and much work that has been done on the bedrock geology is summarized by Burton and others (2005). The work by Lyford and others (2007) describes the LAS model that is the basis for the SAS model in this report.

Study Methods

The SAS model of ground-water flow was constructed and calibrated using data from a monitoring-well network that was installed for this study. Advective particle tracking was used to estimate contributing recharge areas to monitoring wells and water-supply wells and to estimate the distribution of apparent ground-water residence time.

Design of Monitoring Well Network

The design, installation, and sampling of the monitoring-well network (figs. 3 and 4) will be described in detail in a separate report (C.J. Brown, U.S. Geological Survey, written commun., 2007). Selection of monitoring well locations (except for wells WY35 and PSW-1, which existed prior to this study) was based on a preliminary simulation model, the goal of which was to identify a ground-water-flow path in the contributing recharge area to PSW-1 (figs. 3 and 4). Subsequent uncertainty analysis showed that there was a good probability that the source area to PSW-1 included an area off the chosen flow path. Additional wells were drilled to include the off-flow-path areas. In most locations, a nest of two to three wells was installed so that different levels within the aquifer could be monitored and sampled. At some locations, bedrock wells were drilled to characterize the contribution of flow from bedrock to the water produced from well PSW-1.

Development and Application of Simulation Models

Steady-state ground-water flow was simulated using the computer program MODFLOW-2000 (Harbaugh and others, 2000; Hill and others, 2000), which uses a finite-difference method to simulate three-dimensional ground-water flow through a porous medium. MODFLOW-2000 uses a non-linear regression technique to estimate parameter values that result in the best match between observed and simulated values. This model, when combined with boundary and initial conditions, describes three-dimensional ground-water flow in a heterogeneous and anisotropic medium, provided that the principal axes of hydraulic conductivity are aligned with the coordinate directions. Source-water areas and ground-water residence times were delineated using the particle-tracking computer program MODPATH (Pollack, 1994).

Geohydrologic Setting

The ground-water-flow system in the glaciated Northeast is composed primarily of two materials: bedrock and glacial deposits. Although there are large variations of hydrologic properties within each material, the greatest difference with regard to ground-water flow is between them. Ground-water flow in bedrock takes place in fractures in the rock, and flow in the glacial deposits is through the pore spaces between mineral grains.

Geology

Bedrock underlies the entire study area and is one of three types: crystalline rock (primarily gneiss and schist), basalt, or sedimentary rock (primarily arkose and shale). Glacial deposits overlie bedrock everywhere except locally where bedrock crops out at the surface. Glacial deposits are either till or glacial stratified deposits.

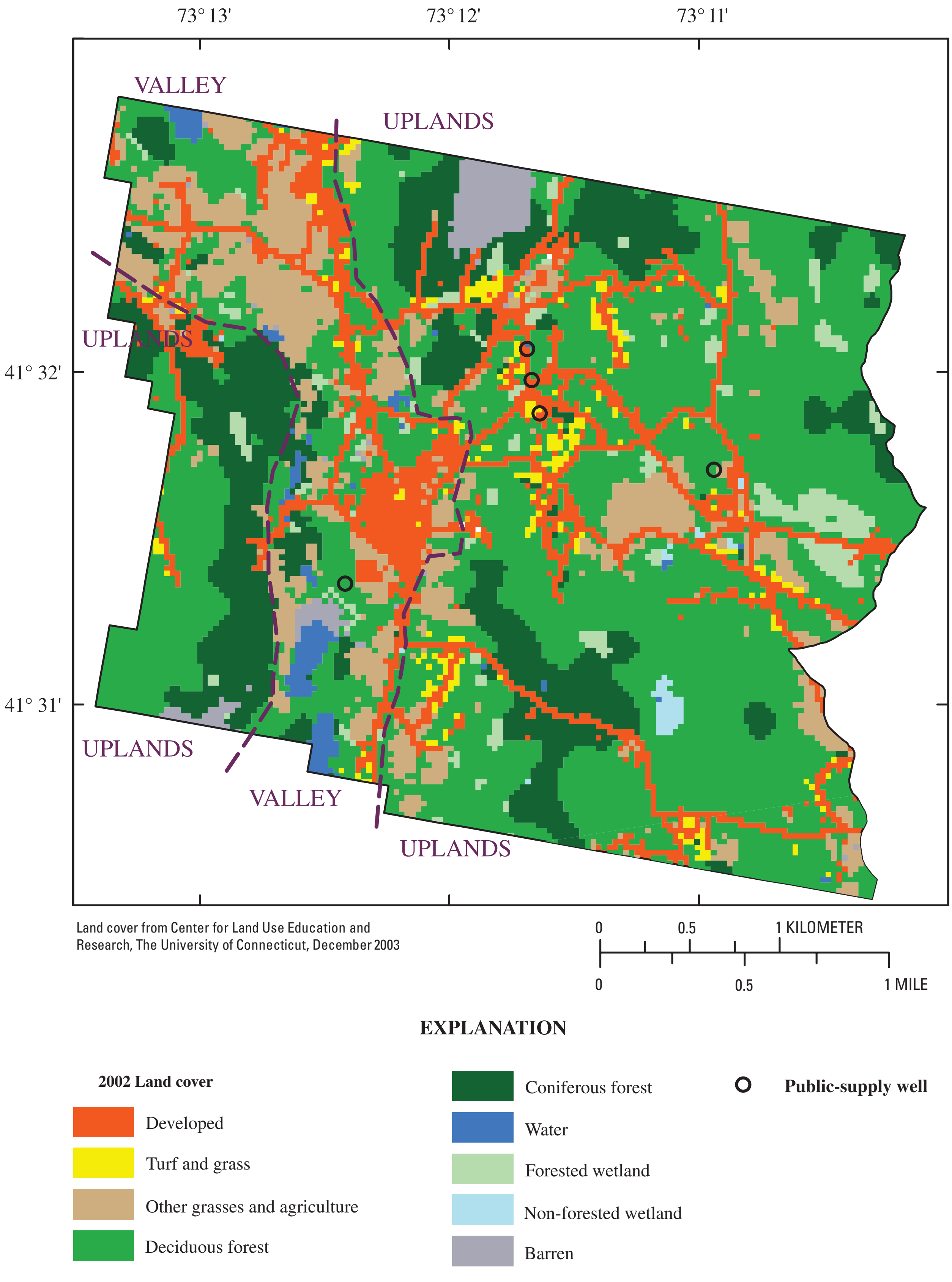


Figure 2. Land cover and public-supply wells in 2002 in part of Woodbury, Connecticut.

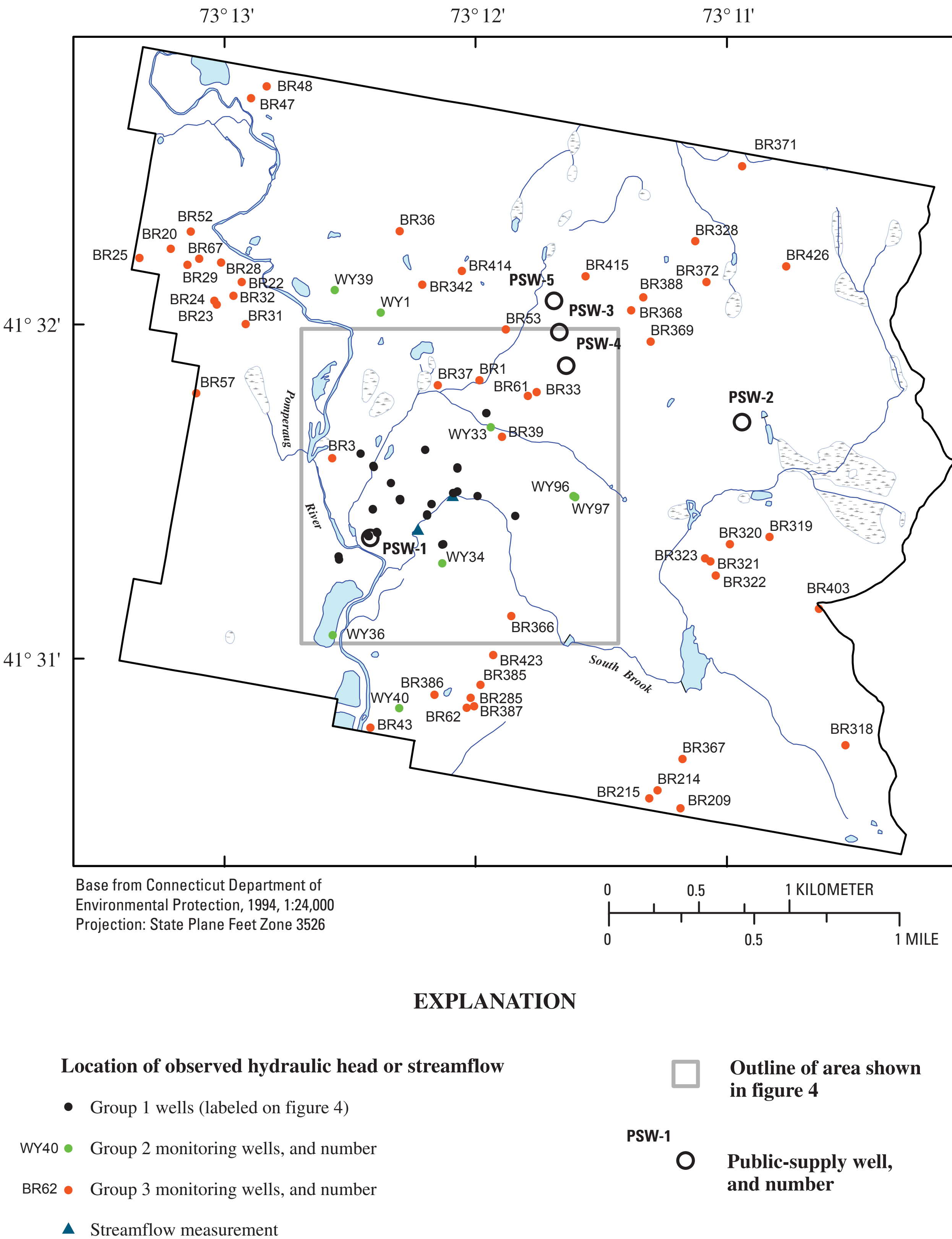


Figure 3. Locations of all observation and monitoring wells used in this study, Woodbury, Connecticut.

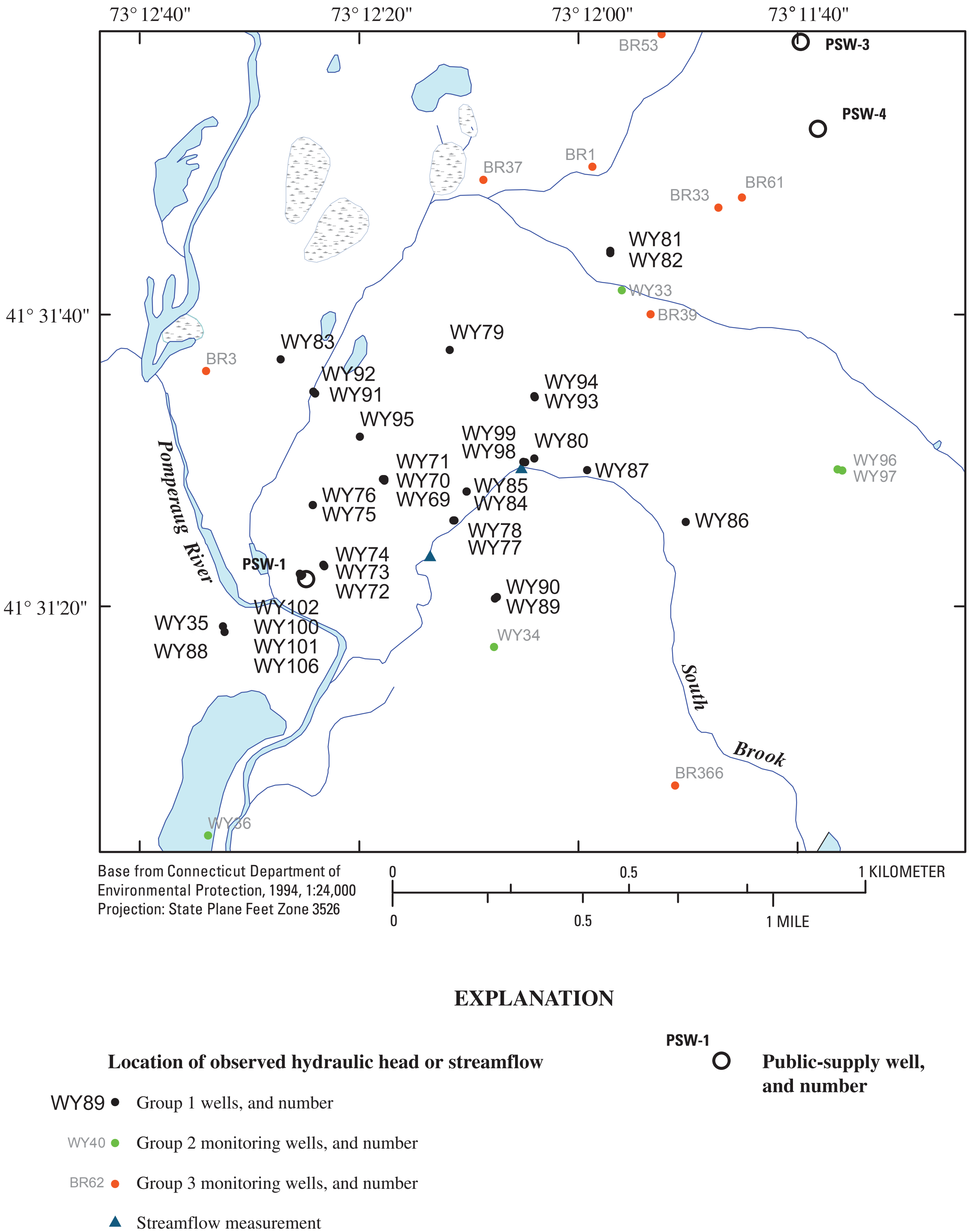


Figure 4. Locations of monitoring wells installed for this study, Woodbury, Connecticut.

Bedrock

Geologic mapping by Burton (2006) identified bedrock in the SAS area of two distinct ages: early Paleozoic and Mesozoic. Early Paleozoic rocks are present primarily in the uplands on the eastern side of the SAS area and consist of schist and gneiss of the Taine Mountain Formation and Waterbury Gneiss. The Mesozoic and Paleozoic rocks are separated by a fault that runs generally north-south and passes between wells WY86 and WY87. Paleozoic rocks close to the fault are highly fractured, but the degree of fracturing likely decreases away from the fault. These fractures are nearly vertical and tend to parallel the fault. The Mesozoic Orenaug Basalt underlies most of the Pomperaug River valley in the study area. Fractures in the Orenaug Basalt tend to be mineralized with calcite, which decreases their permeability; however, at least five near-vertical, north-south faults in the Orenaug Basalt may increase its permeability. Another basalt unit, the South Brook Basalt, is highly fractured and may be more permeable than the surrounding rocks. Mesozoic rocks also include fractured arkose and shale.

Starn and Stone (2005) identified three types of bedrock fractures in Connecticut. The first type—vertical fractures—provides the pathways for ground-water flow vertically into deeper fractures. The second type of fractures comprise nearly horizontal stress-relief fractures that generally diminish in number and aperture with depth. Ground water can potentially flow in any direction in this type of fracture. The third type of fractures cross-cuts vertical fractures and may connect them, but generally is not as laterally continuous as layer-parallel fractures.

The bedrock surface in the SAS area is characterized by two low areas separated by a buried ridge (figs. 5 and 6) of what probably is a resistant layer in the Orenaug Basalt (Burton and others, 2005). Orenaug Basalt is exposed in the streambed of the Pomperaug River and at several hillside locations near the village of Pomperaug. The buried ridge itself is bounded by north-south trending faults. This buried ridge probably provided sufficient resistance to the flow of glacial ice so that the retreating ice margin remained at this position for some time. The low area on the bedrock surface to the northwest of the buried ridge is underlain by easily eroded sedimentary rock. The low area on the bedrock surface to the southeast of the buried ridge is underlain, at least in part, by easily eroded sedimentary rock and basalt. Faulting also could contribute to the low bedrock altitude in this area.

Glacial Deposits

Two major types of glacial deposits are in the SAS area: till and glacial stratified deposits. The differences in composition and distribution of these two types stem from their mode of deposition. Till was deposited directly by glacial ice; is generally nonlayered and nonsorted; and contains a wide range of grain sizes from clay to large boulders. Glacial stratified deposits were laid down by glacial meltwater and are composed of well to poorly sorted layers of sediment including gravel, sand, silt, and clay. Till blankets the bedrock surface in most places in the uplands and commonly is present beneath glacial stratified deposits in valleys. Glacier meltwater was concentrated in valleys as the ice margin retreated; therefore, glacial stratified deposits occur predominantly in valleys.

Till deposited by glacier ice is a nonlayered, nonsorted silty sand or clayey silt-sand mixture containing 5 to 40 percent pebbles, cobbles, and boulders (Melvin and others, 1992). Till deposits in the SAS area include several types: (1) compact till of subglacial origin of Illinoian age that is the predominant material in glacially smoothed hills known as “drumlins,” (2) compact to loose sandy late Wisconsinan-age till that overlies the lower till in drumlins; and (3) a loose sandy, surface till of late-Wisconsinan glacial ablation (melt) origin (Stone and others, 2005). Drumlin till is moderately to very compact and probably has a lower hydraulic conductivity than the overlying sandy till because of compaction beneath the glacial ice. It commonly is finer grained and less stony than the sandy till. The compact till in drumlins is as much as 51 m thick in the SAS area and is generally covered by a thin (less than 4–5 m thick) veneer of sandy till (Stone and others, 2005). The lower part of the sandy till is compact due to its deposition beneath glacial ice. Compact sandy till commonly is found on north-facing slopes of bedrock hills where it may be as much as 10 m thick. The upper part of the sandy till is non-compact, loose, bouldery surface till that forms a thin (less than 2 m thick) discontinuous veneer overlying compact sandy till and bedrock.

The laboratory-determined hydraulic conductivity of till has a strong relation to grain size (Stephenson and others, 1988, p. 306), particularly to the clay content. Hydraulic conductivities are uniformly low above a threshold content of 15 to 20 percent clay. The clay content of sandy late-Wisconsinan tills in southern New England ranges from less than 1 percent to 7 percent, whereas the content of clay-sized particles in drumlin till ranges from 11 to 38 percent (Melvin and others, 1992).

