HYDROGEOLOGY, SIMULATED GROUND-WATER FLOW, AND GROUND-WATER QUALITY AT TWO LANDFILLS IN BRISTOL, VERMONT

By Thomas J. Mack

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CONVERSION FACTORS, VERTICAL DATUM, AND ABBREVIATED WATER-QUALITY AND GEOPHYSICAL UNITS

Multiply	By	To Obtain
acre	4,047	square meter
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
foot (ft)	0.3048	meter
foot per day (ft/d)	0.3048	meter per day
gallon (gal)	3.785	liter
inch (in.)	25.4	millimeter
inch per year (in/yr)	25.4	millimeter per year
mile (mi)	1.609	kilometer
million gallons (Mgal)	3,785	cubic meter
million gallons per day (Mgal/d)	0.04381	cubic meter per second
square foot (ft^2)	929.0	square centimeter
square foot per day (ft^2/d)	0.09290	square meter per day
square mile (mi^2)	2.590	square kilometer

Water temperature is given in degrees Celsius (°C), which can be converted to degrees Fahrenheit (° F) by use of the following equation: °F = 1.8 (°C) + 32

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

ABBREVIATED WATER-QUALITY AND GEOPHYSICAL UNITS USED IN REPORT

In this report, the concentration of a chemical in water is expressed in milligrams per liter (mg/L) or micrograms per liter (µg/L). Milligrams per liter is a unit expressing the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of water; 1,000 µg/L is equivalent to 1 mg/L.

Specific conductance of water is expressed in microsiemens per centimeter at 25 degrees Celsius (µS/cm). This unit is equivalent to micromhos per centimeter at 25 degrees Celsius (µmho/cm), formerly used by the U.S. Geological Survey.

In this report, the electrical conductivity of Earth materials is expressed in millisiemens per meter (mS/m); 10 µS/cm is equivalent to 1 mS/m. This unit (mS/m) is equivalent to millimhos per meter, formerly used by the U.S. Geological Survey. The electrical resistivity of Earth materials is expressed in ohm-meters (ohm-m); 1 ohm-m is equivalent to the inverse of 1 siemen per meter. The frequency of radio transmissions, in thousands of cycles per second, is expressed in kilohertz (kHz).

HYDROGEOLOGY, SIMULATED GROUND-WATER FLOW, AND GROUND-WATER QUALITY AT TWO LANDFILLS IN BRISTOL, VERMONT

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Abstract

A study was done by the U.S. Geological Survey, in cooperation with the Vermont Agency of Natural Resources, to describe the hydrogeology of unconsolidated deposits, simulate ground-water flow, and to describe ground-water quality at two landfills in Bristol, Vermont. The study area is characterized by a glacial delta more than 200 feet thick, having an exposed face about 150 feet high, on the west flank of the Green Mountains. A municipal landfill, more than 20 years old, is atop the delta in a coarse-grained deposit, where depths to water range from 30 to 130 feet below land surface. A private landfill, also more than 20 years old, is at a lower altitude next to the delta in medium- to fine-grained sediments as much as 100 feet thick, where the depth to water is from 10 to 50 feet. A sand, silt and clay lens next to the delta possibly extends into the delta and acts as a confining unit, separating an upper unconfined aquifer and a lower confined aquifer. The horizontal hydraulic conductivity of the coarse-grained deltaic sands and gravels was estimated, by slug and specific-capacity tests, to range from 120 to more than 250 feet per day. The horizontal hydraulic conductivity of fine- to medium-grained sands was estimated to range from 10 to 30 feet per day and that of till to be about 1 foot per day.

Recharge to the glacial aquifers is predominantly from ground-water leakage from the underlying bedrock aquifer and accounts for about 80 percent of the total recharge. Infiltration of precipitation, approximately 14 inches per year, accounts for about 10 percent of total recharge, and inflow from adjacent unconsolidated aquifers accounts for the remaining 10 percent. High upward gradients of 0.03 to 0.3 are evidence that most recharge is as upward ground-water leakage from bedrock.

Surface electromagnetic geophysical terrain surveys, electromagnetic induction and very low frequency methods were used to identify areas underlain by electrically conductive landfill-leachate plumes. Natural-gamma radiation and electromagnetic borehole geophysical logs were used to identify and vertically delineate landfill-leachate plumes in the aquifers. Leachate plumes were near the water table and ranged in thickness from less than 5 to nearly 20 feet.

A two-layer ground-water-flow model of the upper and lower glacial aquifers was developed and calibrated under steady-state conditions, approximated by the conditions observed during April and May 1991. Advective ground-water-flow paths were calculated by a particle-tracking analysis. Simulated paths of landfill leachate are generally concentrated in the upper aquifer, or upper part of an aquifer, at the private and municipal landfills. Ground water from beneath the private landfill discharges as ground-water outflow to the aquifer by way of a gap in the bedrock ridge west of that landfill. Some ground water beneath the private landfill discharges to a brook west of the private landfill. Most ground water beneath the municipal landfill discharges as ground-water outflow to the southwest at a gap in the bedrock ridge west of the private landfill. This ground water follows a longer path before discharging at the gap in the ridge than does ground water from beneath the private landfill.

Ground water beneath the east end of the municipal landfill possibly flows southward to eventually discharge at the southern model boundary in the Bristol Flats area. Flow paths in the upper aquifer beneath both landfills remained in the upper layer, and paths in the lower aquifer generally flow into the upper aquifer.

Water samples degraded by landfill leachate generally had a specific conductance greater than 400 μS/cm (microsiemens per centimeter at 25 degrees Celsius); the median was about 700 μS/cm. The maximum specific conductances were 2,920 μS/cm at the municipal landfill and 1,980 μS/cm at the private landfill. Native ground water had a median specific conductance of 355 μS/cm, and ranged from about 200 to 400 μS/cm.

Leachate degraded water from the landfills contained mean concentrations of common constituents and trace elements that were 1.5 to 10 times the background concentrations. Trace elements detected in leachate from the landfills included copper, nickel, zinc, cobalt, lead, and arsenic. Nickel was the most commonly detected trace element (19 detections), followed by zinc (8 detections). Some chemical constituents in water samples appear to be present in lower concentrations than were found during earlier investigations.

Ten volatile organic compounds were detected at four observation wells associated with the municipal landfill, and three volatile organic compounds were detected at two observation wells associated with the private landfill. No one volatile organic compound was consistently detected, and concentrations generally were at or near detection limits. Volatile organic compounds were detected fewer times and at lower concentrations than in previous investigations. Volatile organic compounds were not detected in samples of native water.

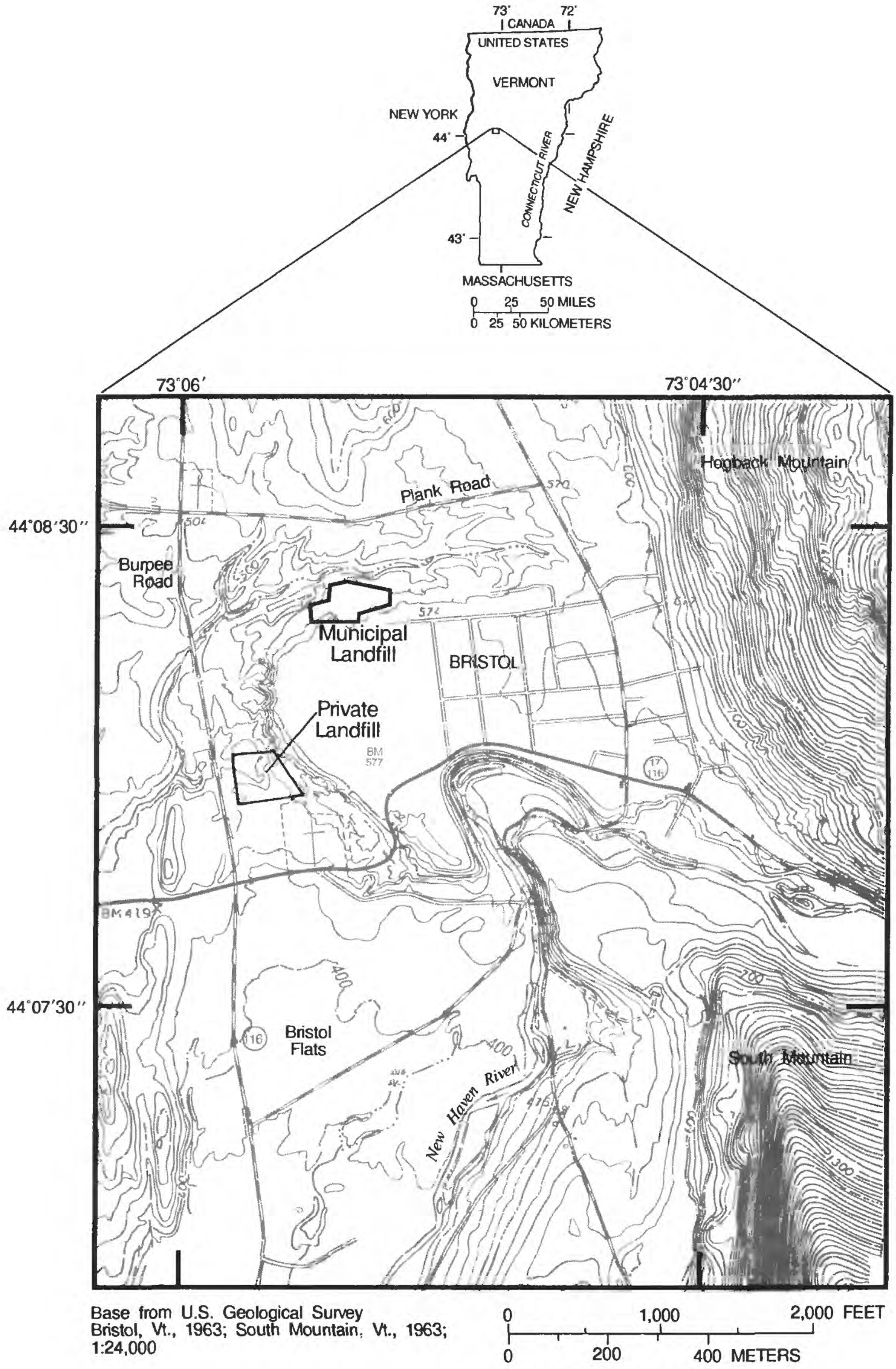
INTRODUCTION

Two solid-waste landfills, one privately owned and one municipally owned, have been used in Bristol, Vermont (fig. 1) for more than 20 years. Most of the solid wastes generated by domestic and industrial activities throughout Vermont have been disposed of in unlined landfills, which can affect the quality of water in underlying aquifers. Other investigations have demonstrated that landfills can degrade local ground-water quality (Kimmel and Braids, 1980; Wexler, 1988a). The demand on Vermont's ground-water resources is continually increasing. To plan for future ground-water use, water managers and planners will need to obtain data on ground-water-flow systems and the effects of landfills on ground-water quality as efficiently and effectively as possible.

The Vermont Agency of Natural Resources, Solid Waste Division (VANR) and the U.S. Geological Survey (USGS) began a cooperative study to examine the hydrogeology and ground-water quality at the two landfills in Bristol. The study, which was done from April 1990 to September 1992, demonstrates the application and analysis of geophysical, hydrogeologic, and numerical ground-water-flow simulation techniques at the landfills.

The study area consists of the two landfills (fig. 1) and a surrounding 2-mi^2 area. The study area is on the west flank of Hogback and South Mountains, which are part of the Green Mountains (fig. 1). The village of Bristol is on a prominent glacial delta; an exposed face of the delta rises more than 150 ft above the New Haven River Valley bottom. The delta was formed by glacial meltwater that once flowed between Hogback and South Mountains into glacial Lake Fort Anne. The private landfill, at the southwestern edge of the delta, is at the base of the deltaic sequence just north of a cemetery in an area marked by sand and gravel excavations. The municipal landfill is within the deltaic sequence at the northwestern edge of the delta.

The private landfill (fig. 1), formerly a sand and gravel excavation pit, has been operated by several owners since it opened in 1968; in 1992 it was approximately 10 acres in area. The landfill received not only municipal wastes from 10 surrounding towns but also industrial wastes from local manufacturing companies. Landfilling proceeded eastward and westward from the center of the former excavation site. During this study, a 1-acre scrap-metal pile, located in a gravel excavation pit a few hundred feet north of the landfill, was considerably reduced in size by trucking offsite. When the study began, the private landfill was inactive but filling resumed in 1991 and the landfill is scheduled to be capped within a few years.



Base from U.S. Geological Survey
Bristol, Vt., 1963; South Mountain, Vt., 1963;
1:24,000

CONTOUR INTERVAL 20 FEET
NATIONAL GEODETIC VERTICAL DATUM OF 1929

Figure 1. Locations of landfills and extent of study area, Bristol, Vermont.

The municipal landfill (fig. 1) has been owned and operated by the town of Bristol since 1968 and may have informally received solid waste before 1968. This 8-acre landfill was also a former sand and gravel excavation pit and received only municipal wastes since its operation by the town. Filling began at the western end of the landfill, and as of 1992, the surface was nearly completed to the natural grade. The eastern end of the excavation pit occupied by the landfill has been used primarily for temporary storage of scrap metals, and in the past, was used as a brush and wood-burning area. A recycling program, begun in 1990, has reduced the rate of refuse brought to the landfill.

Purpose and Scope

This report describes the hydrogeology, simulation of ground-water flow, and water quality of the glacial aquifers that underlie two landfills in Bristol. Specifically, this report (1) describes the use of various hydrogeologic methods for studies of landfills in glaciated terrain, and (2) describes the ground-water-flow patterns and ground-water quality beneath the landfills. The study is limited to the unconsolidated glacial deposits of this area and does not include the underlying bedrock aquifer.

Previous Investigations

Previous investigations of the study area included a study of the private landfill for the VANR (Marshfield Engineering Services, 1979) that was primarily concerned with the design and operation of the landfill but it did address some aspects of hydrogeology and water quality. The investigation included installation of seven monitoring wells, measurements of ground-water levels, and water-quality sampling. Three of the wells installed were downgradient from the landfill, and two were downgradient from the scrap-metal piles north of the landfill. The remaining wells were destroyed before the USGS investigation began.

Investigations of the municipal landfill by the VANR (Hackbarth, 1980) and by a consultant to the town of Bristol (Johnson Company, 1989) also were done. These two studies resulted in design and operation reports that included some hydrogeologic data. The first investigation (Hackbarth, 1980) resulted in the installation of three observation wells at the landfill site. Wells were not installed downgradient from the landfill because drilling is difficult in this area. During the second investigation (Johnson Company, 1989), three wells were installed downgradient from the municipal landfill in the glacial delta deposits. A terrain-conductivity survey was used to determine observation-well placement, and slug tests were done to estimate aquifer characteristics. Both investigations included water-level measurements and water-quality sampling as part of the data collection.

A geologic report on the Burlington-Middlebury region (Stewart, 1973) describes the bedrock geologic setting and surficial geology and gives appraisals of ground-water availability. A ground-water favorability map of the Otter Creek Basin, which includes the study area, was produced by Hodges (1967) and includes Stewart's work (1973). The Bristol delta and surrounding area is less than 15 mi from Middlebury College, in Middlebury, Vt., and has been the subject of senior thesis's by students of the geology department (Brewster Baldwin and John Schmidt, Middlebury College, oral commun., 1991).

Acknowledgments

The author thanks James Surwilo of the VANR for his support and assistance, particularly with water-quality sampling and analysis. The author thanks officials of the town of Bristol and numerous landowners for their cooperation and for granting access to their properties.

METHODS OF STUDY

Methods of study described in the following sections include compilation and analysis of available data and collection and analysis of hydrogeologic data. Data collected includes surficial geologic mapping, surface geophysical surveys, lithologic logs of USGS test holes, borehole geophysical logs, water-level measurements, aquifer tests (slug tests), and streamflow measurements. Water-quality samples were collected and analyzed by the VANR.

The order that the methods of study are discussed in this report are generally the most useful order of application to hydrogeologic contaminant studies in glaciated areas. Each method provides a more focused location of further data-collection activities and more specific or detailed information. Data compilation consolidates available data to locate initial study efforts and avoid redundant efforts. Surficial geologic mapping provides an initial view and insight into the deposits present. Surface geophysical surveys provide insight into aquifer properties or geometry.

Data Compilation

Subsurface data on ground-water levels, stratigraphy, and depth to bedrock (James Ashley, Vermont Department of Environmental Conservation, Water Supply Division, written commun., 1991) were extracted from more than 230 drillers' logs of domestic wells in Bristol reported to the State since 1974. The locations of domestic wells used in this study were checked for accuracy. Other subsurface data included test wells at the Bristol municipal landfill (Hackbarth, 1980; Johnson Company, 1989), test wells at the private landfill (Marshfield Engineering Services, 1979), and test wells for a proposed leachfield (Brent Whitney, Thermo Engineering, written commun., 1991).

Surficial Geologic Mapping

The areal extent of unconsolidated deposits was mapped and described at vertical faces of gravel pits and at hand-dug excavations. Contacts between sedimentary units were traced between pits by projection and interpolation. This information is used in construction of geologic sections and development of conceptual models of the ground-water system underlying the study area.

Surface Geophysical Surveys

Surface geophysical surveys consisted of seismic-refraction profiles and electromagnetic terrain surveys. Seismic-refraction surveys were used to determine the depth to water table and bedrock surfaces. Electromagnetic terrain surveys consisted of electromagnetic terrain conductivity (EM) surveys and very low frequency terrain resistivity (VLF) surveys. Both were used to measure contrasts in electrical properties of aquifer materials and ground water. Interpretation of EM surveys generally requires additional subsurface data.

Seismic Refraction

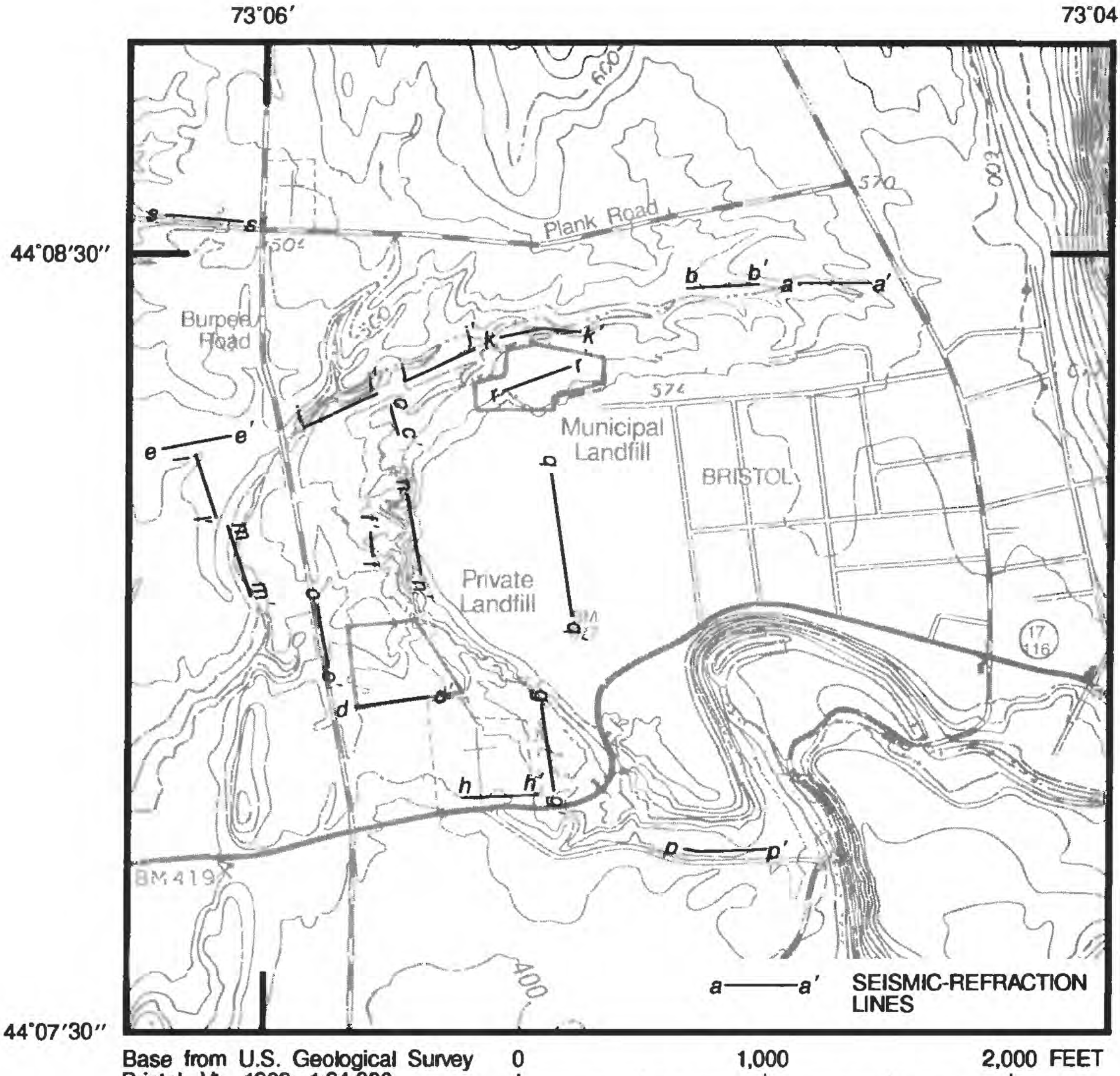
Seismic-refraction surveys were done at 19 sites (fig. 2) to determine depth to water table and bedrock surfaces. The interpreted seismic-refraction profiles of 15 of those 19 sites (appendix 1) provided information for water-table and bedrock-surface maps and were used to help determine placement of test wells and aid in the interpretations of other geophysical surveys. Interpreted seismic-refraction profiles were not completed at four of the survey sites (*d-d'*, *e-e'*, *j-j'*, and *q-q'*; fig. 2) because the hydrogeologic conditions at these sites did not meet the requirements necessary for data interpretation. Two physical requirements are fundamental to a successful seismic survey. The first is that layer velocities increase with depth; the second is that individual layers are thick enough (which generally means that layer thicknesses must increase with depth) for the refracted energy wave to be detected by surface geophones. At sites *j-j'* and *q-q'*, the saturated zone was too thin—less than the thickness of the unsaturated zone—to be detected (such a zone is called a blind zone). At sites *d-d'* and *e-e'*, a high-velocity saturated zone overlies a low-velocity unsaturated zone (this is called a velocity inversion).

The results of seismic-refraction profiles were interpreted by use of the methods described by Haeni (1988). A 12-channel, signal-enhancement seismograph was used to measure the time required for a sound wave to travel from a shot point, usually an explosive charge, to each of the 12 geophones in a geophone spread. Most surveys were done with 50-foot spacing between geophones and with three to five shotpoints per geophone spread. Geophone spacing is critical and must be designed with the depths of interest in mind; however, the number and spacing of shotpoints is also important to the integrity of the seismic data collected. Shotpoint and geophone setups will differ throughout an investigation because depths to water table and bedrock surfaces are rarely uniform.

Data were interpreted by use of a seismic-refraction inverse computer modeling program (SIPT) developed by Scott (1977) and based on a delay-time and iterative ray-tracing technique. Haeni (1988) provides a description and guidelines for use of this program. SIPT is designed to handle complex field setups and to develop interpretations for multilayer, dipping-bed geologic settings.

Electromagnetic Survey Methods and Application

EM surveys measure the combined conductivity, or resistivity, of the earth materials and any water or other fluid present. In general, dry sand and gravel is resistant, whereas silt and clay is conductive. Any given earth material is more conductive when saturated. The specific conductance of distilled water, 1 µS/cm (Hem,



Base from U.S. Geological Survey Bristol, Vt., 1963, 1:24,000

CONTOUR INTERVAL 20 FEET
NATIONAL GEODETIC VERTICAL DATUM OF 1929

Figure 2. Locations of seismic profiles, Bristol, Vermont.

1989, p. 68), is indicative of a resistant substance; specific conductance increases with the addition of ions. Ambient uncontaminated or native ground water underlying the Bristol area has a specific conductance of about 200 to 400 µS/cm. New Haven River water generally has a specific conductance of 100 to 200 µS/cm. Water flowing through a landfill or other contaminant source, typically dissolves and incorporates more ions. Leachate contaminated ground water from landfills may have a specific conductance greater than 1,000 µS/cm (Kimmel and Braids, 1980; Wexler, 1988a). Therefore, because of the contrast between the specific conductance of leachate and native ground water, EM techniques can be used in the detection of contaminant plumes. In the optimal EM survey, measurement points would be arranged in a regular grid pattern; however, this ideal arrangement is not always possible because of cultural interferences (metal fences, buried or overhead cables, pipelines, vehicles, and buildings), limited access, or large topographic relief.

Cultural interferences can hamper an investigation by obliterating the measurements at a specific site. Power transmission lines can strongly affect a survey because they produce interfering magnetic fields. Above-ground structures affect the EM survey because magnetic fields extend in a spherical pattern above and below the ground.

When the stratigraphy of the area surveyed is highly variable, the data interpretation becomes complex and may require modeling to interpret the results. Forward and inverse computer-modeling programs are available for EM terrain conductivity data (Granthem and others, 1987); a forward computer model is available for VLF interpretations (Granthem and others, 1986). Forward models simulate the response of the EM survey equipment to thickness and electrical conductivity of a layered earth model (which may approximate the hydrologic model of the system). In practice, the number of earth layers, layer thickness, and electrical conductivity is adjusted until the simulated instrument response approximates the observed instrument response. In some cases, several combinations of earth layers and electrical conductivities can reproduce the observed instrument response. Forward modeling can produce a unique solution however, at least one variable (such as layer thicknesses or other hydrogeologic information) must be known. Inverse modeling can optimize the interpretation and produce a better simulation with less trial and error. If some hydrogeologic information is known, the model results can be used to determine whether the instrument response at a particular site is affected by the hydrogeology of the area or by a conductive zone of contaminated ground water.

Very Low Frequency Terrain Resistivity

Very low frequency (VLF) radio signals are transmitted from naval communications stations for use in submarine navigation systems. Previously, these signals were used in geophysical surveys for locating ore deposits. VLF resistivity surveys have been used to locate fracture zones in dolomite (Yager and Kappel, 1987) and to detect ground-water contamination at a landfill in Connecticut (Grady, 1989). The VLF equipment used in this investigation consists of a lightweight handheld receiver and two probes that function as receiver coils (Geonics, 1979a, b).

A VLF terrain-resistivity survey uses a 15 to 25 kHz (kilohertz) VLF signal. The transmitting stations used in this study were Cutler, Maine (24.0 kHz), Seattle, Wash. (24.8 kHz), and Annapolis, Md. (21.4 kHz). The VLF signal consists of a horizontal radial electromagnetic field at the Earth's surface: and two components that penetrate the Earth's surface; a vertical electrical field and a horizontal magnetic field. As the terrain resistivity increases, the horizontal electric field increases. VLF equipment measures the ratio of and the phase angle between the horizontal electric field and the horizontal magnetic field and is calibrated to read apparent terrain resistivity in ohm-meters.

The effective depth of exploration depends mostly on terrain resistivity and only slightly on operating frequency. The expected exploration depths for a given terrain resistivity at operating frequencies of 15 to 25 kHz are listed in table 1.

Where the terrain resistivity is high, such as in a thick unsaturated sand and gravel unit, the VLF exploration is effective to considerable depths. The effectiveness of the VLF survey decreases in an electrically conductive zone at the ground surface, such as a surficial clay layer. The VLF survey effectively penetrates the materials present until a conductive zone is reached. Any material beyond that conductive zone is virtually undetected.

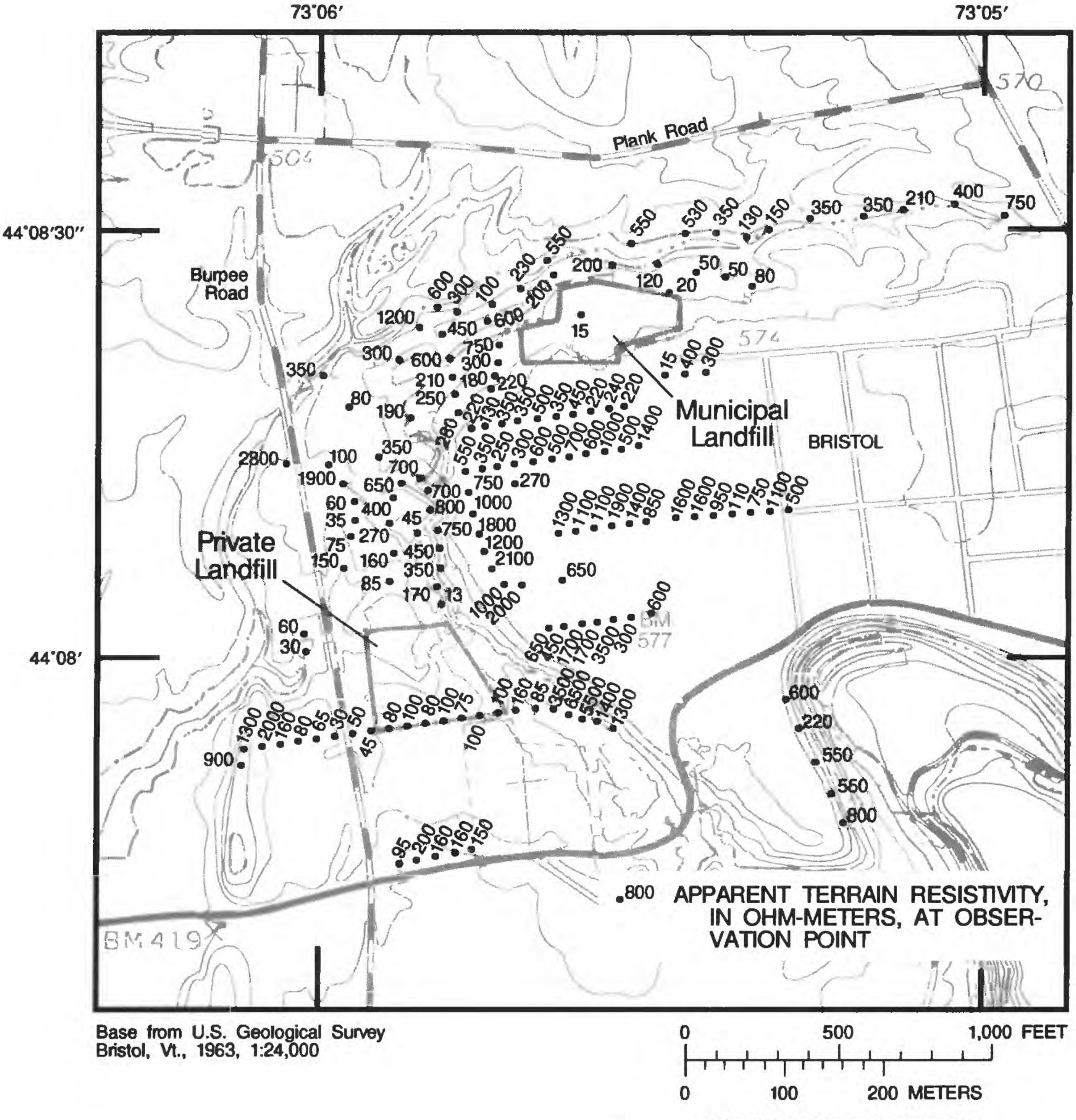
The phase angle between horizontal electric and magnetic fields indicates the type of material stratification. A phase angle less than 45° indicates that conductive materials overlie more resistive materials, a phase angle greater than 45° indicates that resistive materials overlie more conductive materials, and a phase angle of 45° indicates that the material within the exploration depth is electrically homogenous.

VLF terrain-resistivity measurements were made at sites near the two landfills, subject to accessibility and nearby cultural interferences. Measurement sites were generally 50 to 100 ft apart. Apparent VLF terrain resistivities are shown in figure 3. VLF terrain-resistivity measurements were made at other sites throughout the study area to assist in geologic interpretations of the subsurface materials.

Table 1. Effective exploration depths for given terrain resistivities at operating frequencies of 15 to 25 kilohertz

[Data from Geonics, 1979b]

Terrain resistivity (ohm-meters)	Exploration depth (feet)
10,000	1,150
1,000	330
500	260
100	115
50	80
10	33



Base from U.S. Geological Survey
Bristol, Vt., 1963, 1:24,000

CONTOUR INTERVAL 20 FEET
NATIONAL GEODETIC VERTICAL DATUM OF 1929

Figure 3. Locations of measurement sites for very low frequency terrain resistivity.

VLF survey results of an area can be interpreted qualitatively without modeling where the geology is uniform. This was the case on the delta at Bristol, where VLF measurements can be compared for a qualitative interpretation of changes in the subsurface resistivity. Terrain-resistivity measurements greater than 500 ohm-m at this site are generally characteristic of an unsaturated sand and gravel unit that is more than 100 ft thick (fig. 3). Measurements near the southern boundary of the municipal landfill are less than 500 ohm-m. Near the face of the delta, an area of low resistivity (200 to 300 ohm-m) extends southward from the west end of the landfill for a few hundred feet (fig. 3) and probably indicates an area of electrically conductive ground water. Phase angles were generally greater than 45° on the delta deposits; thus, resistive material probably overlies conductive material.

Clay layers (which limit VLF exploration depth) are present to the south and west of the private landfill and pinch out to the north and east. VLF measurements made on the valley floor adjacent to the delta and southwest of the private landfill were limited because of the presence of clay at the land surface. Because of complex geology, widely varying topography, and cultural interferences on Burpee Road (fig. 3), few measurements were made west of the private landfill. Estimates of equipment response, based on forward modeling, indicate that low-conductivity leachate (less than 1,000 μS/cm) may not be distinguishable from clay at the private landfill.

Terrain resistivities on the valley floor adjacent to the delta primarily reflect variations in the local stratigraphy of glacial materials and not changes in ground-water quality. A comparison of adjacent measurements is adequate to assess lateral changes in ground-water quality; therefore, measurements were not contoured in figure 3. Contoured measurements emphasize regional geologic trends. For example, changes in terrain resistivity from west to east along a line immediately south of the private landfill (fig. 3) reflect the resistivity (greater than 1,000 ohm-m) of a dolomite ridge west of Burpee Road (phase angle near 45°), a low resistivity sequence of sand and clay (less than 100 ohm-m) at Burpee Road, a moderately resistive sand and diminishing amounts of clay (greater than 100 ohm-m) to the east near the cemetery, and a resistive largely unsaturated sand and gravel (greater than 1,000 ohm-m).

Terrain resistivities west of the private landfill, were low and could indicate that a clay layer is present or that ground water is conductive in that area. Phase angles were less than 45° adjacent to the delta, an indication that conductive material overlies resistive material. Resistive bedrock is near land surface in this area and accounts for phase angles of less than 45°. Phase angles on the dolomite ridge west of the landfills were all about 45°, an indication of uniformly resistive material with depth.

Electromagnetic Terrain Conductivity

Inductive EM techniques have been widely used for mineral exploration for many years and are now used to detect contaminant plumes (McNeill, 1980a; Mack and Maus, 1987; Grady, 1989). The equipment used consists of a transmitter and transmitter coil and a receiver and receiver coil. The two components are separated by a 33-, 66-, or 131-foot cable during the survey. Physical contact with the ground is not necessary; each coil is simply held upright or set on the ground when a reading is taken.

In inductive EM, an alternating current is passed through a coil to induce a magnetic field that penetrates the ground. This magnetic field induces a current in the ground that generates a secondary magnetic field out of phase with the first. The receiver coil is placed a specified distance from the transmitter coil and receives the secondary magnetic field. At low transmission frequencies, the ratio of the secondary to primary magnetic field is linearly proportional to the conductivity of the terrain (McNeill, 1980a).