The distribution of glacial stratified deposits in the SAS area fits general models of glacial sedimentation in southern New England described by Stone and others (2005) and by Randall (2001). Glacial stratified deposits consist of overlapping deposits (morphosequences) that begin at an ice-margin position and grade to the water levels in a glacial lake or pond downstream from the ice. Morphosequences in the SAS area include collapsed ice-marginal material deposited in kame terraces (Pessl, 1970), and, on the basis of drilling done as part of this study, these deposits overlie compact till. Deltaic deposits in the center of the valley are remnants of an ice-marginal delta or subaqueous fan that was deposited in glacial Lake Pomperaug. The delta or fan is referred to herein as a delta for simplicity. Deltaic deposits consist of well-sorted, layered, non-collapsed coarse and medium sand (Pessl, 1970). Collapsed ice-marginal fluvial deposits in the delta extend to the northeast, where they are overlain by deltaic deposits of a younger morphosequence. The surface

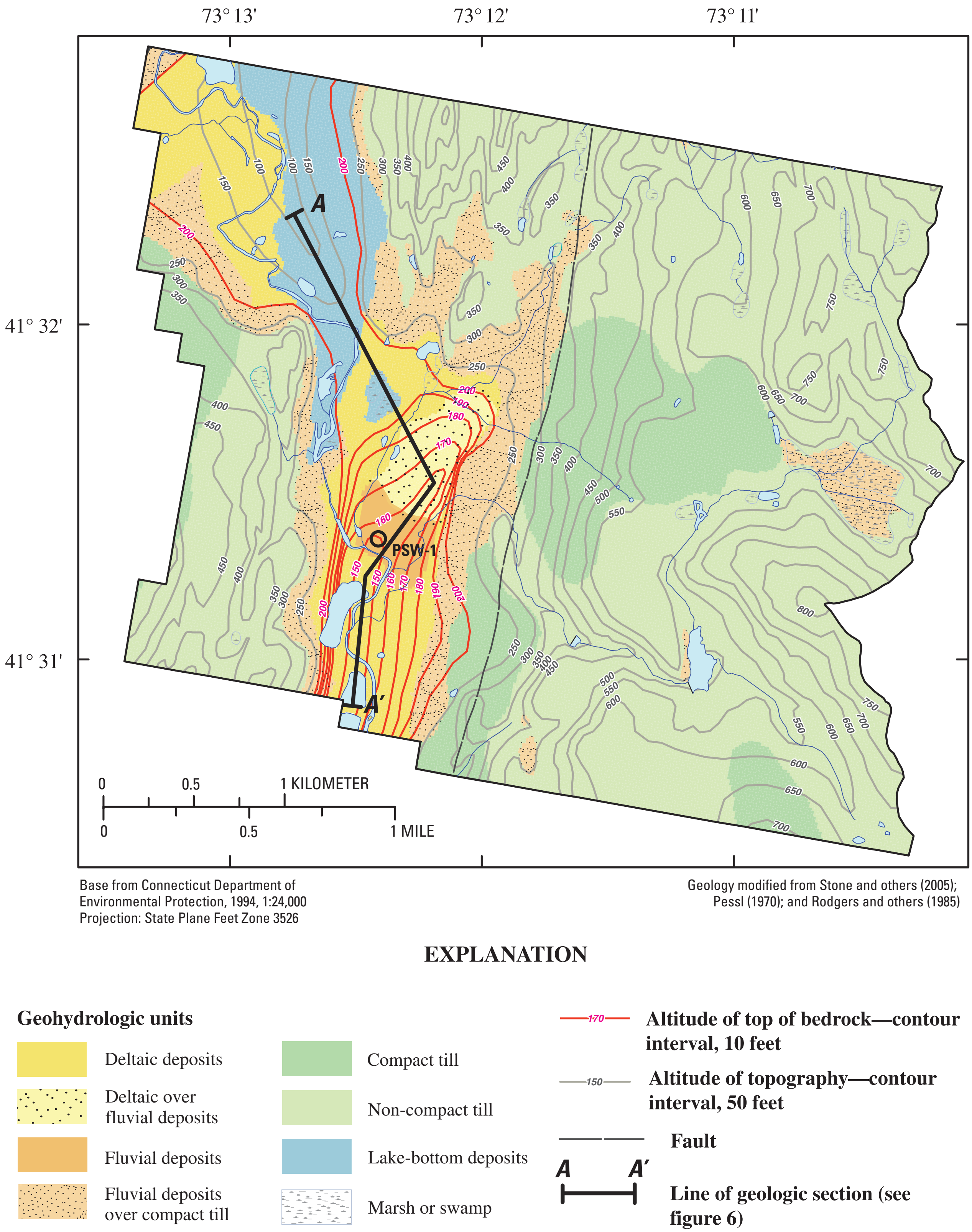


Figure 5. Bedrock contours and surficial geology, Woodbury, Connecticut.

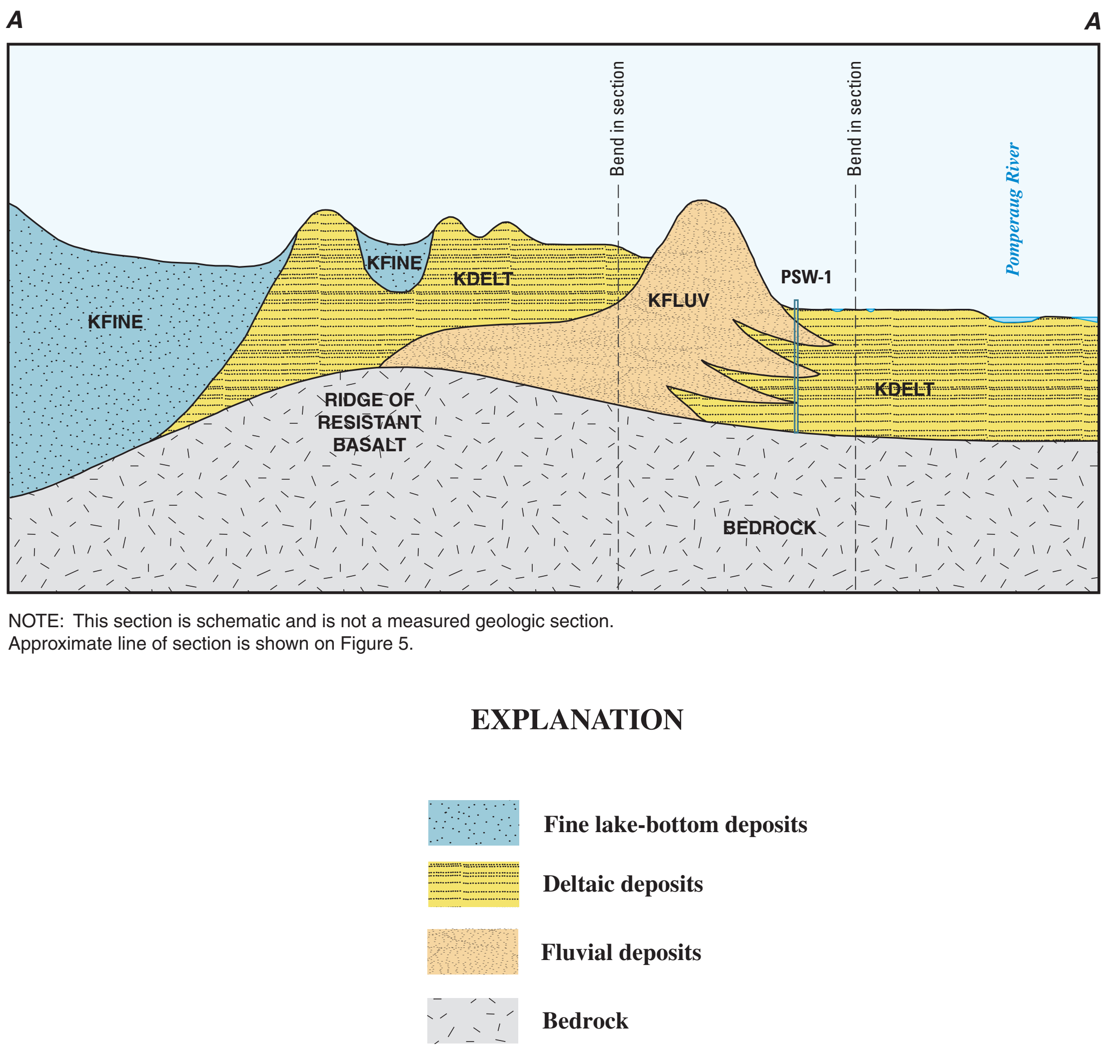


Figure 6. Geohydrologic units, Woodbury, Connecticut.

of the younger deposits is at a higher altitude than the surface south of the delta, indicating that these surfaces were graded to different lake levels (fig. 6). The delta was mined for sand and gravel sometime between 1970 and 1985 and no longer exists as a landform, but the lower stratigraphic section of the delta still remains. Fine-grained lake-bottom sediments are present in the northwestern part of the SAS area.

Hydrology

As discussed previously, the Pomperaug River Watershed receives an average of 117 cm/yr of precipitation. About 57 cm/yr of that precipitation is lost to evaporation or is used by plants, and about 60 cm/yr is runoff that discharges to the Pomperaug River. The runoff component is a combination of direct surface runoff and ground-water recharge. The path of ground-water flow between recharge and discharge areas is influenced by the geologic material, the altitude and spacing of streams and ponds, and by the pattern of water use.

If water levels in a given area are relatively constant over a long period, the flow of water into an area is balanced by the flow of water out of the area. Water levels in well WY1 (in the SAS area) have been measured monthly since 1945 (fig. 7). Water levels in this well respond primarily to seasonal fluctuations in recharge caused by annual cycles of evapotranspiration. Smoothed water levels in WY1 show the effect of the multi-year drought in the 1960s and of a shorter drought in 2001 and 2002. Water levels may have increased slightly since 1985, but more data are needed to see if this is a persistent trend or a short-term phenomenon.

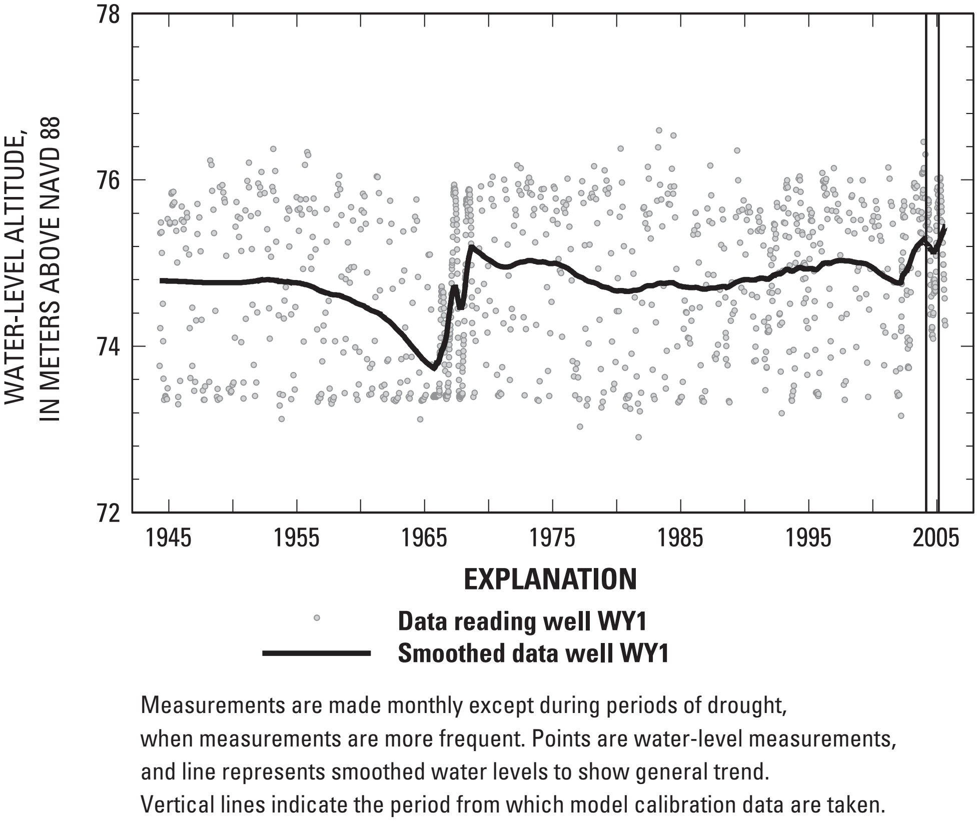


Figure 7. Monthly and smoothed water-level altitude in observation well WY1, Woodbury, Connecticut.

Sources of Ground Water

Sources of ground water are spatially and temporally variable throughout the SAS area. The sources of recharge include precipitation that infiltrates into the ground and flows through the unsaturated zone to the water table (referred to as "recharge"), streamflow that infiltrates the streambed, and ground water that flows into the SAS area from adjacent areas. Runoff from a commercial area flows into drains that directly recharge ground water ("dry wells").

The spatial distribution of recharge depends on surficial geology (till or glacial stratified deposits), land-surface slope, precipitation amount and rate, the areal coverage of impervious surfaces and poorly drained soils (Natural Resources Conservation Service Class D soils), and actual evapotranspiration (D.M. Bjerklie, U.S. Geological Survey, written commun., 2007). Precipitation amount, precipitation intensity, and evapotranspiration also influence temporal differences in the rate of recharge. Generally, there is more recharge in spring and fall than in summer and winter, but the SAS model uses only annual-average recharge as input. The rates of recharge are discussed in the section on model boundaries and stresses.

The conceptual model of recharge in the SAS model is different from that used by Lyford and others (2007), who based their estimates of recharge on two factors: type of glacial deposit (glacial stratified deposits or till) and land-surface slope. Slope was considered conceptually, not quantitatively, in dividing the amount of predicted recharge into areal subunits. In this method, all runoff is either direct runoff to surface water or recharge to ground water. In addition to these two components, water can move as shallow subsurface flow (D.M. Bjerklie, U.S. Geological Survey, written commun., 2007). Shallow subsurface flow is water that moves laterally on top of a low permeability layer, such as in till on top of bedrock or on top of a hardpan soil layer, but it may not represent recharge to the water table that would be available to a water-supply well. For this reason, the recharge estimated by a method that accounts for shallow subsurface flow is lower than that estimated by Lyford and others (2007) (D.M. Bjerklie, U.S. Geological Survey, written commun., 2007).

Ground-Water Flow

Ground water in the uplands flows through glacial till and (or) bedrock. The water moves through pore spaces between material grains in the till and flows laterally to glacial stratified deposits, to streams in till, and downward into fractures in the bedrock. Flow through fractures in the bedrock discharges laterally or upward into glacial stratified deposits in the valley and also directly to surface water where bedrock is exposed. Some ground water in fractures discharges to public water-supply wells and private residential wells.

Ground water in the valley flows through glacial till, bedrock, and glacial stratified deposits. Flow generally is upward from bedrock to glacial stratified deposits, passing through till where till is present beneath glacial stratified deposits. Water in the glacial stratified deposits is from precipitation, from losing reaches of streams, and from subsurface flow through till and (or) bedrock fractures. Ground water discharges to the Pomperaug River, to ponds formed in abandoned gravel quarries south of the study area, and to public water-supply well PSW-1.

Water Use

Water use constitutes about 4 percent of the total runoff in the SAS area. Ground water is withdrawn for human use through five public-supply wells and many residential wells (fig. 2). The largest withdrawal from the glacial stratified deposits from 1997 to 2001 was 391.7 m^3/d (72 gal/min) at public water-supply well PSW-1 (table 1). The water is supplied to customers in the Pomperaug valley. Withdrawals from PSW-1 showed an increase in summer over winter and a slight increasing trend from 1997 to 2001 (fig. 8). This increase in withdrawals is due to a small increase in population and to increasing water use by the existing population of Woodbury. Four other public-supply wells (PSW-2 through PSW-5) serve condominiums and apartments and withdrew a total of 233.9 m^3/d (43 gal/min) from bedrock (Lyford and others, 2007).

Table 1. Simulated pumping rates based on reported water withdrawals at public-supply wells in Woodbury, Connecticut.

[Data from Lyford and others (2007)]

Well	Pumping rate, in cubic meters per day
PSW-1	391.7
PSW-2	97.7
PSW-3	45.5
PSW-4	45.3
PSW-5	45.4

No data are available on actual residential water use, but a rough estimate can be made. The number of residences in the SAS area is estimated to be 517, the average number of people per household in Woodbury is 2.68 (U.S. Census Bureau, 2007b), and the average water use per person in Connecticut is 0.28 m^3/d (Hutson and others, 2005). On the basis of these values, the total residential water use in the SAS area is 388 m^3/d. The distribution of houses is fairly even in the uplands, so there is no area of concentrated pumping. An estimated 20 percent of water pumped is lost to evaporation (Solley and others, 1998), and 80 percent of the water pumped in the SAS area is returned to the ground through septic systems. The net effect of domestic wells is to redistribute water from bedrock fractures to shallow soils. More of the water in the shallow zone evaporates or flows to local streams than would be the case if the water remained in the bedrock. For comparison to the water-use figures, the annual average runoff in the Pomperaug River Watershed is 0.0189 m^3/s/km^2 (Morrison and others, 2006), so over the 15-km^2 area of the SAS area, the average annual runoff is 24,500 m^3/d.

Geohydrologic Units

Seven geohydrologic units were defined for input to the SAS model on the basis of data from monitoring wells drilled for this project (fig. 5; table 2). These data were interpreted in the context of previous published work in the area and by J.R. Stone (U.S. Geological Survey, oral communs., 2005 and 2006). The altitude of bedrock (fig. 5) was subtracted from the altitude of land surface to obtain the thickness of the glacial deposits, which was then used to define layer boundaries in the SAS model, as discussed below. The geohydrologic units are identified in the model by a series capital letters chosen for mnemonic purposes.

Glacial till in the SAS area is represented by the KTCOM and KTNON (table 2) geohydrologic units. KTCOM (fig. 5) consists of compact drumlin till and compact surface till and may be overlain by KFLUV at the valley margins. KTNON consists of loose (noncompact) surface till.

The glacial stratified deposits in the SAS area are represented by the KFLUV, KDELT, and KFINE geohydrologic units (table 2). At the valley margins, KFLUV consists of collapsed, ice-marginal fluvial sequences that overlie KTCOM (fig. 5). In this area, the layer boundary between KFLUV and KTCOM was assumed to be at a depth equal to one-third of the total thickness of the glacial deposits. In the center of the valley, KFLUV consists of collapsed ice-marginal fluvial deposits throughout the entire saturated thickness of the glacial stratified deposits. The contact between fluvial deposits and the underlying deltaic deposits marked the water level in glacial Lake Pomperaug at an altitude of 76.2 m (Pessl, 1970), and the map-view contact between KFLUV and KDELT units (fig. 5) was estimated to be close to the 76.2-m land-surface contour, unless local data indicated otherwise (J.R. Stone, U.S. Geological Survey, oral commun., 2006).

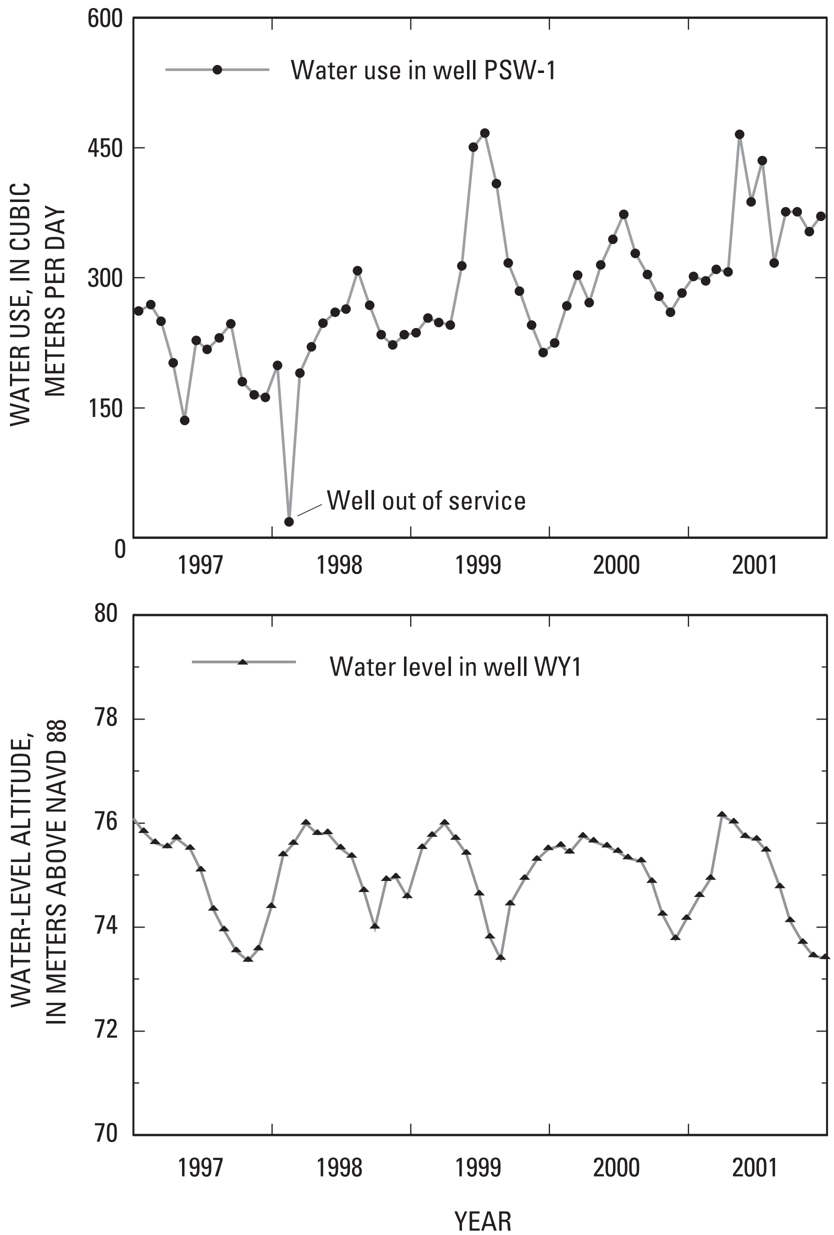


Figure 8. Withdrawals from well PSW-1 and water-level altitude in well WY1, 1997 to 2001, Woodbury, Connecticut.

Table 2. Specified values of hydraulic properties and definition of parameters for the small-area simulation model, Woodbury, Connecticut.

[“Estimated” means that the parameter value was not specified but was estimated (estimated values shown in table 3); m/d, meters per day; DQ, indicates dimensionless quantity]

Parameter name	Parameter description	Units	Value from modified large-area simulation	Parameter values and ranges from Lyford and others (2007)	
				Values used in large-area simulation	Range of reasonable values
KTCOM	Hydraulic conductivity of compact till	m/d	Estimated	0.12	0.003 to 0.3
KTNON	Hydraulic conductivity of non-compact till	m/d	0.3	.09	.03 to 3
KFLUV	Hydraulic conductivity of fluvial deposits	m/d	Estimated	5.7 to 6.3	3 to 52
KDELT	Hydraulic conductivity of deltaic deposits	m/d	Estimated	5.7 to 6.3	3 to 52
KFINE	Hydraulic conductivity of lake-bottom deposits	m/d	2.3	2.8	3 to 52
KMESO	Hydraulic conductivity of Mesozoic bedrock	m/d	Estimated	0.09	0.03 to 1.5
KXLN	Hydraulic conductivity of Paleozoic bedrock	m/d	Estimated	.03	.003 to .3
KVTILL	Ratio of horizontal to vertical hydraulic conductivity in all till	DQ	1	1	1 to 10
KVGSD	Ratio of horizontal to vertical hydraulic conductivity in all glacial stratified deposits	DQ	1	1	1 to 10
KVROCK	Ratio of horizontal to vertical hydraulic conductivity in all bedrock	DQ	1	1	1 to 10
KPOMP	Hydraulic conductivity of Pomperaug River streambed deposits	m/d	0.3	0.3	0.1 to 3
KTRIB	Hydraulic conductivity of tributary streambed deposits	m/d	Estimated	.3	.1 to 3
PTT	Porosity of compact till	DQ	.1	.08	.25
PTILL	Porosity of non-compact till	DQ	.1	.035	.20 to .35
PGSD	Porosity of glacial stratified deposits	DQ	.3	.35	.30 to .45
PBR	Porosity of bedrock	DQ	.001	.02	.005 to .02

KDELT consists of two deltaic sequences having different surface altitudes (fig. 6). In between these areas, KDELT overlies KFLUV (fig. 5), and the layer boundary is assumed to be at a depth equal to two-thirds of the total thickness of the glacial deposits. This zone was delineated on the basis of hydraulic-head data, which show that hydraulic heads in some deep wells (KFLUV) are lower than hydraulic heads in shallow wells at the same location (KDELT). One way for this situation to arise is for there to be more permeable deposits deeper in the glacial stratified deposits.

KFINE consists of a fine-grained sequence of silts and clays deposited in glacial Lake Pomperaug. These deposits are north of the buried bedrock ridge that separates thick glacial stratified deposits north of the study area from thick glacial stratified deposits in the study area.

The bedrock in the SAS model is divided into two units as was done in the LAS (Lyford and others, 2007). KMESO represents Mesozoic bedrock, and KXLN represents Paleozoic bedrock. There are differences from the LAS: (1) the boundary between Paleozoic and Mesozoic rock changed slightly on the basis of analysis of rock samples collected in wells drilled for this study (wells WY86 and WY87), and (2) the Mesozoic bedrock in the study area is now known to be primarily basalt, as opposed to arkose, which had been reported in the LAS (Burton and others, 2005).

Simulations of Ground-Water Flow and Residence Time

The construction, calibration, and results of the steady-state SAS model are described in this section. A steady-state model is one in which ground-water flow between cells does not change with time. In this case, the steady-state SAS model represents average annual conditions, as discussed in the following paragraphs. In order to represent different types of aquifer materials, the model uses parameter zones, that is, the modeled area is divided into large units in which each hydraulic property of the subsurface is considered to be represented by a single characteristic value. The use of parameter zones allows parameter values to be estimated with a small amount of data, but in reality, properties vary within each zone. The initial SAS model used boundary conditions and aquifer-property estimates from a modified version of the LAS model.

Large-Area Simulation of Ground-Water Flow

The LAS model was used to estimate ground-water-flow rates across external boundaries into the SAS area. The layer thickness, recharge, and parameter estimates were modified from the original LAS model described by Lyford and others (2007), in order to make layers correspond between the LAS and SAS models and to incorporate information that became available after the LAS model was completed.

Changes in layer thickness in the LAS model were necessary because that model represented a combination of glacial stratified deposits and bedrock in a single 45.7-m thick layer; therefore, aquifer properties in the LAS model are a combination of properties of bedrock and glacial deposits. More detail was needed for the SAS model because the glacial deposits vary in thickness and are less than 30 m thick. In the modified LAS model, the upper layer was subdivided into three layers, and the lower LAS layer, which represents only bedrock, was left as one layer.

The LAS model also was modified by using a more detailed representation of ground-water recharge (D.M. Bjerklie, U.S. Geological Survey, written commun., 2007). The average annual recharge was used so that the modified LAS model would be compatible with the SAS model. The use of average annual recharge is discussed in the section on model boundaries and stresses.

Other minor changes were made in the modified LAS model, such as replacing the altitude of the top of the upper layer with more accurate values interpolated from 1:24,000 topographic map contours. Another change was that a more accurate estimate of bedrock altitude was calculated and used in the model. This was done by subtracting the interpolated values of the thickness of the glacial deposits (Mazzaferro, 1986a), which were modified slightly to include new data collected for this project, from land-surface altitude. For these reasons, the upper model layers in the modified LAS model represent more homogeneous materials than did the single layer in the original LAS model.

The different layering changed the geohydrologic unit definition, and an accurate estimate of ground-water flow could be obtained only by re-estimating model parameters. The re-estimation was done using the parameter-estimation capability available in MODFLOW-2000 and the same set of calibration data that was used in the LAS calibration. These modifications resulted in slightly different aquifer property estimates than those used in the LAS model (table 2).

Small-Area Simulation of Ground-Water Flow

The purpose of the SAS model was to provide detailed estimates of steady-state ground-water-flow rates and ages in the source area to monitoring wells and to water-supply well PSW-1. The results of the model simulations include estimates of aquifer properties, simulated hydraulic heads and streamflow, water budget, ground-water-flow paths, and ground-water residence times.

Model Characteristics

Steady-state ground-water flow in the study area is simulated by a model constructed of a grid of square cells. The flow of water into and out of each cell is simulated by the model. In the SAS model, the Hydrogeologic Unit Flow module in MODFLOW-2000 was used (Anderman and Hill, 2000). This module allows aquifer properties to be specified independently of the model grid, and geohydrologic units do not need to be in a continuous layer. The aquifer-property boundaries correspond to the geohydrologic units discussed previously in this report. Input parameters for these units are adjusted until the simulation reproduces observed values as closely as possible.

Model Domain and Grid-Cell Dimensions

The model domain of the SAS model is the area within the external boundaries of the model (fig. 9) that includes all areas that could contribute water to well PSW-1 under various scenarios of pumping rates, taking into account the uncertainty in hydrologic and aquifer properties. The external boundaries were selected to be particle tracks from the original LAS model. Particle tracks trace a path line of flow across which there is no flow of water; therefore, the external boundaries of the SAS model initially were specified to be no-flow boundaries. In the course of re-estimating aquifer properties in the modified LAS model, some simulated flow crossed the SAS model's external boundaries. The simulated flow from the modified LAS model was specified as a fixed flow rate for each model cell on the external boundaries of the SAS model.

The top of the uppermost model layer is the water table, but the position of the water table was observed at only a few locations; therefore, the initial top of the upper model layer

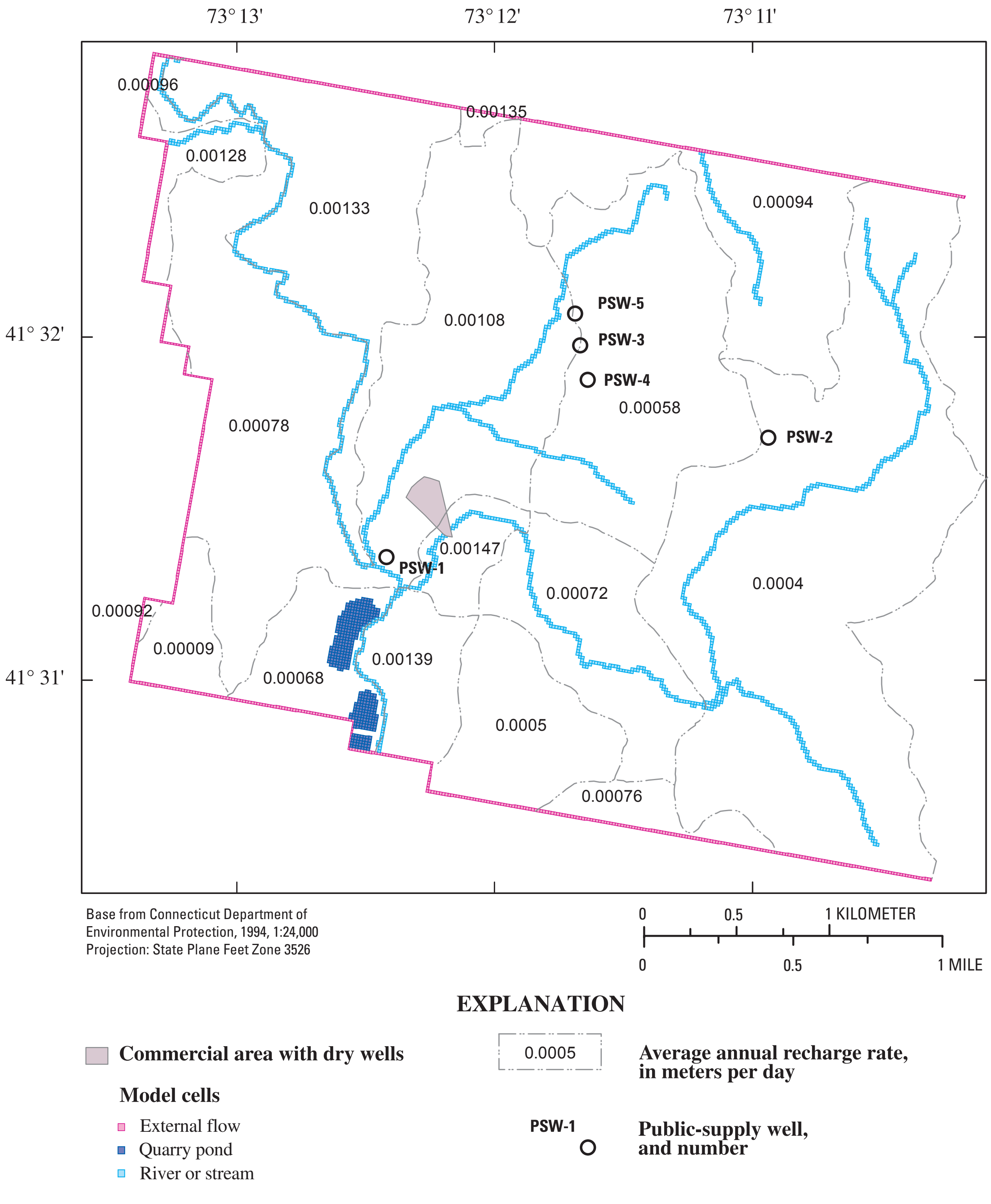


Figure 9. The domain of the SAS model, recharge rates, and public-supply wells, Woodbury, Connecticut.

was taken to be land surface or the simulated water table, whichever was lower. The top altitude was continually refined by replacing the top altitude with the water-table altitude determined from a model run. This process was run iteratively until a stable water-table position was reached. In this way, vertical resistance assigned to model layers takes account of only the saturated part of the subsurface. The lower boundary of the model is a no-flow boundary placed at the depth at which flow through fractures is assumed to be negligible (61 m below the base of the glacial deposits).

In the SAS model grid, there are 241 rows, 322 columns, and 7 layers. Active grid cells in the SAS model cover 15 km^2. The grid has the same orientation as the LAS model grid (the *y*-axis is rotated 10 degrees clockwise from north). The model was initially calibrated at a 61-by-61-m grid-cell size. After an acceptable model was obtained through calibration, the grid cell size was refined to 15.2-by-15.2 m. The upper six model layers were a uniform 3 m thick, and the lowest model layer was 61 m thick.

Boundary Conditions and Model Stresses

Ground-water flow is specified at the external boundaries of the SAS model, except for quarry ponds, which are represented by head-dependent flow boundaries. The eastern boundary has a specified flow rate equal to zero (no flow) because this is a watershed boundary, and ground-water flow across the boundary is assumed to be minimal. Ground-water flow across the other lateral boundaries was accounted for by taking the flow rate across cell faces in the modified LAS, and converting them to a flow per unit length multiplied by the length of the cell in the SAS model. Flow across lateral boundaries was simulated in MODFLOW-2000 using the Flow and Head Boundary (FHB) package.

The boundary condition for the top of the model is a specified flow rate equal to average annual recharge from 1998 to 2004 (fig. 9), as determined using a rainfall-runoff model (D.M. Bjerklie, U.S. Geological Survey, written commun., 2007). The use of an annual average value dampens the influence of variability among years. Also, recharge during this time period is compatible with the hydraulic-head measurements used for parameter estimation (see next section). Recharge to the model was modified in the commercial area, where all surface runoff flows into storm drains (dry wells). In this area, recharge was set to the annual rainfall over the commercial area. This approach slightly overestimates recharge from the commercial area, because some water evaporates before reaching the dry wells.

The boundary condition for the bottom of the model is a specified flow rate equal to zero, because at some depth, ground-water flow in fractures in bedrock is minimal. Alternative versions of the model were run with the base of the model at 500 ft below the base of the glacial deposits, and the model was found to be not sensitive to the depth of the lower model boundary.

Surface-water head-dependent-flow-rate boundaries are of two types. Streams are simulated using the Stream (STR) package in MODFLOW-2000, as was done in the LAS model (Lyford and others, 2007). Ponds near the Pomperaug River are simulated using the General-Head Boundary (GHB) package in MODFLOW-2000. The conductance term for this boundary is calculated using a hydraulic conductivity that is equal to that used for the Pomperaug River, so there is virtually no difference between how ponds are simulated and how the Pomperaug River is simulated. The different packages available in MODFLOW-2000 were used simply to allow for water budgets to be calculated separately for quarry ponds and for the river and tributary streams.

Pumping wells are simulated in the SAS model at the same locations and withdrawal rates as in the LAS (table 1) (Lyford and others, 2007). Wells were simulated using the Multi-Node Well (MNW) package (Halford and Hanson, 2002). The MNW package enables wells to span multiple layers, and the model calculates the amount of total well pumpage assigned to each model layer.

Model Hydraulic Parameters, Observations, and Observation Weights

The hydraulic-property parameters in the SAS model are the nine horizontal hydraulic conductivities of five glacial geohydrologic units, two bedrock geohydrologic units, and the streambeds of the Pomperaug River and of tributary streams. The values of the parameters were taken either from the LAS or were estimated in the SAS model using the parameter-estimation capability of MODFLOW-2000. The values of some of these nine parameters could be estimated because the calibration data contained sufficient information about the parameter (table 2), but other parameter values could not be estimated because either the model was not sensitive to the parameter or the calibration data contained insufficient information about that parameter. Values for parameters that could not be estimated were not changed from the modified LAS values (table 2). The ratio of horizontal to vertical hydraulic conductivity also could not be estimated with the data available. Other model parameters were estimated while the ratio was held at fixed values between 1 and 10, and the best model was obtained with a value of 1. A value of 1 also was determined to produce the best fit to observed values in the LAS model study (Lyford and others, 2007).