Exploration depth depends on intercoil spacing, coil orientation, and transmission frequency (table 2). The two coil orientations are described as vertical dipole or horizontal dipole; in the vertical dipole orientation, the two coils are flat on the ground and in the horizontal dipole orientation, the two coils are upright. Measurements made when the intercoil spacing is 131 ft

Table 2. Inductive electromagnetic exploration depths at three intercoil spacings and low-transmission frequency

[Data from McNeill, 1980a]

Intercoil spacing		Exploration depth (feet) for given coil orientation	
Feet	Meters	Horizontal dipoles	Vertical dipoles
33	10	25	49
66	20	49	98
131	40	98	197

are often difficult to obtain because equipment operation with a large exploration depth becomes increasingly sensitive to cultural interferences. As the survey area increases, the interaction between equipment and cultural interferences increases.

An EM survey done at the municipal landfill by a previous investigation (Johnson, 1989) indicated high terrain conductivity immediately south of the landfill. The largest EM anomalies were found by use of the EM equipment at 131 ft intercoil spacing, vertical dipole orientation. No additional EM measurements were made in this area because the considerable depth to water (130 ft) made the utility of the measurements questionable.

EM measurements of terrain conductivity were made at low areas on the delta, on the valley floor downgradient from the municipal landfill, at selected sites near the private landfill, and at sites throughout the study area for geologic control. EM measurements were not made in a grid pattern in this investigation because of the variable topography in many areas, the considerable depth to water (130 ft) in elevated areas of the delta, and the cultural interferences along Burpee Road. Similar to the VLF survey, the EM survey reflects the different lithologies in the study area. For this reason, analysis of the EM survey was limited to a qualitative comparison of EM measurements in figure 4.

Terrain-conductivity measurements made on the dolomite ridge west of the landfills indicate EM conductivities of 0.3 to 0.9 mS/m (millisiemens per meter). Measurements made on exposed Green Mountain quartzite, about 1 mi east of the study area, were 0.4 to 1.3 mS/m. These measurements enhance EM data interpretations by providing the EM conductivity of only the dolomite and quartzite bedrock because the measurements do not integrate other materials.

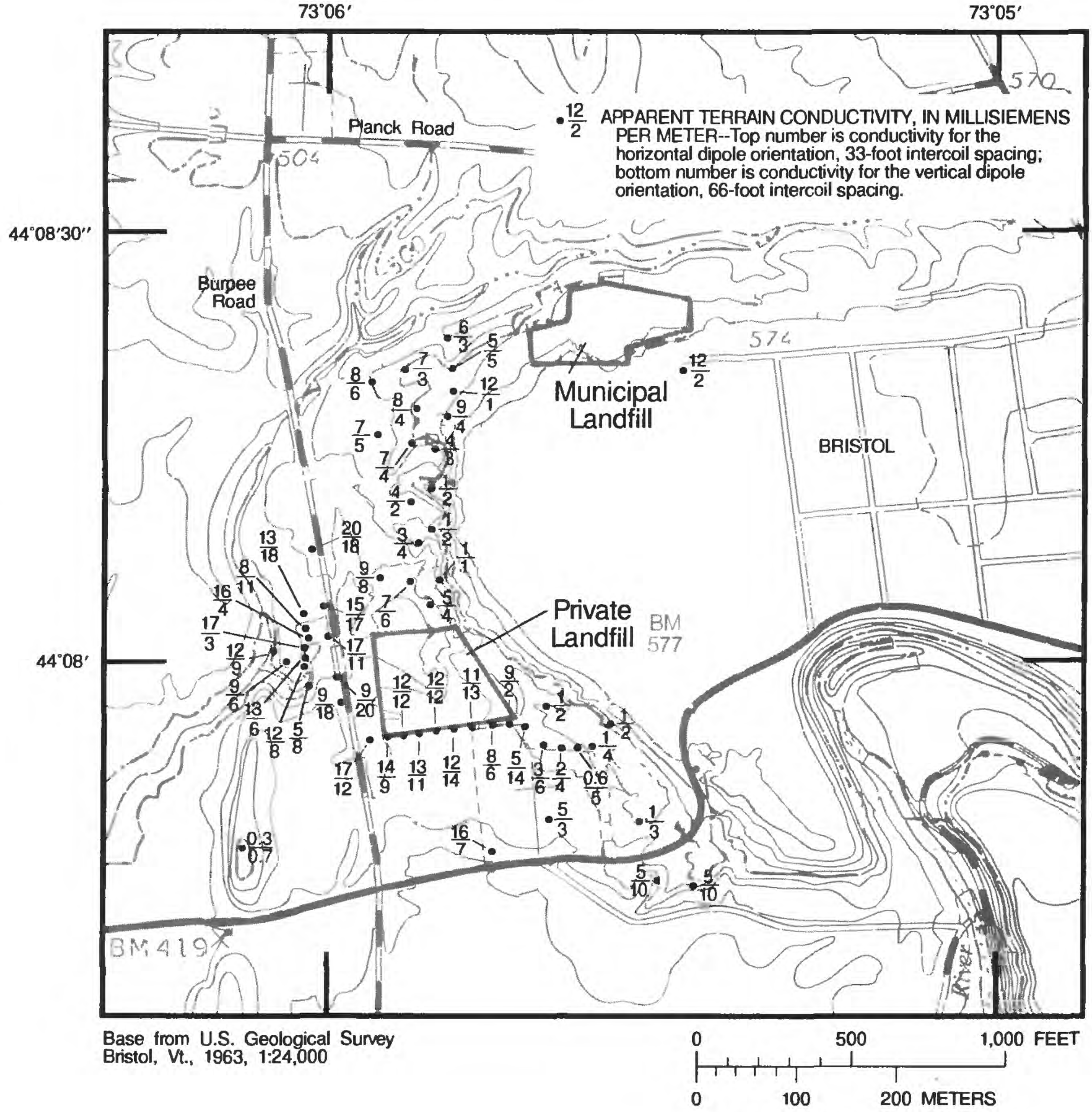
Immediately southwest of the municipal landfill terrain conductivities are high (9 and 12 mS/m) for an area characterized by predominantly thin sand and gravel. Farther west of the landfill, terrain conductivities are moderate (5 to 8 ms/m). The high conductivities immediately west of the municipal landfill indicate the presence of conductive ground water near the land surface, whereas the low conductivity measured with the 66-foot vertical dipole indicates near-surface bedrock.

Terrain-conductivity measurements made near the private landfill (fig. 4) are difficult to interpret because of variations in depth to water (from 0 to greater than 50 ft) and the presence of clay layers. Measurements made west of the landfill along Burpee Road reveal high apparent terrain conductivities. Farther west, measurements were made in a depression containing a small pond when the water surface was frozen. The topographical low area containing the pond is beneath the surficial clay layer, yet high terrain conductivities were measured near the land surface from Burpee Road westward to a brook by use of a horizontal dipole orientation at an intercoil spacing of 33 ft. Estimates of instrument response based on forward modeling indicate that the high apparent terrain conductivities can be the result of clay units in this area. Measurements made immediately south of the private landfill at a cemetery indicate that the surficial and lower clays thin from west to east (similar to the VLF survey results) from Burpee Road to the base of the delta. Results from the VLF and EM surveys indicate that conductive ground water is probably not present southeast of this landfill. However, because of the presence and complexity of conductive clay south and west of the landfill, it is not possible to determine without additional data whether conductive ground water is present south and west of the landfill.

Observation-Well Siting

Observation wells were sited to provide information on the glacial-aquifer lithology, hydraulic characteristics, and horizontal and vertical ground-water-flow directions, and for collection of water-quality samples of native ground water and landfill leachate. Some wells were installed upgradient from the landfills to allow for sampling of ambient, uncontaminated water (native water); the rest of the observation wells were installed downgradient from the landfills to collect data in areas where possible landfill-leachate contamination was indicated by surficial geophysical surveys. Where possible, additional observation wells were installed to supplement available hydrogeologic data and to help determine the lateral extent of leachate contamination.

Observation wells were initially installed as deeply as equipment would allow in the unconsolidated glacial sediments to permit borehole geophysical logging of as much of the aquifer as possible. Wells were then installed adjacent to the first group of wells at predetermined intermediate depths based on results of borehole geophysical logs (described in the section 'Borehole Geophysical Methods'). Vertical hydraulic gradients were determined by installing multiple wells at selected sites and setting well screens at different depths in the aquifer.



•$\frac{12}{2}$ APPARENT TERRAIN CONDUCTIVITY, IN MILLISIEMENS PER METER--Top number is conductivity for the horizontal dipole orientation, 33-foot intercoil spacing; bottom number is conductivity for the vertical dipole orientation, 66-foot intercoil spacing.

Base from U.S. Geological Survey
Bristol, Vt., 1963, 1:24,000

CONTOUR INTERVAL 20 FEET
NATIONAL GEODETIC VERTICAL DATUM OF 1929

Figure 4. Locations of measurement sites for electromagnetic terrain conductivity.

A total of 48 observation wells were installed for this study. All were constructed of flush-threaded polyvinyl chloride (PVC) casing and slotted PVC screens sized according to the aquifer material at the screen zone. Wells were grouted with bentonite seals either along the entire casing length or above the screen and at a few feet below land surface to prevent flow in the borehole. After completion, wells were developed by evacuating the well several times to remove water and any other material introduced during drilling and to improve the hydraulic connection with the aquifer. Well locations are shown in figure 5, well data are listed in appendix 2, and lithologic logs are given in appendix 3.

A 4-inch-diameter hollow-stem auger drill was used to install 30 wells with 2-inch-diameter[1] slotted screens and 6 wells with 1-inch-diameter slotted screens. The hollow-stem auger rig allowed split-spoon samples of relatively undisturbed aquifer materials to be collected ahead of the drill stem for lithologic identification and grain-size analysis. A 6-inch-diameter mud-rotary drill was used to install six 2-inch-diameter PVC wells in the glacial delta where drilling with the auger rig was not possible because of cobbles and large boulders. A small 2-inch-diameter drill was used to install six shallow 2-inch-diameter wells to a depth of 14 to 20 ft.

Borehole Geophysical Methods

Two borehole geophysical methods were used in this study--EM induction and natural-gamma radiation (gamma) logging. The two logging techniques were used comparatively (Mack, 1993) to determine whether electrically conductive zones were indicative of clay or of conductive ground water. The borehole geophysical logs also were used to improve interpretation of lithologic logs. Borehole geophysical logs obtained during this study are shown in appendix 4.

Borehole gamma logging generally is used to delineate silt or clay layers, or lenses, in unconsolidated deposits in a borehole. The EM-induction log technique, referred to as "the EM log," measures the electrical conductance of the formation outside the borehole. The conductivity discussed throughout this report section refers to the electrical conductivity of the formation or specific conductance of water, not the hydraulic conductivity of aquifer materials. The EM log operates on the same principles as the surface EM survey discussed previously and also produces an integrated measure of the effects of the materials present. The total integrated conductance is controlled primarily by the amount of clay minerals relative to coarse-grained materials and the presence and concentration of the dissolved solids in ground water.

Electromagnetic Logs

The EM log was originally designed for use in the oil industry to measure formation resistivity through oil-based drilling mud, where no conductive medium is present between the probe and the formation (Keys, 1988). The principles of the EM log are similar to those of the EM terrain-conductivity equipment; results differ in that EM logging provides detailed information on vertical changes in formation conductivity. The EM probe contains three coils; one for transmitting an electromagnetic field and two for receiving an induced, or secondary, magnetic field (McNeill, 1986). The transmitted electromagnetic field, or primary field, induces a flow of eddy currents in the material in and around the borehole. These currents induce a secondary magnetic field that is received by and causes an induced voltage in the receiver coil. The electrical conductivity of the rock and water outside the borehole is proportional to the magnitude of the secondary magnetic field received. The second receiver coil in the probe is used to negate the effect of the borehole and borehole fluids on instrument response.

The transmitter coil for the equipment used in this investigation transmits at about 39 kHz and has two receiver coils spaced 14 and 20 in. from the transmitter (McNeill, 1986). This coil configuration gives a maximum instrument response from material about 1 ft from the probe and ensures that the instrument responds to material beyond the area disturbed by drilling processes or borehole fluids in a 4- to 6-inch-diameter borehole (McNeill, 1986). A thickness of about 13 ft is required to produce a full instrument response so that the conductivity of the unit can be accurately determined (Taylor and others, 1989). A layer less than 1.6 ft thick can be detected if the conductivity contrast between it and the adjacent units is large; however, determination of layer thickness and conductivity is not possible (Taylor and others, 1989).

An induction log can be used in an uncased borehole or in wells with nonmetallic casing, such as PVC, 2-inch diameter or greater. The induction log is ineffective in a metal-cased well. Metal in or near the borehole,

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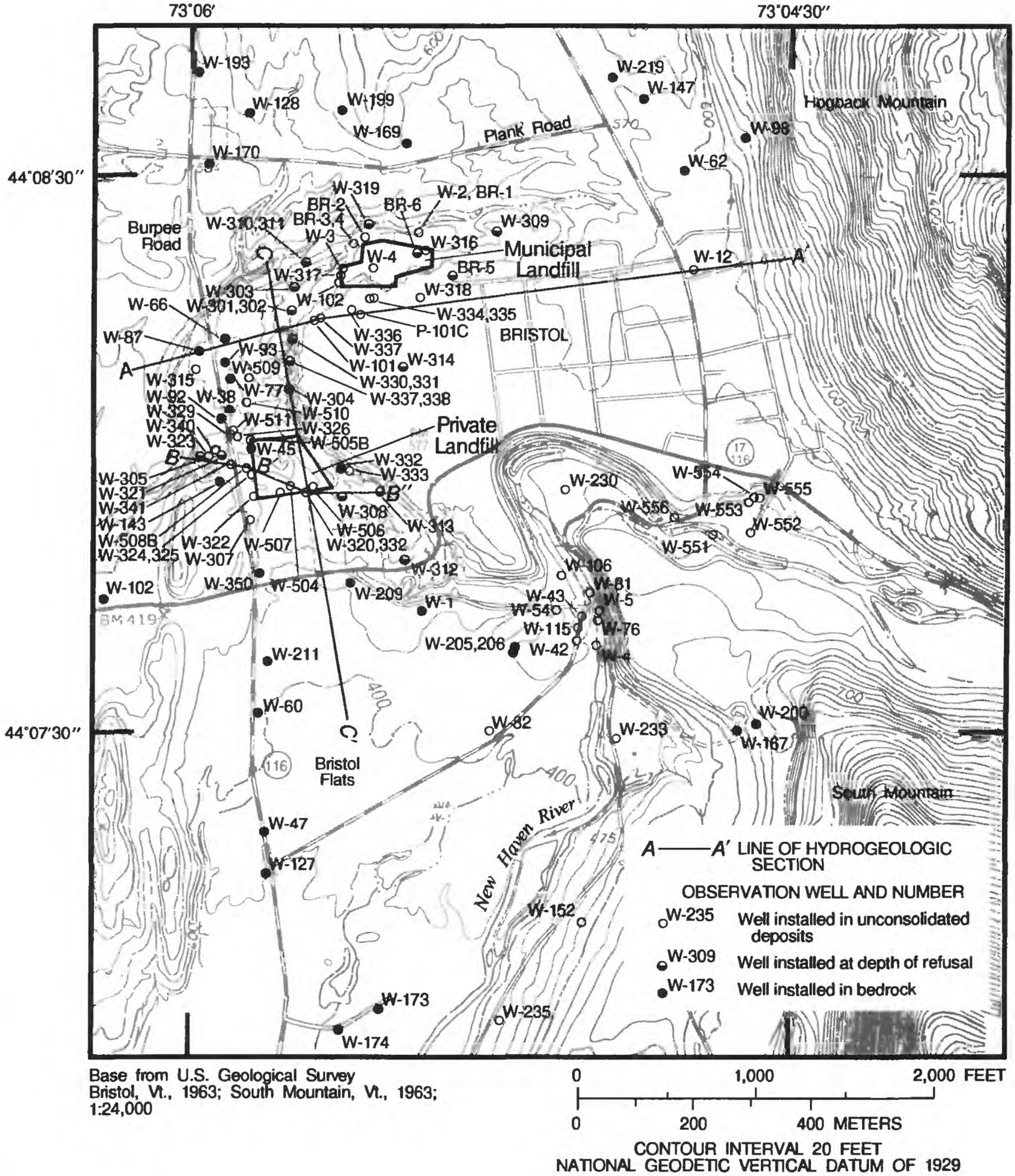
[1] In this report, well diameter is the nominal inside diameter.



Base from U.S. Geological Survey
Bristol, Vt., 1963; South Mountain, Vt., 1963;
1:24,000

Figure 5. Locations of observation wells.

such as a steel well protector used at the surface or metal fragments from a drill bit, will show distinctly on the induction log as a sharply spiked positive or negative response trace that may go beyond normal scale.

Natural-Gamma-Radiation Logs

Gamma logs are used in ground-water studies for identification and correlation of lithology. The gamma log measures the total gamma radiation, measured in counts per second, detected in a borehole within a selected energy range. Gamma-emitting radioisotopes are natural products of uranium- and thorium-decay series and potassium-40. Uranium and thorium are concentrated in most clays by the processes of adsorption and ion exchange (Keys, 1988). Potassium is abundant in some feldspar and mica that decompose to clay (Keys, 1988). Therefore, fine-grained detrital sediments that are rich in clay are generally more radioactive than quartz sand or carbonate rocks and can be readily distinguished from quartz sand or carbonate rocks by use of a gamma log.

A gamma log can be used in uncased boreholes and in PVC or steel-cased wells because gamma radiation will penetrate most casings. A steel casing, however, will attenuate gamma penetration slightly.

Aquifer Tests

Two aquifer tests, slug and specific capacity, were used in the study to determine aquifer characteristics. During a slug test, the water level in a well is raised or lowered rapidly and the water-level response is measured. During a specific-capacity test, water is withdrawn at a constant rate from a well and the response of the water level is measured. Slug tests were done at 28 observation wells and specific-capacity tests were done at 5 wells, results are given in appendix 5.

The slug-test method used in this study is described by Prosser (1981). In practice, the well casing is sealed and pressurized with air to displace the water column. Once the aquifer has stabilized to the increased pressure, the pressure is released instantly by a ball valve and the water-level change to initial conditions is digitally recorded with an electronic pressure transducer. A digital-data recorder collects data at intervals of less than 1 second. Slug-test data that form smooth response curves can be obtained in less than 1 minute from aquifer materials where the hydraulic conductivity is high. An example curve showing a rapid slug-test response obtained by this method is shown in figure 6.

At some wells, it was difficult to adequately pressurize the casing to use the slug-test method described by Prosser (1981) because the well screens extended

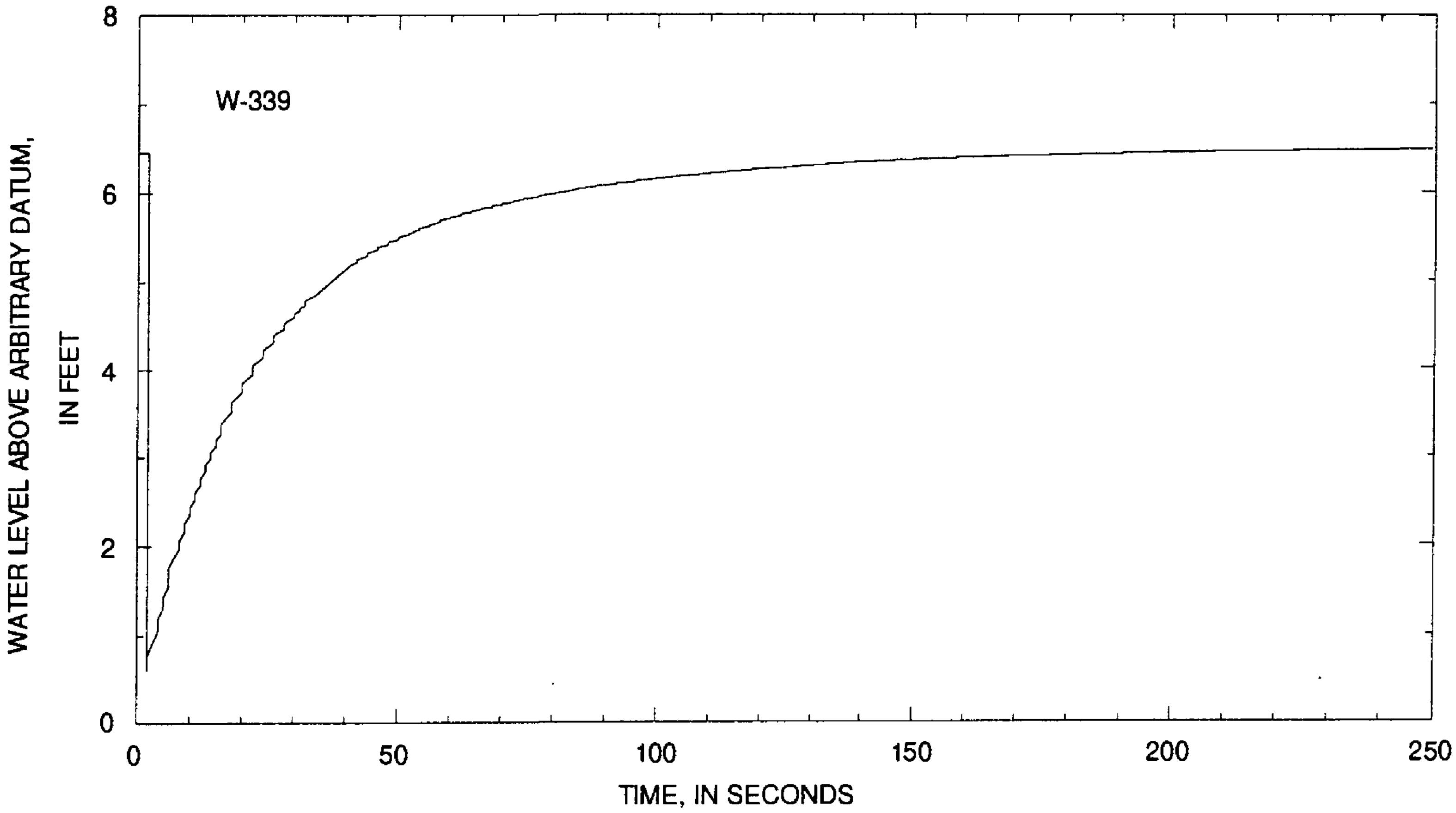


Figure 6. Typical slug-test response curve.

above the water table and or the depth to water was too great. If the depth to water is great, there are more PVC joints from which air can escape than if water is nearer to land surface. Under such circumstances, a weight was used to displace the water level. In this type of slug test, the change in water level is measured when the weight is dropped into the well and the level has stabilized, and then again after the weight was removed from the well. Use of a weight (or, similarly, a slug of water) may not produce accurate results because the initiation of the slug test is not instantaneous, the test response may occur as the weight is dropped or withdrawn, or the weight or slug of water may be too small to create an adequate water-level change.

Slug-test data were analyzed according to the method presented by Cooper and others (1967). Type curves for instantaneous head change in a well of finite diameter (Cooper and others, 1967) can be used to calculate transmissivity at a well screen. According to the method of Cooper and others (1967), dimensionless head (instantaneous head change divided by the maximum head change) is plotted with log time from the first head change. An example of a dimensionless head-change curve for a typical slug test (fig. 6) is shown in figure 7. Calculation of an aquifer storage coefficient by this method is possible, but the resulting value is probably not accurate (Cooper and others, 1967). However, storage coefficient can be estimated to an order of magnitude from grain-size data. This estimate subsequently allows the use of the appropriate type curve from Cooper and others (1967) to calculate the transmissivity of the screen zone.

Where slug tests could not be used, the transmissivity of selected wells was estimated from specific-capacity tests by use of the method of Theis and others (1963). Estimates of transmissivity by this method are only approximate but are considered to be reasonable for a sufficiently long test (at least a few hours). However, specific capacity (discharge divided by drawdown) is affected by well construction, well performance, and pump capacity; deficiencies in any of these can reduce the estimated transmissivity.

Streamflow Measurement

Streamflow was measured at nine stations on the New Haven River and at all the tributary streams that enter and leave the study area. Measurement stations are shown on figure 8 with the exception of site 0 (0.5 mi east of the study area on State Route 17 on the New Haven River) and sites 20 through 22, which drain directly off South Mountain (1 mi south of the study area). Streamflow was measured by use of standard or pygmy current-flow meters on the New Haven River and

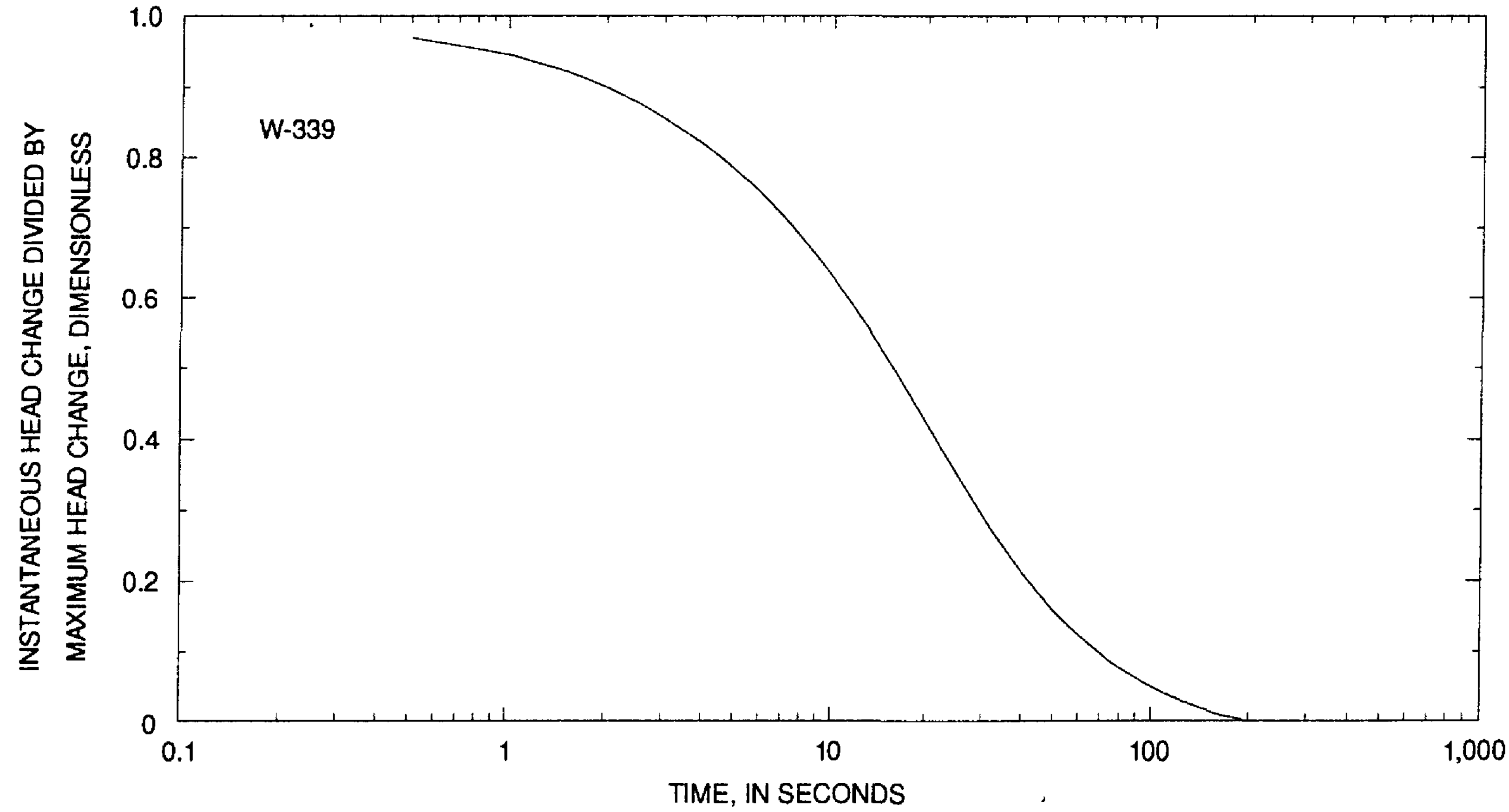
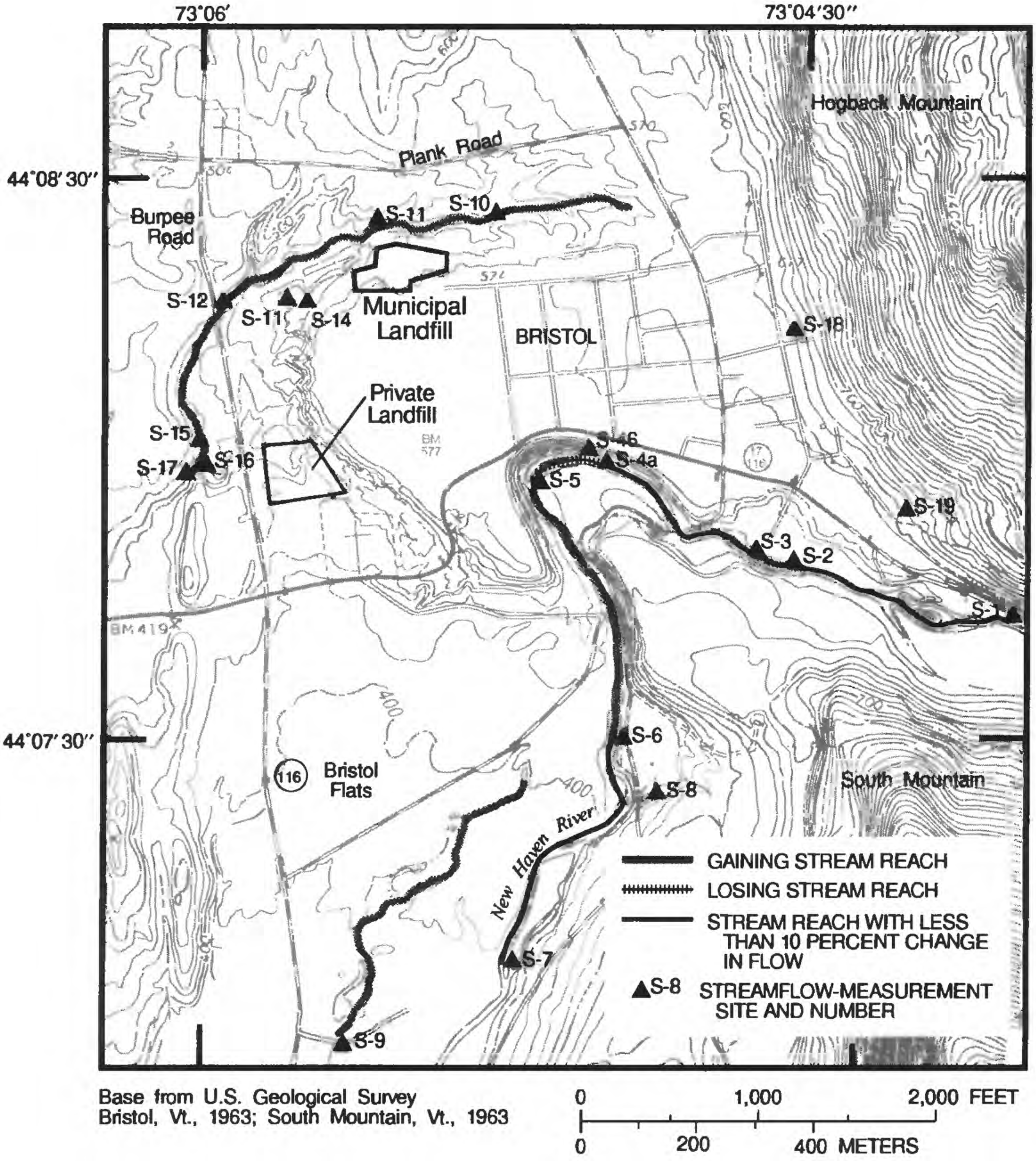


Figure 7. Typical dimensionless head-change curve.



Base from U.S. Geological Survey
Bristol, Vt., 1963; South Mountain, Vt., 1963

CONTOUR INTERVAL 20 FEET
NATIONAL GEODETIC VERTICAL DATUM OF 1929

Figure 8. Locations of streamflow-measurement sites and gaining and losing stream reaches.

on tributaries according to the methods described by Rantz and others (1982a). Tributaries with small streamflows (less than 1 ft^3/s) were measured by use of v-notch weirs and Parshall flumes as described by Rantz and others (1982b). At measurement stations 20 through 22, flow from South Mountain was measured by use of a calibrated bucket and stopwatch.

The flow measurements are summarized in table 3. Measurement sites listed in table 3 are designated as having either streamflow into (in), or out of (out) the study area, or as a site where flow was within (>>) the stream-aquifer system. The net streamflow gain, shown in table 3, is calculated by adding all streamflow into the study area (in) and subtracting all streamflow out of the study area (out). Sites 0, 20, 21, and 22 are outside the study area but were measured to help quantify and analyze runoff from the Green Mountains.

Ground-Water-Flow Modeling

A numerical ground-water-flow model can be used to simulate the flow of water in an aquifer under steady-state or transient conditions. Development of a ground-water-flow model requires that all fluxes into and out of the system and all aquifer characteristics be quantified. Many solutions to modeled flow systems are not unique; that is, various combinations of input parameters can fit

Table 3. Streamflow measured from January-July 1991 in Bristol, Vermont

[All data measured in cubic feet per second; in, out, and >> designates streamflow into, out of, or within the stream-aquifer system at the study area. <, actual value is less than value shown; --, no measurements]

Site No. (fig. 8)	Instantaneous streamflow for selected sampling dates					Relation of stream-aquifer system
	01/15	03/27	05/21	07/03	07/11	
0	--	--	--	18.9	--	(1)
1	--	149	74.8	21.6	--	In
2	--	--	--	20.9	--	>>
3	--	--	95.4	24.8	--	>>
4a	--	--	--	23.9	31.4	>>
4b	--	--	--	--	29.2	>>
5	--	--	79.6	21.1	26.9	>>
6	--	146	84.4	26.5	34.2	>>
7	--	--	83.4	26.1	--	Out
8	--	--	<.01	0	0	In
9	--	--	2.98	2.31	--	Out
10	--	--	0	0	0	>>
11	--	--	0	0	0	>>
12	--	--	<.01	0	0	>>
13	--	--	--	--	--	>>
14	--	--	--	.08	--	>>
15	--	--	.08	.07	--	>>
16	--	--	.05	.01	--	>>
17	--	--	[2].15	.11	--	Out
18	--	--	.26	.05	--	In
19	--	--	--	0	--	In
	--	--	(11.5)	(6.9)	--	(Net out of the study area)
20	0.09	--	[3].12	0	--	(4)
21	.08	--	[3].04	0	--	(4)
22	--	--	[3].08	.02	--	(4)

[1] Upstream from study area.

[2] Estimated flow.

[3] 5/02/91 measurement date.

[4] Not in study-area drainage.

together reasonably and produce acceptable results. During the development of a model, a hypothesis describing the conceptual flow system is tested and refined to improve understanding of the flow system. Reilly and others (1987) discuss the use of simulation to aid solute-transport studies and present a framework for a study of physical mechanisms of transport. The ground-water-flow model consists of the numerical data sets that represent the conceptual model of the ground-water aquifer system. These data sets become input to the computer program known as the numerical model.

The finite-difference numerical model developed by McDonald and Harbaugh (1988), MODFLOW, was used to simulate ground-water flow in the aquifer. Results of simulated water levels are given in appendix 6. MODFLOW consists of a main program and a set of independent subroutines or "packages." Each package handles a specific feature of the hydrologic system, such as wells, recharge, or rivers. This program can be used to simulate three-dimensional flow and to estimate flow at all six faces of rectangular cells that represent parts of the modeled system.

A stream-routing subroutine (Prudic, 1989) was used to simulate stream-aquifer interactions. This subroutine can account for streamflow volume in the simulated stream and allows a stream to dry up if the water level in the aquifer declines below the streambed.

A particle-tracking program developed by Pollock (1989), MODPATH, was used to compute pathlines of ground-water flow. MODPATH is designed as a postprocessor to MODFLOW; MODFLOW output files are used directly in computational schemes. MODPATH uses a semianalytical particle-tracking scheme to compute the path of a particle through the simulated aquifer from any given starting point in a model cell to its point of discharge.

Ground-Water-Quality Sampling

Ground-water samples were collected semiannually from autumn 1990 through autumn 1991 by VANR from as many monitoring wells as possible after development. A few samples were collected at dates between the semiannual samples. Additional monitoring wells were added throughout the study; therefore, the number of wells sampled increased continuously to the end of the study. Although investigation of the bedrock aquifer was outside the scope of this study, the VANR routinely collected water-quality samples from at least five bedrock domestic wells near the landfills when the semiannual samples were collected. Results of laboratory and field analysis are given in appendix 7. Before samples were collected, three or more casing volumes were removed from each well, if possible, to introduce representative formation water into the casing. Because some wells were screened in low-yield till or very fine-grained sediments, it was not always possible to remove three casing volumes of water; however at least one casing volume was removed before any sampling.

Most of the wells were installed with dedicated polyethylene-bailer or polyethylene-tubing samplers to minimize the potential of cross contamination or contamination by cleaning agents used on a nondedicated sampler. The polyethylene-tubing sampler was equipped with a one-way valve at the base of the tubing; water was displaced upward by moving the tubing up and down. This tubing was used for collecting samples from wells where a large amount of water was in the well or the depth to water was great. Polyethylene-tubing samplers were most efficient where the amount of water in the well was greater than a few feet and when recovery was rapid. In wells containing only a few feet of water or less, samples were collected with dedicated bailers. When necessary, wells were purged by use of a bailer or submersible pump, and samples were subsequently collected with a bailer.

Samples were analyzed at the Vermont Department of Environmental Conservation (VDEC) Laboratory in Waterbury, Vt. Analyses included pH, specific conductance, sulfate (SO_4), arsenic, barium, cadmium, calcium, chloride, chromium, copper, iron, lead, magnesium, manganese, nickel, potassium, selenium, silver, sodium, zinc, and 36 volatile organic compounds (VOC's). The VDEC Laboratory participated in the Standard Reference Water Sample Program, a quality-assurance program administered by the USGS (Long and Farrar, 1992).

HYDROGEOLOGY

The hydrogeologic setting of Bristol, Vt., is characterized by unconsolidated glacial deposits in contact with the Green Mountains. The geology of the bedrock and unconsolidated deposits are briefly described in the following report sections. Aquifer characteristics and ground-water flow in the glacial aquifers are described in detail in the report section "Glacial Aquifers."

Geology

The prominent geologic features in the study area (fig. 9) are the west face of the Green Mountains (Hogback and South Mountains), which form a north-south boundary rising more than 1,500 ft above the study area, and a gently sloping terrace of unconsolidated material. This unconsolidated material is believed to be the surface of a glacial delta that formed by glacial meltwater flowing westward from the Green Mountains. The upper surface of the delta is more than 150 ft above the valley floor (fig. 9).

Bedrock

The bedrock geology of the study area (fig. 9) is characterized by the Cheshire Quartzite of Cambrian age of Hogback and South Mountains to the east and the Cambrian Dunham Dolomite to the west (Doll and others, 1961; Stewart, 1973). Bedrock bedding and cleavage strike north-south and dip vertically or steeply westward. The contact between these units is probably beneath the western edge of the delta. Cheshire Quartzite crops out west of Burpee Road and north of Pank Road just outside the study area. Quartzite and dolomite may interfinger by folding and faulting beneath the study area (Stewart Clark, U.S. Geological Survey, oral commun., 1991). Quartzite is locally massive and resistant to physical and chemical weathering and erosion. Dolomite is more susceptible to weathering and erosion than the quartzite. The dolomite, a carbonate rock, is dissolved by acidic water and contains fractures enlarged by solution weathering.

A map of the bedrock surface (fig. 10) shows considerable relief. The bedrock-surface altitude ranges from slightly higher than 300 ft above sea level at the southern landfill to higher than 400 ft at the northern landfill. A substantial depression or trough in the bedrock surface underlies the Bristol Flats area (fig. 10). A bedrock ridge trending north-south (fig. 9) is roughly parallel to Vermont route 116 near Bristol Flats, and crosses Burpee Road and extends to the north near a cemetery. The ridge generally consists of dolomite and possibly a quartzite at its base south of Plank Road. The ridge forms a westward boundary for the study area but is eroded near the pond adjacent to Burpee Road. The break in the ridge is most likely an area of predominantly dolomite and is referred to as the "dolomite gap" throughout this report. The bedrock surface dips sharply west of the ridge to less than 200 ft above sea level (fig. 10).

Unconsolidated Deposits

The unconsolidated deposits throughout most of the study area consist of coarse-grained deltaic sands and gravels (fig. 9). The deltaic deposits are bounded by fine- to coarse-grained glaciolacustrine deposits and till (fig. 9). Representative geologic sections (*A-A*', *B-B*'', and *C-C*') showing the stratigraphy of glacial deposits are in figures 11 and 12. A sequence of glaciofluvial sand, gravel, and boulders more than 200 ft thick (figs. 11 and 12) represent a delta built into proglacial Lake Vermont, which formed as retreating continental ice blocked the northward drainage of the Champlain Valley (Chapman, 1937; Stewart and MacLintock, 1969; Wagner, 1972). Bedding structures typical of deltaic sequences can be seen in the exposed west flank of the delta. Discontinuous silt and clay lenses are present locally in the deltaic sequence (W-337, appendix 4). Glaciolacustrine deposits consisting of an alternating sequence of silt, clay, sand, and gravel are between the delta and the ridge along Burpee Road (sections *A-A*' and *B-B*'', fig. 11) and in the Bristol Flats area. Throughout the valley floor, a clay layer is found at the land surface above the water table. Below the surficial clay, a medium- to coarse-grained sand unit is found in the gravel pits along the west face of the delta. This sand forms part of the upper glacial aquifer (discussed later).

A saturated silt, sand, and clay layer, ranging in thickness from about 5 ft (W-324, appendix 4) to greater than 15 ft (W-322, appendix 4), lies below the upper sand unit (section *C-C*', fig. 12). The clay in this unit is present as lenses, a few inches thick, between lenses of fine to medium sand with some silt. In lithologic logs, the thickness of this silt, sand, and clay unit is not readily distinguished from the unit below without the aid of the borehole geophysical logs shown in appendix 4. This silt, sand, and clay constitutes a confining unit and appears to be continuous west and south of the private landfill (section *B-B*'', fig. 11) from approximately the middle of the study area south into Bristol Flats. The unit was not found immediately north of the private landfill (well W-304) but it was found farther to the north in the logs of wells MW-101, MW-102d, W-309, and W-310. The easternmost observations of this unit (at the private landfill) are in the logs of wells W-308 and W -504 (MW-4; Marshfield Engineering Services, 1979). The unit is either discontinuous near the eastern side of the landfill or may have been excavated before the landfill was created. Drill cuttings from well W-332, installed east of the landfill, did not appear to contain the

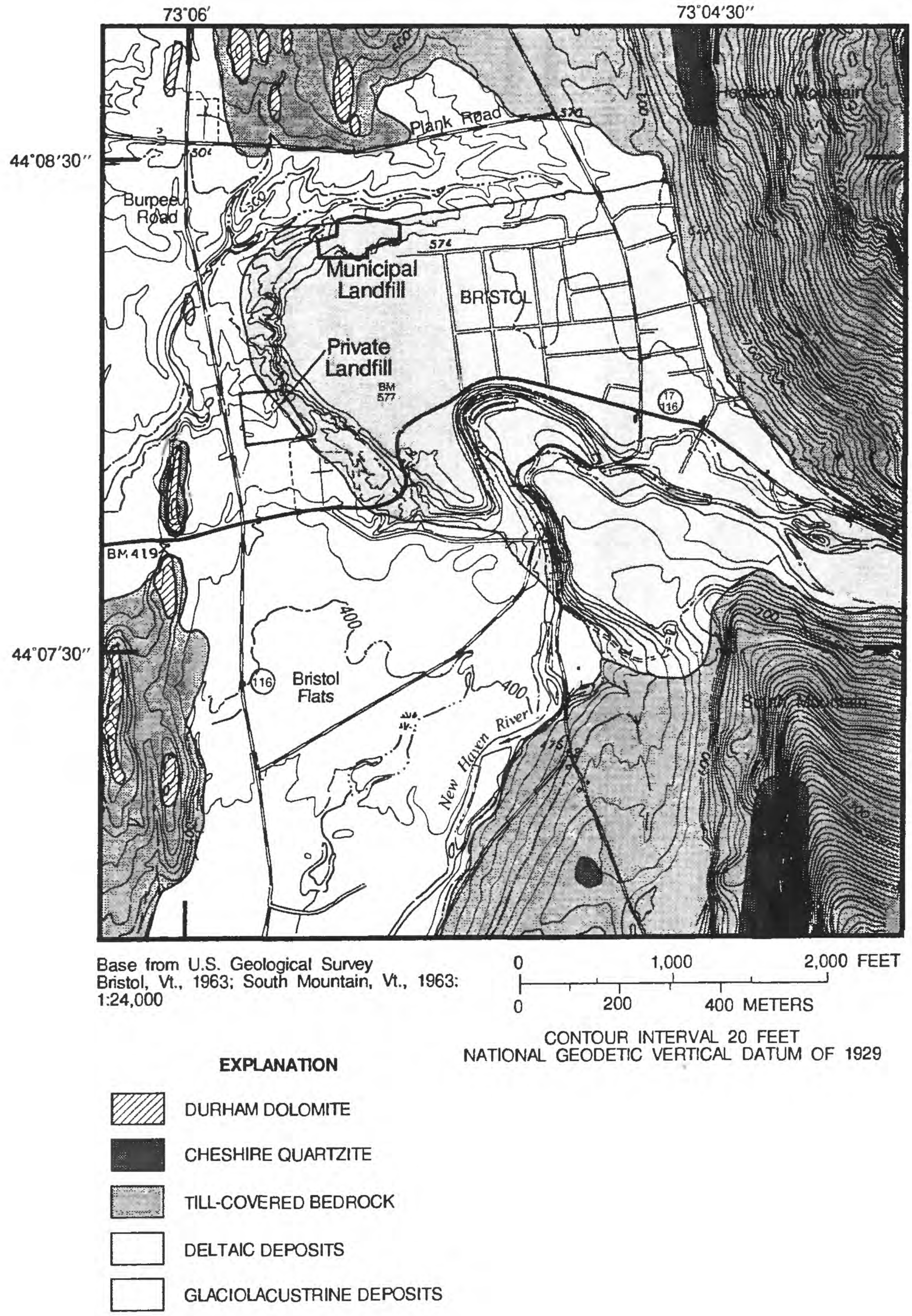
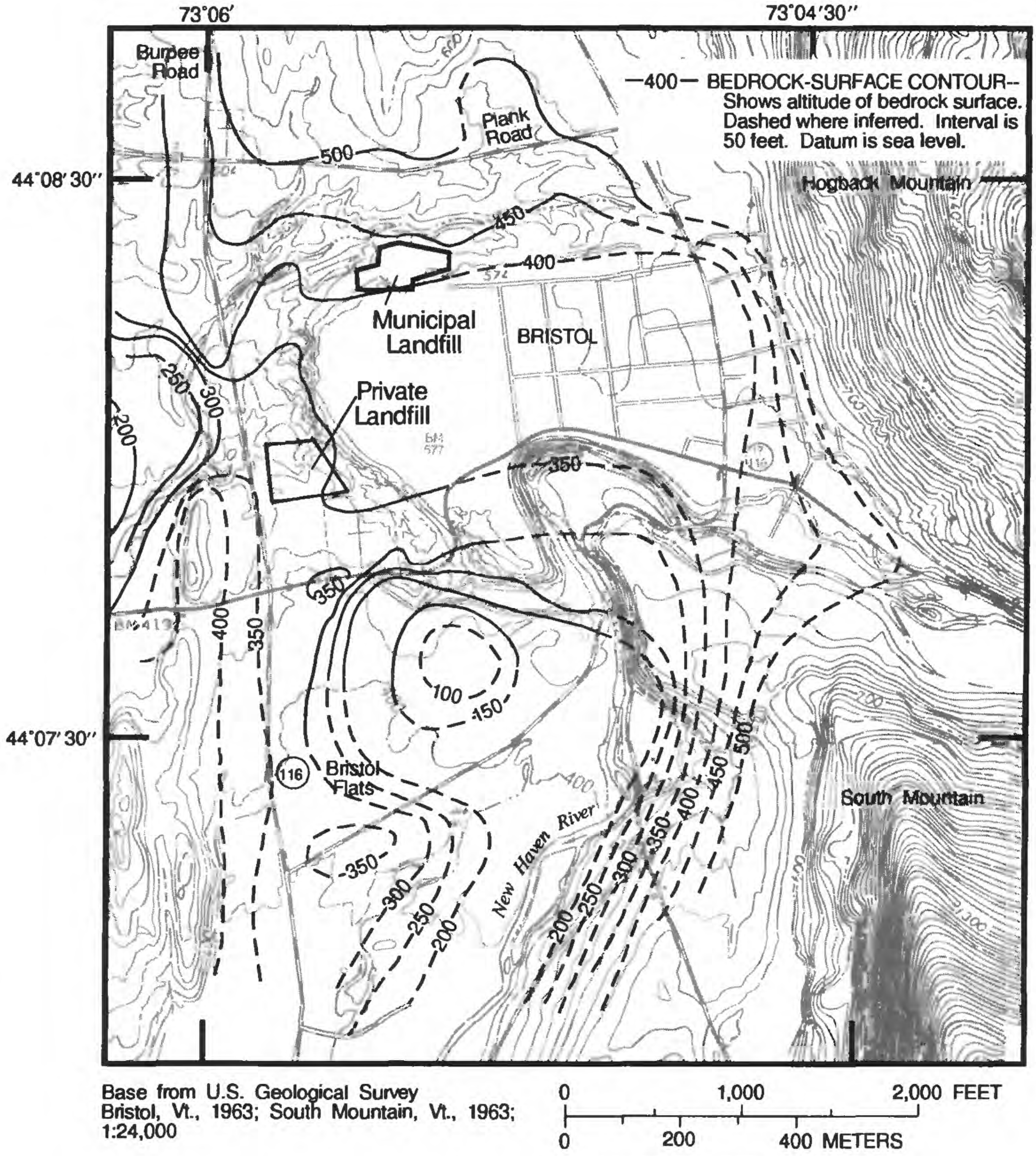


Figure 9. Generalized surficial geology in the study area.



Base from U.S. Geological Survey
Bristol, Vt., 1963; South Mountain, Vt., 1963;
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NATIONAL GEODETIC VERTICAL DATUM OF 1929

Figure 10. Altitude of the bedrock surface in the study area.

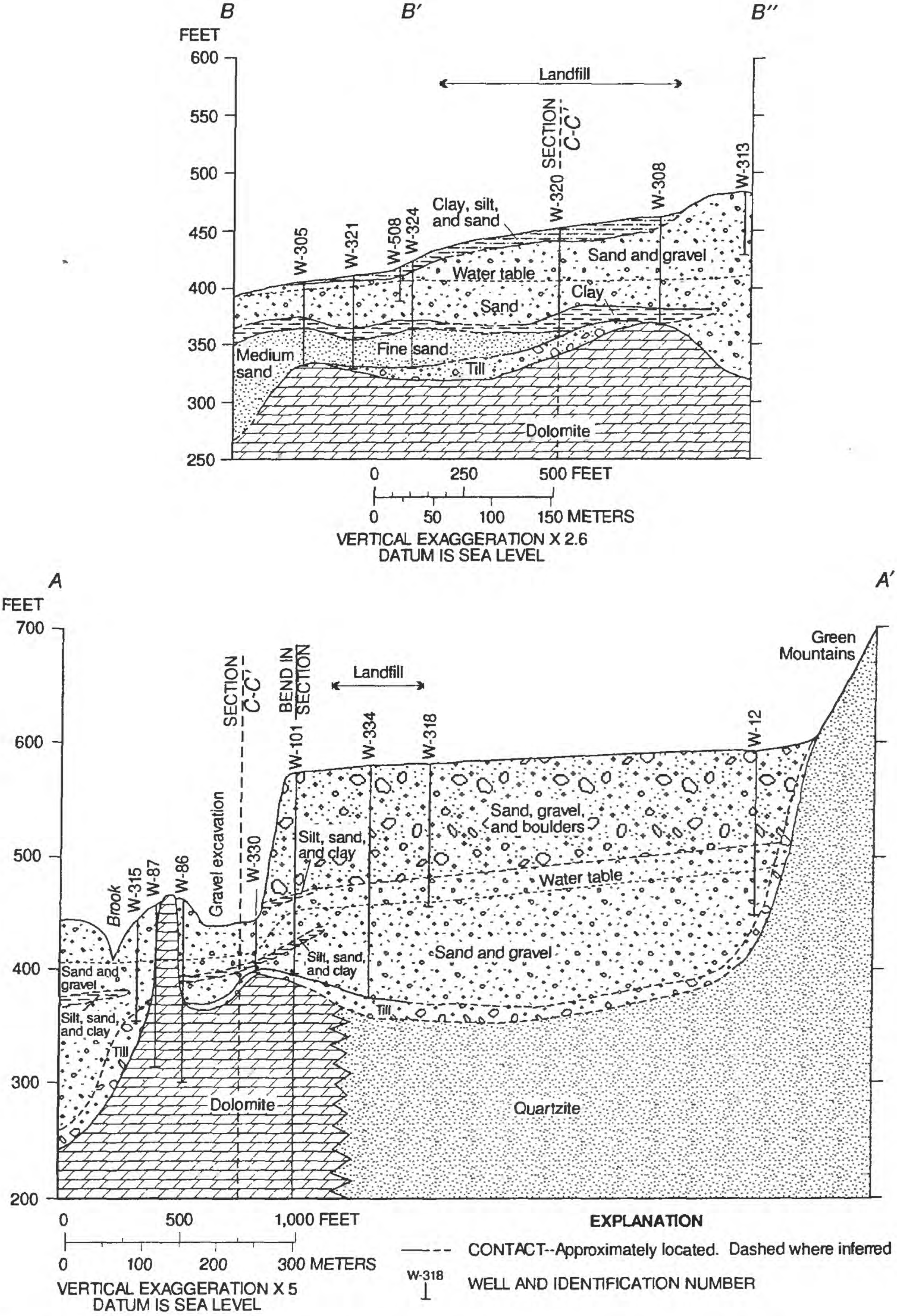


Figure 11. Hydrogeologic sections *A-A′* and *B-B′*. (Lines of section shown in fig. 5)

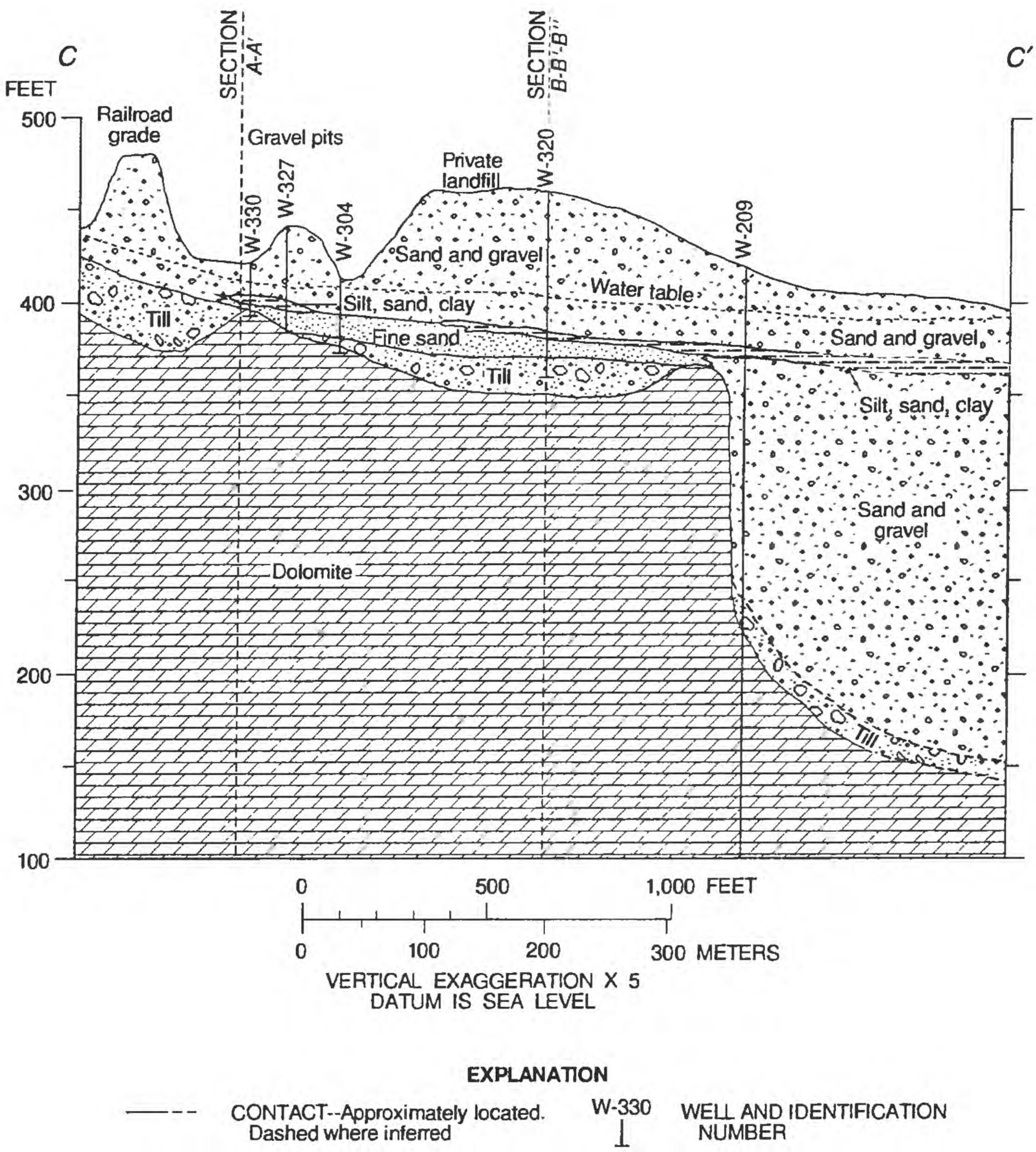


Figure 12. Hydrogeologic section *C-C'*. (Line of section shown in figure 5)

silt, sand, and clay unit; however, a geophysical log of well W-332 would be necessary for positive identification of clay at this location.

Below the sand, silt, and clay confining unit is a fine- to medium-grained sand unit, with some silt that extends from the private landfill southward into Bristol Flats (section *C-C'*, fig. 12). The materials composing this unit are very similar to the sand and silt in the confining unit but without the clay lenses. Drillers' logs indicate that this lower unit becomes coarser and the amount of gravel increases towards the Bristol Flats area. Glacial till, a dense mixture of silt, clay, sand, and angular to subrounded pebbles, directly overlies bedrock throughout most of the study area. Till crops out at the surface, north of the delta, near Plank Road. Till is not present in all areas and can range in composition from predominantly clay to predominantly coarse sand.

Glacial Aquifers

The upper and lower sand units, previously described, constitute an upper glacial aquifer under water-table conditions and a lower glacial, generally confined aquifer (figs. 11 and 12). The upper glacial water-table aquifer—herein referred to as the upper aquifer—is extensive and generally consists of medium- to coarse-grained sands and gravel. The lower glacial aquifer—herein referred to as the lower aquifer—is found predominantly west and south of the delta and generally consists of fine-grained sands. The confining unit that separates the two aquifers pinches out in the delta in an eastward direction, and is herein considered part of the lower aquifer. The lower aquifer extends into the delta where it grades into coarse material and is not distinguishable from the upper aquifer. Domestic well logs from the Bristol Flats area indicate that the lower aquifer consists of medium- to coarse-grained sand and gravel.

Aquifer Characteristics

Aquifer characteristics were determined by interpretation of drillers' logs, grain-size analyses, borehole geophysical logs, and aquifer tests at observation wells. Borehole geophysical logs were used to enhance the interpretation of drillers' logs and to aid in the delineation of upper and lower aquifers. Grain-size analysis of split-spoon samples from borings drilled during this investigation were used to characterize material types and the degree of sorting. Results from aquifer tests were compared to lithologic material types obtained at the same screen zones so that estimated hydraulic conductivities could be extrapolated and applied to other areas of similar lithology.

Estimated hydraulic conductivities and lithologic descriptions of material at well screens where aquifer tests were conducted are listed in appendix 5. The correlation of estimated hydraulic conductivities and associated material type depends not only on the formation at the screen zone but also on the response-test method and material description. For example, results from a slug test are representative of a small volume of aquifer material, whereas a specific-capacity test yields hydraulic conductivities that are representative of a larger aquifer volume.

Estimated horizontal hydraulic conductivity and mean grain size of aquifer materials are summarized in table 4. Similar hydraulic conductivities have been estimated for sand and gravel aquifers in southern New Hampshire by use of an empirical grain-size relation developed by Olney (1983) and supported by aquifer-test data (Harte and Mack, 1992). The horizontal hydraulic conductivities in table 4 are generalized estimates grouped by material types. The range in hydraulic conductivity for glacial sand and gravel is considerable and varies with the degree of sorting.

Table 4. Estimated horizontal hydraulic conductivity of aquifer materials determined from slug and specific capacity tests, predominant material type, and mean grain size in Bristol, Vermont

[mm, millimeter; ft/d, foot per day; <, actual value is less than value shown; >, actual value is greater than value shown]

Material type	Mean grain size (mm)	Estimated horizontal hydraulic conductivity (ft/d)	
		Median	Range
Till	<0.01-0.5	1	<1-4
Sand, fine	.1	10	1-30
Sand, medium	.3	30	10-60
Sand, coarse	.7	120	60-200
Gravel	2.0-4.0	250	150->250

The sand and gravel of the upper aquifer and the delta is highly permeable. Median horizontal hydraulic conductivities were about 100 ft/d and ranged from 10 to greater than 200 ft/d (table 4). Interpretation of an aquifer test of a well screened in a similar sand and gravel aquifer, 8 mi south of the study area on the west flank of the Green Mountains, resulted in a hydraulic conductivity of 95 ft/d (D.L. Maher Co., 1985). The fine to medium sands of the lower aquifer have a hydraulic conductivity that ranges from 1 to 30 ft/d. Glacial till generally has a hydraulic conductivity of less than 4 ft/d; the median is about 1 ft/d. Till that has a predominantly coarse sandy matrix has a hydraulic conductivity of 3 to 4 ft/d. Silt and clay can be expected to have an hydraulic conductivity less than 0.1 ft/d. At a well (W-323) screened in the lower confining unit (sand, silt, and clay), the hydraulic conductivity was 30 ft/d as estimated from results of the slug-test method. The relatively high hydraulic conductivity is a result of preferential flow through sand lenses, which function as horizontal pathways within the confining unit, and is the basis for inclusion of this unit as part of the lower aquifer.

No data were available to estimate the vertical hydraulic conductivities of both aquifers; however, values of one-tenth the horizontal hydraulic conductivity are assumed to be reasonable. Other investigators who have studied glacial sand and gravel aquifers in the Northeast have found vertical hydraulic conductivity to be approximately one-tenth (Childress and others, 1991; Harte and Mack, 1992), one-tenth to one sixtieth (Getzen, 1977) and one-fifth (de Lima and Olimpio, 1989) of the horizontal hydraulic conductivity. The hydraulic conductivity of the confining unit is vertically anisotropic; the horizontal hydraulic conductivity is probably 100 to 1,000 times greater than the vertical hydraulic conductivity. The effective horizontal hydraulic conductivity of the confining unit is strongly affected by sand and pebble lenses. The vertical hydraulic conductivity of the confining unit is affected by the hydraulic conductivity of the clay. Clay has an hydraulic conductivity of about 1×10^{-5} to 1×10^{-7} ft/d (Freeze and Cherry, 1979). Wexler (1988b) summarized vertical hydraulic conductivities reported for a clay and sand confining unit on Long Island as ranging from 7.0×10^{-2} to 2.2×10^{-5} ft/d. Childress and others (1991) used 5×10^{-3} ft/d as the vertical hydraulic conductivity for a clay unit in Ohio.

Ground-Water-Flow System

Water levels were measured monthly in the observation-well network and at a USGS observation well 8 mi south of the study area. Water-level measurements were used to determine fluctuations in ground-water levels and general directions of ground-water flow. An estimation of the long-term average ground-water level is necessary for use in ground-water-flow simulation. Discharge measurements along the New Haven River and on tributary streams in the area were used to identify gaining and losing stream reaches and to estimate recharge.

Ground-Water Levels and Flow Directions

Fluctuations in ground-water levels are caused principally by variations in recharge to and discharge from the aquifer and by variations in evapotranspiration. Water levels from an observation well (MGW-11) open to the bedrock and in a similar physiographic area 8 mi south of the study area were used to identify long-term and seasonal trends in water levels in the bedrock and glacial aquifers in the region (fig. 13) (David Butterfield, Vermont Agency of Natural Resources, written commun., 1992). Average monthly water levels, monthly precipitation, and long-term average monthly precipitation for August 1990 through November 1991 are included in figure 13.

A slight rise in the water table from snowmelt recharge and increased precipitation was observed in March 1991. Recharge is reduced by the onset of evapotranspiration during the growing season; however, water levels decline as aquifer drainage exceeds recharge rates. The average annual range in water level at this well is 4 ft; from November 1990 through May 1991, the water level was the most stable and only slightly above the long-term average annual level. In April 1991, the water level was about 1 foot above the long-term average (1981-91) but below the monthly average water level (fig. 13); therefore, the April 1991 water level is similar to a long-term average water level. Water levels in May 1991 matched the long-term average level; however, water levels in wells in Bristol were generally highest for the year in May. Average precipitation in April approximates the long-term average precipitation (fig. 13).

The annual range in ground-water-level fluctuations in the aquifers in Bristol was from less than 2 ft to nearly 5 ft and was generally 2 to 3 ft throughout the year. Smaller fluctuations were observed at wells in coarse-grained sediments where water was far below land surface, such as in the delta. For example, well W-318 (fig. 14), which had a water-level range of only 1.2 ft, is set in coarse-grained sand, and depth to water was 100 ft. Under such conditions, precipitation filters slowly through the thick unsaturated zone, thus smoothing out pulses of recharge. Ground-water-level fluctuations were greater at wells screened in fine-grained materials, such as at wells W-305 and W-306 (fig. 14). High water levels were observed in late autumn 1990, when evapotranspiration was reduced and recharge increased because of increased precipitation, and generally were the highest in May 1991 because of increased precipitation and snowmelt. Low water levels were observed during summer 1991 when precipitation was below normal. The ground-water-flow system is probably never in steady-state condition (no change in storage); however, the period of least change and close-to-average water levels during the study was April 1991.

A water-table map for April 18, 1991, a period of approximately average water levels, is shown in figure 15. The potentiometric surface of the lower aquifer is similar to but higher than the water-table surface. Ground-water flow throughout the study area is predominantly from northeast to southwest. At the delta face southwest of the municipal landfill, a steep head gradient

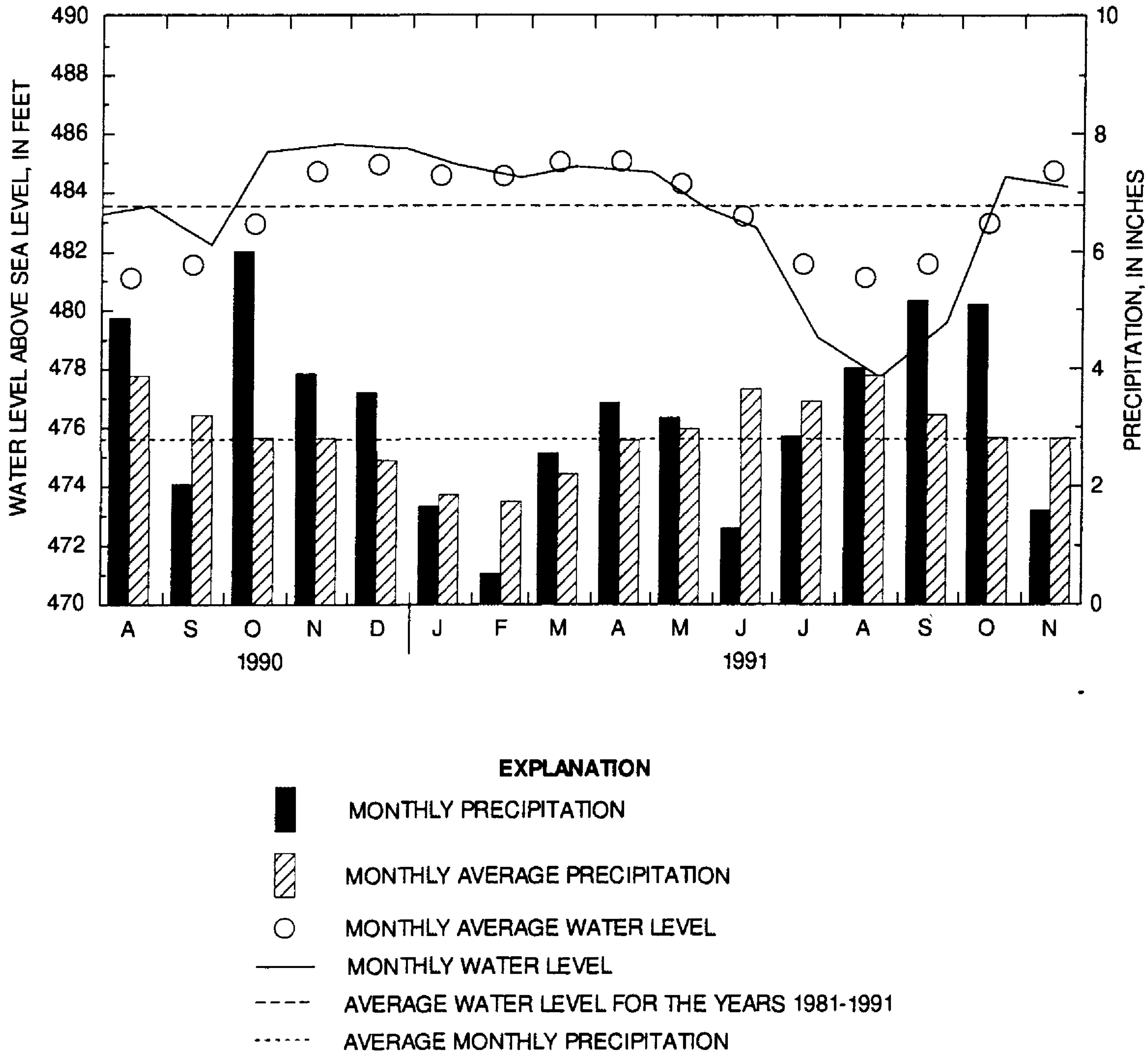


Figure 13. Hydrographs showing monthly water level in well MGW-11 at Middlebury and precipitation at Burlington, Vermont.

results from a reduction in transmissivity caused by a facies change to finer grained sediments and by a reduction in aquifer thickness. At the private landfill, a ground-water-flow divide at the south side of the landfill results in a somewhat radial ground-water-flow pattern. Ground water flows westward, discharging to a small brook and through the dolomite gap a few hundred feet west of the landfill; ground water also flows southward to the aquifers at Bristol Flats.

Ground water in the upper aquifer is unconfined. Depth to the water table ranges from 10 to 55 ft at the private landfill and from 25 to 75 ft at the municipal landfill. Whereas the depth to the water table from the delta surface is as much as 130 ft, elsewhere, the water table near the two landfills emerges as springs at the land surface.