Although not required for the flow simulation, effective porosity values are required for the advective particle tracking that is used to calculate residence times. The SAS model uses four porosity groups (identified in the model by the capital letters in parentheses): glacial stratified deposits (PGSD), non-compact surface till (PT), compact till (PTT), and bedrock (PBR) (table 2). The range of porosities assigned to glacial sediments is from 0.20 (till) to 0.45 (glacial stratified deposits), a maximum to minimum ratio equal to 2.5. The range of porosities assigned to bedrock is from 0.005 to 0.02, a maximum to minimum ratio equal to 4.0 (Lyford and others, 2007). The simulated residence time is proportional to the porosity, so porosity variation in glacial deposits has less effect on residence time than variation in bedrock. Residence time of water in wells completed in glacial deposits will be sensitive to a combination of porosity in bedrock and glacial deposits because some water may flow through bedrock before entering the glacial stratified deposits.

Parameters were estimated in the SAS model using three types of observed data ("observations"). The primary type of observation is hydraulic head. In the 34 wells drilled for this study (Group 1 wells; figs. 3 and 4), either monthly or continuous water-level measurements were available. In both cases, the average water level from March 1, 2004 through February 28, 2005 was used in parameter estimation. This period was chosen because it had the most water-level measurements available. In addition, this period roughly represents average conditions, in part because precipitation in the period was at the 58th percentile for annual total precipitation for all years from 1950 to 2005. Although some variability in water levels is seasonal (fig. 8), levels increased slightly from 1985 to 2005 at WY1 (fig. 7) and a possible short-term rise occurred from 1998 to 2005. If this trend is real, SAS model results are more representative of conditions in the last 20 years than they are of conditions longer ago, when average water levels might have been lower.

Hydraulic-head observations in wells from two other sources (fig. 3) were used in parameter estimation—water levels in wells used in the LAS model (from Mazzaferro, 1986b) (Group 2 wells) and water levels in domestic wells measured at the time of well completion (Group 3 wells; from Burton, 2006). Group 2 also includes two wells (WY96 and WY97) that were drilled for this study but were not surveyed to the same accuracy as Group 1 wells. The latter two groups of hydraulic-head data are less accurate than those measured for this study, but they are important in areas where data are sparse.

The second type of observation is streamflow gain and (or) loss. In order to constrain leakage from tributary streams, streamflow was gaged at two locations on South Brook. South Brook was identified as a possible source of water in the LAS model (Lyford and others, 2007). The difference between the two measurements represents the rate of water gained or lost from the stream. The median of five measurements made during low flow periods in 2003 and 2004 was used in parameter estimation. The third type of observation used is ground-water age. Hydraulic conductivity values were determined first using parameter estimation, then porosities were adjusted manually to achieve a good fit between tritium/helium age estimates and simulated residence times.

If observation data used in parameter estimation have different units (hydraulic head in meters and flow in cubic meters per second, for example) or the data have different measurement errors, the data can be weighted to emphasize more reliable or more accurate values while allowing all values to be included in the regression. A method for calculating weights (Hill, 1998) was followed initially in the SAS model. MODFLOW-2000 requires the user to specify one of the following: the standard deviation, variance, or coefficient of variation for each observation. Weights used in the regression are calculated internally in MODFLOW-2000 from the user-specified value, and are the inverse of the variance. Weights on observations were subsequently changed within reasonable ranges during parameter estimation to achieve an approximately normal probability distribution of residuals.

Hydraulic heads measured in Group 1 wells are assumed to be within 0.003 m of their true value. Reference point altitudes were surveyed to within 0.0003 m, and water levels were measured to within 0.003 m. To relate the measurement to a normal probability distribution, a 95-percent confidence interval was used. Weights were assigned on the basis of the statement that 95 percent of all measurements are within 0.003 m of the true value. The initial standard deviation from which weights were calculated is 0.00155 m, and the final value is 0.0762 m. This large increase in the estimated standard deviation could reflect variation of hydraulic conductivity, which affects hydraulic head, within parameter zones. In other words, variation of hydraulic head within a parameter zone is affected by sources of error other than purely measurement error, on which the initial standard deviation was based.

Observed hydraulic heads in Group 2 wells are assumed to be within 1.5 m of their true value because the reference points were estimated from a topographic map having a 10-ft (3.048-m) contour interval. A 90-percent confidence interval was assumed for these wells. The initial and final standard deviation from which weights were calculated is 0.924 m. Observed hydraulic heads in the Group 3 wells are assumed to be within 4.5 m of their true value. Their reference points were estimated from the same topographic map as were Group 2, but the locations were less certain and the terrain was steeper, so that the reference points are more uncertain. In addition, these hydraulic heads were not measured at the same time and may not have completely stabilized after drilling. A 90-percent confidence interval was assumed for these wells. The initial standard deviation from which weights were calculated is 2.77 m and the final value is 3.11 m.

The difference between streamflow measured at two locations on South Brook (fig. 4) also was used as an observation. The median measured difference in streamflow was a loss of 0.0017 m^3/s on July 7, 2004. The error of the individual measurements was assumed to be 10 percent, and the total variance was calculated by adding the measurement variances. The initial and final coefficients of variation (standard deviation divided by the mean) from which weights were calculated was 0.94 (dimensionless). Because the standard deviation and the mean are almost equal, there is a large amount of uncertainty in the streamflow data, and the weight assigned is two orders of magnitude lower than the weight assigned to hydraulic-head observations. Even so, streamflow loss was used in the final parameter estimation because it does improve the model fit slightly.

Estimation of Model Parameters

In order to have confidence in model simulation results, the model should produce reasonable parameter values; hydraulic heads, streamflow, and ground-water residence times that are close to observed data; and residuals that are normally distributed and uncorrelated. In this study, the parameter values that were estimated are reasonable when compared with previous estimates (tables 2 and 3). The standard error of the regression is 1.85 m, but the regression-based calibration minimizes the sum of squares of weighted residuals, and the observations have different weights, so a simple measure like the standard error of the regression is not straightforward to interpret; therefore, other aspects of model fit will be discussed.

The parameter estimation should produce weighted residuals (observed minus simulated values) that are small and unbiased, as indicated by a random distribution of weighted residuals. Also, both the sum and the mean of residuals will be close to zero. In the SAS model, there are more observations of hydraulic head than other types of observations; therefore, unweighted head residuals offer the most intuitive qualitative assessment of model fit.

Weighted model residuals range from -4.8 to 5.2 m (table 4). The mean weighted residual is 0.30 m, and there are four dry wells in the simulation. Group 1 weighted residuals also ranged from -4.8 to 5.2 m (the minimum and maximum for all data were in Group 1), and the mean was 0.012 m, indicating a good fit between observations and simulated values. The mean absolute residual is 1.80 m for the Group 1 wells and is an indication of how far weighted simulated heads are from weighted observed heads. The sum of weighted residuals for Group 1 was 0.44 m, indicating that there is little bias in the model. Weighted model residuals also can be analyzed graphically to detect potential problems with the regression (fig. 10A). Residuals are distributed approximately randomly when plotted against simulated values, although shallow and deep pairs of wells (WY77/WY78 and WY96/WY97) tend to have residuals of equal magnitude but opposite sign. This probably means that some geologic feature is not represented in the model, as discussed in the following paragraph. Most weighted residuals tend to lie on a line having a 1:1 slope when plotted against observed values (fig. 10B).

Unweighted water-level residuals range from -15 to 45 m (table 4; fig. 10C). The mean unweighted residual is 2.9 m, and there are four dry wells in the simulation. The extreme values for unweighted residuals are at Group 3 wells (table 4). The location and reference altitudes for these wells is less certain than for other wells, so there is less weight placed on these data. For the 34 hydraulic-head observations in Group 1 wells, the unweighted residuals ranged from -1.2 to 1.3 m, and the mean was 0.0030 m, indicating a good fit between observations and simulated values. The mean absolute residual is 0.46 m for the Group 1 wells and is an indication of how far simulated heads are from observed heads. The sum of unweighted residuals for the Group 1 wells was 0.11 m, indicating that there is little bias in the model.

Although the Group 1 residuals are small and indicative of a good model fit, their pattern does reveal some aspects of the model with respect to the position of geologic contacts that could be improved. Residuals from the area where KDELT overlies KFLUV are too high in the upper, deltaic deposits and too low in the lower, fluvial deposits. Also, the absolute magnitude of these residuals is higher than elsewhere. This model, which uses the superposition of these two geohydrologic units to help explain the head differences between shallow and deep intervals in the glacial stratified deposits has less bias and “splits the difference” better than models that do not include this feature. Other models were constructed to simulate this feature, for example by introducing a confining layer between the shallow and deep parts of the glacial stratified deposits, but these models did not have a geologic basis and did not remove the bias in residuals. Perhaps the absolute magnitude of residuals in this area could be lowered by moving the position of the glacial stratified deposits/till contact, but no geologic evidence supported this, and it was not clear that this would improve the model.

The observed value of streamflow loss, 147 m^3/d, was the same as the simulated value, 147 m^3/d, although the uncertainty in the measurements was almost as large as the difference in the streamflow measurements. Thus, the weight on this observation was small and the observation did not affect the regression to the same extent as the observed values of hydraulic head.

In the SAS model, the correlation coefficient of weighted residuals against a normal probability plotting position is 0.972, but the critical value at the 5-percent significance level is 0.975, which would lead to the rejection of the assumption of normality. However, a regression produces residuals that are correlated for observed values that are close to one another. An overprediction at one location is likely to produce an overprediction at a nearby location. Residuals can be plotted against randomly generated residuals that have the same degree of correlation as expected from the model to see if the assumption of normality of residuals can be accepted (Cooley and Naff, 1990). The SAS model did produce residuals that are normally distributed based on this type of analysis.

After the hydraulic parameters were estimated, porosities were varied systematically in order to achieve the best fit between simulated residence times and measured tritium/helium age dates. The range of reasonable porosity values is small (table 2), and porosity in the glacial stratified deposits that is in the contributing recharge area to PSW-1 probably varies in an even smaller range than that given in table 2 (values in table 2 are for many types of glacial stratified deposits in many locations). Porosity for the glacial stratified deposits in the contributing recharge area is probably in the range 0.20 to 0.35, or a decrease and increase of 3 percent and 17 percent, respectively, from the porosity used in the SAS model (0.30). Residence times are a linear function of porosity, so the residence times for water that recharges the

Table 3. Estimated optimal parameter values for the small-area simulation model, Woodbury, Connecticut.

[All units are meters per day]

Parameter name	Parameter description	Optimal parameter value
KTCOM	Hydraulic conductivity of compact till	0.0099
KFLUV	Hydraulic conductivity of fluvial deposits	18
KDELT	Hydraulic conductivity of deltaic deposits	6.2
KMESO	Hydraulic conductivity of Mesozoic bedrock	.14
KXLN	Hydraulic conductivity of Paleozoic bedrock	.035
KTRIB	Hydraulic conductivity of tributary streambed deposits	.039

Table 4. Statistics of residuals (observed minus simulated values) for the small-area simulation model, Woodbury, Connecticut.

[All residuals in meters]

Statistic	All wells	Group 1	Group 2	Group 3
Number of water levels	97	34	8	51
Number of dry wells	4	0	0	4
Weighted water-level residuals				
Minimum water-level residual	-4.8	-4.8	-1.7	-1.5
Mean water-level residual	.30	.012	.073	.52
Median water-level residual	.28	.46	.22	.067
Maximum water-level residual	5.2	5.2	1.7	4.4
Standard deviation of water-level residual	1.8	2.4	1.3	1.3
Sum of water-level residuals	28	.44	.57	27
Mean absolute residual	1	1.80	1.00	1
Unweighted water-level residuals				
Minimum water-level residual	-15	-1.2	-5.2	-15
Mean water-level residual	2.9	.0030	.22	5.2
Median water-level residual	.19	.12	.67	.67
Maximum water-level residual	45	1.3	5.2	45
Standard deviation of water-level residual	10.4	.6	3.8	14
Sum of water-level residuals	274	.11	1.7	272
Mean absolute residual	6	.46	3.0	10

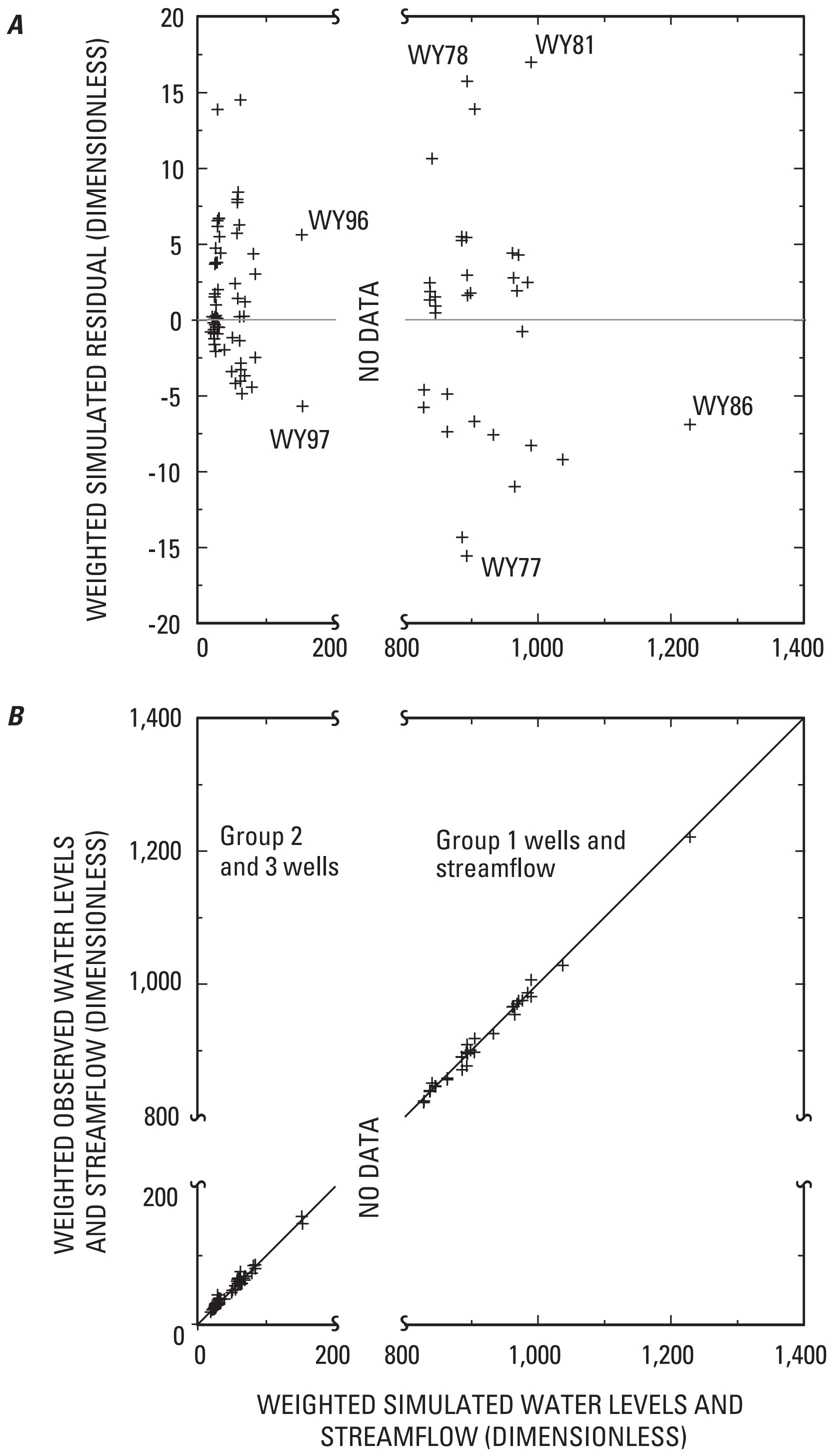


Figure 10. Weighted simulated residuals, weighted observed water levels and streamflow, weighted simulated water levels and streamflow, and unweighted observed and simulated water levels for the small-area simulation model, Woodbury, Connecticut.

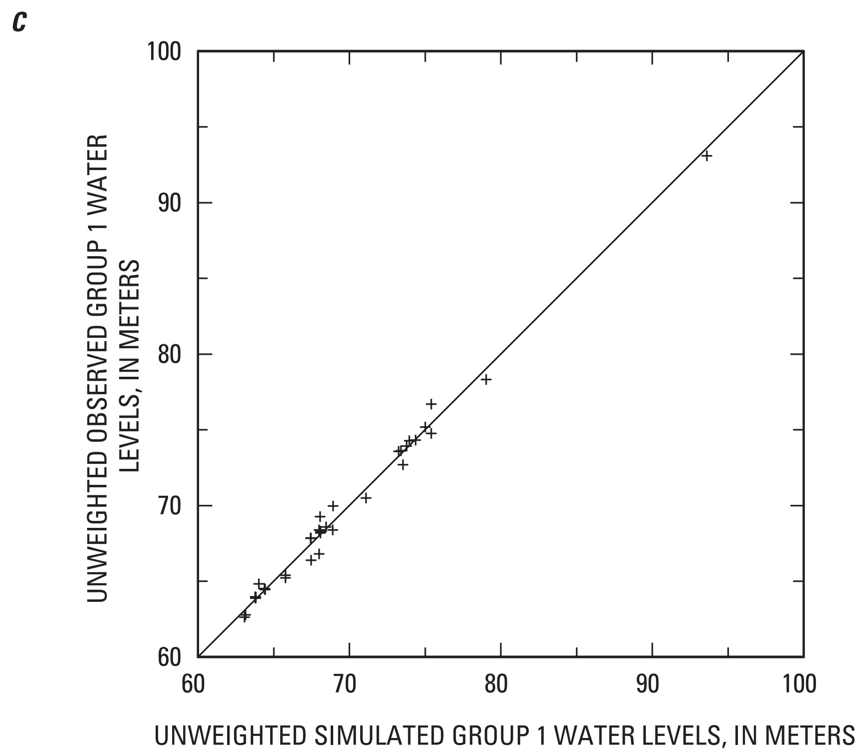


Figure 10. Weighted simulated residuals, weighted observed water levels and streamflow, weighted simulated water levels and streamflow, and unweighted observed and simulated water levels for the small-area simulation model, Woodbury, Connecticut.—Continued

glacial stratified deposits could be expected to vary by -33 percent to +14 percent of the reported values. Water that recharges the uplands follows paths that include both types of till as well as bedrock, and the residence time is a function of the porosity and the amount of time spent in all those units, so no generalization is possible. Residence times did not match well for bedrock wells, as discussed later in this report, but the final porosity used for bedrock (0.001) was reasonable based on published values, and resulted in the best overall match for all wells. An attempt to match residence times in bedrock wells resulted in a much poorer match for other wells. The final porosities reported in table 2 produced the best fit between measured and observed residence times for all wells.

The use of the SAS model developed and described here is deemed acceptable for the purposes of this study. The model generally produces unbiased and optimal estimates of the most important model parameters. The hydraulic heads, streamflows, and parameter values estimated using the model are within reasonable ranges. Simulated residence times show the same magnitude and trends with depth as measured residence times, except for bedrock for reasons that are discussed in the next section.