Ground water in the lower aquifer is confined; the head in this aquifer is generally 2 ft above the water-table surface but has been measured at some wells as much as 12 ft above the water-table surface. The saturated thickness of the lower aquifer ranged from 0 to the north, where the aquifer pinches out or grades into till, to greater than 100 ft in the Bristol Flats area. In the delta, no lithologic distinction can be made between the upper and lower aquifers. Although the confining unit is not continuous, the aquifer in the delta is distinguished as an upper and a lower aquifer in this report because of the large head differences between these aquifers in most locations.

At one well cluster east of the private landfill (W-332 and W-333), the head in the lower aquifer was 12 ft below that of the upper aquifer. It is possible that

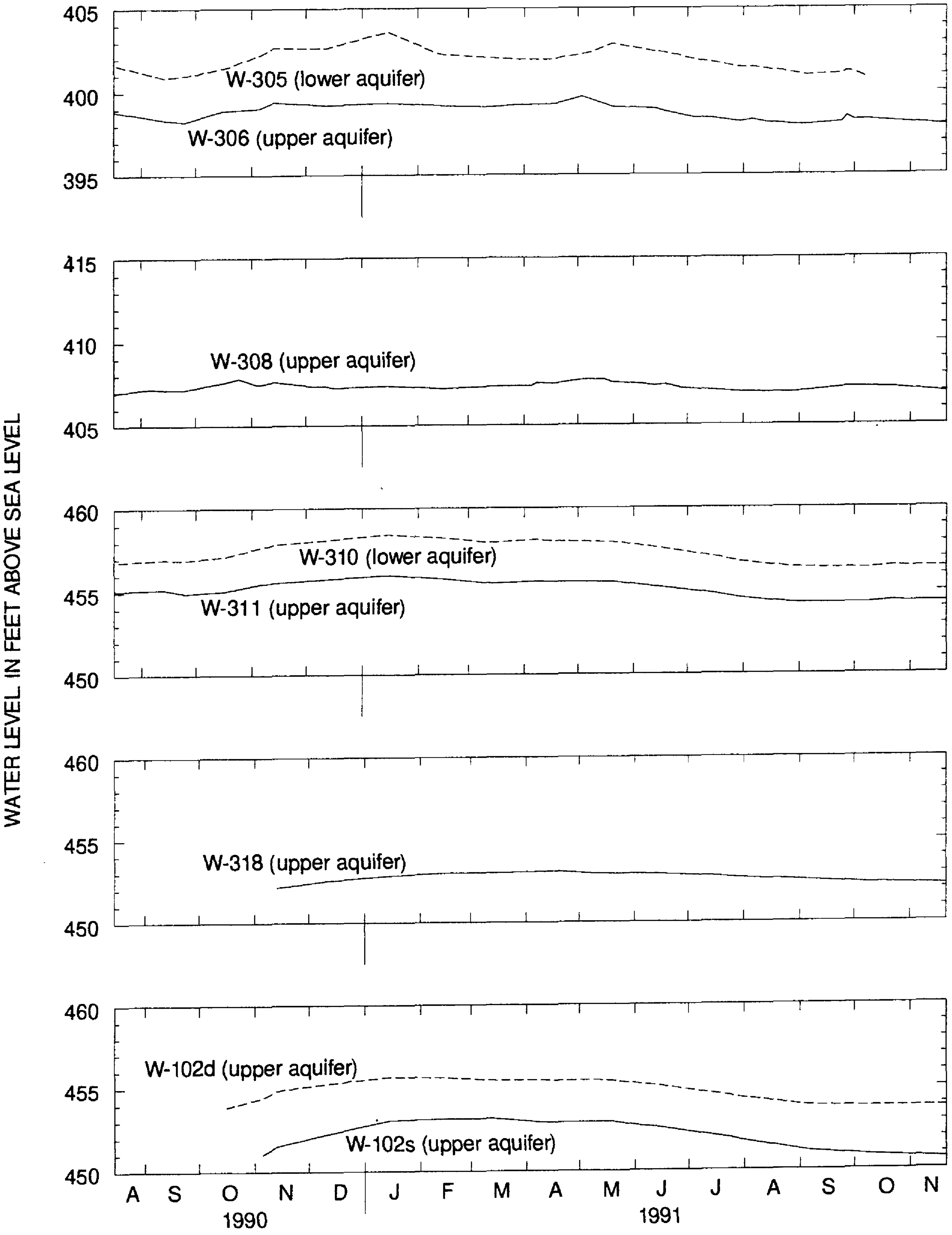
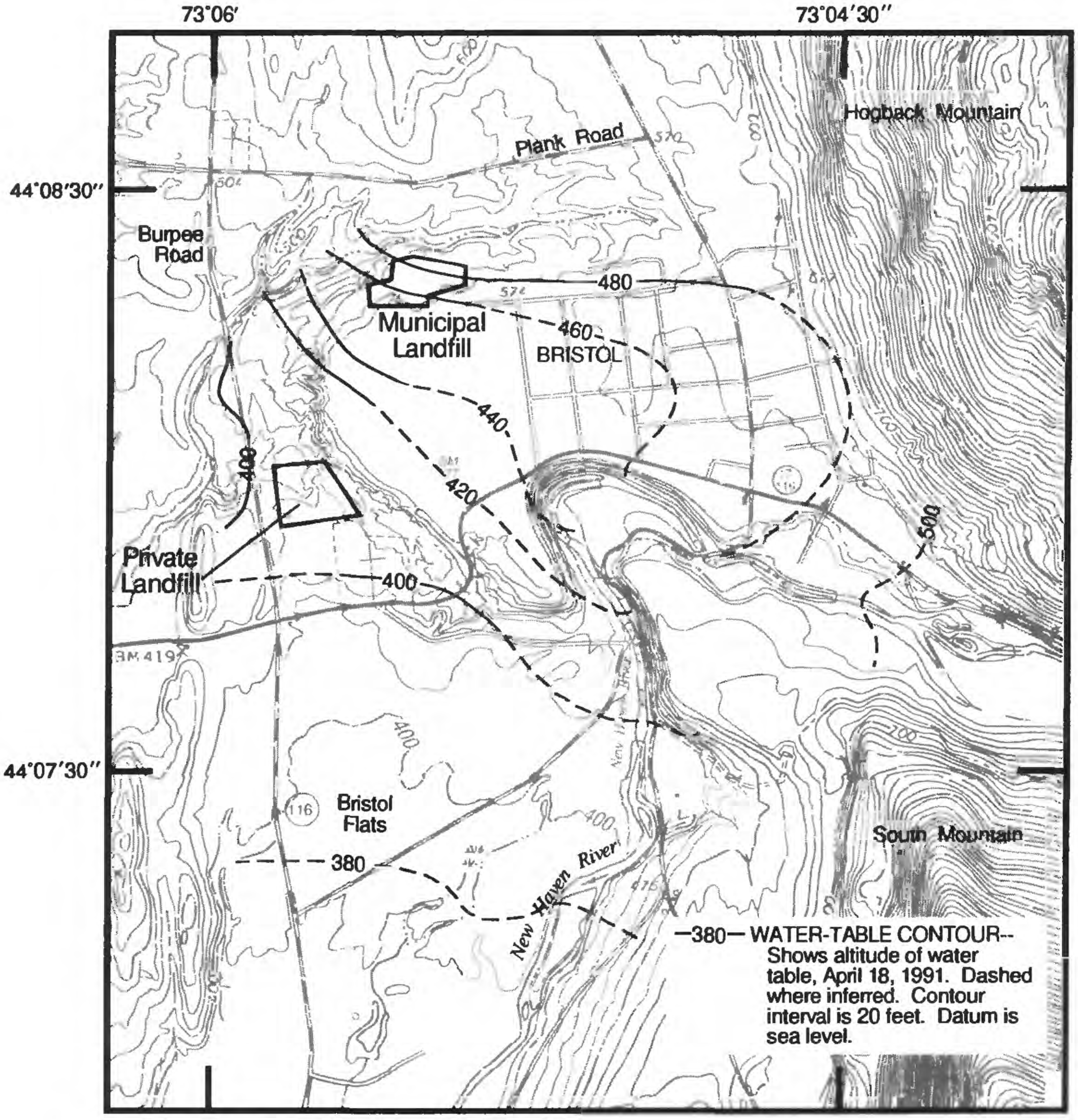


Figure 14. Water-level hydrographs for selected wells in Bristol, Vermont.



—380— WATER-TABLE CONTOUR-- Shows altitude of water table, April 18, 1991. Dashed where inferred. Contour interval is 20 feet. Datum is sea level.

Base from U.S. Geological Survey Bristol, Vt., 1963; South Mountain, Vt., 1963; 1:24,000

CONTOUR INTERVAL 20 FEET
NATIONAL GEODETIC VERTICAL DATUM OF 1929

Figure 15. Altitude of water table in Bristol, Vermont, April 18, 1991.

the shallow well (W-333) is either screened in a locally perched aquifer or poorly connected to the upper aquifer. After well W-333 was pumped, recovery took more than 8 hours; such a slow recovery was atypical of other wells in the area. Wells W-332 and W-333 were installed near the end of the study and, because the validity of water-level measurements at well W-333 could not be confirmed, water levels from this well were not used in the study.

Recharge and Discharge

Recharge to the upper aquifer is derived from three main sources: infiltration of precipitation directly on the aquifer, inflow from upland till or bedrock, and leakage from streams that cross the aquifer. Under steady-state conditions, recharge by infiltration of precipitation, called direct recharge in this report, is equal to the precipitation minus evapotranspiration and surface runoff. Surface runoff is usually negligible in areas underlain by sand and gravel. Direct recharge, therefore, can be approximated as precipitation minus evapotranspiration (Lyford and Cohen, 1988). Precipitation measured at a gage in Burlington, Vt., 22 mi to the north, was 38.0 in. during water year 1991; the long-term annual average is 33.7 in. The precipitation gage in Burlington, at an altitude of 332 ft, is closer to the altitude of the study area (380 to 580 ft) than to the altitude of a gage in Lincoln (2,020 ft), about 10 mi southeast of the study area. Long-term average-annual precipitation at the Lincoln gage is about 54 in.

In the Northeast, precipitation varies from year to year, but annual evapotranspiration remains nearly constant (Lyford and Cohen, 1988). Therefore, if the long-term annual evapotranspiration equals long-term rainfall minus long-term streamflow, then long-term annual potential recharge should equal long-term annual streamflow. On the basis of this method, Lyford and Cohen estimated a long-term annual streamflow of about 20 in/yr for west-central Vermont. The mean streamflow measured in the New Haven River at New Haven, 6 mi southwest of the study area, was 220 ft^3/s, or 25 in., in water year 1991 (Kenneth Toppin, U.S. Geological Survey, written commun., 1992). Precipitation in water year 1991 was 27 percent above the long-term average; therefore, the long-term maximum potential recharge probably is about 20 in/yr.

Direct recharge to a stratified-drift aquifer can be estimated by the method of Rasmussen and Andreasen (1959), where the sum of the water-level rises in a water-table well is multiplied by the specific yield. By this method, ongoing recession, or drainage, from the aquifer is accounted for by extending the recession limb before recharge and adding the total rise from the extended recession. Specific yield of the glacial-aquifer sediments in Bristol was assumed to average about 0.30. Analysis of hydrographs of water-table wells near the landfills (fig. 14) yield estimates of recharge ranging from about 12 to 15 in/yr. The lower estimates were from hydrographs of wells at sites with a thick unsaturated overburden, such as W-308 and W-318 (fig. 14). At such wells, water-table rises and recession limbs in the hydrograph could not easily be distinguished. A representative recharge for this area during 1991 is about 14 in/yr (3.7×10^{-8} ft/s).

Ground-water discharge from the upper and lower aquifers occurs as seepage to streams, ground-water outflow to adjacent aquifers, and leakage to the underlying bedrock aquifer. Recharge to the upper aquifer occurs as leakage from tributary streams that cross the aquifer and from overland flow from adjacent bedrock and till-covered mountains to the east. Additional recharge occurs as upward ground-water flow from the bedrock to the lower aquifer. To estimate these fluxes and to assess stream-aquifer interactions, streamflow was measured during extended periods of little or no rainfall in the New Haven River and the tributaries entering or leaving the aquifer.

The net streamflow gain in the part of the New Haven River that crossed the study area was about 11.5 ft^3/s in May 1991 and 6.9 ft^3/s in July 1991 (table 3). A streamflow gain of 20 ft^3/s (a 26-percent increase) in May and 4 ft^3/s (a 20-percent increase) in July was measured at site S-3 at the face of Hogback and South Mountains. Most of the gain was then lost to the aquifer (table 3) further downstream. Gains also were measured between sites S-5 and S-6 during May and July. Streamflow-measurement error was estimated to be no more than 5 percent of the measurement. The magnitude of the gains observed, 20 and 26 percent of streamflow, are not likely to be accounted for by measurement error or to have been repeatable under different conditions if measurement error was significant. Springs with a constant temperature of 8 to 9°C were observed in this area of the New Haven River and were distinguishable from the river, which had a temperature of 21°C in July. The springs were observed throughout the year and caused sections of the New Haven River to remain uncharacteristically free of ice during winter. The glacial aquifer upgradient from this measurement site is no more than half a square mile in area and is highly unlikely to contribute this amount of water to the river. The source of the large gain between S-2 and S-3 is believed to be ground-water discharge from bedrock, in the Hogback and South Mountain upland areas to the east (fig. 1), flowing through the glacial aquifer. Precipitation on the mountains is more than 60 percent greater than precipitation in the valley that includes the study area. Ground-water-flow simulations, discussed in "Simulation of Ground-Water-Flow," also indicate that recharge other than direct recharge is necessary to cause the hydrologic conditions observed.

Because of the steepness of the west flank of the Green Mountains in Bristol (fig. 1), very few streams flow westward off the mountains to recharge the aquifers in and around the study area. These streams in the study area, sites S-18 and S-19 (fig. 8) on Hogback Mountain, contributed less than 3 percent of the total streamflow gains measured in the New Haven River (about 0.4 ft^3/s in May and 0.05 ft^3/s in July; table 3). Similar streams flowing off the west flank of South Mountain immediately south of the study area, S-20, 21, and 22 (fig. 8), were measured for comparison. Streamflow in these streams generally was less than 1 ft^3/s at different times of the year and virtually zero during low-flow periods (table 3).

The small brook upgradient from the municipal landfill (fig. 8) flows when the water table is high enough to intercept the base of the brook. At its outlet from the study area, streamflow in this brook is less than 1 ft^3/s most of the year. Most of this brook is usually dry during summer when its start-of-flow is generally near site S-15.

The tributary flowing south off Bristol Flats (S-9, fig. 8) is sustained by the ground-water discharge from the aquifer at Bristol Flats, and streamflow is fairly constant throughout the year. In 1991, streamflow was 3.0 ft^3/s in the spring and 2.3 ft^3/s during the summer (table 3).

Ground water also discharges as ground-water outflow to adjacent aquifers. Ground-water outflow is primarily at two locations: at the gap in the dolomite ridge west of the private landfill and through the sand and gravel aquifer at Bristol Flats. Ground-water inflows and outflows were calculated from Darcy's law:

$$Q = KiA, \tag{1}$$

where

Q is discharge,
K is hydraulic conductivity,
i is hydraulic gradient, and
A is area through which discharge occurs.

Ground-water outflow at the gap in the dolomite ridge was calculated to be 1.3 ft^3/s. The mean hydraulic conductivity of the glacial sediments was about 80 ft/d, the average hydraulic gradient was 0.015, and the cross-sectional area was about 95,000 ft^2. Ground-water outflow at the study area boundary at Bristol Flats was estimated to be 5 ft^3/s. The hydraulic conductivity of the glacial sediments (estimated from drillers' logs) was 100 ft/d, the average hydraulic gradient was 0.008, and the cross-sectional area was about 500,000 ft^2.

At the east boundary of the aquifer, ground water enters where the New Haven River flows between Hogback and South Mountains. The hydraulic gradient was estimated to be similar to the gradient of the river surface at this location (0.011), the horizontal hydraulic conductivity was estimated to be 175 ft/d, and the cross-sectional saturated area was about 75,000 ft^2. Ground-water inflow at the east boundary of the aquifer is about 1.7 ft^3/s.

The total water budget for spring 1991 is summarized in table 5. The remainder of all inflow minus outflow is assumed to be equal to upward leakage from bedrock to the overlying glacial aquifer. Some ground water probably discharges to the underlying bedrock; however, because hydraulic gradients are mostly upward throughout the study area, this discharge is likely to be small. About 0.27 Mgal/d (0.4 ft^3/s) of recharge from domestic leach field systems enters the aquifer at the delta (Scott Powell, Bristol Water Department, oral commun., 1991). This water is added to the budget because this water originates from outside the study area. Most of the ground water ultimately discharges as streamflow (table 5).

Table 5. Water budget for the study area for April and May 1991 for Bristol, Vermont

[ft^3/s, cubic foot per second; Mgal/d, million gallons per day; in/yr, inch per year]

Water-budget component	Rate	
	ft^3/s	Mgal/d
Inflows:		
Infiltration of precipitation (14 in/yr).	1.8	1.2
Ground-water flow	1.7	1.1
Leach field	.4	.3
Outflows:		
Net streamflow	11.5	7.4
Ground-water flow	6.3	4.1
Net difference (bedrock leakage inflow).	13.9	8.9

SIMULATION OF GROUND-WATER FLOW

Ground-water flow was simulated in the study area to further the understanding of the local and regional ground-water-flow system. A particle-tracking simulation provided detailed analysis and understanding of the local ground-water-flow system underlying the two landfills.

Simulation of ground-water flow at the two landfills is presented by descriptions of the conceptual model and development of the ground-water-flow model, the model grid and boundary conditions, and selection of model-input parameters. Accuracy and use of the model is described by a calibration and sensitivity analysis, ground-water-flow simulation, and model appraisal.

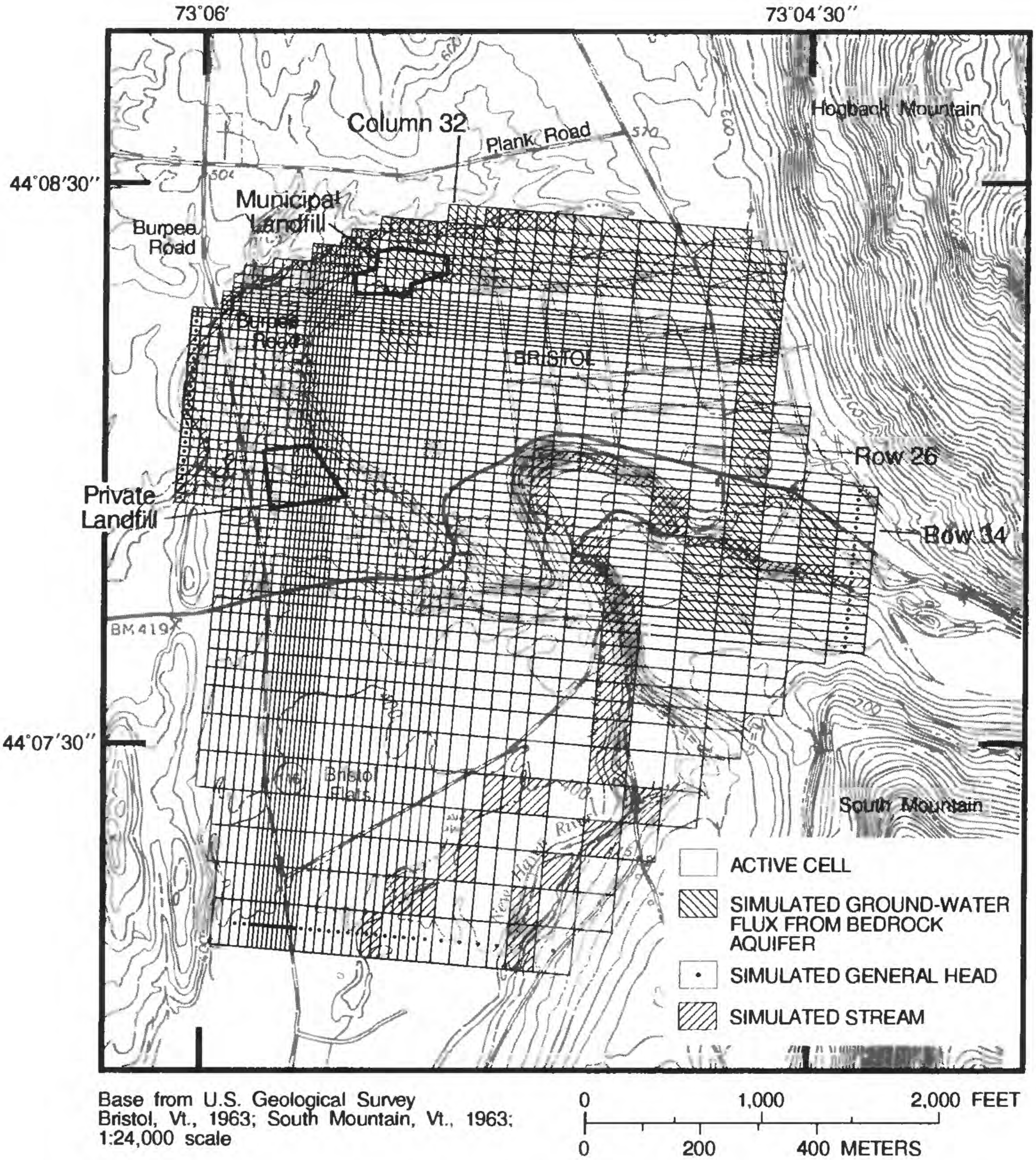
Description of Conceptual and Numerical Models

The conceptual model of the ground-water-flow system, for the purposes of the numerical model, is a simplified representation that integrates all known features of the system and characterizes the aquifer into similar hydrogeologic units by means of assigned representative parameters. The conceptual model represents the complex hydrologic and geologic environments in nature. In a conceptual model, the appropriate level of hydrologic detail depends on the purpose and scope of the investigation, the amount and quality of the data, and the sensitivity of the numerical model to increases in the level of detail. A two-layer conceptual model of the aquifer system requires boundary conditions that are simplifications of the hydrogeologic system. The effects of simplifying assumptions on the ground-water-flow model differ with numerical model complexity and application. Useful discussions of simplifying assumptions are provided by Morrissey (1983), deLima and Olimpio (1989), and Mack (1991). The boundaries in the aquifer system at Bristol and the effects of the simplifications applied to the model developed are discussed in the following paragraphs.

The glacial delta and outlying areas are described by a two-aquifer system: an upper coarse-grained aquifer under unconfined conditions and a lower, fine- to coarse-grained aquifer under confined conditions, which are separated in places by a sand, silt, and clay confining unit. The conceptual model of this system is based on the geologic sections shown in figures 11 and 12. The dominant ground-water-flow paths are in the upper and lower aquifers, although horizontal flow occurs in the sand, silt, and clay confining unit. The low vertical hydraulic conductivity of the confining unit, although not quantitatively measured, is evident by the large vertical gradient between the upper and lower aquifers. The primary effect of the confining unit is the restriction of vertical flow. Because the confining unit contains significant thicknesses of sand, which have hydraulic characteristics similar to those of the lower aquifer; flow in the confining unit was simulated as part of the lower aquifer; that is, the confining unit itself was not modeled as a separate layer. The thickness of the confining unit was incorporated into the lower aquifer model layer, and the confining effects of this unit were simulated by a low vertical hydraulic conductivity to restrict upward flow.

Model Grid and Boundary Conditions

A rectangular finite-difference model grid was superimposed over a map of the study area to discretize the conceptual model into individual model cells. The outline of the active area for the upper model layer is shown in figure 16. The model grid is divided into cells that range in size from 50 by 50 ft to 400 by 500 ft (fig. 16). Because of the steep valley walls throughout most of the study area, the lower model layer represents nearly the same area as the upper layer. The north and southwest model boundaries for the lower layer consist of a few less rows and columns of cells. Fine discretization was used for the ground-water-flow simulation of the two landfills not only because these are areas of interest but also because the density of data is greater for the landfills than for the surrounding areas. The finest



Base from U.S. Geological Survey
Bristol, Vt., 1963; South Mountain, Vt., 1963;
1:24,000 scale

CONTOUR INTERVAL 20 FEET
NATIONAL GEODETIC VERTICAL DATUM OF 1929

Figure 16. Location of model grid and cell types for the aquifer system in Bristol, Vermont.

discretization (50 by 50 ft) was used southwest of the municipal landfill because of the high hydraulic gradients near the face of the delta (fig. 16). Coarse discretization was used in the eastern and southern model areas (fig. 16), where the objective of the model is to simulate regional boundary fluxes to the glacial aquifers and where the hydraulic gradient is the lowest.

The upper boundary of the model is the unconfined calculated head surface (water table), which is simulated as a specified-flux (recharge) boundary. The New Haven River and small streams in the model are simulated as head-dependent-flux boundaries.

The lower boundary of the model is the bedrock surface (fig. 10) and is simulated as a specified-flux and a no-flow boundary. In some ground-water-flow simulations in the Northeast, it is assumed that little or no ground water flows from the bedrock that underlies the glacial aquifer because the permeability of bedrock is generally orders of magnitude lower than that of the glacial aquifer; however, as described in the "Recharge and Discharge" section, the water budget (table 5) shows a net gain of 14 ft^3/s that can be explained only as upward leakage from the bedrock aquifer. Because this flow forms a significant amount of the overall aquifer budget, leakage from bedrock was incorporated into the model by simulating a specified flux at the lowest active-cell boundaries in the northern and eastern parts of the modeled area (fig. 16). Initial hypothesis-testing simulations indicated that a uniform flux from the bedrock aquifer could not approximate the observed ground-water-flow system. The flux was distributed in the areas shown in figure 16 because streamflow data indicate that a large amount of the flux from the bedrock aquifer originates near the west face of the Green Mountains. Additional hypothesis testing indicated that a flux from the bedrock aquifer in the northern part of the modeled area, near the municipal landfill, was necessary to maintain the heads measured in the lower part of the glacial aquifer. Stewart (1973) noted solution-weathered enlargements of fractures in dolomite in the Bristol area that probably function as conduits for leakage from the bedrock aquifer to the glacial aquifers. Springs flow from the solution-weathered fractures in the dolomite outcrops (fig. 9) north of Plank Road (Brewster Baldwin, Middlebury College, written commun., 1990). Solution-enlarged fractures could be related to northeast-southwest trending fracture traces in the Bristol area (Johnson Company, 1989). Although cells representing recharge fluxes are somewhat arbitrarily located, measured vertical gradients and potential solution-enlarged fractures provide support for placement of the fluxes shown in figure 16. No data are available to proportion flux from the bedrock aquifer in the remainder of the model area; therefore, a no-flow boundary was used.

Evidence for a large flux from the bedrock aquifer includes the following observations: (1) heads in the bedrock aquifer at wells along Burpee Road were above those in the glacial aquifers (appendix 2); (2) the head at bedrock well W-233 (fig. 5) was a few feet above the land surface; (3) a large upward vertical gradient, 0.3 ft/ft (a 12-foot head difference over an aquifer thickness of 40 ft), was observed at clustered wells BR-6 and W-316 screened in the glacial delta at the north end of the modeled area, where no confining unit is present; (4) streamflows off the west flank of the Green Mountains in the study area were negligible (table 3); and (5) large streamflow gains from the aquifer system (table 3) were repeatedly measured in the New Haven River.

Lateral model boundaries are simulated as shown in figure 16. The Green Mountains to the east are treated as a specified-flux boundary, as described above, and form the east model boundary. Near the intermittent stream south of Plank Road, thin till and bedrock (fig. 9) forms the northern model boundary and is treated as a specified-flux boundary. Initially, model simulations included the till-covered hillside at Plank Road; however, high hydraulic gradients in the till require the use of a finer model grid than was warranted for the scope of this study. This area, therefore, was not simulated explicitly. The bedrock ridge west of Burpee Road (fig. 9) forms the west model boundary and is treated as a no-flow boundary (fig. 16), although some ground water probably leaks to or from the bedrock. The gap in the dolomite ridge directly west of the private landfill (fig. 9), along the western model boundary, is simulated as a general-head boundary.

The valley-fill aquifer beneath Bristol Flats, south of the landfills, extends beyond the study area. A south model boundary was selectively placed far enough from the landfills to have little effect on simulation of flow in those areas and at a point in the aquifer where hydraulic gradients could be reasonably estimated and streamflows measured. This boundary is simulated as a general-head boundary to provide for flow out of the aquifer (fig. 16).

Selection of Input Parameters

The parameters assigned to each cell are assumed to be constant and represent an average value within each cell. Parameters that must be assigned to cells include hydraulic conductivity, cell bottom, cell top for the lower cells, recharge, discharge, and stream characteristics. Parameters are assigned initial input values that are based on estimated or known data. Initial-input values are modified or accepted during calibration to become the final model-input values.

Horizontal hydraulic conductivity was estimated from grain-size distributions and from slug and specific-capacity tests; initial input values are listed in table 4. Vertical hydraulic conductivity was assumed to be a fraction of the horizontal hydraulic conductivity; 0.1 for the coarse materials and 0.001 for the confining unit. Vertical hydraulic conductivity generally is less significant than horizontal hydraulic conductivity in models simulating nonstressed flow systems (Harte and Mack, 1992).

The depth and width of streams were measured at streamflow-measurement sites and at other locations. The altitude of streams were either measured at streamflow-measurement sites or estimated from topographic-map contours. The hydraulic conductivity of streambeds, based on examination of the streambed materials, was estimated to be 5 ft/d for the New Haven River bottom and 1 to 3 ft/d for tributaries and the pond west of Burpee Road. Childress and others (1991) measured similar values at streams in a glaciated area of Ohio. Streambed thickness used was 2 ft for the New Haven River bottom and 1 ft for all other streams. Recharge from infiltration of precipitation to the aquifer was set at 3.7 × 10-8 ft^3/s (14 in/yr), as discussed in the section “Recharge and Discharge.”

Calibration

Calibration is the process of adjusting initial-input values until the difference between measured heads and flows and simulated heads and flows is within acceptable limits. Model-input values generally were changed for a group of cells, or zones, rather than on a cell-by-cell basis to keep the conceptual representation of the aquifer system intact and to avoid adjusting input values at individual cells until simulated heads or ground-water flows matched measured values.

In general, model-input values were similar to initial-input values based on data for April and May 1991. A large part of calibration consisted of adjustments of initial-input values between data points and the distribution of allocations of fluxes at the appropriate model boundaries. Calibrated vertical and horizontal hydraulic conductivities were similar to the initial-input values and the conductivities listed in appendix 4 and discussed in "Aquifer Characteristics." Model-input, streambed hydraulic conductivities were the same as the initial-input values.

The flow model was calibrated by observing the difference between simulated and measured water levels (appendix 6) at model cells and estimated and measured-head data. Heads were estimated at new wells that had been installed after April 1991 by comparing water levels at nearby sites and estimating head fluctuations in the new wells on the basis of fluctuations measured in nearby wells. Ground-water levels for April were estimated at well sites where wells had not yet been installed by adding 2.2 ft to the November 1991 head for the same well site.

The standard mean difference between calculated and estimated or measured heads (appendix 6) compares favorably for simulation of an aquifer system with high hydraulic and vertical gradients and complex fluxes. The mean head difference, simulated head minus measured and estimated heads, was 0.6. The absolute mean head difference of 2.5 ft is an accurate measure of total error and indicates a favorable comparison between simulated and measured heads overall. Most of the simulation error is produced at the delta face and at the eastern model boundary. Simulated vertical gradients (the vertical-flow component) compared favorably with measured or estimated vertical gradients (appendix 6). The absolute error between measured- and simulated-head gradients was small (2.0 ft). The high upward vertical gradient (11.2 ft) measured at the municipal landfill (between wells W-316 and BR-6) was reproduced (11.9 ft) by a simulated flux from the bedrock aquifer. The downward vertical gradient measured between wells MW-101 and W-337 (indicated by a negative-head difference in appendix 6) was not reproduced by the simulation because of the complex hydrogeology at this location. The simulated head at W-337 also is the largest error in calculated head (-9.5 ft, appendix 6). Well W-337 is at the delta face, where the measured hydraulic gradient is very high: more than 30 ft of head change in a distance of 300 ft. Simulation of this high hydraulic gradient at the delta face was improved (initial head errors were greater than 20 ft) by reducing the cell discretization to 50 by 50 ft. Finer discretization allows model nodes to be centered closer to observation wells and allows for better representation of heads by representing smaller aquifer areas.

The simulated water table is shown in figure 17, and the simulated, potentiometric surface in the lower aquifer (layer 2) is shown in figure 18. Although comparisons between simulated and measured head surfaces are difficult to make because of the 20-foot-contour interval of the measured water-table surface (fig. 15) and the large hydraulic gradient across the model area, the simulated water table (fig. 17) compares favorably to the measured water table (fig. 15). Important features of the water table and potentiometric surface are reproduced by the ground-water-flow simulation; ground-water-flow directions throughout the model are correct, vertical gradients are similar to measured values, and simulated water-table contours appropriately indicate gaining and losing reaches of the New Haven River.

The simulated hydraulic gradient at the delta face (fig. 17) is slightly offset and not as large as the measured gradient. A better match of simulated to measured heads near the delta face and at the eastern model boundary could be achieved by use of an even finer discretization and by adjustments to recharge and hydraulic conductivity of the current model. However, because acceptable simulations of hydrologic conditions are produced and model parameters are within realistic ranges, for the purpose of this study, further refinements to the model are not warranted.

A relatively large head-difference error of 9.4 ft was measured near the east model boundary at well W-554. This error is not considered significant because (1) this is not an area of emphasis in the simulation, and (2) the match of measured to simulated heads at nearby wells (W-12, W-551, W- 552, W-553, W-555, W-556) was favorable. A much better match at well W-554 could be obtained by use of finer discretization in this area and by adjustments to or a redistribution of boundary fluxes; however, these improvements are not necessary for the purpose of this study.

Simulation errors greater than 3 ft at the two landfills are of a greater concern than simulation errors greater than 5 ft elsewhere in the model. Areas of significant errors between measured and simulated heads include the north model boundary, the till-stratified-drift interface, and along the face of the delta. The errors resulted for various reasons. Along the north model boundary, simulated heads were dependent on the simulated flux from bedrock to reproduce the measured heads and large upward vertical gradient. Without knowledge of bedrock-fracture locations and hydraulics, the estimated flux from the bedrock aquifer cannot be distributed accurately. Along the delta face, errors are inherently the result of significant lithology contrasts and large head changes (laterally and vertically).

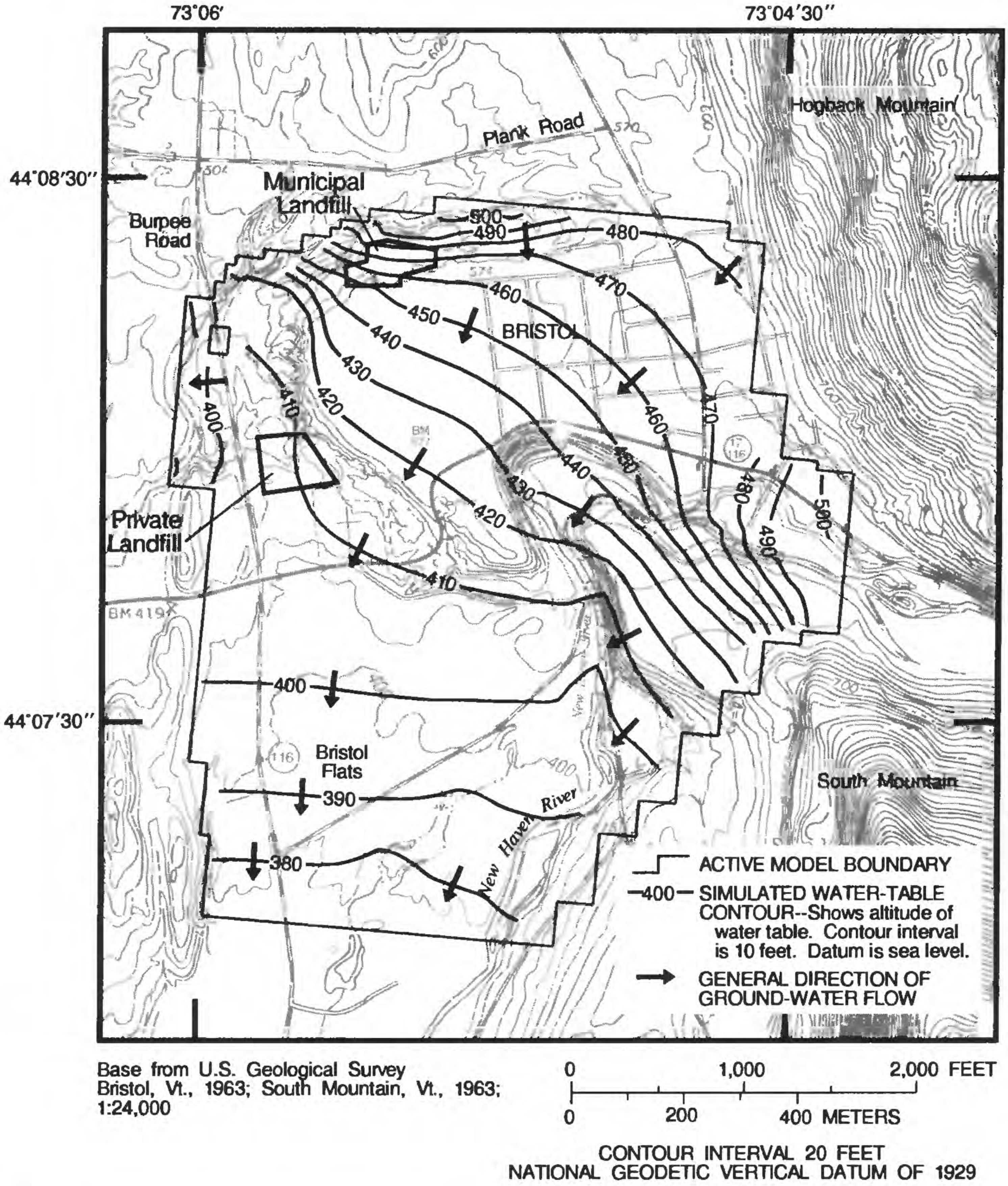
An estimated and simulated water budget for the study area is shown in table 6. Parameters such as direct recharge and leakage from bedrock were set at or near the estimated rates. Model fluxes such as streamflow leakage and ground-water inflow and outflow are simulated by head-dependent boundaries and not at a rate specified in a given model input-data set.

The calibrated model reproduces the estimated total water flux (18 ft^3/s) adequately. The major difference between the model and estimated budgets is that the model simulates less streamflow and more ground-water flow out of the aquifer system than is estimated. Most of the differences in flux are associated with the streamflow of the New Haven River and ground-water outflow from the aquifers at Bristol Flats (table 6). The simulation differences are not critical to the overall flow simulation or to flow at or near the landfills because the simulated total discharge at the south model boundary (14 ft^3/s) and the simulated total budget amount (17 ft^3/s) are plausible. The simulated ground-water seepage to the New Haven River (3.8 ft^3/s) is low because of the difficulties in reproducing the extreme gain (20 ft^3/s) between measurement sites 1 and 3 (table 3) and difficulties in proportioning the large upward discharge from the bedrock aquifer that produces the streamflow gain.

Specified fluxes are adequately simulated at the gap in the dolomite ridge at the west model boundary. Fluxes at this boundary affect flow at the private landfill and the municipal landfill. Simulated streamflow is slightly greater than estimated amounts.

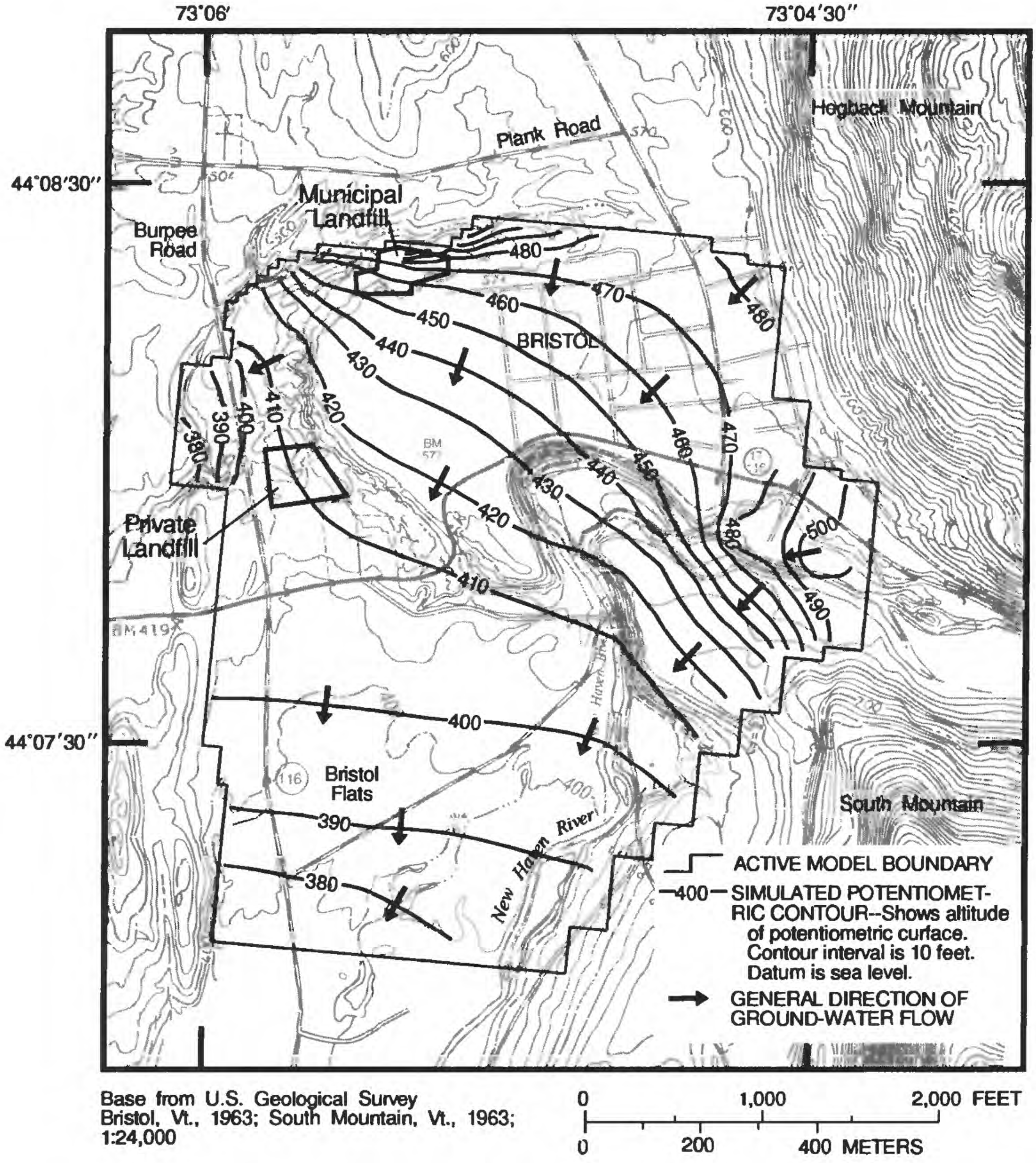
Sensitivity Analysis

The sensitivity of the model was analyzed to assess the relative effects of model-input parameters and boundary conditions on results of the simulations. The principal model parameters—direct recharge, streambed hydraulic conductivity, horizontal hydraulic



Base from U.S. Geological Survey Bristol, Vt., 1963; South Mountain, Vt., 1963; 1:24,000

Figure 17. Simulated water-table configuration of the upper aquifer (model layer 1).



Base from U.S. Geological Survey Bristol, Vt., 1963; South Mountain, Vt., 1963; 1:24,000

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Figure 18. Simulated potentiometric surface of the lower aquifer (model layer 2).

Table 6. Estimated and simulated ground-water budget for the study area in Bristol, Vermont

[Rates are in cubic feet per second. --, not simulated]

Water-budget component	Estimated or measured rate	Simulated rate
Recharge to aquifer:		
Direct recharge (precipitation)	1.8	1.7
Streamflow leakage	.3	--
Leach fields	.4	--
Ground-water inflows:		
Eastern boundary at New Haven River.	1.7	1.7
Leakage from bedrock	13.9	13.7
Total	18.1	17.1
Discharge from aquifer:		
Discharge to streams:		
Northwest Brook	.2	1.3
Bristol Flats Brook	3.0	2.3
New Haven River	8.6	3.8
Subtotal	11.8	7.4
Ground-water discharge		
Western boundary at dolomite gap.	1.3	1.8
Southern boundary at Bristol Flats.	5.0	7.9
Subtotal	6.3	9.7
Total	18.1	17.1

conductivity, flux to the glacial aquifer from the bedrock aquifer, and general-head-boundary hydraulic conductivity—were changed by 50 percent above and below the model input or calibrated values. Vertical hydraulic conductivity was changed by factors of 0.1 and 10 times the calibrated values. Results of the sensitivity analysis are summarized in figure 19. Simulation results are shown as boxplots of water-level residuals (simulated head minus measured head) for the 58 wells and checkpoints listed in appendix 6, and as simulated discharge to streams and from ground-water discharge at the west model boundary for each of 13 model simulations.

The model is most sensitive to changes in horizontal hydraulic conductivity and recharge from bedrock (fig. 19). Investigators (de Lima and Olimpio, 1989; and Harte and Mack, 1992) have shown that ground-water-flow models are most sensitive to reductions in horizontal hydraulic conductivity. The model is least sensitive to adjustments of vertical hydraulic conductivity and the general-head-boundary hydraulic conductivity. The model was also minimally sensitive to changes in areal recharge and streambed hydraulic conductivity. None of the sensitivity simulations indicate that changes to any one parameter could produce an overall improved simulation. Recharge, streambed conductivity, and general-head-boundary conductivity values increased by 50 percent and produced head distributions similar to the calibrated simulation but with larger overall error. A reduction in streamflow at the west model boundary, which compares favorably with estimated streamflow, can be produced by a reduction of the flux from bedrock; however, this is not a viable simulation because the resultant simulated heads are too low.

Ground-Water Flow

Analysis of the ground-water flow in the upper and lower aquifers is based on calibrated-model simulations of head (appendix 6) (figs. 17 and 18), the water budget (table 6), and flow-path projections based on results from a particle-tracking analysis. Ground-water-flow paths at the two landfills were calculated by placing particles at various locations in the upper and lower aquifers beneath the landfills.

Regional Flow

The regional ground-water flow in the study area is unusual because it is largely controlled by flow from the underlying and adjacent bedrock aquifer, which is the dominant recharge component to the aquifer system. Recharge from bedrock leakage accounts for about 80 percent of the total simulated recharge, whereas recharge from precipitation accounts for only about 10 percent of the total recharge. This large flux from bedrock leakage is indicated by large upward head gradients throughout the aquifer system and relatively high heads in the northern part of the glacial aquifer (figs. 17 and 18, appendix 6).

The New Haven River is a major feature in the Bristol aquifer system, and, although large gains and losses are noted in the measured streamflow, the river generally is a ground-water drain (table 3). A river entering a coarse-grained glacial aquifer and having a head higher than that in the aquifer loses water to the glacial aquifer (Harte and Mack, 1992). At Bristol, however, large streamflow gains were measured as the river crossed the aquifer because of the ground-water leakage from bedrock. Small streams flowing off the flank of the mountains commonly lose water to the aquifer;

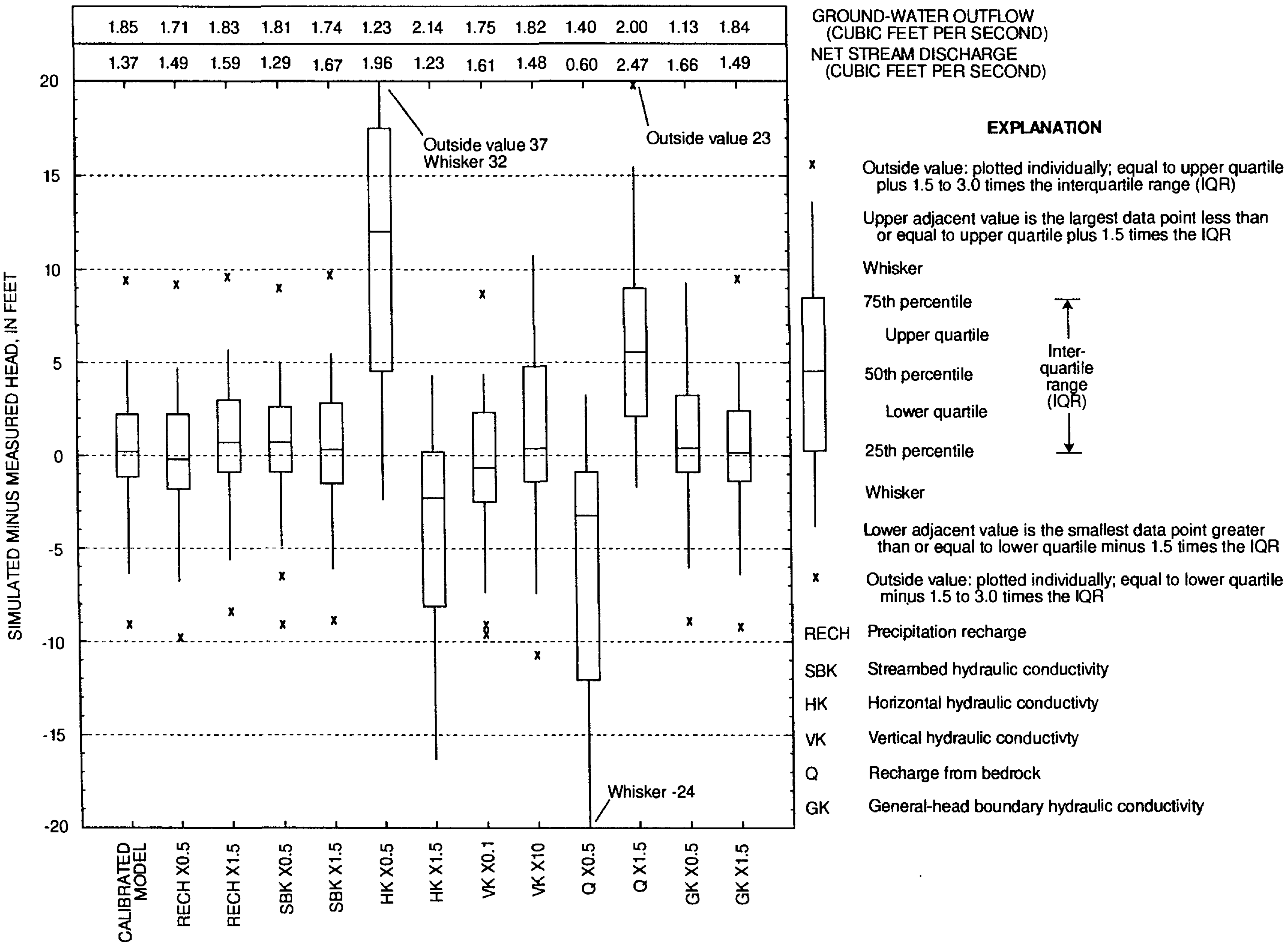


Figure 19. Statistical distribution of the differences between measured and simulated heads for sensitivity tests for the ground-water-flow model of Bristol, Vermont.

however, the stream at Bristol Flats and the northwestern stream are gaining streams and thus drain the glacial aquifer. Simulated ground water discharged to streams accounts for about 43 percent of the total aquifer discharge (table 6). Measured discharges to streams account for about 65 percent of the total discharge.

Ground water in the simulated glacial-aquifer system discharges primarily through the south model boundary at Bristol Flats, which represents 80 percent of the total ground-water discharge. Ground-water discharge at the south boundary accounts for 45 percent of the total simulated aquifer system (combined stream water and ground water) discharge. The remaining 20 percent of the total ground-water discharge is at the gap in the dolomite ridge along the west model boundary.

Flow at the Landfills

The simulated path of ground-water flow in the aquifers beneath and near the landfills are shown in figures 20 and 21. Flow lines represent the simulated path of a particle placed at specific locations in a model cell (25 and 75 percent of the cells thickness) and tracked forward (in the direction of flow) from the landfills. Flow paths do not represent the concentration of a chemical constituent but simply the average path that water from a specific location can be expected to take through the aquifers.

Private Landfill

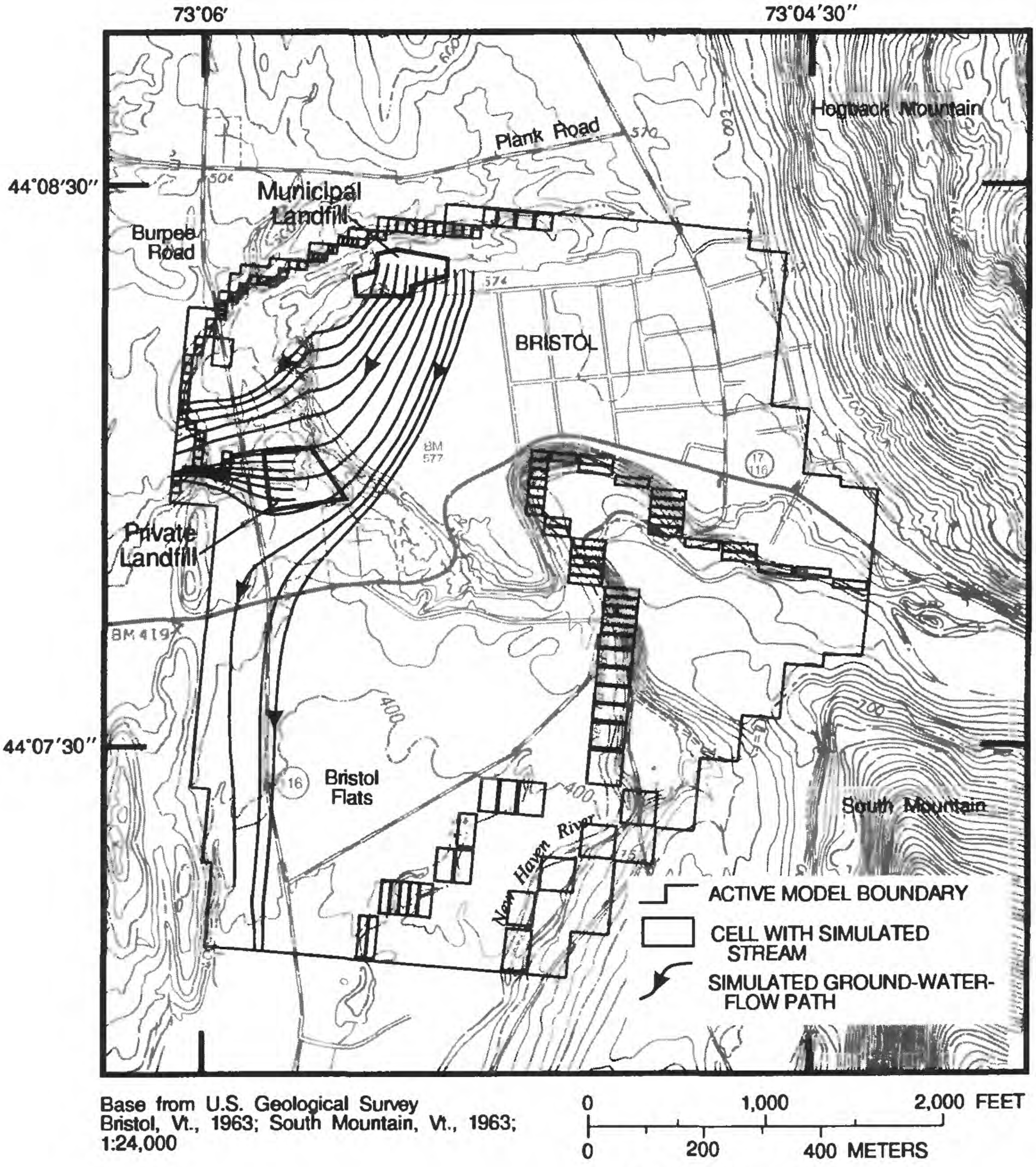
Particle-tracking analysis indicates a ground-water divide immediately south of the private landfill in the upper aquifer, where flow is either west to the gap in the dolomite ridge (figs. 20 and 21) or south towards Bristol Flats. The analysis indicates that ground water at the southeast edge of the private landfill may eventually flow southward. Ground water beneath the remainder of the landfill discharges to the dolomite gap area, although the ground-water flow in the lower aquifer beneath the south boundary of the private landfill apparently flows southward before discharging to the west.

A sectional view of the flow path of ground water in the aquifers beneath the private landfill is shown in figure 22. The section shown is a MODPATH approximation of a dipping, variable-thickness aquifer section, represented here by horizontal rectangular cells and local coordinates as a nonsloping uniform aquifer. The line of cross section (row 34, fig. 22) was selected where the direction of ground-water flow is predominantly parallel to the row direction. Pathlines are constructed by placing a few particles at selected starting locations in the upper and lower aquifer along model row 34 and initiating a forward pathline analysis. Most of the resultant pathlines are not parallel to the model grid; that is, there is a horizontal component of flow into or out of the page. As a result, those flow paths are projected through the aquifer onto the model row shown to illustrate the general flow paths along this section of the aquifer. Particle paths that leave a row and travel in an adjacent row cannot be shown without such projection. Because of this path deviation, only a few particle paths are shown to reduce overlapping pathlines and to avoid obscuring the figure.

Water that enters the aquifer at the surface of layer 1 at the private landfill follows a relatively short flow path and discharges to the aquifer and stream at the dolomite gap (fig. 22). Distinctions should not be made in the upper aquifer (model layer 1) between flow-path discharges to the stream and ground-water flow past the stream through the dolomite gap. MODPATH cannot accurately partition flow paths to the sinks in a cell (in this case the stream and the aquifer) because all sinks are uniformly distributed within the cells (Pollock, 1989). Although the total flow budget is acceptable, model layer discretization is probably not fine enough to accurately represent vertical partitioning of flow at the stream cells in layer 1. Ground water near the top of the lower layer flows into the upper layer as it moves westward towards the dolomite gap (fig. 22). Ground water deepest in the lower aquifer apparently flows westward to the general-head-discharge boundary at the dolomite gap with less upward movement.

Municipal Landfill

The particle-tracking analysis shows that ground water from the municipal landfill flows southward and westward (figs. 18 and 19). The flow paths from this landfill cover a greater amount of the study area than the simulated flow paths for the private landfill. Ground water from the west end of the municipal landfill follows a relatively short flow path to discharge at the north end of the dolomite gap. Ground water from the middle to the eastern parts of the municipal landfill follows a much longer flow path beneath the private landfill and discharges at the south end of the dolomite gap. Ground



Base from U.S. Geological Survey
Bristol, Vt., 1963; South Mountain, Vt., 1963;
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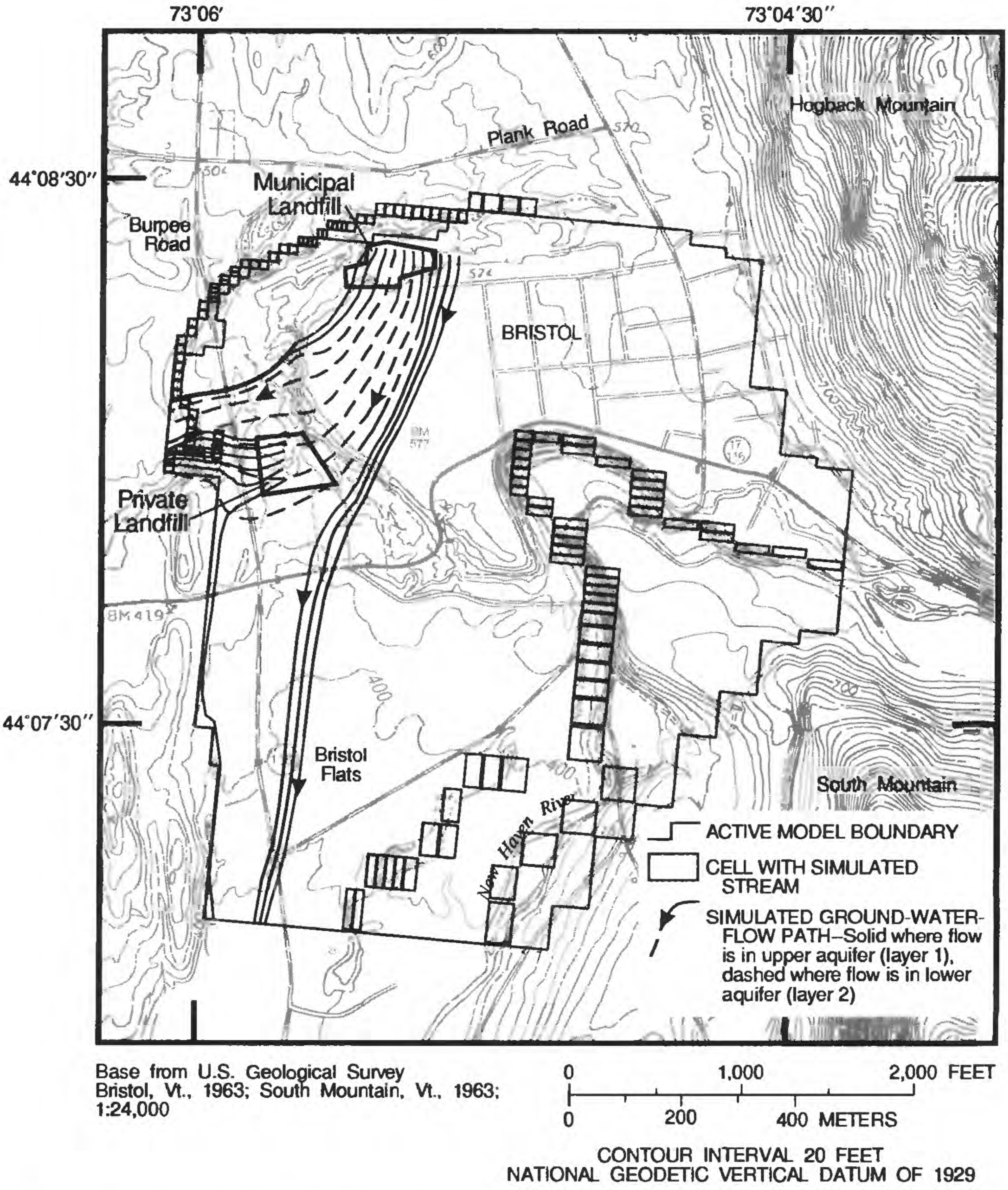
Figure 20. Simulated ground-water flow paths for water in the upper aquifer (model layer 1) beneath the landfills.



Base from U.S. Geological Survey
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Figure 21. Simulated ground-water flow paths for water in the lower aquifer (model layer 2) beneath the landfills.

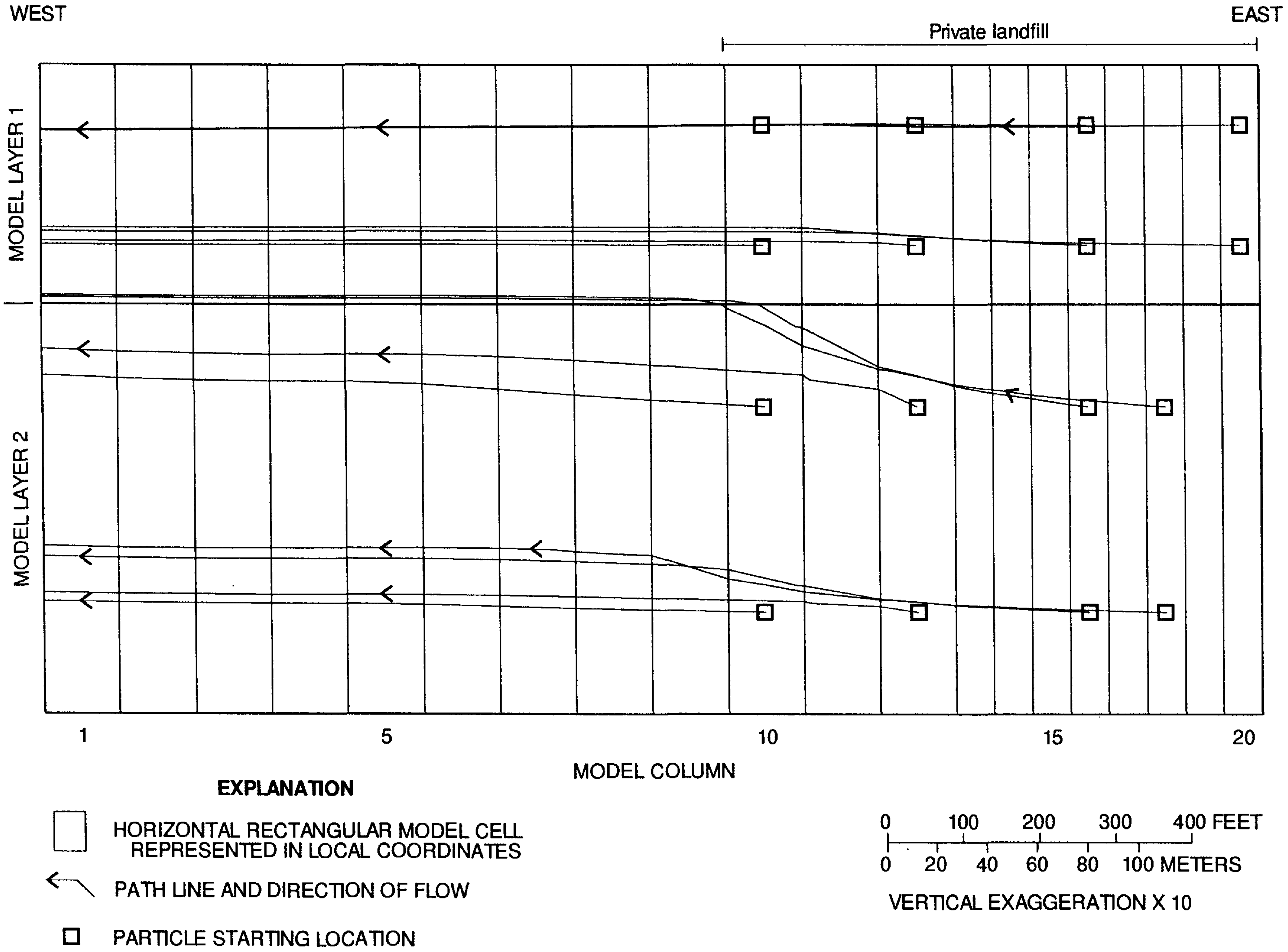

Figure 22. Section showing ground-water flow path along model row 34 at the private landfill in Bristol, Vermont. (Row location shown in fig. 16)

water beneath the easternmost part of the municipal landfill flows just east of the private landfill and then discharges southward through Bristol Flats (fig. 18).

Sectional views of the ground-water-flow paths from the municipal landfill are shown in figure 23 along model row 26 and in figure 24 along column 32. Row 26 and column 32 were selected to examine vertical and horizontal flow because lateral flow was closely aligned in the row and column directions. A few particles were placed selectively in model cells to reduce overlapping pathlines, and flow paths were projected onto row 26 or column 32 to produce readable figures.

The flow paths projected along column 32 (fig. 24) indicate an upward gradient beneath the landfill and show that most of the simulated ground water flows into the upper aquifer (layer 1) and discharges at the dolomite gap (rows 23-36). Ground-water flow near the top of the upper aquifer (layer 1) remains at the water-table surface. Ground water beneath the eastern part of the landfill follows a simulated flow path to Bristol Flats (fig. 17). Ground water flowing westward from the municipal landfill along row 26 (fig. 23) follows a similar vertical flow pattern. Most ground water at this site flows to the upper aquifer and discharges to either the stream or the boundary at the dolomite gap. Few flow paths remain in layer 2. Some flow paths originating at column 32 (fig. 24) and ending in rows 34 and 35 layer 1, appear to stop in the middle of the figure, but they actually follow a westerly path perpendicular to the figure.

Model Appraisal

An important consideration of a numerical ground-water-flow simulation is the uniqueness of the solution. Various combinations of reasonable model input parameters and boundary conditions can produce similar or improved simulations of ground-water flow. This is particularly applicable for a complex-flow simulation such as the one presented in this report. The number of nonunique solutions can be reduced if estimates of input parameters and fluxes are as accurate as possible and if proposed and known hydrologic concepts of the flow field are tested in preliminary simulations in order to refine the conceptual model.

A comparison of measured and estimated heads to simulated heads ("Calibration" Section) indicates that the simulation of ground-water heads in the upper and lower aquifers is acceptable. The simulation error—absolute mean-head difference of 2.5 ft and standard mean-head difference of 0.6 ft—is acceptable for a complex aquifer system such as that underlying Bristol with steep head gradients. Large residuals are noted at an observation well near the delta face; however, this error may not be significant because of the steep head gradients (fig. 17) at this location. Simulated differences in head between the upper and lower aquifer ranged from 1 to 11 ft and compared favorably to measured head differences. A comparison of the estimated and simulated ground-water budget for the model (table 6) indicates that the simulation reproduces the estimated budget closely. The primary error in the simulation is that more ground water and less surface water discharges from the aquifer at the southern model boundary than was estimated. However, the simulated combined ground- and surface-water discharge closely approximates the combined estimated discharge. Because the southern model boundary is not the primary area of interest in this study, the effect of this discrepancy is considered negligible.

GROUND-WATER QUALITY

Ground-water samples from 53 observation wells and 7 springs and surface-water sites were collected from September 1990 to October 1991 for analysis of common inorganic and organic constituents. All water samples were collected and analyzed by the VANR in accordance with USEPA sampling guidelines, and results were compared to State and Federal primary drinking-water regulations established for chemical constituents that can produce adverse health effects (Vermont Agency of Natural Resources, 1988; U.S. Environmental Protection Agency, 1991). Analytical results are presented in appendix 7.