Simulated Water Budget

The simulated water budget is similar to that produced by Lyford and others (2007) in that the dominant source of water in the uplands is recharge from precipitation, whereas recharge from precipitation accounts for only about one-third of the total flow in the glacial stratified deposits. In both the LAS and SAS models, contributions of ground water from uplands to the glacial stratified deposits are significant. The Pomperaug River is the main discharge area, but some ground water flows into tributary reaches as well. Tributaries can gain or lose ground water, depending on the local relation of stream stage and ground-water level. A more detailed description of the simulated water budget provides some insight into the relation of upland and glacial stratified deposits parts of the ground-water flow system (table 5).

Table 5. Simulated water budget for the small-area simulation, Woodbury, Connecticut.

[--, not applicable; budgets do not always sum to 100 percent because of rounding; negative flow is out of unit; positive flow is into unit; m³/d, cubic meters per day]

Component of water budget and MODFLOW module (in parentheses)	Water in, in m³/d	Water in, in percent	Water out, in m³/d	Water out, in percent	Net flow, in m³/d
FLOWS IN AND OUT OF MODEL					
Entire simulated area					
Multi-node wells (MNW)	--	--	626	4	-626
General Head-Dependent Boundaries (GHB), representing quarry ponds	251	2	1,280	8	-1,029
Recharge from precipitation on land surface (RCH)	11,600	70	--	--	11,600
Specified Flows (FHB), representing flows across external boundaries	3,960	24	560	3	3,400
Stream Leakage (STR)	793	5	14,100	85	-13,307
Valley (Glacial stratified deposits)					
Multi-node wells (MNW)	--	--	392	3	-392
General Head-Dependent Boundaries (GHB), representing quarry ponds	252	2	1,280	8	-1,030
Recharge from precipitation on land surface (RCH)	5,070	32	--	--	5,070
Specified Flows (FHB), flow across external boundaries	2,920	19	210	1	2,710
Stream Leakage (STR)	781	5	11,500	74	-10,700
Uplands ([1]Upper 18 meters of till and bedrock)					
Multi-node wells (MNW)	--	--	128	2	-128
Recharge from precipitation on land surface (RCH)	6,480	77	--	--	6,480
Specified Flows (FHB), flow across external boundaries	411	5	166	2	245
Stream Leakage (STR)	12.4	--	2,580	31	-2,570
Uplands ([1]Bedrock deeper than 18 meters)					
Multi-node wells (MNW)	--	--	106	2	-106
Specified Flows (FHB), flow across external boundaries	625	9	184	3	441
FLOWS BETWEEN UNITS IN MODEL					
Valley (Glacial stratified deposits)					
Upper 18 meters of till and bedrock	1,330	8	203	1	1,130
Bedrock deeper than 18 meters	5,300	34	2,041	13	3,260
Uplands ([1]Upper 18 meters of till and bedrock)					
Glacial stratified deposits	203	2	1,330	16	-1,130
Bedrock deeper than 18 meters	1,330	16	4,250	50	-2,920
Uplands ([1]Bedrock deeper than 18 meters)					
Glacial stratified deposits	2,040	30	5,300	77	-3,260
Upper 18 meters of till and bedrock	4,250	61	1,330	19	2,920

[1] Depths measured from water table.

Recharge to the water table in upland areas is dominated by precipitation (table 5). Discharge from the upper 18 m of till and bedrock in the uplands is to bedrock deeper than 18 m (50 percent), streams (31 percent), and glacial stratified deposits (16 percent). Minor amounts (3 percent) of upland ground water flows to wells and external model boundaries. Bedrock greater than 18 m deep receives water from till, glacial stratified deposits, and external boundaries. Most water from deep bedrock discharges upward into glacial stratified deposits, with minor amounts to till and wells.

The glacial stratified deposits receive ground water through recharge from precipitation (32 percent), upward flow from bedrock (34 percent), and flow across external boundaries (19 percent), as well as minor amounts from streams, till, and quarry ponds (fig. 11). Although net percentages (as presented above) give an overall water budget, losing stream reaches account for 5 percent of the flow into the glacial stratified deposits. Most of the streamflow loss occurs in the lower reach of South Brook in the study area (fig. 12). Losing reaches of streams are common in the New England glacial-aquifer setting. Evidence that South Brook loses water to the aquifer was observed on August 26, 2005, when the stream went dry over the predicted losing reach because of a lack of rainfall. Water was flowing in South Brook above and below this reach. The dry wells in the commercial area (fig. 9) also are a source of recharge to the glacial stratified deposits (about 80 m^3, or 1.6 percent of the natural recharge from precipitation to the glacial stratified deposits). Most of the ground water in the glacial stratified deposits discharges to the Pomperaug River (74 percent) and quarry ponds (8 percent). Water in the quarry ponds eventually discharges back into the Pomperaug River south of the study area. Only about 3 percent of the ground water in the glacial stratified deposits is used for public-water supply.

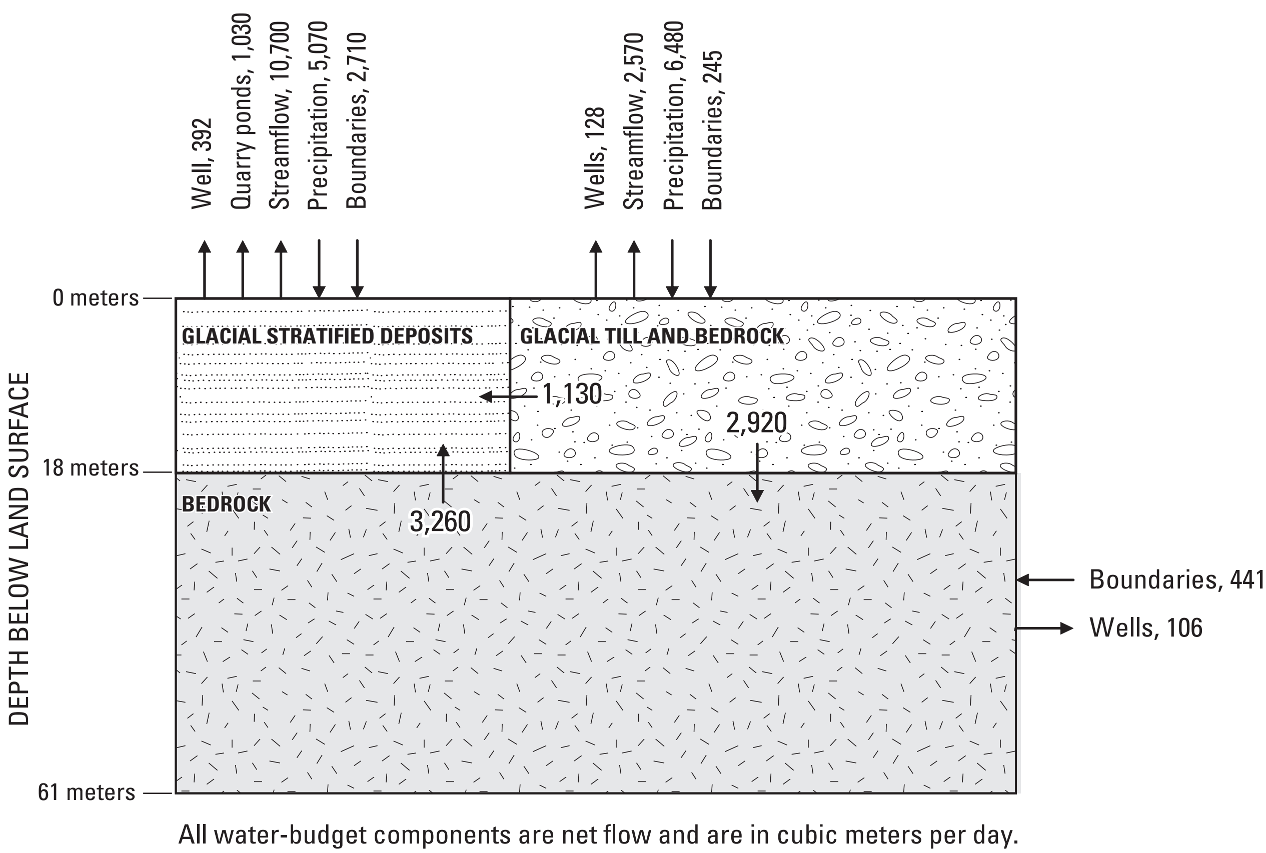
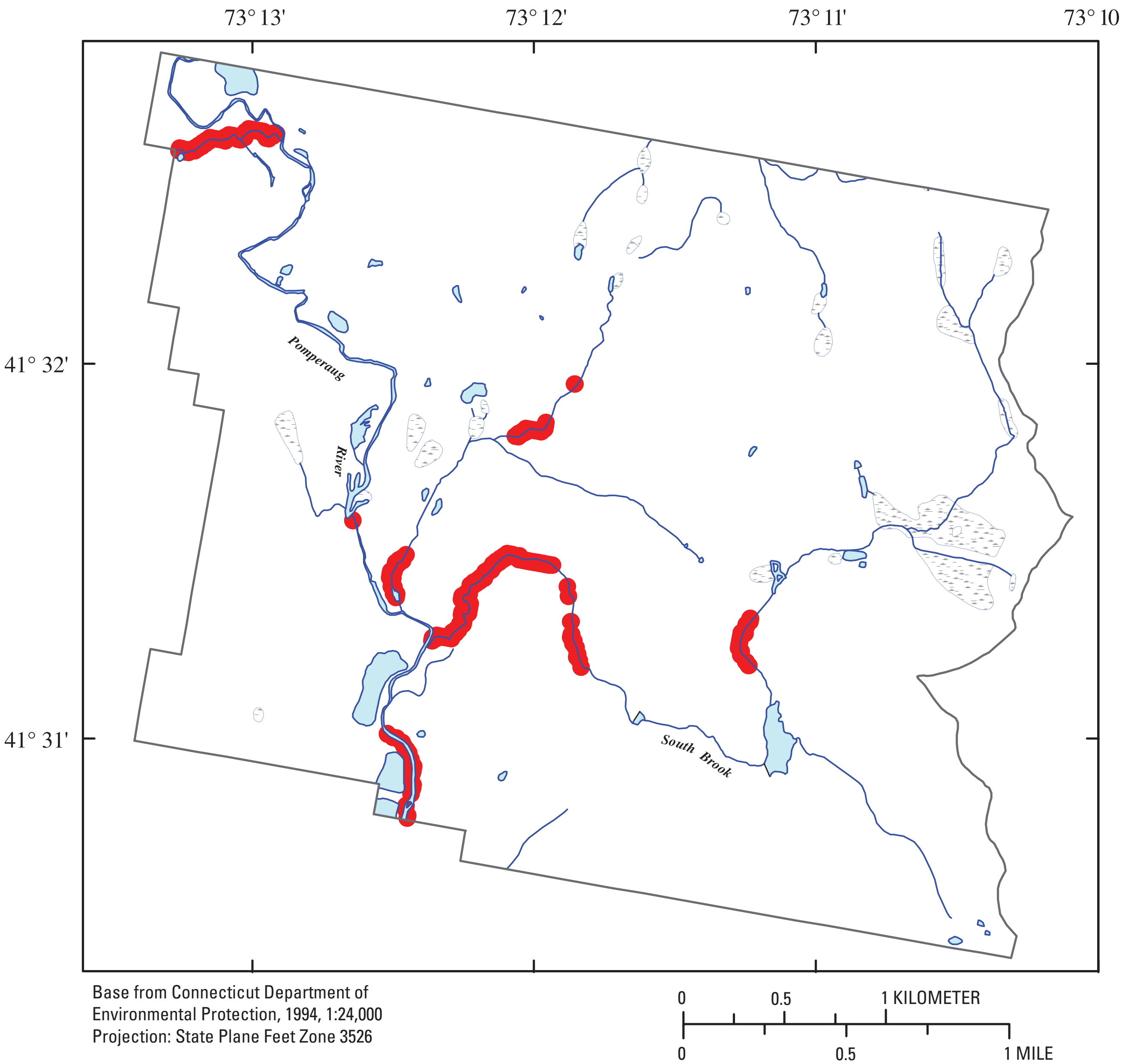


Figure 11. Water budget for the small-area simulation, Woodbury, Connecticut.



EXPLANATION

Boundary of Small-Area Simulation area

Losing stream reach

Figure 12. Simulated losing stream reaches in the small-area simulation model, Woodbury, Connecticut.

Simulation of Ground-Water Residence Time

The particle-tracking program MODPATH (Pollock, 1994) was used to simulate ground-water residence time, which was in turn used to relate natural and anthropogenic factors to water quality. Land use is related to shallow ground-water quality (Grady, 1994); for any given land use in a contributing recharge area to a well, an associated set of water-quality characteristics moves with the ground water toward the well. If land use changes, a new characteristic water quality develops that also moves toward the well. The residence time is related to the time it takes for that new water-quality characteristic to be detected in a water sample from the well. Residence time also can be used in the analysis of chemical processes that depend on contact time between the water and the solid matrix through which the water flows.

Residence Time of Water in Monitoring Wells

Residence times and contributing recharge area were simulated for monitoring wells in order to refine understanding of the relation between land use and water quality. Well PSW-1 integrates water quality from a large area containing a variety of land uses and geohydrologic units, but monitoring wells, which are pumped only when being sampled, have a smaller contributing recharge area and therefore are affected by fewer land uses.

Residence times were simulated for monitoring wells by reverse particle tracking using the SAS model. The reason for using reverse particle tracking is that water is not normally pumped from monitoring wells, so ground-water-flow paths do not converge on the well, and only a limited number of particles pass forward through the well. The best way to ensure that particles pass through the well is to “start” them at the well and track them back to their source area.

Concentrations of tritium and helium-3 were measured in samples from selected wells and converted to residence times for comparison with simulated ground-water residence times. Details of tritium/helium dating procedures are summarized in Schlosser and others (1988) and Solomon and Cook (1999). Measured and simulated residence times generally are similar (table 6). Simulated residence times at monitoring wells ranged from less than 0.1 years to 10.7 years. As expected, water from deeper wells generally had longer simulated residence times. In some medium-depth and deep wells (WY70, WY73, WY75, and WY102), the simulated residence times are shorter than the measured values. This could be because of error either in the configuration of flow paths predicted by the model or in the simulated porosity distribution. In some cases, a slight change in the value of hydraulic conductivity causes shifts in the flow paths to shallower or deeper zones, thus changing the simulated residence time at a fixed depth in a monitoring well. The measured residence time for the sample from bedrock well WY87 (9.9 years) is longer than simulated (2.6 years). The measured residence time can be long for at least two reasons: the actual flow path can be longer than the simulated flow path because of tortuous paths through a fracture network (fracture networks are not explicitly included in the simulation); or, the sample is a mixture of water of a short residence time (through a permeable fracture) and water of a long residence time (from dead-end fractures or from fractures so small that Darcy’s Law does not apply). The simulated residence times in bedrock tend to be short because of the low bulk-rock porosity, based on previous studies (table 2), that is used in the advective particle tracking.

Residence Time of Water in PSW-1

Residence times were simulated for PSW-1 by forward particle tracking using the SAS model. Forward tracking allows particles to be tagged with a recharge volume based on where the particle starts, and flow-weighted residence times at the supply well can be calculated. Flow-weighted residence time is more important for the public-supply well than for the monitoring wells because as water is pumped from the public-supply well, residence times are mixed in the well in proportion to the flow rate over each interval in the well screen.

The contributing recharge area to PSW-1 shows the effect of geology on ground-water-flow direction and residence time (fig. 13). Within the glacial stratified deposits, the contributing recharge area is relatively narrow. The longest residence time and greatest spread of flow paths is in the upland area. About 40 percent of the water withdrawn from PSW-1 originated as upland recharge before flowing through the glacial stratified deposits. The mean residence time for water recharged in the uplands is 7.8 years. About 44 percent of the water withdrawn from PSW-1 originated as recharge in either KFLUV (mean residence time 6.9 years) or KDELT (mean residence time 4 years). About 16 percent of the water withdrawn from PSW-1 comes from dry wells in the commercial area. The residence time for water that is recharged through dry wells ranges from 1.6 to 3.8 years, and the mean residence time is 2.5 years. Dry wells are a fast pathway for water to enter the aquifer and provide a significant amount of water to PSW-1; therefore, PSW-1 is more susceptible to contamination in runoff from the commercial area, which enters the dry wells, than to recharge elsewhere in the SAS area.

Ground-water residence time has been used as a surrogate measure of an aquifer’s susceptibility to contamination (Eberts and others, 2005), and wells producing water with short residence times commonly are judged to be more susceptible to surface contamination than wells producing water with long residence times. If the susceptibility is calculated using age tracers that reflect a flow-weighted average of residence times in the aquifer, the full range of residence times in the water is not revealed. Water withdrawn from a well is a mixture of waters with different residence times, and a single residence time does not fully characterize the susceptibility of the well to recent contamination. The mean simulated flow-weighted residence time from PSW-1 is 5.9 years, which compares reasonably well with the residence time measured

Table 6. Tritium/helium age and simulated residence time at monitoring wells in the small-area simulation model, Woodbury, Connecticut.