Chemical properties of the ground- and surface-water samples varied greatly. On the basis of the analytical results, ground water in the Bristol area can be divided into two categories: (1) native ground water, or water that has been little affected by human activities or landfill processes and (2) water that can be associated with landfill processes, such as landfill leachate and leachate-degraded water. Landfill leachate has high concentrations of chemical constituents resulting from the percolation of water through landfilled materials and (or) the ongoing chemical reactions between leachate and aquifer materials. Leachate-degraded water could have concentrations of chemical constituents that exceed background concentrations but are less than

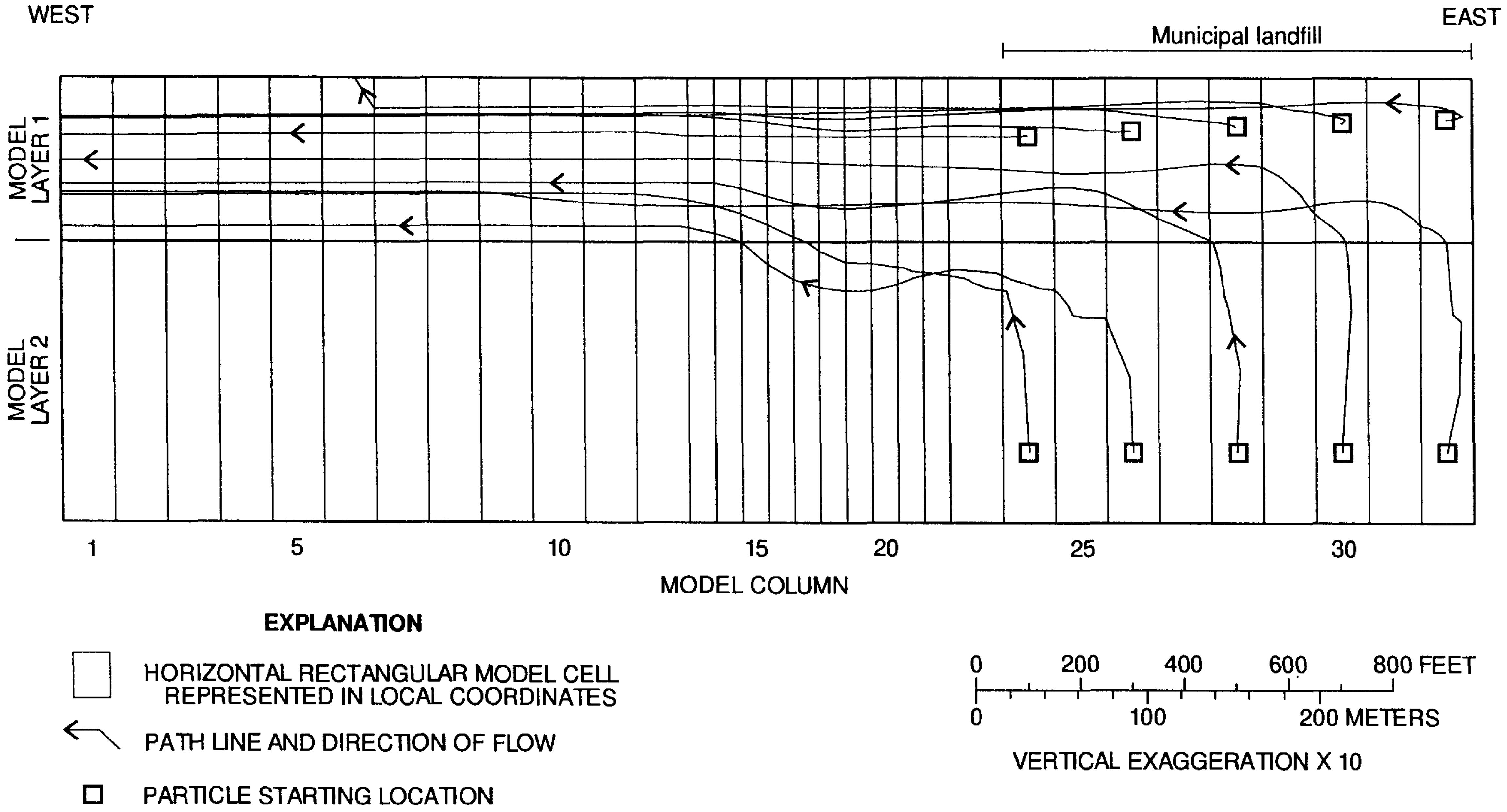

Figure 23. Section showing ground-water flow path along model row 26 at the municipal landfill in Bristol, Vermont. (Row location shown in fig. 16.)

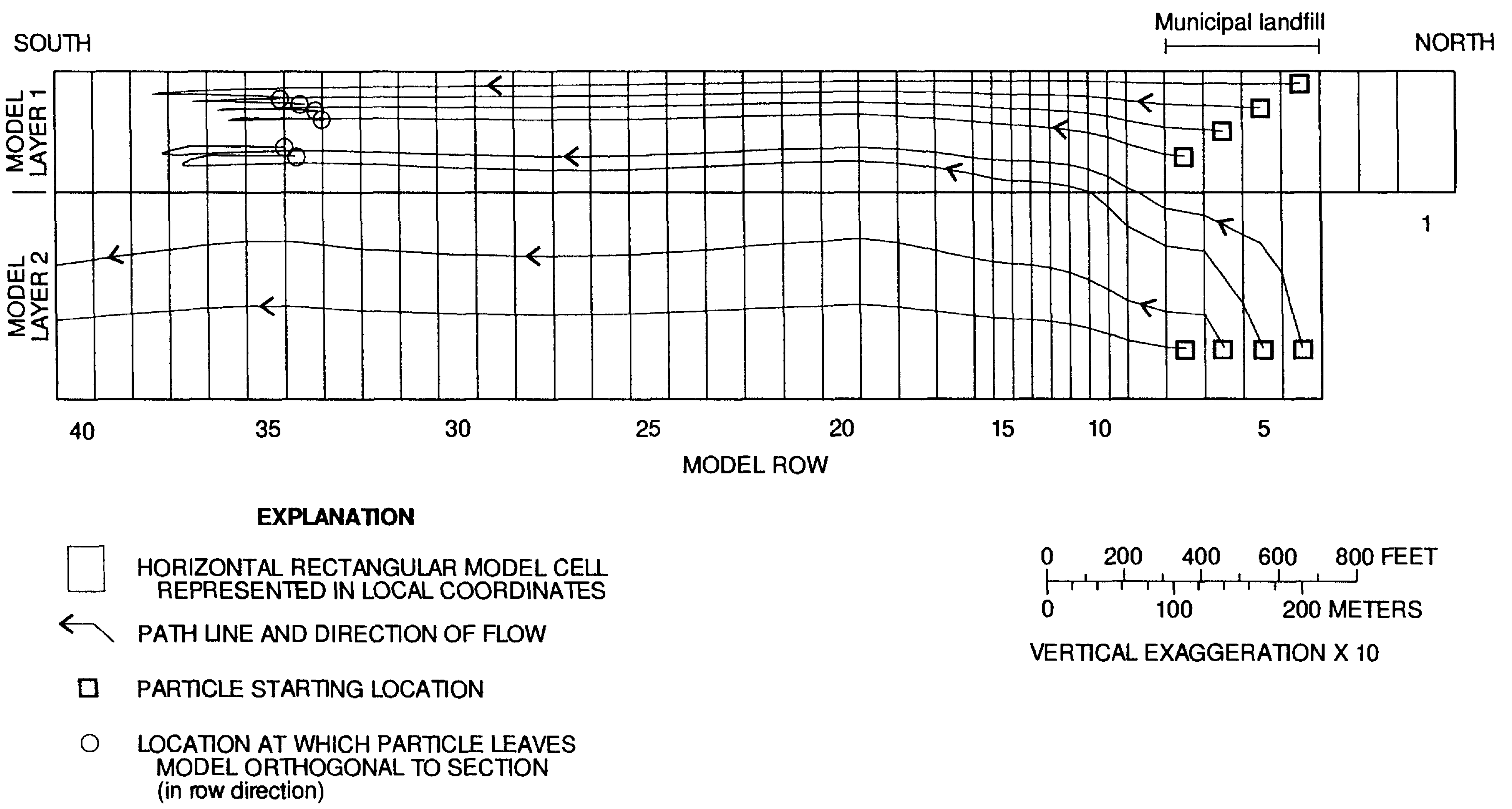

Figure 24. Section showing ground-water flow path along model column 32 at the municipal landfill in Bristol, Vermont. (Row location shown in fig. 16.)

those measured for landfill leachate. This water is probably the result of diffusion and mixing of leachate with native ground water. Wells, springs, and surface-water sites sampled are listed in table 7; sites are grouped into native or landfill-degraded categories. Samples from some sites had an unusual detection of chemical constituents above background concentrations and were identified as possibly affected by landfill leachate (table 7).

Quality of Native Water

Water in wells not degraded by landfill leachate (table 7) was sampled to obtain information on background concentrations of chemical constituents in native ground water. Ranges and median concentrations of those chemical constituents found in native water from wells and springs are listed in table 8. Native water in the glacial aquifers is generally soft (low calcium and magnesium concentrations) and moderately low in dissolved solids, as indicated by a median specific conductance of 354 µS/cm.

Table 7. Observation wells, springs, and surface-water sites sampled, by location relative to landfill and by water category, Bristol, Vermont

[Well locations: See figure 5. Spring locations: See figure 8. Prefixes W-, BR-, MW- indicate wells; prefix S- indicates spring or surface-water site]

Native water				
Municipal landfill		Private landfill		Uncategorized
W-303	S-10	W-304	W-321	W-12
W-309	S-11	W-305	[1]W-322	[2]W-312
[1]W-310	S-13	W-306	W-323	W-315
[1]W-311		[1]W-307	[1]W-324	
W-318		[1]W-308	W-329	
[3]W-319		[1]W-320		
Leachate-degraded water				
Municipal landfill		Private landfill		
W-301	W-2	W-507	W-338	
W-302	W-4	W-508	W-339	
W-316	BR-2	W-509	W-340	
W-317	BR-6	W-510	W-341	
W-327	MW-101	W-511	W-342	
W-328	MW-102s	W-325	W-343	
W-330	MW-102	W-326	Pond	
W-331	W-334	[3]W-332	S-15	
W-335	[3]W-336	[3]W-333	S-16	
W-337	S-14			

[1] Possible landfill-leachate effects.

[2] Possible road-salt contamination.

[3] Sampling was insufficient or not possible; additional sampling is necessary to accurately characterize water from this well.

The primary cations in native water are calcium, magnesium, sodium, and sulfate (table 8). Other ions commonly found at small concentrations include iron, manganese, and barium. Concentrations of iron ranged from less than 10 to 2,890 µg/L, the median concentration was 42 µg/L.

In general, the metals that were analyzed for were at concentrations less than detection limits (table 8). Fifteen of 41 samples (appendix 7) had concentrations of dissolved copper that exceeded the detection limit (10 µg/L); however, the estimated 75th percentile (12 µg/L) only slightly exceeded the detection limit. Concentrations of lead, nickel, and zinc exceeded the detection limits in a few samples (appendix 7). Concentrations of cadmium were found at or near the detection limit (4, 2, and 2 µg/L) in three native-water samples (W-310, W-322, and W-307) on separate occasions. Concentrations of chromium, cobalt, arsenic, selenium, and silver did not exceed the detection limit in native-water samples.

Water from some wells had low concentrations of many constituents; yet, these results may not completely represent native water. For example, two water samples from well W-334, which is screened in the lower aquifer downgradient from the municipal landfill, had significant concentrations of nickel (23 and 13 µg/L) and elevated concentrations[2] of sodium (48.2 mg/L) and copper (76 µg/L). Similarly, water from well W-308. which is adjacent to the private landfill but not directly downgradient from it, had significant concentrations of copper (21 and 17 µg/L) in two samples and elevated concentrations of zinc (112 and 72 µg/L), and lead (73 µg/L) in three samples. Water from well W-324, screened in the lower aquifer adjacent to the private landfill, had low concentrations of all constituents analyzed for, with the exception of two volatile organic compounds (VOC's) that were slightly above detection limits (53 µg/L of acetone and 56 µg/L of 2-butanone). Well W-324 was installed later in the investigation, and only one sample was collected and analyzed; therefore, it is uncertain if this one detection is an anomaly. Further sampling would be necessary at these wells to determine if the water sampled is affected by landfill leachate.

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[2] In this report, the term "elevated concentration" refers to a concentration above regulatory limits or median concentrations.

Table 8. Regulatory limits and summary statistics for physical properties and chemical constituents in native water from wells and springs in Bristol, Vermont, 1990-91

[SMCL, Secondary maximum contaminant level set by the U.S. Environmental Protection Agency (1988c). MCL, Maximum contaminant level set by the U.S. Environmental Protection Agency (1988a). Enforcement standard that requires notification of the Vermont Agency of Natural Resources (1988) and written communication, 1994. Statistics for mean, standard deviation, 25th percentile and 75th percentile are calculated by methods described by Helsel and Cohn (1988). µS/cm, microsiemen per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter. <, actual value is less than value shown; -- not applicable or too few to calculate statistics]

Physical property or chemical constituent	Number of samples	Number less than detection limit	SMCL	MCL	Enforcement standards	Mean	Standard deviation	25th percentile	Median	75th percentile	Maximum detection
Physical properties											
Specific conductance (µS/cm)	52	--	None	None	100	347	62	307	354	384	465
pH (units)	10	--	6.5-8.5	None	None	--	7.15	7.52	7.60	--	--
Major constituents (mg/L)											
Calcium, dissolved	36	0	None	None	--	42.4	16.1	36.3	39.6	44.4	118
Magnesium, dissolved	37	0	None	None	None	20.4	6.1	16.8	20.8	22.1	43.8
Sodium, dissolved	48	0	None	None	10	1.56	8.10	4.88	9.02	13.05	48.2
Potassium, dissolved	36	0	None	None	--	.34	.55	1.11	1.30	1.46	3.80
Sulfate, dissolved	53	0	250	400	250	17.84	7.88	13.05	16.54	20.35	43.0
Chloride, dissolved	52	0	250	None	250	9.9	9.8	2.5	6.7	15.0	40.9
Trace constituents (µg/L)											
Arsenic, dissolved	38	38	None	50	50	--	--	--	--	--	<5
Barium, dissolved	38	0	1,000	2,000	2,000	56	44	24	38	75	203
Cadmium, dissolved	53	50	None	5	5	--	--	--	--	--	4
Chromium, dissolved	47	47	50	100	100	--	--	--	--	--	<10
Cobalt, dissolved	24	24	None	None	None	--	--	--	--	--	<10
Copper, dissolved	41	26	1,000	1,300	1,300	12	27	1	4	12	178
Iron, dissolved	54	10	300	None	300	227	465	14	42	254	2,890
Lead, dissolved	53	38	None	[1]5	15	--	--	--	--	--	73
Manganese, dissolved	52	6	50	None	50	196	228	47	114	242	1,100
Nickel, dissolved	38	35	None	100	100	--	--	--	--	--	23
Selenium, dissolved	38	38	None	50	50	--	--	--	--	--	<5
Silver, dissolved	38	38	100	50	50	--	--	--	--	--	<5
Zinc, dissolved	53	46	5	None	5,000	--	--	--	--	--	308

[1] MCL under review (U.S. Environmental Protection Agency, 1991)

Extent of Contaminated Ground Water

Leachate contaminated ground water is produced by the percolation of recharged water through landfilled waste. Water in the waste is displaced by additional recharge to the landfill. Chemical processes involved in the production of leachate include oxidation, reduction, dissolution, precipitation, ion exchange, and sorption (Baedecker and Back, 1979). Solids, liquids, and gases from the waste are incorporated into (or) suspended in the leachate. Bacterial action produces metabolic carbon dioxide that dissolves in water and decreases pH. Leachate-solvent capability increases by addition of bacterially generated organic acids. The resulting acidic condition favors increased hardness and increased concentrations of solids and metals—particularly iron and manganese—compared to background concentrations. Chloride and sulfate salts of sodium, potassium, calcium, and magnesium are highly soluble and are generally leached directly from wastes (Nicholson and others, 1983). Leachate generation is a complex process that varies with waste composition, hydrogeology, and landfill age and design. Elevated concentrations of constituents are expected only in new unlined landfills that have a rapid rate of early-stage biodegradation (Myers and Bigsby, 1990).

Specific-conductance data have commonly been used to map the extent of landfill-leachate contamination plumes (Kimmel and Braids, 1980; Wexler, 1988a). In the Bristol area, ground water contaminated by leachate is readily distinguishable from native ground water, which generally has a specific conductance less than 400 µS/cm. The extent of leachate contamination near the landfills is indicated in figure 25 by lines of equal specific conductance in ground water. Surface and borehole geophysical data were used to infer contours of specific conductance where water samples were not available. Leachate contamination was detected with application of borehole geophysical logging (Mack, 1993).

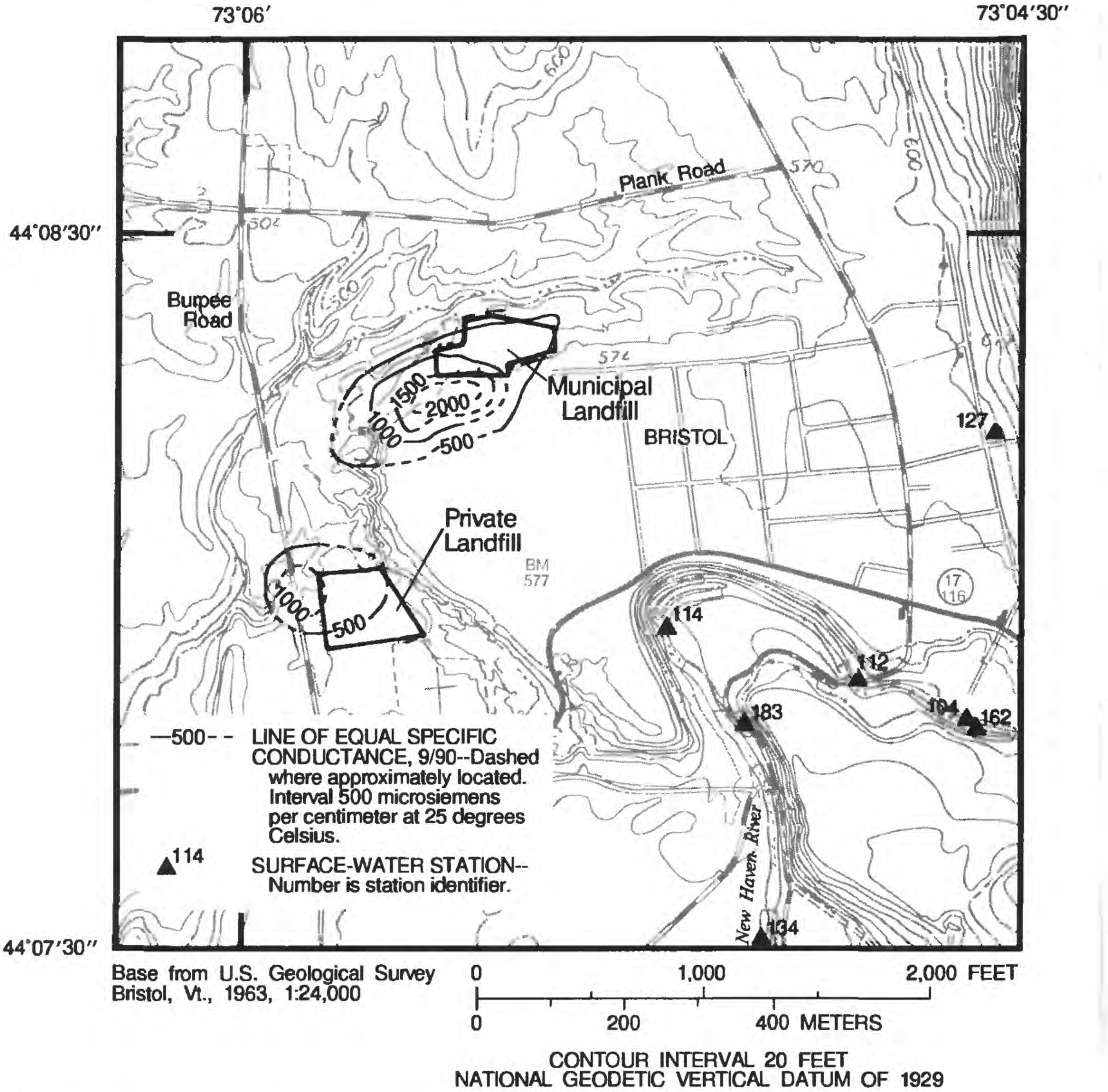

Figure 25. Specific conductance of ground water in Bristol, Vermont, September, 1990.

Private Landfill

The leachate plume, an area of ground water degraded by leachate, emanating from the private landfill (fig. 25) does not extend downgradient as far as the plume from the municipal landfill and appears to be wide for the size of the private landfill. Factors such as the contour of the water table, horizontal hydraulic conductivity, and permeability and surface-water runoff of the landfill probably contribute to a laterally dispersed plume at the private landfill. The water table at the private landfill has a radial-flow component. The ground-water-flow simulation, and the particle-tracking analysis (figs. 18 and 19) indicate that ground water in the upper aquifer (model layer 1) flows southwestward before flowing westward to the dolomite gap. The borehole geophysical logs of wells W- 320, W-322, and W-307 (appendix 4) indicate such a flow pattern from the area of the landfill by the presence of a thin zone of conductive ground water at the water-table surface at wells W-320 and W-322 but not farther southwest at well W-307. Additionally, horizontal hydraulic conductivity was lower at the private landfill than at the municipal landfill site. Low hydraulic conductivity can increase seasonal water-level fluctuations, which then affects ground-water flow and increase lateral dispersion of chemical constituents. Surface runoff could disperse the leachate plume from the private landfill because surface runoff is channeled and directed by a culvert to an area across the ground-water-flow gradient near observation well W-510 (northwest of the landfill). After significant rainfall, runoff having a specific conductance that exceeds 2,000 μS/cm is ponded in an area adjacent to well W-510. The ponding of the runoff probably altered the original extent of the leachate plume produced from this landfill.

Water samples collected from observation wells W-332 and W-333 had elevated specific conductances and elevated concentrations of sodium. Samples from well W-333 may not be representative of native ground water because the well may not have been fully developed and the local ground water might still contain drilling mud. This well cluster is less than 150 ft upgradient from the landfill; it is uncertain whether leachate could be moving with radial flow or whether leachate reached this location by lateral movement through the 80 ft thick unsaturated zone. Further sampling would be necessary for adequate characterization of the area upgradient from the private landfill.

Figure 26 shows the vertical extent of the plume in the downgradient direction along section *B-B'* near the private landfill. This geologic section is constructed from an analysis of borehole geophysical and lithologic logs of wells along this line of section (Mack, 1993). The EM logging was more sensitive than the surface EM techniques to elevated specific conductances associated with leachate contamination (Mack, 1993). For example, the elevated specific conductance at well W-339 was not distinguishable from the aquifer materials by surface EM methods. The leachate plume is concentrated near the water-table surface near the landfill and is deeper in the upper aquifer as the distance downgradient from the landfill increases. The leading edge of the plume is near observation well W- 339, in a zone less than a few feet thick about 17 ft below the water-table surface. The particle-tracking analysis shows that ground water in the upper aquifer (model layer 1) remains in the upper aquifer downgradient from the landfill (fig. 22).

Municipal Landfill

The leachate plume at the municipal landfill is indicated in figure 25. The size and location of the plume is a result of the size of the landfill and the directions of ground-water flow beneath the landfill (figs. 17, 18, 20, and 21). The leachate plume is only slightly wider than the width of the landfill and extends about 500 to 700 ft from the landfill in the direction of ground-water flow. As a result, the longitudinal dispersion of the plume is greater than the lateral dispersion, a finding that is consistent with those of Kimmel and Braids (1980) for two landfills in New York. The most concentrated leachate contamination, as indicated by the specific conductance of the ground water (well W-335, fig. 25), is directly downgradient (southwest) from the western half of the landfill and is concentrated in the upper aquifer, as indicated by the simulated ground-water-flow paths (figs. 23 and 24).

Surface-geophysical measurements from the area near well W-335 indicate that leachate contamination decreases in the direction of ground-water flow; however, further testing in this area was not possible. The ground-water-flow simulation indicates that ground water from beneath the eastern part of the municipal landfill remains in the upper aquifer and flows past the north side of the private landfill. Ground water beneath the gravel pits immediately west of the landfill is slightly affected by landfill leachate. Borehole geophysical logs indicate that well W-301 is the only well adjacent to the

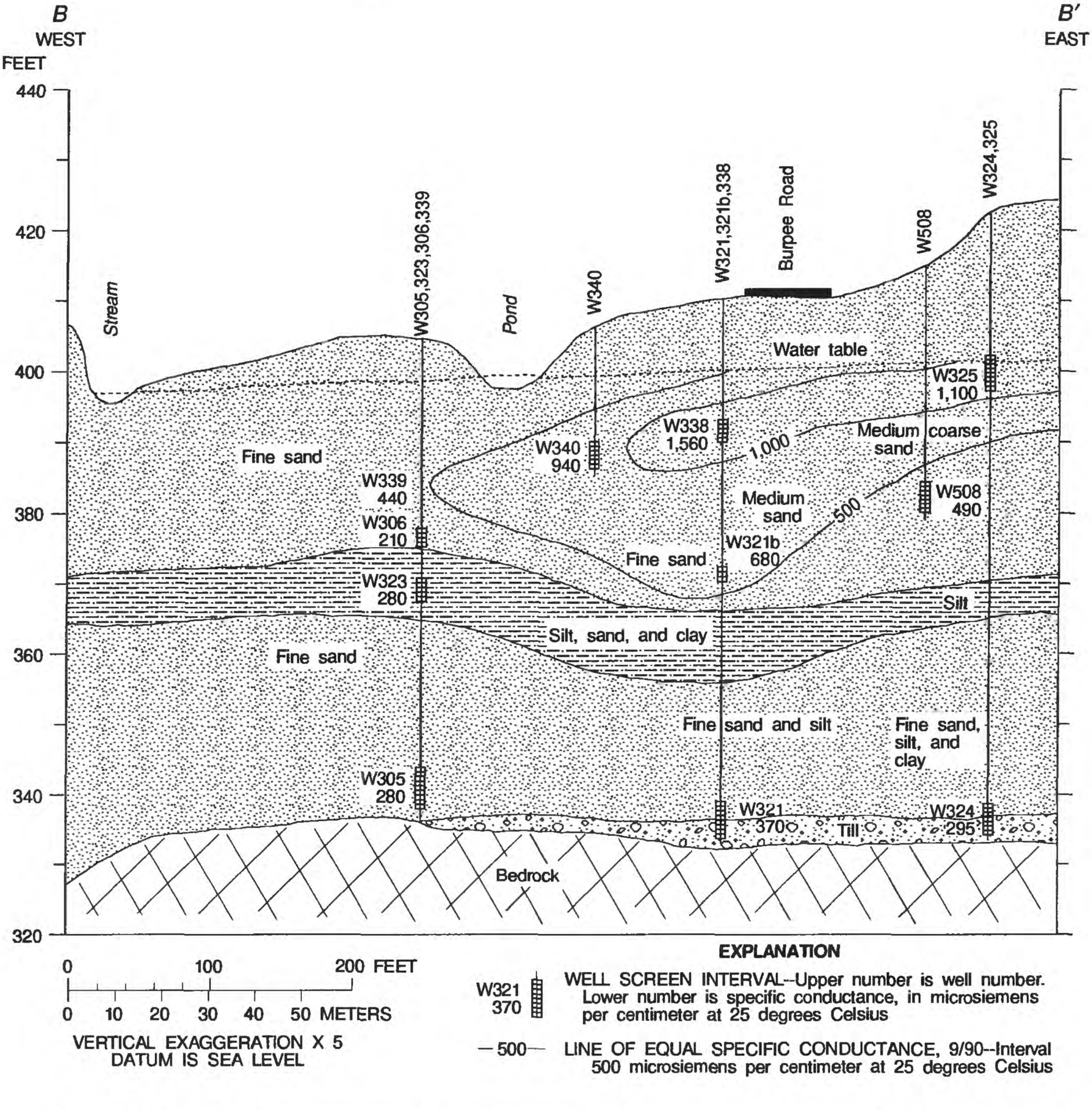

Figure 26. Hydrogeologic section *B-B′* showing specific conductance of ground water at the private landfill. (Line of section shown in figure 5)

delta that is affected by leachate from the municipal landfill. Borehole geophysical logs of wells W-310 and W-303 do not indicate leachate contamination northwest of the landfill (appendix 4).

Surface-geophysical measurements indicate that leachate contamination probably disperses immediately west of wells W-301, W-331, W-327, and W-304. Surface-geophysical measurements near W-304 and borehole-geophysical measurements of W-304 did not indicate the presence of leachate contamination from the municipal landfill immediately north of the private landfill. Contaminants that typically make up leachate are either dilute (less than background concentrations) or not present in ground water north of the private landfill. Ground water from the east end of the municipal landfill follows a long flow path (fig. 18) east of the private landfill. As a result, any elevated concentrations of chemical constituents that can be attributed to the landfill leachate have probably been reduced to background concentrations by the time this water reaches Bristol Flats.

Characteristics of Contaminated Ground Water

Physical Properties and Major Constituents

Analyses of ground-water samples collected from wells screened near the water-table surface at the two landfills were examined for evidence of contamination by leachate (appendix 7). Samples from a spring at the face of the delta in a gravel pit directly west of the municipal landfill were included in the analyses for the municipal landfill. Samples from a spring at the bank of a small stream about 800 ft west of the private landfill were included in the analysis for the private landfill. Springs were included with the ground-water analyses because they could be sampled at the source where water emerged from the ground. Other surface-water samples were not included in this analysis. Summary statistics of chemical analyses of ground water at or near the municipal and private landfills are presented in tables 9 and 10. The results indicate elevated concentrations of several chemical constituents.

Concentrations of physical properties and major constituents in ground water at the landfills were compared to those in native water (fig. 27). Specific conductance, an indicator of dissolved-solids concentration, exceeded background levels for ground water underlying both landfills. The median specific conductance (fig. 27) was greater at the private landfill (655 µS/cm) than at the municipal landfill (491 µS/cm); however, more wells were installed and sampled in the leachate plume at the private landfill than at the municipal landfill. The highest specific conductance (2,920 µS/cm) and concentration of sodium (432 mg/L) were at a well downgradient from the municipal landfill (W-335). Medians for these constituents, however, were higher for the private landfill (tables 9 and 10) than for the municipal landfill.

Median concentrations of dissolved calcium were greater in water associated with leachate contamination from the private landfill than in water associated with leachate contamination from the municipal landfill. Median calcium concentrations from samples at the private landfill were about twice the background concentrations (fig. 27); however, the 75th percentiles of all sample concentrations were approximately the same for both landfills and were more than twice the 75th percentile for native water samples. At wells near the landfills, samples with elevated concentrations of calcium were similar in concentration, but samples having elevated calcium concentrations were more numerous at the private landfill than at the municipal landfill. Median concentrations of potassium at both landfills (fig. 27) were only slightly higher than background concentrations (1 mg/L); however, analysis of samples degraded by landfill leachate reveal a range of potassium concentrations from 0.84 to 23.9 mg/L. The potassium concentrations in leachate degraded water from the two landfills were similar.

Median concentrations of sulfate in ground water contaminated by leachate were similar to background concentrations (16 mg/L, fig. 27); however, samples from wells W-332, MW-101, W-316, and W-317 had higher sulfate concentrations (50 to 100 mg/L) than samples from other wells. Water from well W-333 had a sulfate concentration of 248 mg/L (appendix 7). Because only one sample was collected from this well, additional sampling would be required to determine if this sample is representative of ground water at this site. The origin of the sulfate in ground-water samples is difficult to determine, because atmospheric deposition of sulfate through precipitation can constitute a significant loading to ground water. (Atmospheric sulfate originates primarily from stack emissions of fossil-fueled industrial facilities.)

Examination of chloride data from wells at the two landfills, collected before and during this investigation, revealed some trends. Water from wells BR-2, W-2, and W-4 at the municipal landfill and M-507, W-508, and W-510 at the private landfill had concentrations of chloride that ranged from 4.3 to 43.4 mg/L, which were similar to concentrations found in previous investigations; however, chloride concentrations in water samples from wells MW-101, MW-102s, and MW-102d at the municipal landfill and well W-511 at the private landfill decreased. This trend is shown in figure 28 for wells MW-102s, MW-102d, and W-511. Concentrations of chloride measured in water samples from wells installed for this study, wells W-338 and W-325 at the private landfill and well W-335 at the municipal landfill, were

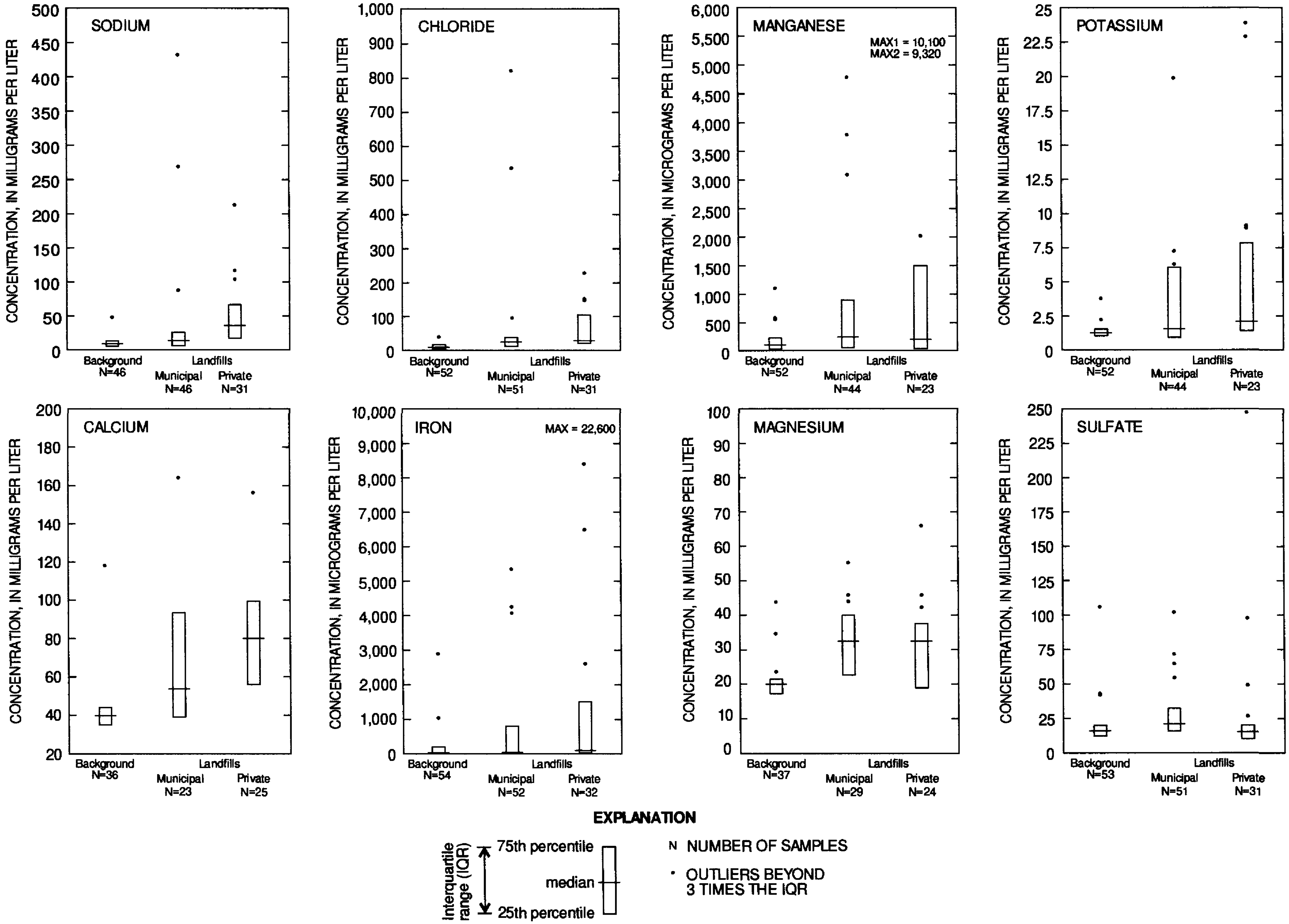

Figure 27. Distribution of concentrations of selected dissolved chemical constituents in native ground water and in samples from near landfills in Bristol, Vermont.

higher than concentrations of water samples from wells MW-101, MW-102s, MW-102d, and W-511. Given a trend of decreasing chloride concentration (fig. 28), concentrations in the area of wells W-338 and W-325 may have been higher before 1991 than those measured in this study.

Chloride is a common constituent in wells near roads subjected to deicing salt (Hall, 1975). Groundwater samples from wells screened in the upper aquifer and downgradient (west) from Burpee Road had sodium and chloride concentrations greater than those in native water; however, water samples from wells immediately upgradient from Burpee Road also had elevated sodium and chloride concentrations. Median concentrations of chloride and sodium in native water were 9 and 7 mg/L, respectively. Samples collected from well W-508 had chloride concentrations that ranged from 24 to 34 mg/L, and samples collected from well W-325 had chloride concentrations of 104 and 111 mg/L. The pond immediately west of Burpee Road, which receives runoff from Burpee Road, had chloride concentrations that ranged from 19 to 33 mg/L and sodium concentrations of about 17 mg/L. Water from wells screened about 17 ft below the water table west of Burpee Road (W-338, W-340, W-341, and W-342) had chloride and sodium concentrations from 25 mg/L to greater than 200 mg/L. Landfill leachate probably contributes most of the sodium and chloride in shallow ground water in the upper aquifer west of Burpee Road. Water from well W-312, adjacent to State Routes 17 and 116 (fig. 1), had concentrations of chloride and sodium of 26 and 41 mg/L (appendix 7); this road receives more deicing salt than Burpee Road.

Trace Elements

Trace elements commonly found in landfill leachate include arsenic, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, selenium, strontium, and zinc. With the exception of lead, all are either in insoluble, elemental form or are in complex form as specific industrial wastes.

Many samples collected during this investigation did not contain trace elements in detectable concentrations at the detection limits (tables 9 and 10). Iron is commonly present in higher concentrations than other metals. In the Bristol area, the median background iron concentration was 42 µg/L (table 8).

Concentrations of iron in ground water degraded by landfill leachate generally are higher than the background concentration. At the municipal landfill, the median concentration of iron in ground water was 97 µg/L (table 10) and at the private landfill was 114 µg/L (table 9). Samples collected near the two landfills had iron concentrations greater than 1,000 µg/L and three samples collected near the private landfill had iron concentrations of about 20,000 µg/L (appendix 7). Median concentrations of magnesium at the landfills were 1.5 to 2 times greater than the median background concentration of 21 mg/L (table 8). Median concentrations of manganese in ground water contaminated by leachate were about 2 times the median background concentration of 114 µg/L (table 8). Ground-water concentrations of manganese and iron vary considerably between locations and with time (appendix 7), as indicated by the high standard deviations of concentrations in native water samples. In general, samples with high manganese concentrations also had high iron concentrations. Concentrations in most samples, including native water, exceeded State and Federal drinking-water limits (Vermont Agency of Natural Resources, 1988; U.S. Environmental Protection Agency, 1988a).

Iron and manganese are common in water from sand and gravel aquifers and indicate geochemical reactions with aquifer materials. These metals have been found naturally in high concentrations in sand and gravel aquifers in New Hampshire (Mack and Lawlor, 1992). For example, two of three water samples from W-312 (appendix 7), which is far removed from the effects of either landfill, had elevated concentrations of iron and manganese. The origin of naturally occurring iron and manganese in ground water is difficult to determine without further water-quality information.

The metals most commonly detected in ground water near both landfills were copper, nickel, and zinc. Copper, a common metal, is readily dissolved by acidic water. Copper was detected in 22 of 35 water samples from wells at and near the municipal landfill and in 6 of 25 samples for the private landfill. Median concentrations of copper were less than 10 µg/L in ground water from both landfills. Nickel was detected in 10 of 34 samples from the municipal landfill (table 9), at a maximum concentration of 29 µg/L (wells W-317, and MW-102s) and in 10 of 25 samples at the private landfill (table 10) at a maximum concentration of 51 µg/L (well W-507). Nickel was detected in samples from two wells screened

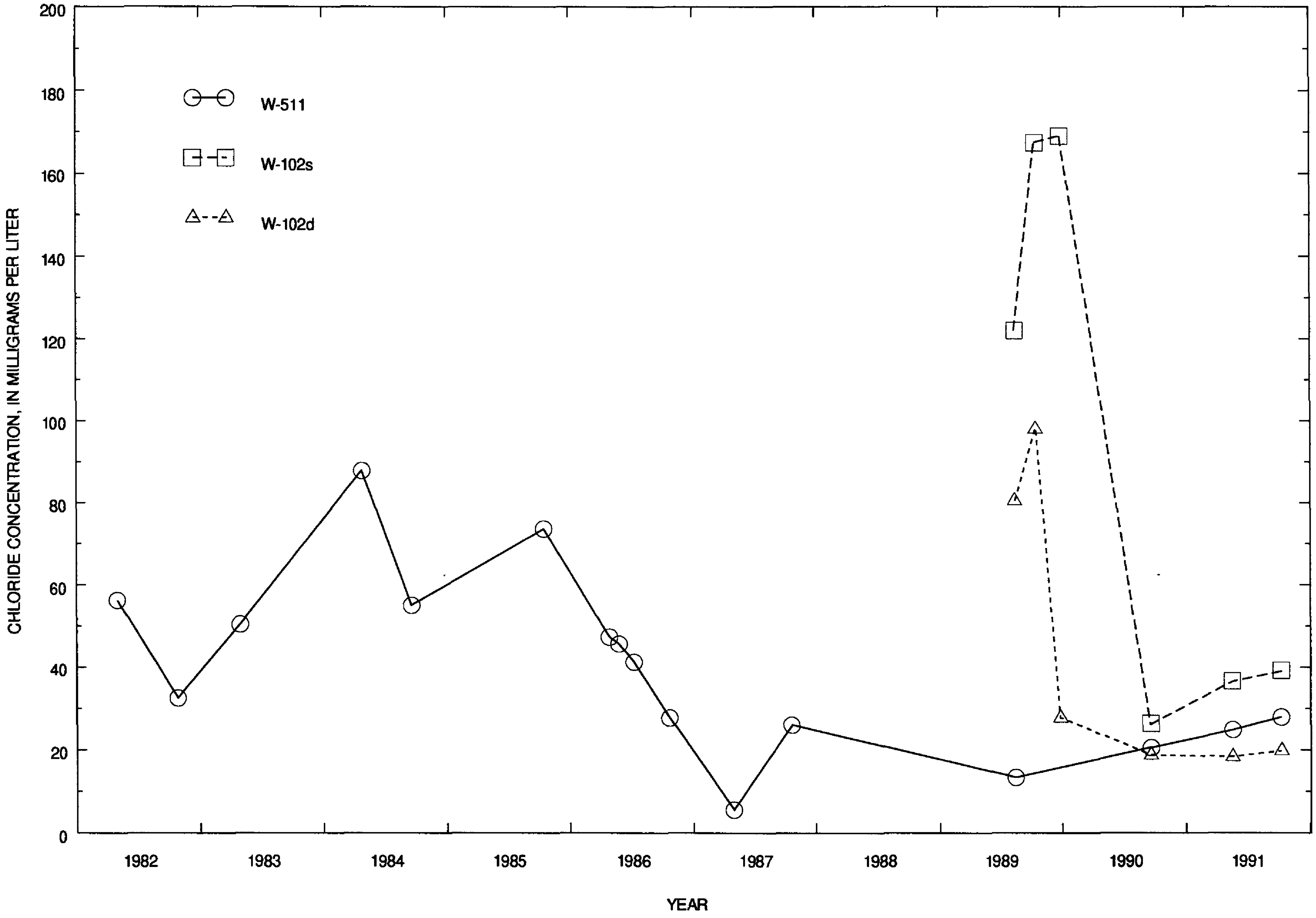

Figure 28. Chloride concentrations at wells W-511, MW-102s, and MW-102d. (Locations of wells shown in figure 5.)

Table 9. Summary statistics for physical properties and chemical constituents in water from wells and springs near the private landfill Bristol, Vermont, 1990-91

[Statistics for mean, standard deviation, 25th percentile, median, and 75th percentile are calculated by methods described by Helsel and Cohn (1988). µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligrams per liter; µg/L, micrograms per liter. <, actual value is less than value shown; --, not applicable or too few to calculate statistics]

Property or constituent	Number of samples	Number less than detection limit	Detection limit	Mean	Standard deviation	25th per-centile	Median	75th per-centile	Maximum detected
Physical properties									
Specific conductance (µS/cm at 25°C)	41	--	--	738	351	439	655	944.07	1,980
pH (units)	8	--	--	--	--	6.62	6.80	7.07	--
Major constituents (mg/L)									
Calcium, dissolved	25	0	--	79.3	37.1	57.4	80.2	99.2	156
Magnesium, dissolved	24	0	--	30.7	13.8	19.2	32.7	38.5	66.0
Sodium, dissolved	31	0	--	48.06	43.68	17.30	36.00	67.40	213
Potassium, dissolved	25	0	--	5.42	6.09	1.55	2.18	7.75	23.9
Sulfate, dissolved	31	1	0.20	24.29	45.16	9.50	14.90	20.20	248
Chloride, dissolved	31	0	--	60.4	54.8	20.3	27.9	104.0	228
Trace constituents (µg/L)									
Arsenic, dissolved	24	24	5	--	--	--	--	--	<5
Barium, dissolved	25	0	--	88	84	34	58	120	310
Cadmium, dissolved	30	30	2	--	--	--	--	--	<2
Chromium, dissolved	38	37	10	--	--	--	--	--	103
Cobalt, dissolved	19	18	10	--	--	--	--	--	24
Copper, dissolved	25	19	10	8	9	2	4	10	38
Iron, dissolved	32	1	10	2,758	6,223	33	114	1,520	22,600
Lead, dissolved	30	30	10	--	--	--	--	--	<10
Manganese, dissolved	23	3	10	1,041	1,955	40	201	1,710	10,100
Nickel, dissolved	26	16	10	10	11	3	6	12	51
Selenium, dissolved	24	24	5	--	--	--	--	--	<5
Silver, dissolved	24	24	1	--	--	--	--	--	<1
Zinc, dissolved	32	27	40	26	11	19	33	81	87

in the lower aquifer (table 10), in one sample downgradient from the private landfill (W-321), and in two samples downgradient from the municipal landfill (W-334). The presence of nickel in these samples is probably related to landfill leachate. Nickel is used extensively in stainless steel and other alloys and can be found in many waste materials. Concentrations of nickel in native water is generally less than a few micrograms per liter (Hem, 1989). Zinc was detected in three ground-water samples from wells near the municipal landfill and in five ground-water samples from wells near the private landfill. Zinc is used in brass and bronze, in galvanized metals, and as a white pigment in paint (Hem, 1989). Concentrations of copper, nickel, and zinc were detected in native water samples from wells considered as background near both landfills (appendix 7). Some of these wells (W-310 and W-311 near the municipal landfill and W-307, W-308, W-320, and W-322 near the private landfill) possibly are affected by landfill leachate (table 7).

Table 10. Summary statistics for physical properties and chemical constituents in water from wells and springs near the municipal landfill, Bristol, Vermont, 1990-91

[Statistics for mean, standard deviation, 25th percentile, median, and 75th percentile are calculated by methods described by Helsel and Cohn (1988). µS/cm, microsiemen per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter. <, actual value is less than value shown; --, not applicable or too few to calculate statistics]

Property or constituent	Number of samples	Number less than detection limit	Detection limit	Mean	Standard deviation	25th percentile	Median	75th percentile	Maximum detected
Physical properties									
Specific conductance (µS/cm)	50	--	--	648	455	367	491	814	2,920
pH (units)	16	--	--	--	--	6.72	7.12	7.40	--
Major constituents (mg/L)									
Calcium, dissolved	23	0	--	67.8	34.4	38.6	54.6	95.5	107
Magnesium, dissolved	29	0	--	32.1	12	22.8	32.8	41.3	59
Sodium, dissolved	46	0	--	32.1	72.5	6.17	13.9	26.3	432
Potassium, dissolved	23	0	--	4.64	5.77	.97	1.62	6.31	19.9
Sulfate, dissolved	51	0	0.20	32.11	31.25	15.90	22.20	34.10	71.8
Chloride, dissolved	51	0	--	53.5	132	13.3	27.2	41.5	820
Trace constituents (µg/L)									
Arsenic, dissolved	34	33	5	--	--	--	--	--	5
Barium, dissolved	37	0	--	85	50	40	89	119	190
Cadmium, dissolved	47	47	2	--	--	--	--	--	<2
Chromium, dissolved	47	47	10	--	--	--	--	--	<10
Cobalt, dissolved	24	23	10	--	--	--	--	--	15
Copper, dissolved	35	13	10	11	10	5	9	15	32
Iron, dissolved	52	5	10	616	1,163	24	97	838	5,350
Lead, dissolved	44	42	10	--	--	--	--	--	16
Manganese, dissolved	44	5	10	824	1,173	82	251	908	4,790
Nickel, dissolved	34	24	10	--	--	--	--	--	29
Selenium, dissolved	34	34	5	--	--	--	--	--	<5
Silver, dissolved	34	33	1	--	--	--	--	--	6.0
Zinc, dissolved	46	43	40	--	--	--	--	--	198

Concentrations of cadmium ranging from 5 to 7 µg/L exceeded the U.S. Environmental Protection Agency (1988a) and Vermont Agency of Natural Resources (1988) enforcement standard in water samples from wells MW-101, MW-102s, W-4, and W-2 at the municipal landfill (Johnson Company, 1989). Cadmium is used in electroplating and in manufacturing batteries, pigments, ink, plastics, and fluorescent and video tubes. Cadmium was not detected in water samples from wells where it had been detected during previous studies. Cadmium was detected at low concentrations in only three water samples (appendix 7) from wells that may be affected by landfill leachate. Water from wells W-307 and W-322, near the private landfill, had cadmium concentrations at the detection limit (2 µg/L). Water from one well (W-310) near the municipal landfill had a cadmium concentration of 4 µg/L. When concentrations are near the detection limit, it is difficult to determine whether the concentration is the result of contamination or the result of sampling or analytical error. Cadmium is probably not a dominant constituent in the leachate; it has perhaps been leached out of landfilled wastes and is presently in ground water in low concentrations.

Arsenic, used in some pesticides (Hem, 1989), was found in one water sample from well BR-2 at the detection limit of 5 µg/L. Cobalt was found at a concentration of 19 µg/L in water from well BR-2 near the municipal landfill and 24 µg/L in water from well W-325 near the private landfill. Silver, commonly used in photographic materials and present in some industrial wastes (Hem, 1989), was detected at a concentration of 6.0 µg/L in water from well W-2. Selenium is present in leachate from disposal of waste ink and rubber. Selenium was not detected above the detection limit in any water samples.

Organic Compounds

Water samples were analyzed for the 36 volatile organic compounds (VOC's) listed in table 11. During the study, a few VOC's were detected at observation wells associated with the two landfills, but no VOC's were detected in native water, surface water, or springs. VOC's detected in water samples collected during this study, sample concentrations, and sampling dates are grouped by wells associated with the two landfills and are listed in table 12. VOC's were detected 10 times in water samples from four observation wells near the municipal landfill and 3 times at two observation wells near the private landfill. The VOC's detected were at concentrations at or near detection limits. As with metals, concentrations of VOC's detected at or near the detection limit could be a result of sampling or analytical error.

Water from observation well W-324, at the private landfill, had the highest VOC concentrations: 53 µg/L of acetone and 56 µg/L of 2-butanone. These concentrations, however, are just above the detection limit for these constituents of 50 µg/L. Only one sample was collected from this well, which was installed late in the investigation, and it is not known if the sample is representative of water in the aquifer. VOC's were not detected in other nearby observation wells (W-320 and W-321) in the lower aquifer at the private landfill. Toluene was detected at a concentration of 2 µg/L (detection limit of 2 µg/L) in water from well W-507 screened in the base of the upper aquifer at the private landfill. Previous detections of VOC's at the private landfill were for water samples from wells W-508, W-510, and W-511 (all screened in the upper aquifer) and for surface-water samples (in 1984) from the drainage culvert at the landfill and at the pond west of the landfill across Burpee Road (James Surwilo, Vermont Agency of Natural Resources, written commun., 1992). The drainage culvert at the private landfill originally directed runoff from the landfill to the pond west of the landfill; however, runoff has been diverted for some time to an area north of the landfill near observation well W-510 (fig. 5).

More VOC's were found in water samples from observation wells at the municipal landfill than in samples from wells at the private landfill (table 12). The greatest number of VOC detections in ground water (five during this study) was from well MW-102d; the most prevalent compound found was 1,2-dichloroethene (two detections). Well MW-102d is screened near the base of the aquifer at the west boundary of the municipal landfill (fig. 5). Three VOC's were detected at W-317 (table 12), which is at the same location as MW-102d but screened at the water table. Water from well MW-102s, at the same location but screened at an intermediate depth in the aquifer, did not contain detectable concentrations of VOC's during this study. No patterns of the compounds detected were indicated at the well clusters (table 12).

Relation of Water Quality to Geohydrologic Characteristics

Leachate contamination has been detected in ground water at or near both landfills. Borehole geophysical logs and ground-water samples indicate that the leachate contamination is concentrated near the water table within the upper aquifer downgradient from both landfills. Concentrations of common inorganic constituents in water samples degraded by landfill leachate are higher than those in native water in the upper aquifer underlying both landfills. Common inorganic constituent concentrations are generally at or near background concentrations in the lower aquifer near the two landfills. Background concentrations of common constituents and trace metals are generally less than detection limits; this finding indicates that lower detection limits may be needed for accurate quantification of constituent concentrations. Concentrations of some trace metals in the upper aquifer near the landfills are greater than their detection limits; however, concentrations in the lower aquifer are generally lower than their detection limits. The most common trace metals in water associated with landfill leachate were copper, nickel, and zinc. VOC's were detected in the upper and lower aquifers at the western end of the municipal landfill. The VOC detections in water from well MW-102d indicate that some contaminants migrate to the lower aquifer despite a large upward vertical gradient; however, the confining unit is

Table 11. Volatile organic compounds for which water samples were analyzed, Bristol, Vermont

[μg/L, microgram per liter]

Compound	Detection limit (μg/L)
Vinyl chloride	10
Chloromethane	10
Bromomethane	10
Chloroethane	10
Trichloroflouromethane	10
Acetone	50
1,1-Dichloroethene	2
Carbon disulfide	2
Methylene chloride	2
Methyl-t-butylether (MTBE)	10
1,2-Dichloroethane	2
1,1-Dichloroethane	2
Vinyl acetate	50
2-Butanone	50
Chloroform	2
1,1,1-Trichloroethane	2
Carbon tetrachloride	2
Benzene	2
Trichloroethene	2
1,1,2,2,-Tetrachloroethane	2
1,2-Dichloropropane	2
Bromodichloromethane	2
4-Methyl-2-pentanone	20
cis-1,3-Dichloropropane	2
Toluene	2
trans-1,3-Trichloropropene	2
1,1,2-Trichloroethane	2
2-Hexanone	20
Tetrachloroethane	2
Dibromochloromethane	2
Chlorobenzene	2
Ethylbenzene	2
Xylenes	2
Styrene	2
Bromoform	2

not continuous at the municipal landfill. At the private landfill, VOC's may be limited to the upper aquifer as a result of the upward head gradient and the confining unit. Although VOC's were detected in the lower aquifer at the private landfill, the reliability of the data was questionable because the VOC's were near the detection limits and no other constituents were elevated.

Some constituents were found immediately upgradient from the landfills in previous investigations and during this investigation. Analytical results for water samples from observation wells W-332 and W-333 (less than 150 ft from the private landfill) were of questionable quality but could be affected by lateral contaminant migration through unsaturated sediments as much as 80 ft thick. Contaminants also were detected in water samples from observation wells MW-2 and BR-2, which are less than 25 ft upgradient from the municipal landfill. For example, previous investigators (Johnson Company, 1989) found VOC's and trace metals at well BR-2. During this study, however, trace metals were detected, but VOC's were not (appendix 7). Detections of contamination at well BR-2 may have been caused by drainage from the landfill surface or lateral contaminant migration through unsaturated sediments towards this well, particularly during wet seasons. Previous contaminant detections at wells BR-2 and MW-2 could also be the result of a more highly concentrated leachate in the landfill than was present during this study. A stream less than 150 ft to the north drains the area during wet seasons. Well W-319 was installed upgradient from well BR-2, midway to the stream, to provide additional data on the hydrogeology and water quality in this area; however, well W-319 did not yield enough water for adequate samples to be taken.

Table 12. Volatile organic compounds detected in observation wells, grouped by landfill, Bristol, Vermont, September 1990-October 1991

[Maximum contaminant level (MCL) set by the U.S. Environmental Protection Agency (1988a). Vermont Agency of Natural Resources (1988) and written communication (1994). µg/L, microgram per liter. none, a standard has not been established]

Well	Compound	Sample date	Concentration (µg/L)	Maximum contaminant level (µg/L)	Enforcement standard (µg/L)
		MUNICIPAL LANDFILL			
MW-102d	1,2-Dichloroethene	Sept. 1990	3	none	70
MW-102d	1,2-Dichloropropane	Sept. 1990	2	5.0	0.50
MW-102d	Methylene chloride	May 1991	2	none	5.0
MW-102d	1,2-Dichloroethene	May 1991	4	none	70
MW-102d	1,2-Dichloroethene	Oct. 1991	3	none	70
MW-101	Benzene	Oct. 1991	2	5.0	1.0
MW-101	Toluene	Oct. 1991	5	1,000	2,420
W-317	1,1-Dichloroethane	Feb. 1991	2	5.0	5.0
W-317	1,1,1-Trichloroethane	Oct. 1991	3	200	200.0
W-337	Benzene	Aug. 1991	3	5.0	1.0
		PRIVATE LANDFILL			
MW-507	Toluene	Oct. 1991	2	1,000	2,420
W-324	Acetone	Oct. 1991	53	none	700
W-324	2-Butanone	Oct. 1991	56	none	none

The concentrations of some water-quality constituents in ground water at both landfills appear to have decreased since earlier investigations. Some chemical constituents may have been leached from the landfilled waste, thereby resulting in a less concentrated leachate. For example, concentrations of cadmium and sodium at the municipal landfill and sodium at the private landfill are generally less than concentrations previously detected. Although trends indicate decreases in chemical constituents in leachate from both landfills, solid-waste disposal was resumed in 1991 at the private landfill after a period of inactivity. Waste disposal is still underway at the municipal landfill. These water-quality trends could possibly represent the effects of recycling and waste screening to reduce the influx and types of solid waste to municipal landfill. These trends could represent short-term processes that could change if the characteristics of the emplaced waste change with time.

SUMMARY AND CONCLUSIONS

The Bristol, Vermont area is characterized by a glacial delta more than 200 ft thick, having an exposed face about 150 ft high, which was deposited on the west flank of the Green Mountains. A municipal landfill, more than 20 years old, overlies this coarse-grained deltaic deposit; depths to water range from 30 to 130 ft below land surface. A private landfill, also more than 20 years old, is at a lower altitude adjacent to the delta in medium- to fine-grained glaciolacustrine sediments up to 100 ft thick; depths to water range from 10 to 50 ft. A sand, silt, and clay unit, found throughout most of the study area adjacent to the delta and possibly extending into the delta in places, functions as a confining unit separating the glacial aquifer into an upper unconfined and a lower confined aquifer. The estimated horizontal hydraulic conductivity of the coarse-grained deltaic sands and gravels, determined by slug and specific-capacity response tests, ranged from 60 to greater than 250 ft/d. The horizontal hydraulic conductivity was estimated to be 1 to 60 ft/d for fine- to medium-grained sands and about 1 to 4 ft/d for till.

Recharge to the glacial aquifers is predominantly from upward ground-water leakage from the underlying bedrock aquifer and accounts for about 80 percent of the total recharge. Infiltration of precipitation (about 14 in/yr) accounts for about 10 percent of total recharge, and inflow from adjacent glacial aquifers accounts for the remaining 10 percent. Large upward hydraulic gradients, ranging from 0.03 to 0.3, between the upper and lower glacial aquifers, and within aquifers, are evidence that most recharge is as upward ground-water leakage from bedrock.

Surface electromagnetic geophysical terrain surveys were used to identify areas underlain by electrically conductive ground water and (or) earth materials and to place observation wells. Natural-gamma radiation and electromagnetic borehole geophysical logs were used to identify and vertically delineate landfill-leachate plumes within aquifer materials and to place well-screen settings. Leachate plumes were found near the water table and in thickness that range from less than 5 to nearly 20 ft.

A three-dimensional numerical ground-water-flow model of the glacial aquifer was developed by use of a fine grid at or near the landfills and a coarse grid to extend to significant hydrogeologic boundaries. The model was constructed as two layers to simulate the upper and lower glacial aquifers. To simulate the confining unit, its thickness was incorporated into the thickness of the lower layer and an appropriate vertical hydraulic conductivity was assigned between the two layers.

The model was calibrated under steady-state conditions, approximated by the conditions observed during April and May of 1991. Measured and estimated hydraulic-head data from 58 observation wells were compared with simulated heads. Simulated heads matched measured and estimated heads with a standard mean difference of 0.6 ft and an absolute mean difference of 2.5 ft. Large head-difference errors (greater than 5 ft) were noted at one location near the model boundary, where a coarse grid spacing was used, and at some locations at the delta face, where high hydraulic gradients were observed. The total ground-water budget simulated by the model (17 ft^3/s) reproduced the total estimated ground-water budget (18 ft^3/s) and specific streamflows and ground-water outflows reasonably closely.

Advective ground-water-flow paths from beneath the landfills to discharge locations at the model boundaries were simulated by use of a particle-tracking analysis. Ground water from beneath the private landfill leaves the study area as ground-water outflow through a gap in the bedrock ridge west of the landfill. Some ground water beneath the private landfill discharges to a brook west of the private landfill. Most ground water beneath the municipal landfill also discharges as ground-water outflow to the southwest at the gap in the bedrock ridge. Ground water follows a much longer path from beneath the municipal landfill than the private landfill and ultimately discharges at the gap in the ridge. Ground water beneath the east end of the municipal landfill possibly flows southerly to eventually discharge at the south model boundary in the Bristol Flats area. Upward vertical gradients were simulated throughout the landfill areas. Simulated flow paths in the upper aquifer beneath both landfills remained in the upper aquifer throughout the area. Simulated flow paths in the lower aquifer beneath both landfills generally moved to the upper aquifer.

Water samples from 54 wells and 7 surface-water and spring sites were collected and analyzed by the Vermont Agency of Natural Resources. Results were grouped into native and landfill-degraded water. Native water had a median specific conductance of 354 µS/cm, and ranged from approximately 200 to 465 µS/cm. The primary cations in native water are calcium, magnesium, sodium, and sulfate. Other cations commonly found at low concentrations include iron, manganese, and barium. Concentrations of most trace elements, such as chromium, cobalt, selenium, and silver, were below detection limits in native water samples. Copper, lead, nickel, and zinc were detected near detection limits at some observation wells.

Water samples degraded by landfill leachate generally had a specific conductance greater than 400 µS/cm; the median was about 700 µS/cm. The maximum specific conductances were 2,920 µS/cm at the municipal landfill and 1,980 µS/cm at the private landfill.

Leachate from the landfills contained mean concentrations of many common constituents and metals that were 1.5 to 10 times the mean background concentrations. The elevated constituents included calcium, potassium, sodium, chloride, iron, magnesium, and manganese. Trace metals detected in leachate from the landfills included copper, nickel, zinc, cobalt, and lead. Nickel was the most commonly detected trace element (20 detections) followed by zinc (8 detections).

Water samples were analyzed for 36 volatile organic compounds (VOC's) during the study. Seven VOC's were detected at four observation wells associated with the municipal landfill and three VOC's were detected at two observation wells associated with the private landfill. No single VOC was consistently detected, and concentrations were generally at or near detection limits. VOC's were not detected in background samples. One sample, from a well adjacent to the private landfill, had concentrations of acetone and 2-butanone near the detection limits. It is uncertain whether the water was contaminated during sampling (no other chemical constituent in this sample indicate the presence of leachate) or if the sample was degraded by landfill leachate.

Some chemical constituents in water samples seem to be present in lower concentrations than those observed in previous investigations. For example, concentrations of chloride appeared to be decreasing at both landfills, and cadmium (which was detected at the municipal landfill in a previous investigation) was not detected during this investigation. VOC's were detected fewer times and at lower concentrations in this investigation than in previous investigations. Operations are changing at both landfills. Without additional sampling it is not possible to determine if the water-quality trends observed in this investigation are short term or if the concentration of leachate constituents will continue to diminish. What effect changes in the operation of both landfills will have on the concentration of leachate constituents is unknown.

Ground water contaminated by landfill leachate was concentrated in the upper glacial aquifer. Hydraulic gradients between and within the glacial aquifers are upward throughout most of the study area and were duplicated in the numerical flow simulations. Measured heads in the bedrock aquifer at domestic wells along Burpee Road also indicate an upward vertical gradient to the glacial aquifers. Examination of specific ground-water fluxes between the bedrock and glacial aquifers at domestic well sites was beyond the scope of this investigation.

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APPENDIX 1

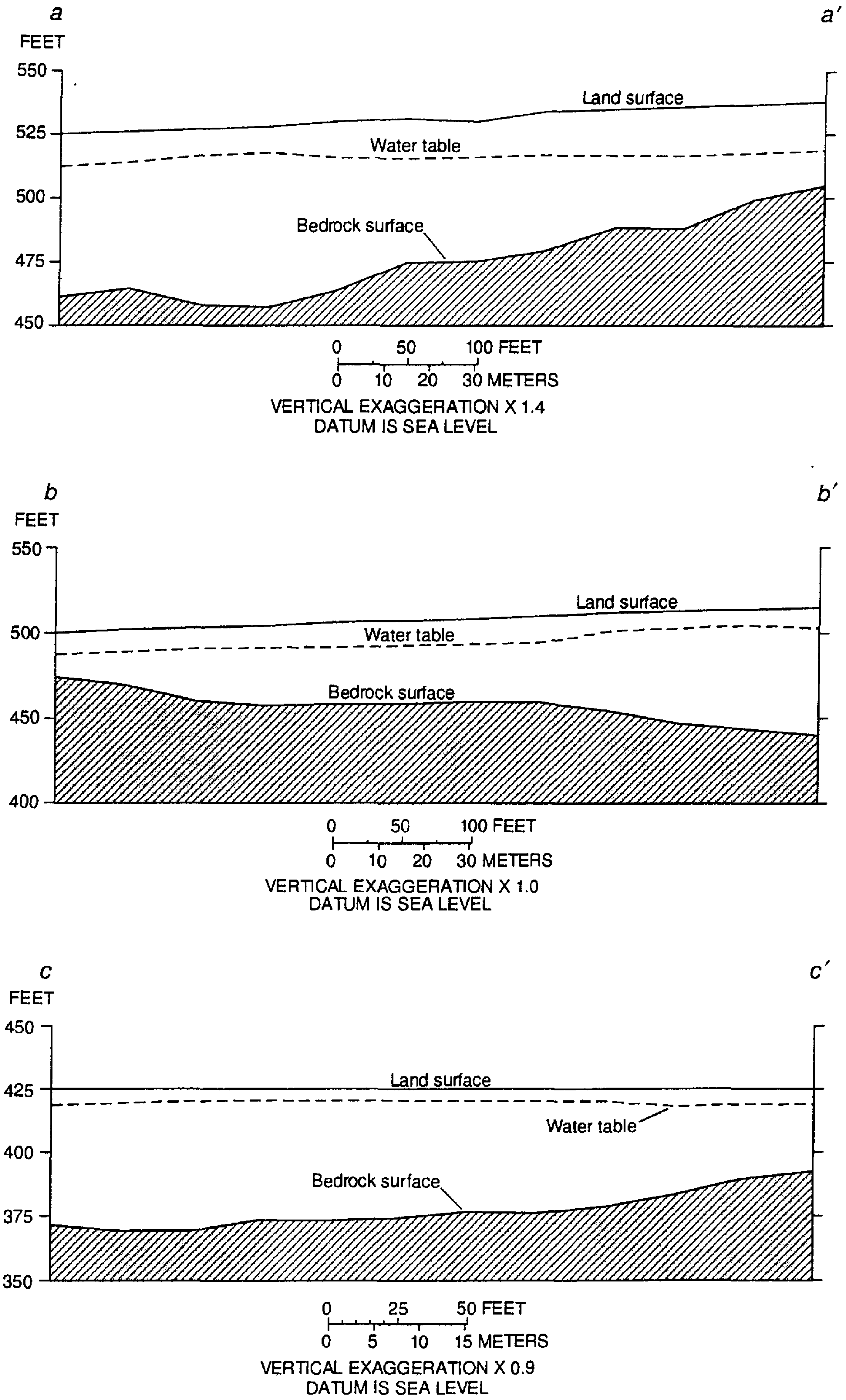

Figure 1-1. Seismic-refraction profiles for a-a′, b-b′, and c-c′ (locations shown in fig. 2).

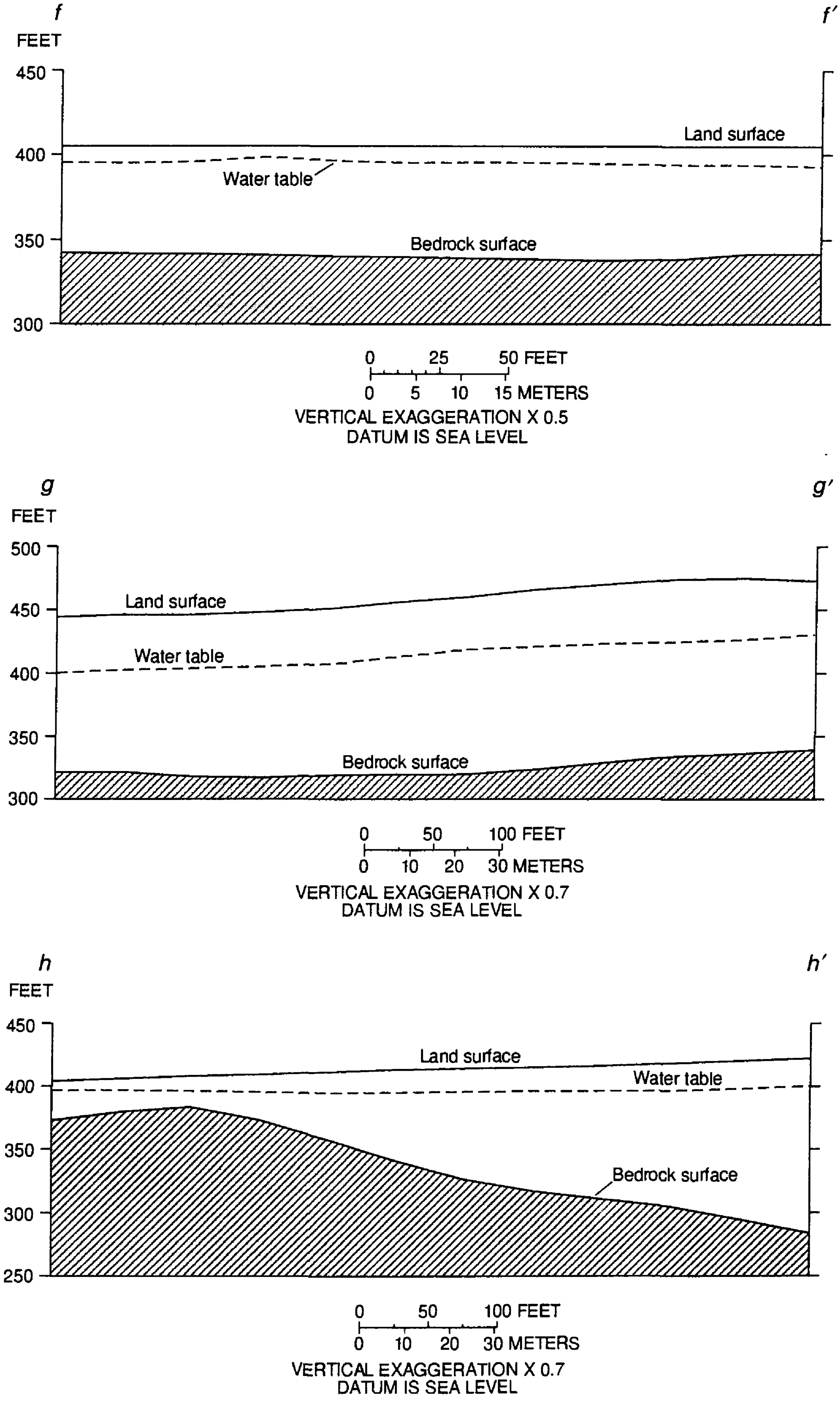

Figure 1-2. Seismic-refraction profiles for f-f′, g-g′, and h-h′ (locations shown in fig. 2).