[--, no tritium/helium analysis or residence time not simulated; blank line separates clusters of wells; vertical well-screen position: Deep wells are screened in the lower third of glacial deposits; middle wells are screened in the middle third of glacial deposits; shallow wells are screened in the upper third of glacial deposits; bedrock means well is screened in bedrock]

Well name	Tritium/helium age, in years	Simulated residence time, in years			Vertical well-screen position
		Minimum	Average	Maximum	
PSW-1	5.5–6.6	--	5.9	--	Deep
WY69	5.8	5.6	6.3	8.3	Deep
WY70	4.9	1.3	1.4	1.5	Middle
WY71	1.8	--	--	--	Shallow
WY72	8.4	7.4	8.5	8.9	Deep
WY73	4.0	2.6	2.8	2.9	Middle
WY74	.7	--	--	--	Shallow
WY75	10.0	5.9	6.2	6.6	Deep
WY76	.9	--	--	--	Shallow
WY77	--	.6	.7	.8	Deep
WY78	--	--	--	--	Shallow
WY79	--	1.2	1.3	1.5	Shallow
WY80	--	.5	.5	.5	Deep
WY81	--	.3	.4	.4	Shallow
WY82	--	2.3	4.3	6.7	Deep
WY83	--	.2	.4	.5	Middle
WY84	3.4	3.3	4.6	6.2	Deep
WY85	--	.8	.8	.8	Shallow
WY86	--	3.1	3.2	3.3	Bedrock
WY87	9.9	2.6	2.6	2.6	Bedrock
WY88	--	.1	.3	.5	Deep
WY89	--	.6	.7	.8	Deep
WY90	--	.3	.4	.4	Deep
WY91	--	7.7	8.4	9.1	Middle
WY92	--	1.6	2.0	2.4	Shallow
WY93	--	5.5	5.5	5.6	Deep
WY94	--	.3	1.2	2.5	Middle
WY95	--	.5	.5	.6	Shallow
WY96	--	--	--	--	Middle
WY97	--	2.8	3.0	3.3	Bedrock
WY98	--	.3	.4	.4	Deep
WY99	--	0.0	.1	.1	Middle
WY100	4.9	10.0	10.4	10.7	Middle
WY101	5.4	6.7	7.5	9.4	Deep
WY102	5.6	.4	.6	.9	Shallow
WY106	--	5.9	5.9	5.9	Bedrock

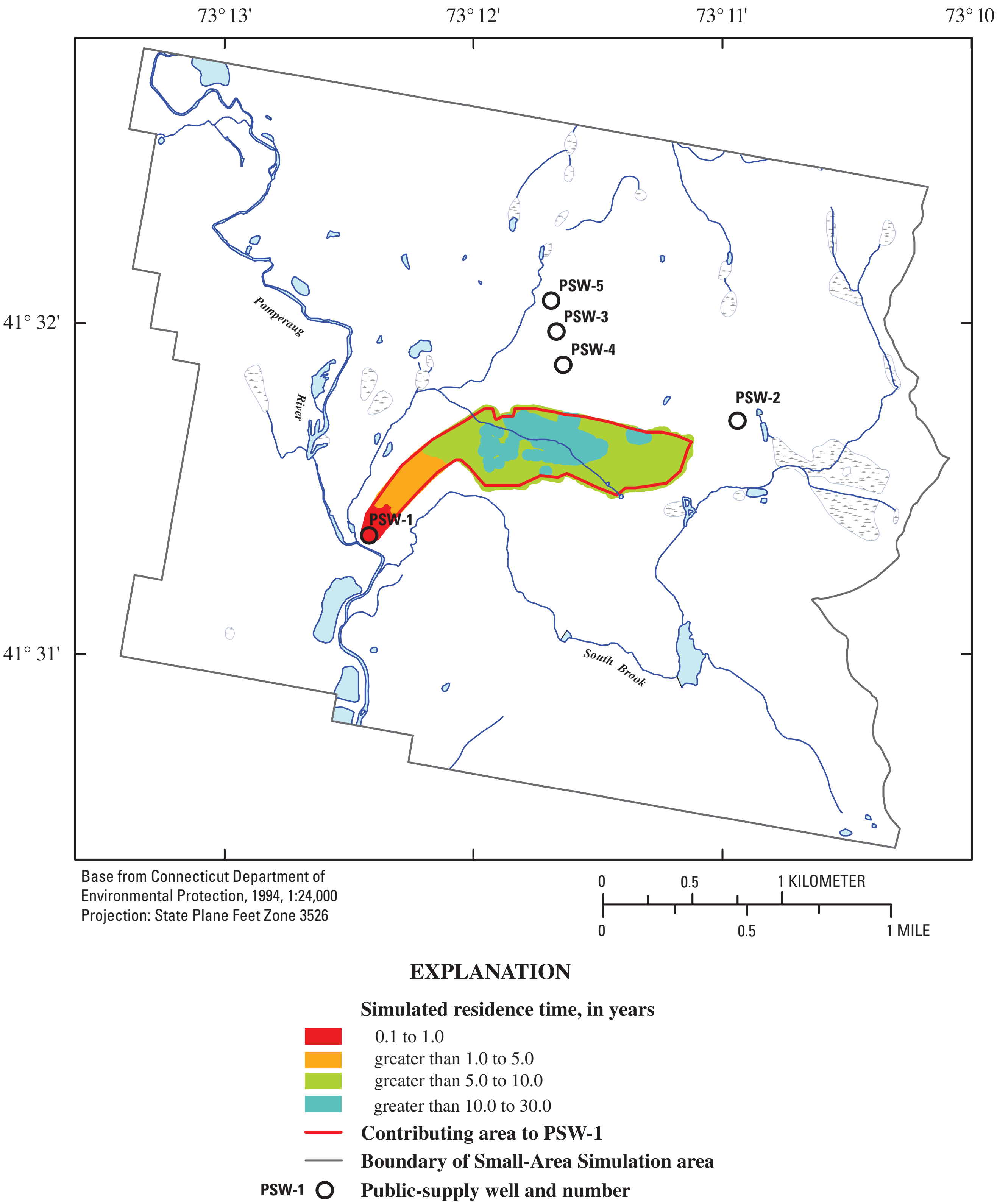


Figure 13. Contributing recharge area to well PSW-1 and ground-water residence time, Woodbury, Connecticut.

using tritium/helium data of 5.5 and 6.6 years (samples for age dating were collected twice from this well). Although the observed and simulated residence times are similar, ground water pumped from PSW-1 contains a range of residence times (figs. 13 and 14 combined). Of the water pumped from this well, 10 percent is less than 1.9 years old, and 10 percent is more than 9 years old (table 7). The residence times are shortest (about 1 year) in KFLUV near PSW-1. There are at least two modes to the distribution of ages, one mode with residence times less than 5 years and one mode with residence times greater than 5 years. About 34 percent of the ground-water in PSW-1 is younger than 5 years and 56 percent of the water is between 5 and 9 years (fig. 14).

Residence Time, Land Cover, and Water Quality

Particle tracking was used to relate land use/land cover and residence time in the contributing recharge area to the water quality in well PSW-1. Each of many particles represents a parcel of water whose chemical quality is determined in part by the land use/land cover in the area of recharge. Each parcel of water reaches PSW-1 after its characteristic residence time, where it mixes with water of different residence times and quality.

Several potential sources of contamination in the contributing recharge area to PSW-1 can be related to land use. The most common source of contaminants to the ground water in unsewered residential areas is septic systems, which receive domestic waste, wastewater, and discarded household chemicals (Grady, 1994). Other potential contaminants include lawn and garden chemicals, wastes from domestic animals, and leaky fuel tanks. Concentrations of many water-quality constituents, such as chloride, dissolved solids, and nitrate, are significantly higher in residential areas than in undeveloped areas; however, ground-water quality in residential areas is highly variable. Grady (1994) found similar concentrations of these water-quality constituents in sewered and unsewered residential areas. The reasons include leaky sewer lines and the heavy use of lawn fertilizer in densely populated sewered areas. The use of road de-icing chemicals affects water quality in developed areas as well as in undeveloped areas, to the extent that roads are present in the undeveloped areas. Impervious surfaces, such as those in the commercial areas in the study area, also are sources of contaminants to the ground water where runoff enters the aquifer through dry wells. Contaminants come from materials that accumulate on impervious surfaces, including detritus, litter, and dust that contain chemicals from parking lot seal coat; motor vehicle emissions; road wear; decaying vegetation and animal waste; litter and garbage; de-icing chemicals; leaks and spills from vehicles; and rainfall (Grady, 1994).

Land-cover changes in Connecticut over the past two decades are documented in a series of satellite images (University of Connecticut, 2003) from 1985, 1990, 1995, and 2002. Land cover, when interpreted from satellite images, does not indicate what the land is used for, so land cover and land use are different but related concepts (table 8). The available geographic data are for land cover, and the available water-quality data are for land use. The "developed" land-cover category includes residential, commercial, institutional, and industrial land, transportation and infrastructure easements, golf courses, parks, and cemeteries. Areas of turf and grass also are considered developed because they are often associated with residential land.

Developed land in the SAS area is primarily a mixture of low-to-medium density residential areas, commercial areas, and transportation easements. Undeveloped land includes deciduous and coniferous forest, water, wetlands, quarries, utility rights-of-way, and low-density residential areas that have a tree cover. Water that recharged in developed land is the dominant ground water at shallow depths (table 9 and figs. 15–18). Water deeper in the aquifer tends to be a mixture of water that recharged in the uplands in undeveloped, developed, and agricultural land cover.

Land cover is not static, as discussed in the Introduction of this report, and changes in land cover complicate the relation between the residence time and chemical quality of ground water (figs. 15–18). From 1985 to 2002, developed land in the contributing recharge area to PSW-1 increased from 28 to 34 percent of the contributing recharge area, a process illustrated by the conversion of green and red symbols to black symbols between the left and right graphs on figures 15 to 18. Because of different rates in recharge in the contributing recharge area, the percentage of recharge to PSW-1 from developed land increased from 38 percent in 1985 to 45 percent in 2002. The percentage of agricultural land in the contributing recharge area was relatively constant from 1985 to 2002, about 25 percent, and the percentage of water from agricultural land in PSW-1 was about 20 percent.

Residence time indicates when water from a particular land use reaches PSW-1. Vertical profiles of shallow ground-water residence time generally show an increase of residence time with depth (figs. 15–18). Deeper in the aquifer, where the source of water is primarily the uplands, the profiles generally show a more uniform residence time with depth. Deviations from the general patterns are the result of different flow paths intersecting a well screen, and age inversions in wells (older water over younger water) can occur (figs. 15–18). Sampling water from any well yields a mixture of water recharged beneath different land covers and of different ages. In general, water is stratified by residence time and by land cover.

Historical concentrations can be reconstructed and projected into the future using information on land-cover changes and residence-time distributions. Examples are given here for nitrate and dissolved solids. Nitrate concentrations are highest in ground water beneath agricultural land, intermediate in developed areas, and lowest in undeveloped areas (table 10), and as land is developed, nitrate concentrations in shallow ground water can change. The conversion of agricultural land to developed land can cause a decrease in nitrate concentrations, whereas the conversion of undeveloped land to developed land can result in an increase. The

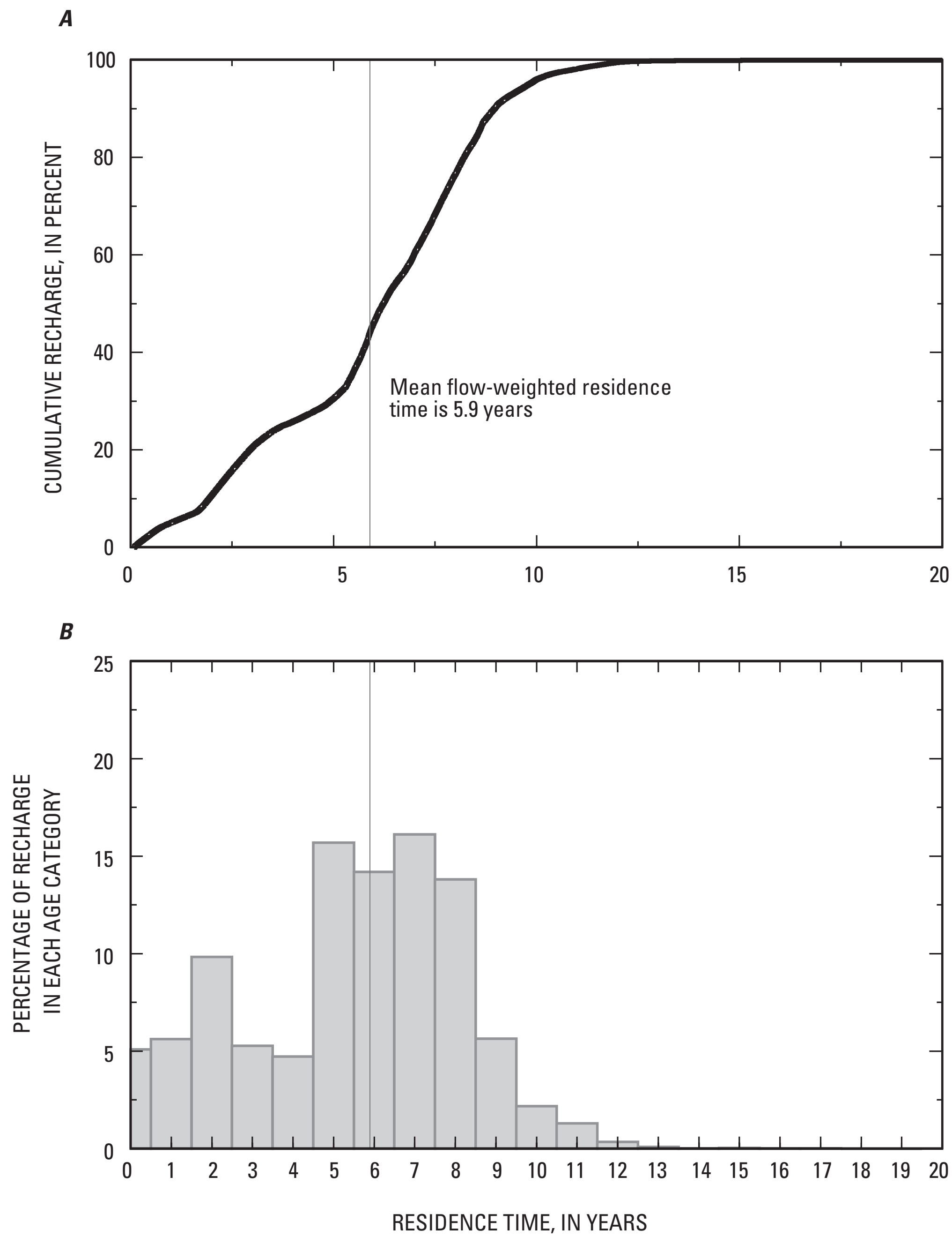


Figure 14. Cumulative frequency distribution and frequency distribution of simulated residence times, Woodbury, Connecticut.

Table 7. Selected characteristics of the contributing recharge area to well PSW-1, Woodbury, Connecticut.

Percentile or mean	Residence time of water, in years	Flow path length, in kilometers	Velocity of water, in meters per day	Residence time of water in glacial deposits, in years	Residence time of water in bedrock, in years
10th	1.93	0.34	0.02	1.30	0.00
25th	3.77	.51	.03	2.37	.00
50th	6.21	.98	.03	3.88	.46
75th	7.91	1.45	.04	5.91	3.11
90th	8.98	1.75	.05	7.31	4.91
Mean	5.91	1.01	.03	4.20	1.71

concentration of dissolved solids is a widely used indicator of overall water quality, and the median concentration of dissolved solids in ground water from developed areas is twice that in water from undeveloped areas (table 10). As land is developed, the concentration of dissolved solids in shallow ground water is likely to increase.

The changes in land cover in the contributing recharge area to PSW-1 are small, and the predicted changes in water quality are smaller than what can be measured reliably, but the following approach nonetheless illustrates the effect of changes in land cover on the quality of public-drinking water. Simulated concentrations of nitrate in water from PSW-1 decreased (though almost indistinguishably on fig. 19) from 1985 to 1995 because of the conversion of agricultural land to developed land, but then began to increase after 1995, also almost imperceptibly, because of the conversion of undeveloped land to developed land (fig. 19; gray line). A similar but slightly more pronounced effect can be shown by using the median nitrate concentrations reported by Grady (1994) for commercial land (fig. 19; red line). This provides a reasonable lower and upper bound on nitrate concentrations, because the contributing recharge area to well PSW-1 is mixed commercial and residential land. Concentrations of dissolved solids, on the other hand, increase over the entire period (fig. 20) as undeveloped and agricultural land is developed. Concentrations of other water-quality constituents such as chloride, alkalinity, and sulfate show patterns similar to those for dissolved solids.

The nitrate concentrations (Grady and Mullaney, 1998) assigned to general land-cover categories are not representative of individual sources of contamination. To extrapolate the contribution of nitrate from individual septic systems in the contributing recharge area, nitrate loading for each household was calculated. First, the nitrate load at PSW-1 was calculated. PSW-1 pumps 391.7 m^3/d, and the mean nitrate concentration measured during this study was 1.9 mg/L (C.J. Brown, U.S. Geological Survey, written commun., 2007), yielding a nitrate load of 744 g/d. If, in the worst case, septic systems were the only source of nitrate, each of the approximately 42 septic systems in the contributing recharge area contributes about 18 g/d of nitrate. Next, the average rate at which water flows through a septic system was calculated. Each household discharges about 0.60 m^3/d to their septic system, based on a household water use of 0.28 cubic meters per person per day (Hutson and others, 2005), 2.68 people/household (U.S. Census Bureau, 2007b), and a factor of 0.80 to account for consumptive losses, such as evaporation (Solley and others, 1998). The average concentration of nitrate needed to supply 18 g/d of nitrate at this flow rate is 30 mg/L, which is higher than that in samples from wells that are downgradient of septic-system drain fields (17 mg/L at WY90 and 19 mg/L at WY85). The calculated concentration is higher than the measured concentrations for two possible reasons: some of the nitrate load in PSW-1 is not from septic sources (which is likely, in part), or that the measured concentrations of nitrate are not representative of shallow ground water downgradient from septic systems. Water

Table 8. Relation of land-cover and land-use categories as used in this report, Woodbury, Connecticut.

Land cover	Land use
Developed	Urban
Turf and grasses	Urban
Other grasses and agriculture	Agriculture
Deciduous forest	Undeveloped
Coniferous forest	Undeveloped
Forested wetland	Undeveloped
Non-forested wetland	Undeveloped
Barren	Undeveloped

Table 9. Percentage of backward-tracked particles started from monitoring well screens by land-use category for the small-area simulation model, Woodbury, Connecticut.

[--, simulated position of well screen is dry; blank line separates clusters of wells; vertical well-screen position: Deep wells are screened in the lower third of glacial deposits; middle wells are screened in the middle third of glacial deposits; shallow wells are screened in the upper third of glacial deposits; bedrock means well is screened in bedrock]

Well name	Percentage of particles by land use			Vertical well-screen position
	Developed	Agriculture	Undeveloped	
WY69	90	0	10	Deep
WY70	100	0	0	Middle
WY71	--	--	--	Shallow
WY72	20	0	80	Deep
WY73	100	0	0	Middle
WY74	--	--	--	Shallow
WY75	100	0	0	Deep
WY76	--	--	--	Shallow
WY77	100	0	0	Deep
WY78	--	--	--	Shallow
WY79	100	0	0	Shallow
WY80	10	0	90	Deep
WY81	100	0	0	Shallow
WY82	60	30	10	Deep
WY83	100	0	0	Middle
WY84	90	0	10	Deep
WY85	100	0	0	Shallow
WY86	100	0	0	Bedrock
WY87	0	0	100	Bedrock
WY88	--	--	40	Deep
WY89	0	0	100	Deep
WY90	0	0	100	Deep
WY91	80	0	20	Middle
WY92	100	0	0	Shallow
WY93	0	0	100	Deep
WY94	0	0	100	Middle
WY95	100	0	0	Shallow
WY96	--	--	--	Middle
WY97	100	0	0	Bedrock
WY98	80	0	20	Deep
WY99	100	0	0	Middle
WY100	100	0	0	Middle
WY101	80	0	20	Deep
WY102	0	30	70	Shallow
WY106	0	0	100	Bedrock

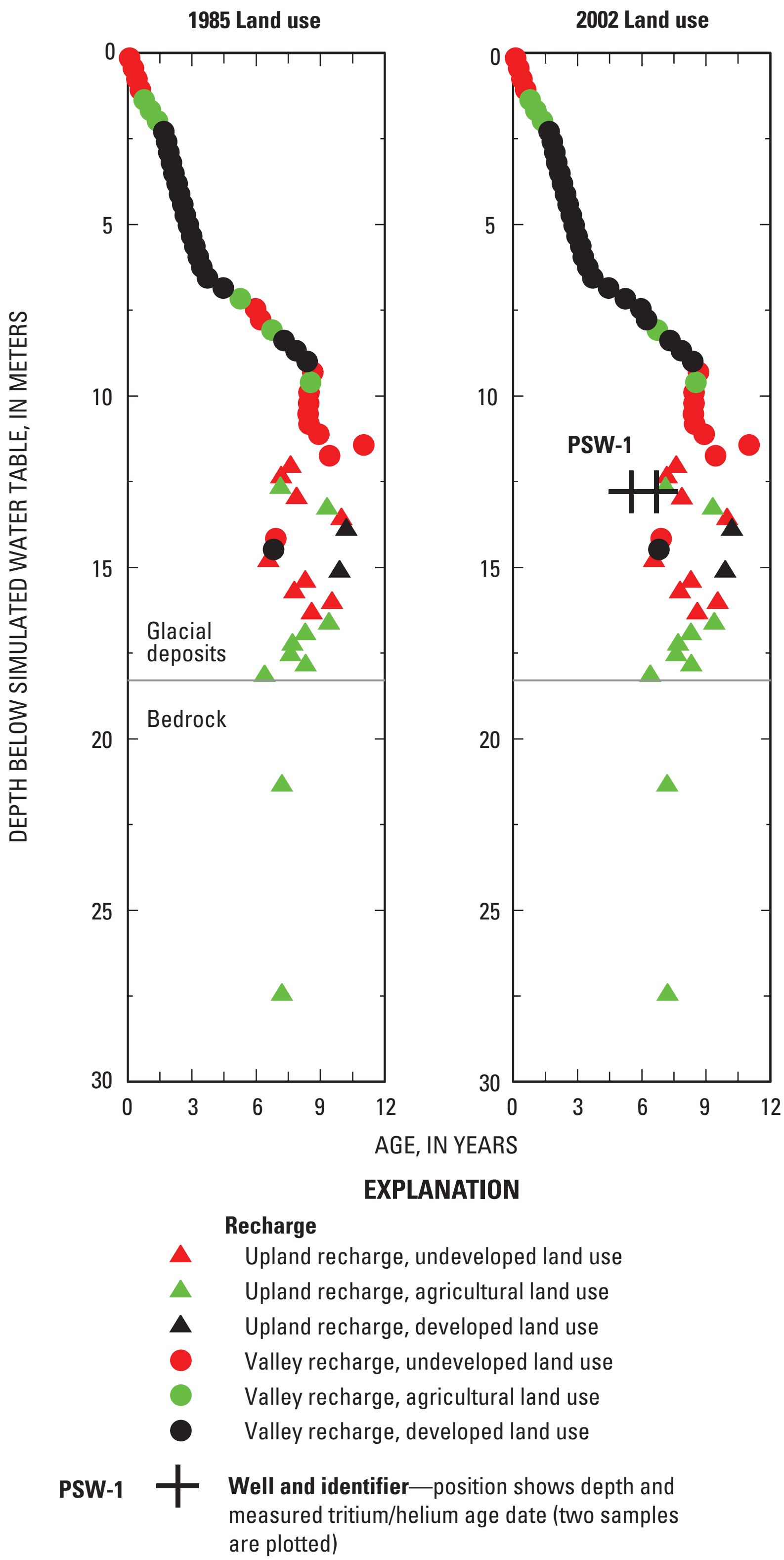


Figure 15. Relation of residence time, land use, and source of water for PSW-1, Woodbury, Connecticut.

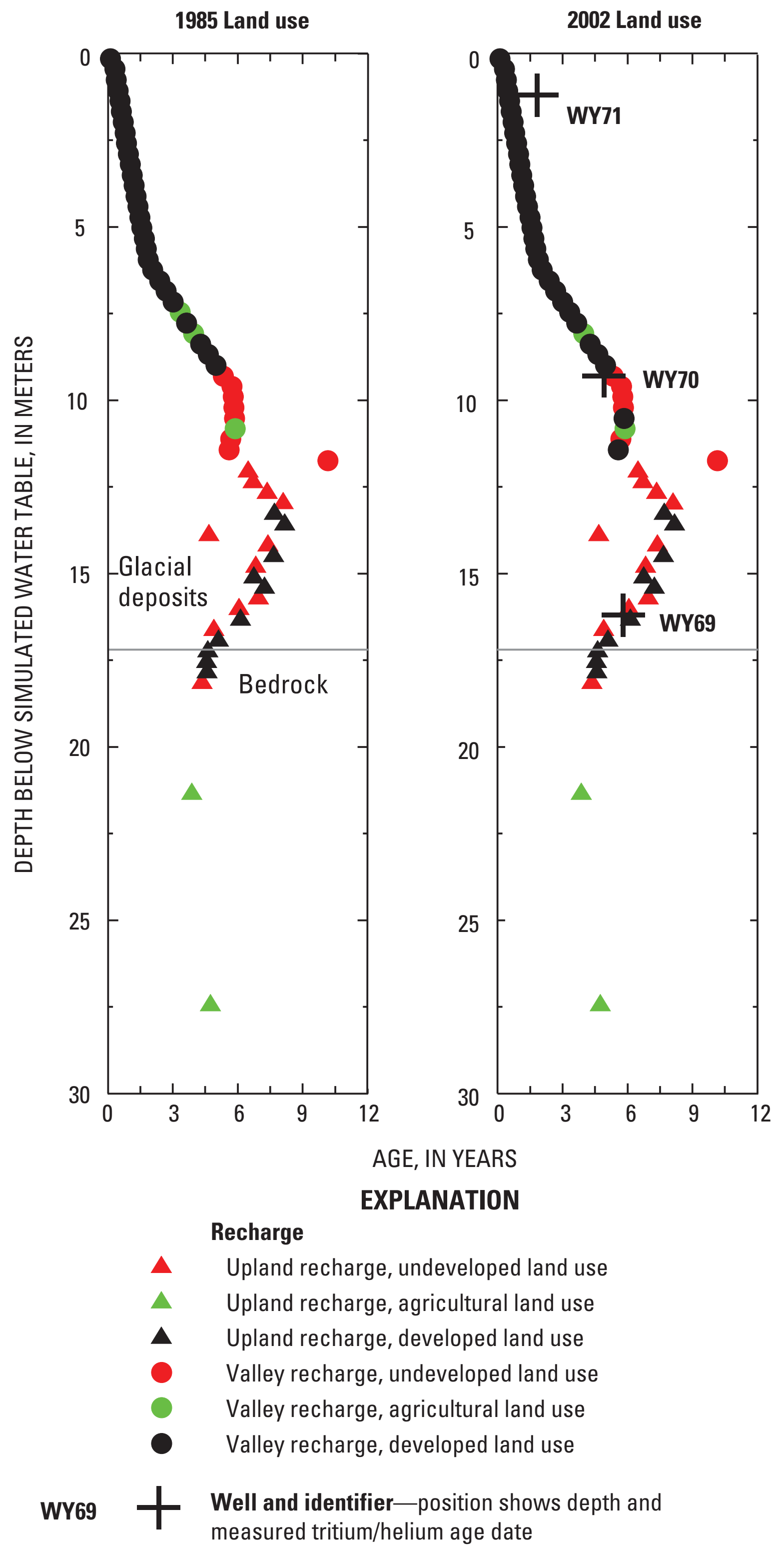


Figure 16. Relation of residence time, land use, and source of water for cluster of shallow, medium, and deep wells WY71, WY70, and WY69, Woodbury, Connecticut.

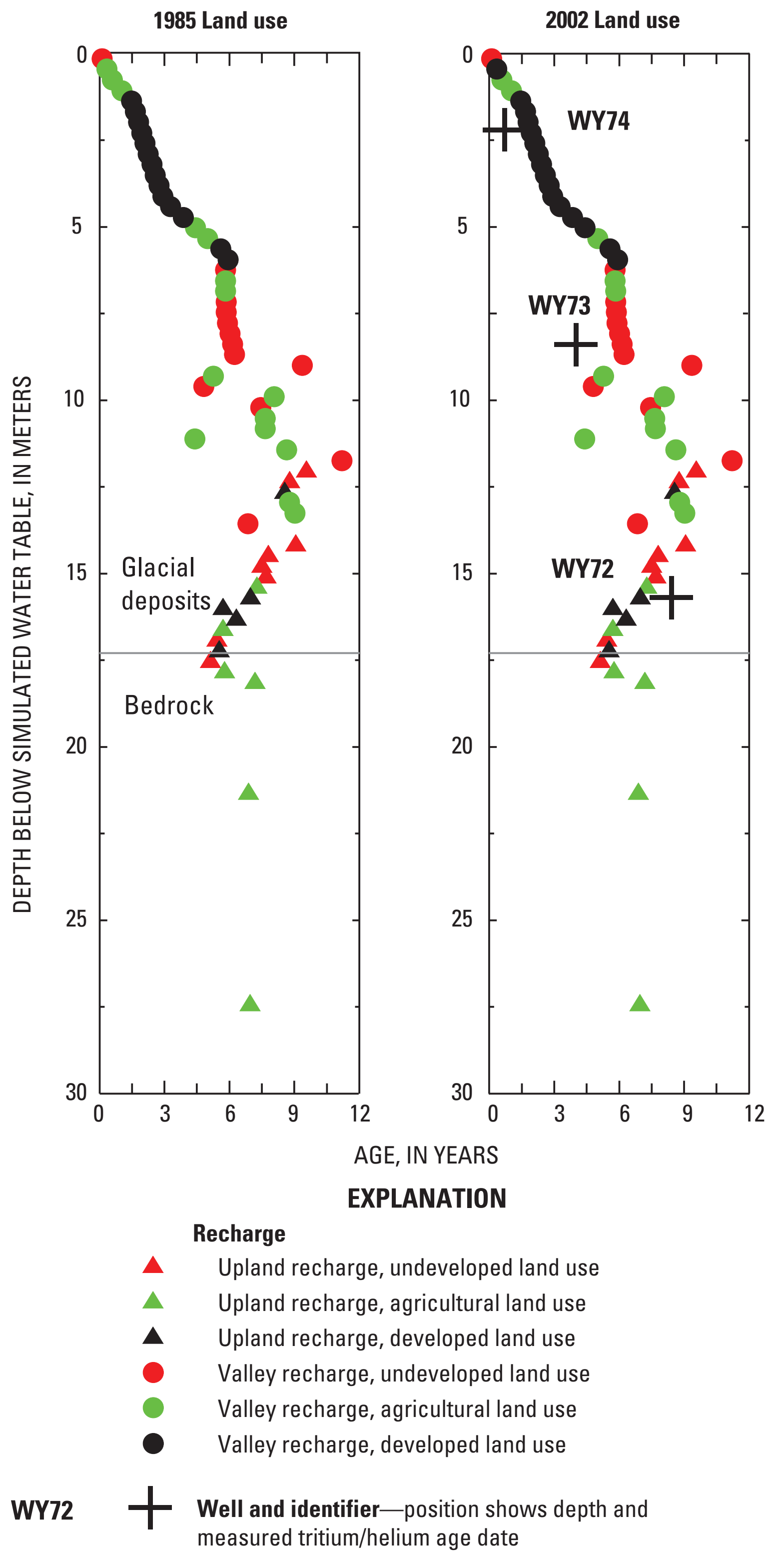


Figure 17. Relation of residence time, land use, and source of water for cluster of shallow, medium, and deep wells WY74, WY73, and WY72, Woodbury, Connecticut.

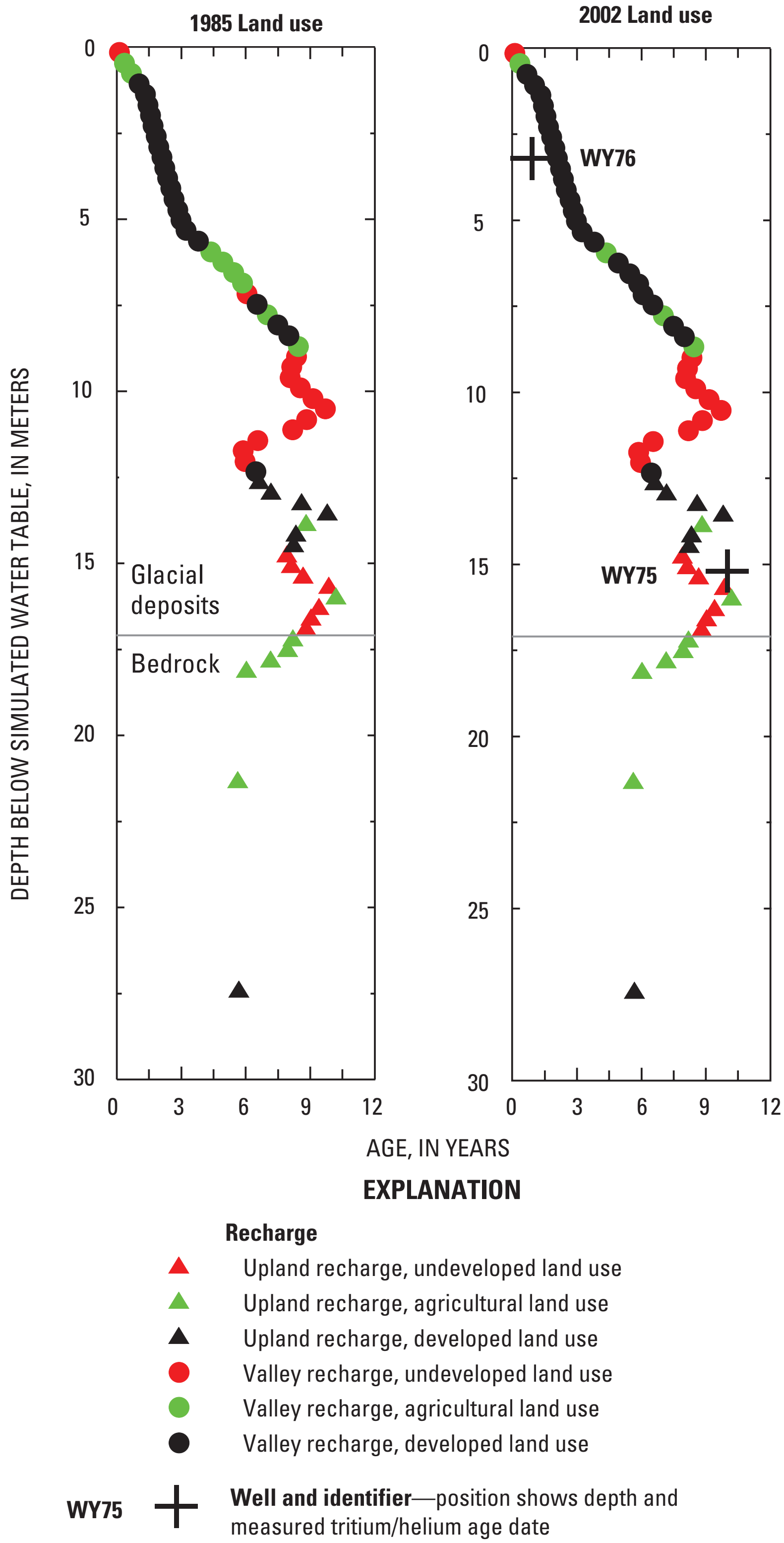


Figure 18. Relation of residence time, land use, and source of water for cluster of shallow and deep wells WY76 and WY75, Woodbury, Connecticut.

Table 10. Median concentrations of selected constituents used to represent various land uses, Woodbury, Connecticut.

[All concentrations in milligrams per liter; NR, not reported]

Constituent	Undeveloped	Agriculture	Developed	Commercial
Alkalinity	10	15	16	NR
Chloride	3.1	12	29	36
Nitrate as nitrogen	.14	3.8	1.1	3.4
Sulfate	48	58	104	19.5
Dissolved solids	98	162	217	241.5

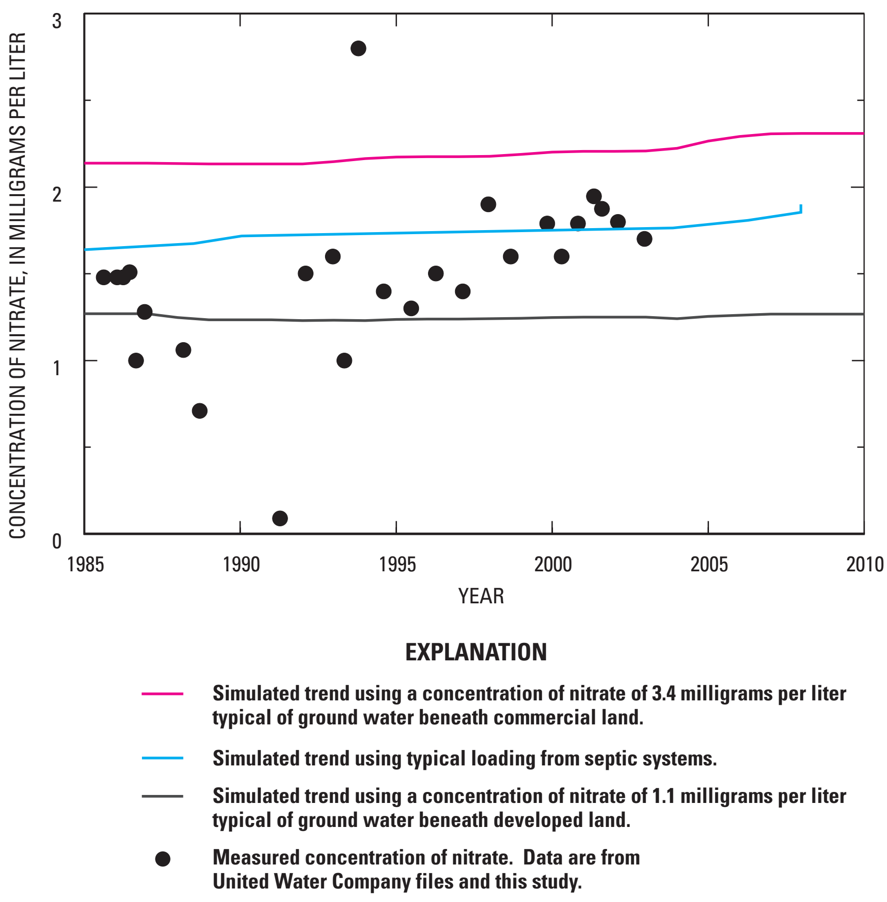


Figure 19. Simulated and predicted trend in concentration of nitrate in PSW-1, Woodbury, Connecticut.

samples from glacial aquifers at other sites downgradient from septic-system drain fields have higher nitrate concentrations. Robertson and others (1991) measured concentrations of 33 and 39 mg/L of nitrate at two domestic septic-system sites, and Harman and others (1996) measured concentrations of 50 mg/L of nitrate at a school septic-system site. Although nitrate concentrations can be lowered through denitrification in an anoxic environment, the aquifer system in the SAS area is mostly oxic (McMahon and others, in press), and the rate of denitrification can be assumed to be small. If septic systems are assumed to be the only source of nitrate in PSW-1, each additional septic system in the contributing recharge area contributes 0.045 mg/L of nitrate to PSW-1. The number of houses that could have septic systems (based on land records in the town of Woodbury) in the contributing recharge area follows the general population trend (fig. 21). The population and number of septic systems remained low until about 1950, when the rate of growth increased. The rate of increase may have leveled off in recent years as developable land has become scarcer. In recent years, septic systems have been added at the approximate rate of three systems every 4 years (fig. 21). If this rate continues in the future, nitrate concentrations can be predicted based on the individual system contribution of 0.045 mg/L nitrate (fig. 19).

The projected nitrate trends bracket most measured nitrate concentrations (fig. 19); however, there appears to be a trend in measured nitrate that is not captured by the predictions. Possible explanations include: (1) any real trend in the data is obscured by sampling methods and seasonal variation in nitrate values (for example, the concentrations of 0.9 and

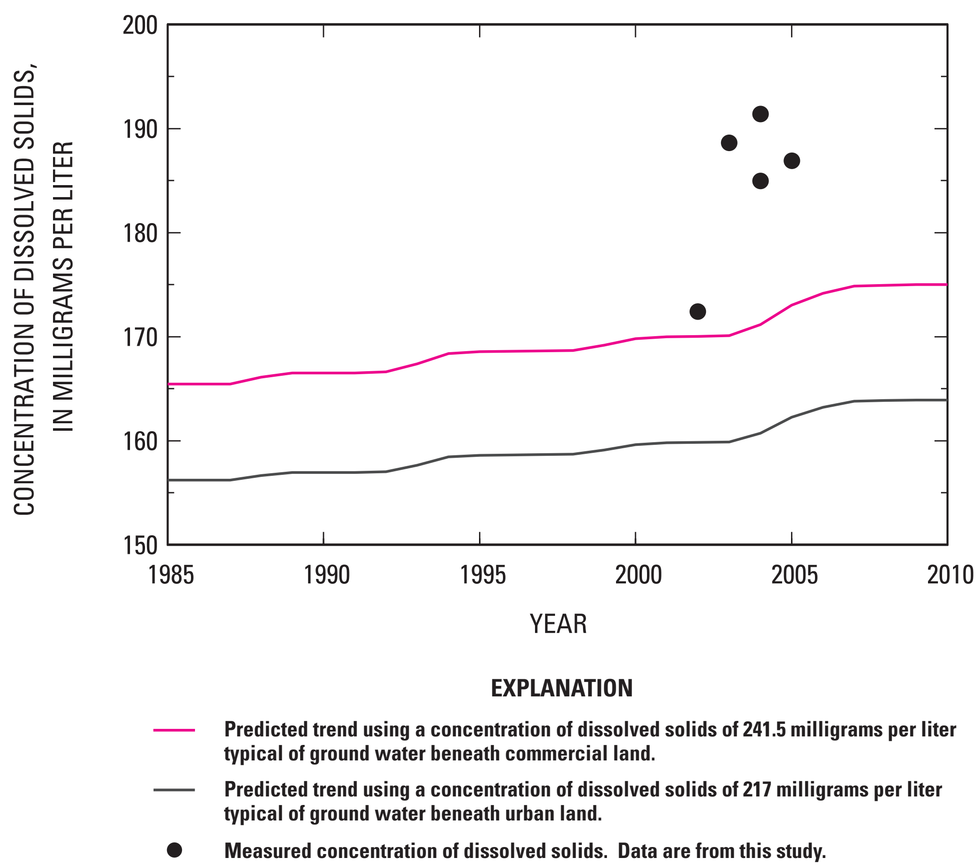


Figure 20. Simulated and predicted trend in concentration of dissolved solids in PSW-1, Woodbury, Connecticut.

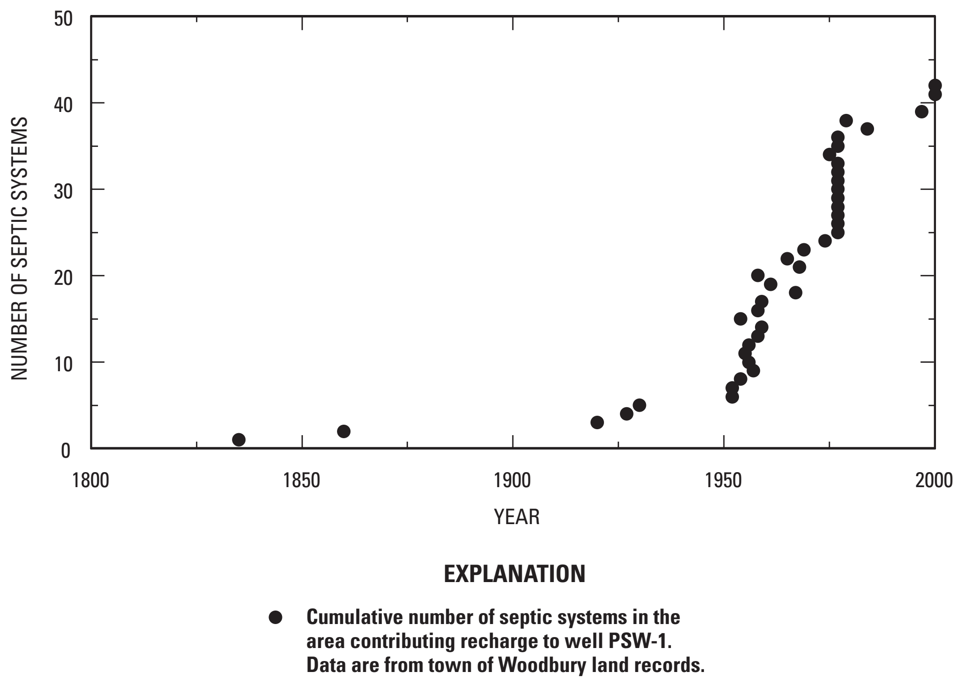


Figure 21. Number of septic systems in the contributing recharge area to well PSW-1, Woodbury, Connecticut.

3.0 shown are substantially outside the range of other values and may be in error), (2) there is a source of nitrate that is not accounted for (for example, a higher mean nitrate concentration in recharge from the multiple-residence septic systems that are present in the area), or (3) despite good indications that the model reproduces actual conditions well, small changes in the model could lead to simulations that predict disproportionate changes in water quality (for example, a large community septic system located just outside the contributing recharge area might be included in the contributing recharge area if the model were changed slightly). In any case, the predicted trends give reasonable estimates of nitrate concentration in PSW-1, and the predicted trends are small.

Model Uncertainties and Limitations

The discussion in the previous section is based on results of the SAS model, which was calibrated using a nonlinear regression technique. The regression produces information about how well the model is calibrated, in other words, about uncertainty in the estimated values. This uncertainty can be propagated through the contributing recharge area analysis using a Monte Carlo technique, as described by Starn and others (2000). The result of this procedure is a probabilistic contributing recharge area. The results show a greater degree of certainty in the delineation of the contributing recharge area near PSW-1, and, as one moves backward in time and space toward the recharge areas, the uncertainty in the model increases (the effect of uncertainty in model parameter estimates causes increasing uncertainty in the boundaries of the contributing recharge area) (fig. 22). The area that could possibly contribute water to PSW-1 is much larger than the contributing recharge area delineated in figure 13. The effect of this uncertainty on the reconstruction of nitrate history is that the 0.045 mg/L of nitrate that may be added by each additional septic system could include many more houses than were originally included in the contributing recharge area. Considering the probabilistic contributing recharge area in figure 22, the number of land parcels, and therefore, potential septic systems, could be double the initial estimate (fig. 23).

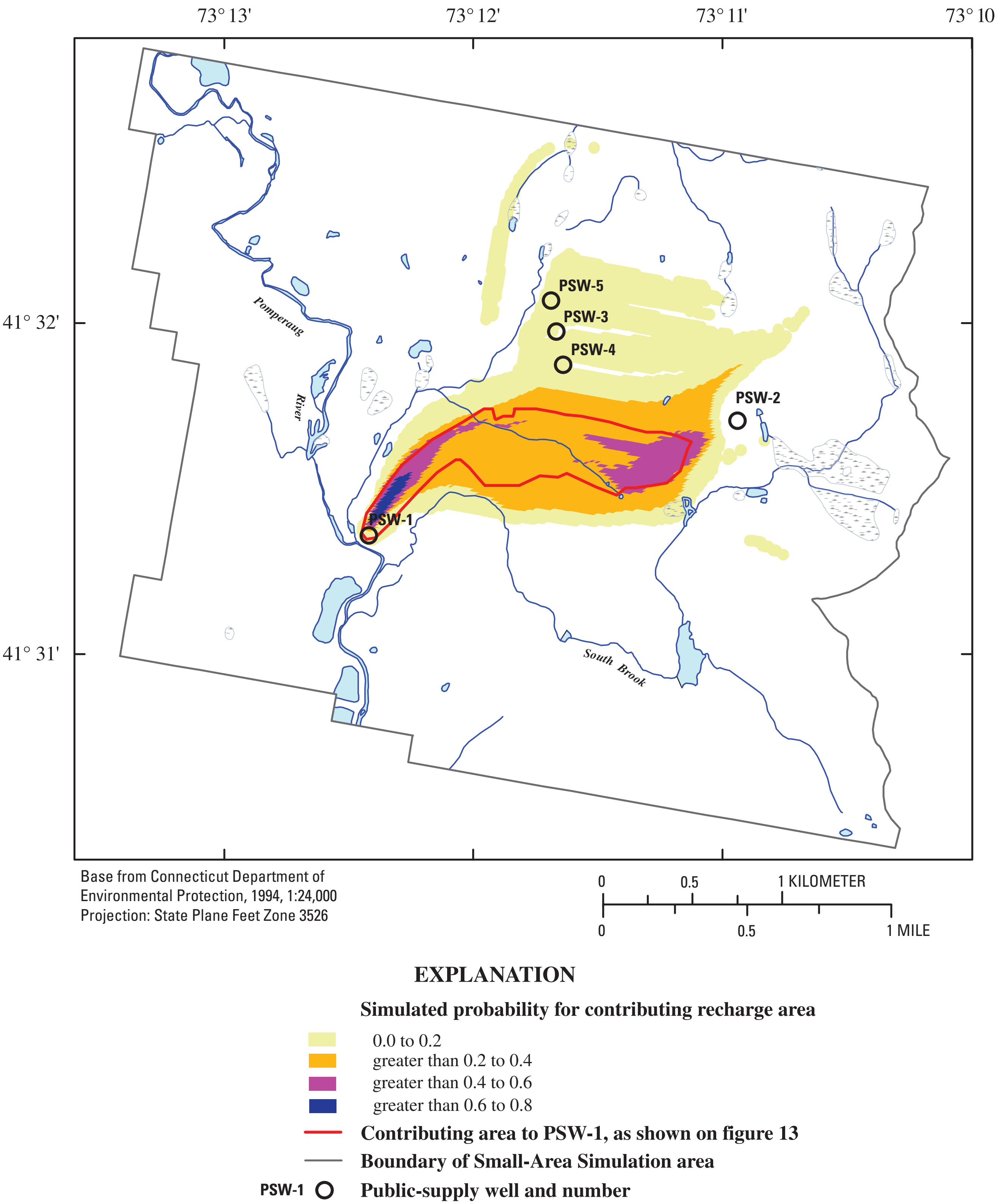


Figure 22. Probabilistic contributing recharge area to well PSW-1 for the small-area simulation model, Woodbury, Connecticut.

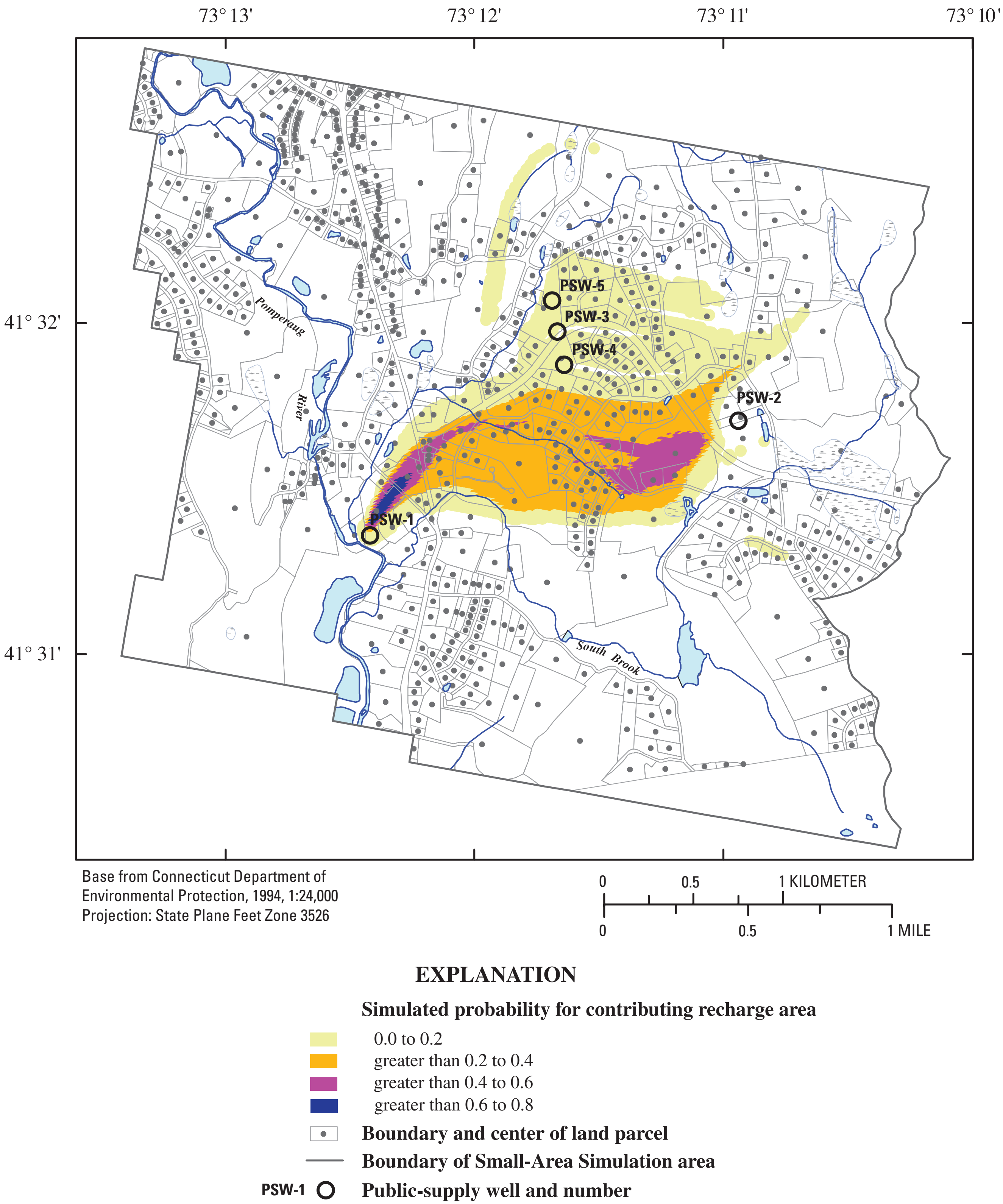


Figure 23. Probabilistic contributing recharge area and land parcels for the small-area simulation model, Woodbury, Connecticut.

Summary and Conclusions

In 2001, the U.S. Geological Survey's National Water-Quality Assessment (NAWQA) Program began an intensive study to assess the vulnerability of public-supply wells to contamination from a variety of compounds. As part of that study, a simulation model of ground-water flow in the aquifer at Woodbury, Connecticut, was constructed to estimate hydraulic parameters, water-budget components, and ground-water residence times. Advective particle tracking was used to estimate contributing recharge areas to monitoring wells and water-supply wells and to estimate the distribution of apparent ground-water residence time.

Hydraulic conductivities estimated with the model ranged from 6.2 to 18 meters per day (m/d) in glacial stratified deposits, 0.01 to 0.04 m/d in compact till, and 0.04 to 0.14 m/d in bedrock. Hydraulic conductivities that could not be estimated with the model were set at fixed values of 2.3 m/d in gravel overlying lake-bottom deposits, 0.3 m/d in loose surface till, and 0.3 m/d in Pomperaug River streambed material. Porosities also were set at fixed values of 0.3 in glacial stratified deposits, 0.1 in till, and 0.001 in bedrock.

Recharge to the uplands is dominated by precipitation. Fifty percent of upland recharge flows downward to bedrock, 31 percent to streams, and 16 percent to glacial stratified deposits in the valley. Recharge to glacial deposits in the valley is from precipitation (32 percent), upward flow from bedrock (34 percent), and flow across external boundaries (19 percent). Dry wells (storm drains that discharge to ground water) and losing stream reaches also are a source of ground water to the glacial deposits in the valley.

Water withdrawn from a well is a mixture of waters with different residence times, and a single residence time does not fully characterize the susceptibility of the well to recent contamination. The mean simulated flow-weighted residence time in water from PSW-1 is 5.9 years, which compares reasonably well with the apparent residence time measured using tritium/helium data of 5.5 and 6.6 years (samples for age dating were collected twice from this well). Water from PSW-1 is a mixture of water recharged in glacial stratified deposits in the valley that is younger than 5 years (34 percent) and water from the upland that is 5 to 20 years old (66 percent).

Vertical profiles of shallow ground-water residence time generally show an increase of residence time with depth. Deeper in the aquifer, where the source of water is primarily the uplands, the profiles generally show a more uniform residence time with depth. Deviations from the general patterns are the result of different flow paths intersecting a well screen, and age inversions in wells (older water over younger water) can occur. Sampling water from any well yields a mixture of water recharged beneath different land covers and of different ages. In general, the ground water is stratified by residence time and by land cover.

About 16 percent of the water withdrawn from PSW-1 comes from dry wells in a commercial area. The residence time for water that originated as recharge in the dry wells ranges from 1.6 to 3.8 years, and the mean residence time is 2.5 years. Dry wells are a fast pathway for water to enter the aquifer and provide a significant amount of water to PSW-1.

Changes in land cover in the contributing recharge area to PSW-1 are small, and predicted changes in water quality are smaller than what can be measured reliably, but the approach illustrates the effect of changes in land cover on the quality of public-drinking water. For example, nitrate concentrations decreased from 1985 to 1995 because of the conversion from agricultural land to developed land, but then began to increase after 1995 because of the conversion of undeveloped land to developed land. Concentrations of dissolved solids, on the other hand, increase over the entire period as land is developed.

One of the main sources of nitrate in the area is septic-tank drain fields from single- and multiple-family developments. The estimated nitrate loading rate from a single family septic system is 18 grams per day. If each household in the contributing recharge area to PSW-1 contributes nitrate at that loading rate to the well PSW-1, each additional septic system in the contributing recharge area is responsible for a 0.045-milligrams per liter increase in nitrate at PSW-1.

Uncertainty in the SAS model can be propagated through the contributing recharge area analysis using a Monte Carlo technique. There is a greater degree of certainty in the delineation of the contributing recharge area near PSW-1, and as one moves toward the recharge areas, the uncertainty in the model increases. The area that possibly contributes water to PSW-1 is much larger than the contributing recharge area delineated using the optimal parameter estimates.

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