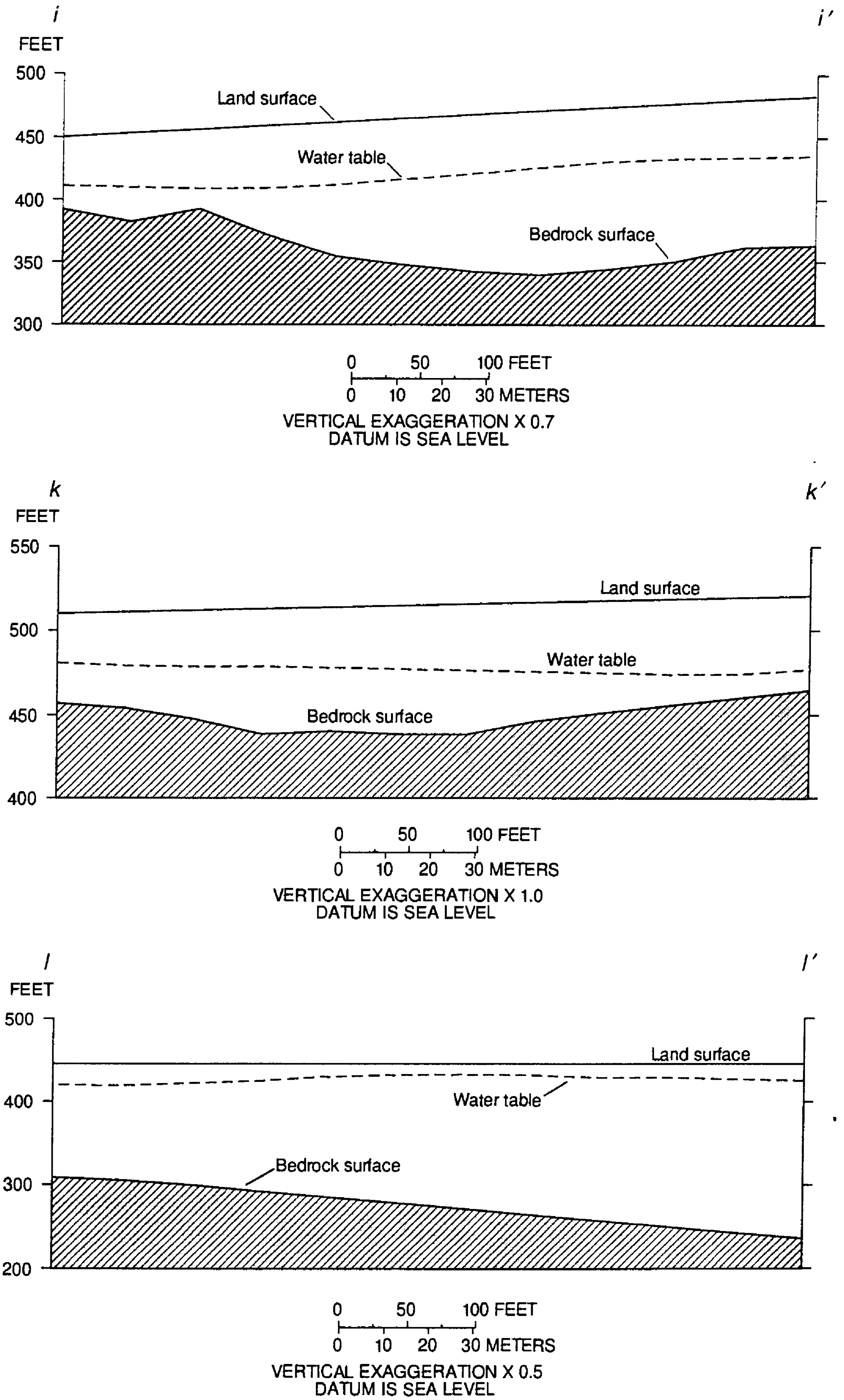

Figure 1-3. Seismic-refraction profiles for i-i′, k-k′, and l-l′ (locations shown in fig. 2).

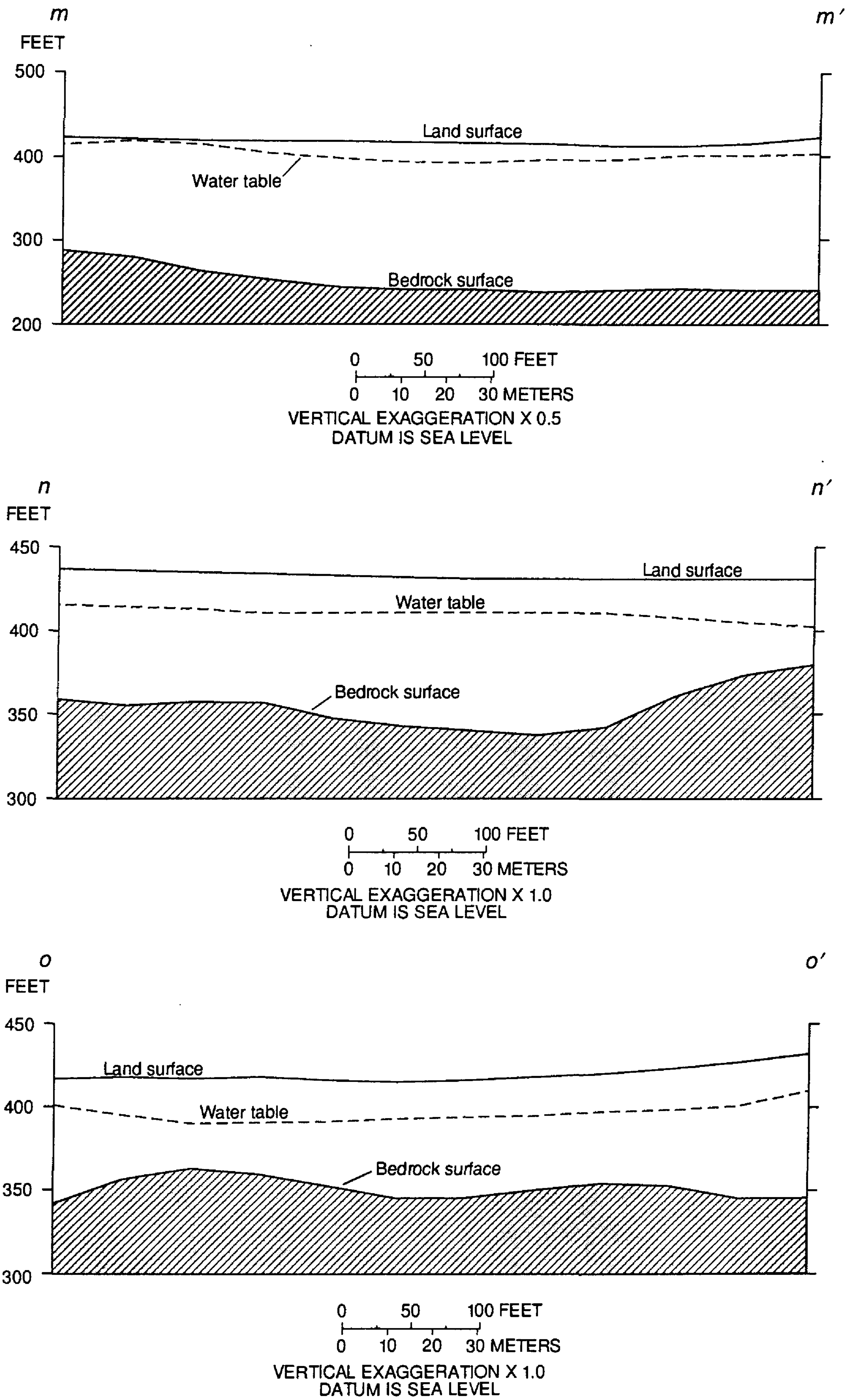

Figure 1-4. Seismic-refraction profiles for m-m′, n-n′, and o-o′ (locations shown in fig. 2).

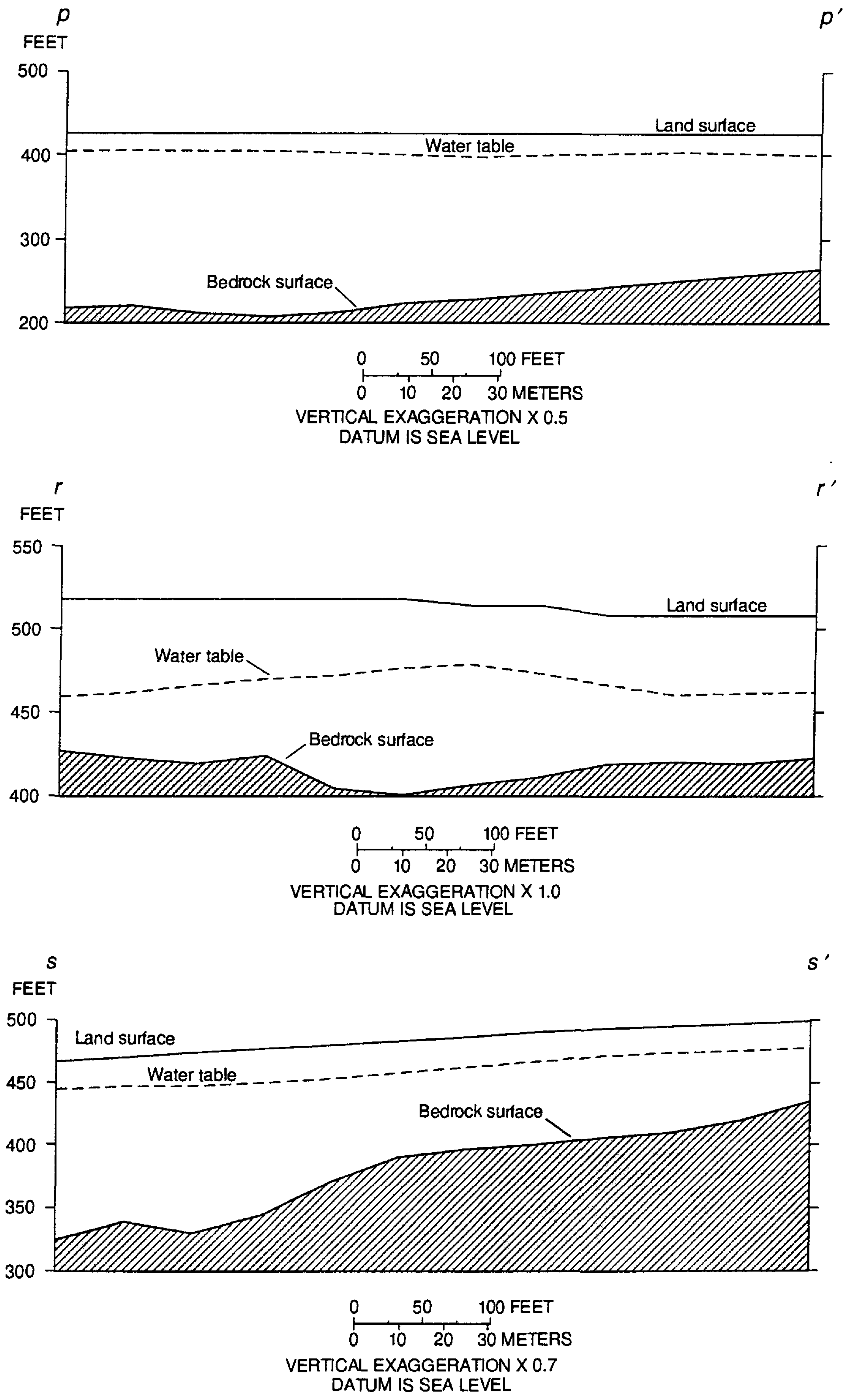

Figure 1-5. Seismic-refraction profiles for p-p′, r-r′, and s-s′ (locations shown in fig. 2).

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APPENDIX 2

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Appendix 2. Well-depth and screened-interval data for the study area in Bristol, Vermont

[All depths in feet from measurement point. Altitudes in feet above sea level or land surface. ft, foot; --, not applicable]

Well No.	Other identifier	Latitude ° ′ ″	Longitude ° ′ ″	Water-level altitude, April 18, 1991 (ft)	Depth to screened interval (ft)	Depth to end of well or casing (ft)	Height of measurement point above land surface (ft)	Altitude of measurement point (ft)
B2W-301	--	44 08 15	073 05 46	[1]435.0	35-37	37	2	430.84
B2W-302	--	44 08 15	073 05 46	428.9	8-9	9	2	430.88
B2W-303	--	44 08 17	073 05 46	444.4	28-30	31	3	448.79
B2W-304	--	44 08 06	073 05 48	406.5	35-37	37	3	410.27
B2W-305	--	44 08 00	073 05 58	401.0	65-69	69	2	404.47
B2W-306	--	44 08 00	073 05 58	399.3	31-33	33	2	405.00
B2W-306b	--	44 08 00	073 05 58	399.3	9-11	11	2	405.06
B2W-307	--	44 07 53	073 05 52	405.1	78-80	100	0	438.40
B2W-308	--	44 07 55	073 05 38	407.5	66-68	84	2	460.83
B2W-309	--	44 08 24	073 05 20	488.4	93-99	99	2.5	525.48
B2W-309b	--	44 08 24	073 05 20	484.1	69-70	70	2.5	525.34
B2W-310	--	44 08 20	073 05 43	458.0	66-68	68	2.5	487.65
B2W-311	--	44 08 20	073 05 43	455.8	37-39	39	2.5	487.65
B2W-312	--	44 07 49	073 05 29	408.0	42-44	44	2	444.41
B2W-315	--	44 08 05	073 06 00	[1]398.4	70-72	100	2	442.30
B2W-316	--	44 08 21	073 05 27	[1]470.9	40-42	42	2.5	506.54
B2W-317	--	44 08 18	073 05 40	453.0	77-82	82	3	527.57
B2W-318	--	44 08 18	073 05 29	453.1	115-117	117	--	552.83
B2W-319	--	44 08 24	073 05 39	484.1	10-16	16	2	497.56
B2W-320	--	44 07 56	073 05 44	406.1	88-92	92	2	450.50
B2W-321	--	44 08 00	073 05 56	404.8	77-79	80	2	411.31
B2W-321b	--	44 08 00	073 05 56	402.3	40-42	42	2	411.28
B2W-322	--	44 07 56	073 05 52	404.1	78-80	100	2	431.21
B2W-322b	--	44 07 56	073 05 52	404.1	40-42	42	2	431.28
B2W-323	--	44 08 00	073 05 59	[1]399.8	35-37	37	2	404.97
B2W-324	--	44 08 00	073 05 53	[1]404.8	89-91	91	2	424.50
B2W-325	--	44 08 00	073 05 53	[1]403.2	21-25	25	2	423.55
B2W-326	--	44 08 01	073 05 54	[1]404.9	17-19	19	2	420.21
B2W-327	--	44 08 09	073 05 43	[1]411.9	53-55	55	2	442.05
B2W-328	--	44 08 09	073 05 43	[1]408.6	33-35	35	2	442.50
B2W-329	--	44 08 02	073 05 54	[1]402.9	68-72	72	2	413.53
B2W-330	--	44 08 11	073 05 47	[1]415.3	30-32	32	2	424.20
B2W-331	--	44 08 11	073 05 47	[1]413.7	10-12	12	2	424.27
B2W-332	--	44 07 59	073 05 39	[1]414.1	126-130	135	2	487.36
B2W-333	--	44 07 59	073 05 39	[1]416.	76-80	80	1.5	487.10
B2W-334	--	44 08 15	073 05 36	[1]455.	178-182	187	0	574.
B2W-335	--	44 08 15	073 05 36	[1]447.	141-145	145	0	574.
B2W-336	--	44 08 16	073 05 40	[1]446	121-125	135	0	574.09
B2W-337	--	44 08 14	073 05 42	[1]446.9	137-143	143	0	574.38
B2W-338	--	44 08 00	073 05 56	[1]399.	18-20	20	1	409.8

Appendix 2. Well-depth and screened-interval data for the study area in Bristol, Vermont--*Continued*

Well No.	Other identifier	Latitude ° ′ ″	Longitude ° ′ ″	Water-level altitude, April 18, 1991 (ft)	Depth to screened interval (ft)	Depth to end of well or casing (ft)	Height of measurement point above land surface (ft)	Altitude of measurement point (ft)
B2W-339	--	44 08 00	073 05 59	[1]398.	18-20	20	1	403.0
B2W-340	--	44 08 01	073 05 56	[1]399.	18-20	20	1	405.1
B2W-341	--	44 07 59	073 05 56	[1]396.	19-21	21	1	410.0
B2W-343	--	44 08 03	073 05 54	[1]402.	13-14	14	0	415.0
B2W-351	[2]W-1	44 07 51	073 04 44	[1]483.9	14-24	24	2	500.82
B2W-352	[2]W-2	44 07 52	073 04 39	[1]485.3	15-25	25	2	503.07
B2W-353	[2]W-3	44 07 54	073 04 40	[1]470.2	31-41	41	2	505.96
B2W-354	[2]W-4	44 07 55	073 04 40	[1]465.5	31-41	41	2	506.69
B2W-355	[2]W-5	44 07 55	073 04 39	[1]472.5	31-41	41	2	510.60
B2W-356	[2]W-6	44 07 53	073 04 50	[1]463.9	40-50	50	2	498.04
B2W-507	[3]MW-7	44 08 01	073 05 48	404.8	--	53	1	435.96
B2W-508	[3]MW-8	44 08 09	073 05 53	402.7	--	30	0	416.37
B2W-509	[3]MW-9	44 08 18	073 05 53	403.1	--	25	2	412.15
B2W-510	[3]MW-10	44 08 13	073 05 53	1.8	--	27	2	404.13
B2W-511	[3]MW-11	44 08 02	073 05 58	401.7	--	16	1	411.81
BR-2	[4]BR-2	44 08 22	073 05 39	480.8	--	38	2	509.00
MW-2	[4]MW-2	44 08 22	073 05 27	486.4	--	51	2	522.66
MW-4	[4]MW-4	44 08 20	073 05 38	461.5	--	67	3	514.95
BR-6	[4]BR-6	44 08 21	073 05 27	482.1	68-76	76	3	505.15
MW-102s	[5]MW-102s	44 08 18	073 05 40	452.9	81-96	96	2	527.99
MW-102d	[5]MW-102d	44 08 18	073 05 40	455.4	119-124	124	2	528.13
MW-101	[5]MW-101	44 08 14	073 05 42	445.4	167-177	177	0	574.19

[1] Estimated water-level altitude.

[2] Bristol leachfield well (Thermo Engineering, written commun., 1991).

[3] Bristol Waste Management Landfill (Marshfield Engineering Services, 1979).

[4] Bristol Municipal landfill (Hackbarth, 1980).

[5] Bristol Municipal landfill (Johnson Company, 1989).

(page 80 follows)

APPENDIX 3

Appendix 3. Lithologic logs of wells in Bristol, Vermont

[Altitudes are given in feet above sea level. Well number shown in bold. ft, foot; --, no data]

Description of material	Top of unit or sample (ft)	Bottom of unit or sample (ft)
W-1, Altitude 370 ft		
Boulders and hardpan	0	100
Silt	100	261
Bedrock	261	--
W-4, Altitude 420 ft		
Boulders and gravel	0	26
Sand, coarse; cobbles	26	50
Sand, fine; silt	50	69
Clay; end of hole 70 ft	69	--
W-5, Altitude 420 ft		
Gravel; end of hole 68	0	--
W-12, Altitude 590 ft		
Gravel; end of hole 150 ft	0	--
W-28, Altitude 535 ft		
Sand	0	6
Bedrock	6	--
W-37, Altitude 400 ft		
Gravel; end of hole 40 ft	0	--
W-38, Altitude 436 ft		
Clay; sand	0	92
Bedrock	92	--
W-42, Altitude 435 ft		
Sand and boulders	0	12
Boulders	12	--
W-43, Altitude 440 ft		
Sand and boulders	0	12
Sand and gravel	12	--
W-45, Altitude 420 ft		
Clay	0	50
Sand and gravel	50	100
Bedrock; end of hole 200 ft	100	--
W-47, Altitude 370 ft		
Gravel	0	7
Hardpan (till)	7	22
Bedrock, end of hole 298 ft	22	--
W-54, Altitude 435 ft		
Boulders and gravel	0	23
Sand and gravel; end of hole 37 ft	23	--
W-60, Altitude 400 ft		
Clay, hardpan (till?)	0	78
Bedrock (limestone); end of hole 249 ft	78	--
W-62, Altitude 590 ft		
Clay and sand	0	20
Limestone; end of hole 225 ft	20	--
W-76, Altitude 420 ft		
Boulders	0	12
Sand, silt	12	30
Boulders	30	50
Gravel; end of hole 60 ft	50	--
W-77, Altitude 445 ft		
Clay	0	15
Sand and gravel	15	87
Bedrock; end of hole 173 ft	87	--
W-81, Altitude 430 ft		
Gravel; end of hole 41 ft	0	--
W-82, Altitude 400 ft		
Boulders	0	25
Sand	25	90
Sand, clay	90	148
Gravel, packed, (till?); end of hole 198 ft	148	--
W-86, Altitude 460 ft		
Sand and gravel	0	95
Bedrock	95	--
W-87, Altitude 451 ft		
Sand and gravel	0	34
Bedrock	34	--
W-92		
Clay	0	43
Sand	43	55
Clay	55	95
Bedrock; end of hole 200 ft	95	--
W-93, Altitude 446 ft		
Clay	0	40
Sand	40	55
Clay; sand, coarse	55	70
Bedrock; end of hole 350 ft	70	--
W-98, Altitude 570 ft		
Boulders and hardpan	0	44
Bedrock; end of hole 248 ft	44	--
W-102, Altitude 415 ft		
Unconsolidated deposits, undifferentiated	0	105
Bedrock; end of hole 175 ft	105	--
W-106, Altitude 445 ft		
Boulders	0	15
Gravel; end of hole 62 ft	15	--
W-115, Altitude 430 ft		
Boulders	0	39
Bedrock; end of hole 123 ft	39	--

Appendix 3. Lithologic logs of wells in Bristol, Vermont--*Continued*

Description of material	Top of unit or sample (ft)	Bottom of unit or sample (ft)
W-127, Altitude 375 ft		
Gravel, coarse	0	17
Clay	17	46
Till	46	66
Bedrock; end of hole 450 ft	66	--
W-143, Altitude 440 ft		
Sand and gravel	0	55
Bedrock, limestone; end of hole 205 ft	55	--
W-147, Altitude 575 ft		
Unconsolidated deposits, undifferentiated	0	10
Bedrock	10	--
W-152, Altitude 460 ft		
Hardpan (till?) and boulders	0	110
Gravel; end of hole 145 ft	110	--
W-167, Altitude 520 ft		
Gravel	0	14
Bedrock; end of hole 77 ft	14	--
W-169, Altitude 550 ft		
Sand	0	38
Bedrock; end of hole 270 ft	38	--
W-170, Altitude 505 ft		
Sand	0	32
Bedrock; end of hole 160 ft	32	--
W-173, Altitude 370 ft		
Clay, gravel, cobbles	0	160
Bedrock; end of hole 280 ft	160	--
W-174, Altitude 370 ft		
Clay, gray, and sand	0	140
Bedrock; end of hole 180 ft	140	--
W-193, Altitude 525 ft		
Sand and gravel	0	34
Bedrock; quartzite; end of hole 200 ft	34	--
W-199, Altitude 560 ft		
Clay and stones	0	20
Bedrock; end of hole 520 ft	20	--
W-200, Altitude 570 ft		
Boulders	0	15
Sand	15	55
Bedrock; end of hole 180 ft	55	--
W-205, Altitude 405 ft		
Gravel	0	18
Sand	18	26
Sand, silty; clay	26	46
Gravel, coarse	46	72
Sand	72	174
Hardpan (till)	174	230
W-205 *Continued*		
Bedrock; end of hole 274 ft	230	--
W-206, Altitude 410 ft		
Boulders	0	22
Clay, blue, solid	22	38
Sand	38	44
Sand, coarse; gravel; end of hole 46 ft	44	--
W-209, Altitude 405 ft		
Gravel and boulders	0	18
Clay	18	42
Sand	85	140
Gravel	140	195
Gravel, sandy (till?)	195	209
Bedrock; end of hole 324 ft	209	--
W-211, Altitude 405 ft		
Sand and boulders	0	12
Sand, with clay layers	12	82
Gravel; end of hole 100 ft	82	--
W-219, Altitude 570 ft		
Gravel (till?)	0	40
Bedrock	40	--
W-230, Altitude 470 ft		
Fill	0	16
Gravel and boulders	16	28
Gravel; end of hole 64 ft	28	--
W-233, Altitude 400 ft		
Gravel	0	45
Sand	45	50
Clay	50	160
Sand	160	210
Gravel; end of hole 220 ft	210	--
W-235, Altitude 410 ft		
Gravel	0	60
Gravel; clay	60	180
Gravel; end of hole 223 ft	180	--
W-301, Altitude 430.84 ft		
Sand, medium, some coarse	0	9
Clay, dark gray; some silt	12	14
Silt; clay	17	19
Sand, fine	19	22
Silt, fine sand, clay, pebbles (till);coarse sand and gravel below	22	24
Silt and clay; fine sand and pebbles below	27	29
Till, coarse pebbles; fine to coarse sand; refusal 35 ft	32	--

Appendix 3. Lithologic logs of wells in Bristol, Vermont--*Continued*

Description of material	Top of unit or sample (ft)	Bottom of unit or sample (ft)
W-303, Altitude 448.7 ft		
Sand and pebbles	0	22
Sand, clayey; pebbles; possibly till; refusal 28.5 ft	22	--
W-304, Altitude 410.27 ft		
Gravel, cobbles, and boulders	0	17
Sand, coarse; some pebbles	17	19
Sand, coarse; some pebbles	22	24
Sand, coarse, pebbles (till); refusal 35 ft	27	--
W-305, Altitude 404.4 ft		
Sand, fine	7	9
Sand, medium	12	14
Sand, fine	17	19
Sand, fine; silt	22	24
Sand, very fine, gray	27	29
Silt; clay, laminated	32	34
Sand, fine, and pebble lenses; silt; clay	37	39
Sand, fine; pebbles	42	44
Sand, very fine; some pebbles	47	49
Sand, fine; pebbles	52	54
Till; sand, fine gray; pebbles; silt.	57	59
Till; sand, medium to coarse, gray; pebbles refusal 67.5 ft	62	--
W-307, Altitude 438.40 ft		
Clay; silt and fine sand	0	16
Sand, fine; silt	21	34
Sand, fine to medium	34	67
Clay; silt; sand fine,	67	80
Sand, fine; silt; clay	80	100
Till; end of hole 100 ft	100	--
W-308, Altitude 460.83 ft		
Sand, fine, yellow; some silt	0	17
Clay, sandy, gray-brown	17	22
Sand, coarse; pebbles	22	27
Pebbles; sand, fine to medium	27	32
Sand, very coarse; pebbles; gravel, fine	37	52
Gravel, fine, and pebbles; clay lens	52	54
Sand, coarse	57	59
Sand, medium to very coarse	62	74
Clay, dark gray; silt, gray-green; refusal 85.5 ft	77	--
W-309, Altitude 525.48 ft		
Sand, medium to coarse; pebbles	0	43
Sand, coarse; pebbles	43	45
Sand, fine, brown	48	55
W-309 *Continued*		
Sand, very fine; silt and clay layers, gray	58	60
Sand, medium	63	65
Sand, fine, brown; some silt and clay layers	68	75
Clay, gray; some silt	78	80
Sand, very fine, gray; clay, gray	83	85
Till, sand, clay, pebbles; refusal 100 ft	93	--
W-310, Altitude 487.65 ft		
Sand, fine; some silt stringers	0	24
Sand, very coarse; gravel, coarse	37	39
Sand, medium; some pebbles	42	44
Sand, fine to medium	47	49
Clay, gray; some silt; at base till-sand, clay, pebbles	52	54
Till-sand, pebbles, gray clay	57	59
Sand, coarse, gray, some fine; some rounded pebbles	62	64
Till-sand, coarse, gray; pebbles; clay; refusal 70 ft	67	--
W-312, Altitude 444.41 ft		
Sand, fine; pebbles	0	37
Sand, fine to coarse; refusal 43 ft	37	--
W-313, Altitude 504.92 ft		
Cobbles, gravel, boulders; end of hole 52 ft	0	--
W-314, Altitude 574.69 ft		
Cobbles, coarse sand; end of hole 59 ft	7	--
W-315, Altitude 442.30 ft		
Sand, fine; silt; clay, gray	0	14
Sand, very fine to medium; some silt	17	29
Clay, gray; sand, fine	32	37
Sand, coarse to very coarse; some pebbles	37	69
Till-sand, coarse, gray; pebbles; clay; refusal 100 ft	77	--
W-316, Altitude 506.54 ft		
Gravel and landfill material	0	13
Pebbles and sand	13	25
Sand, fine, uniform; end of hole 39 ft	25	--
W-317, Altitude 527.57 ft		
Sand, fine; pebbles	0	48
Cobbles	48	50
Sand	50	77

Appendix 3. Lithologic logs of wells in Bristol, Vermont--*Continued*

Description of material	Top of unit or sample (ft)	Bottom of unit or sample (ft)
W-317 *Continued*		
Sand, medium; end of hole 79 ft	77	--
W-318, Altitude 552.34 ft		
Sand, medium; pebbles; end of hole 112 ft	0	--
W-319, Altitude 497.46 ft		
Silt and fine sand	0	10
Sand, silt, and cobbles; refusal 15 ft	10	--
W-320, Altitude 450.50 ft		
Clay and silt; some sand, fine	0	12
Silt, clay, and fine sand	12	25
Sand, fine	25	45
Sand, medium	45	70
Clay, silt, and sand	70	81
Till, gray, sandy, angular pebbles; end of hole 90 ft	81	--
W-321, Altitude 411.31 ft		
Sand, fine; silt; clay	0	15
Sand, fine to medium; some silt	15	22
Sand, fine to medium	22	45
Sand, silt, clay	45	77
Till; refusal 80 ft	77	--
W-322, Altitude 431.21 ft		
Sand, silt, clay	0	25
Sand, medium	25	60
Silt and clay	60	75
Sand, silt, clay	75	83
Till, silty, sandy, angular pebbles; end of hole 100 ft	83	--
W-324, Altitude 424.50 ft		
Sand, very fine; silt	0	15
Sand, coarse	15	25
Sand, fine to medium	25	39
Sand, medium	39	53
Sand, fine; silt, clay	53	55
Sand, medium	55	77
Sand, fine; silt; some clay	77	90
Till; end of hole 90 ft	90	--
W-326, Altitude 420.21 ft		
Fill--cobbles, trash	0	15
Sand, coarse; pebbles; silt; end of hole 17 ft	15	--
W-327, Altitude 442.05 ft		
Sand, medium to very coarse; cobbles	0	37
Silt; clay, solid	37	39
Sand, fine; pebbles; refusal 58.5 ft	39	--
W-329, Altitude 413.53 ft		
Sand, gravel, and cobbles	0	10
Sand, silt, clay	10	13
Sand	13	17
Clay	17	19
Sand, fine, uniform	19	50
Sand, very fine; clay	50	65
Sand, fine; refusal 70 ft	65	--
W-330, Altitude 424.20 ft		
Sand, medium to coarse	0	22
Silt and clay	22	25
Till, coarse; end of hole 30 ft	25	--
W-332, Altitude 495.15 ft		
Gravel, sand and cobbles	0	56
Cobbles and gravel	56	83
Sand, possibly fine with clay	83	95
Sand, medium	95	105
Gravel and cobbles; sand, medium to fine	105	127
Till	127	131
Bedrock, dolomite; end of hole 135 ft	131	--
W-334, Altitude 574 ft		
Fill--trash; sand, fine; cobbles	0	10
Sand, gravel, cobbles	10	70
Boulders, cobbles	70	85
Sand, possibly medium	85	127
Sand, medium; cobbles; boulders	127	180
Clay, possibly	180	185
Till, sand, clay, boulders; end of hole 200 ft	185	--
W-336, Altitude 574 ft		
Fill--trash	0	5
Sand, coarse; gravel; boulders	5	83
Sand, fine	83	130
Sand, fine; pebbles; some clay; end of hole 168 ft	130	--
W-337, Altitude 500.82 ft		
Fill--trash	0	4
Gravel, cobbles, boulders, fine sand; end of hole 150 ft	4	--
W-338, Altitude 411 ft		
Sand, medium; silt; end of hole 20 ft	0	--
W-339, Altitude 405 ft		
Sand, fine; end of hole 20 ft	0	--
W-340, Altitude 407 ft		
Sand, fine to medium; end of hole 20 ft	0	--

Appendix 3. Lithologic logs of wells in Bristol, Vermont--*Continued*

Description of material	Top of unit or sample (ft)	Bottom of unit or sample (ft)
W-351, Altitude 574.38 ft		
Gravel and sand; end of hole 24 ft	0	--
W-352, Altitude 503.07 ft		
Cobbles and sand; end of hole 25 ft	0	--
W-353, Altitude 505.96 ft		
Coarse sand, gravel, cobbles; end of hole 41 ft	0	--
W-354, Altitude 506.69 ft		
Sand and gravel; end of hole 43 ft	0	--
W-355, Altitude 510.60 ft		
Sand and gravel; end of hole 41 ft	0	--
W-356, Altitude 498.04 ft		
Sand and gravel; end of hole 57 ft	0	--
W-504, Altitude 424.49 ft		
Sand, medium	0	24
Sand, fine to medium	24	37
Clay, blue; end of hole 44 ft	37	--
W-505, Altitude 420.10 ft		
Silty sand, trace clay	0	13
Sand, coarse	13	20
Gravel, medium; sand, fine; trace silt	20	24
Sand, very fine to coarse, predominantly medium; end of hole 38 ft	24	--
W-507, Altitude 443.73 ft		
Silt, clay, sand	0	35
Gravel, medium; sand, silty	35	48
Sand, fine, trace silt; end of hole 58 ft	48	--
W-508, Altitude 417.48 ft		
Silty sand, trace clay	0	8
Clay, some silt, little sand	8	17
Sand, little silt; end of hole 28 ft	17	--
W-509, Altitude 412.15 ft		
Gravel, little silt	0	15
Gravel, coarse, much silt	15	20
Till	20	--
W-510, Altitude 404.13 ft		
Gravel; end of hole 30 ft	0	--

Description of material	Top of unit or sample (ft)	Bottom of unit or sample (ft)
W-903, Altitude 510 ft		
Sand, silty	0	5
Silt, sandy	5	10
Sand, fine	10	15
Silt, sandy; end of hole 20 ft	15	--
W-904, Altitude 510 ft		
Sand, fine to silty	0	20
Sand, fine; silt	20	30
Silt, fine sand; gravel at 34 ft	30	39
Bedrock, weathered, soft; refusal 44 ft	39	--
W-905, Altitude 510 ft		
Landfilled material	0	5
Gravel, small stones; refusal 33 ft	5	--
BR-2, Altitude 505.56 ft		
Sand, silty	0	20
Silt	20	25
Silt, clayey	25	35
Silt; refusal 40 ft	35	--
BR-6, Altitude 503.05 ft		
Sand and gravel; garbage	0	5
Sand and gravel	5	30
Sand, medium	30	60
Sand; silt layers	60	70
Till or gravel	70	78
MW-101, Altitude 574.19 ft		
Sand, medium; gravel; some boulders	0	40
Gravel, medium, gray; boulders	40	81
Gravel, small; sand	81	130
Silt, sandy, gray	130	135
Sand, fine to medium, brown	135	178
Clay, silty, gray; end of hole 178 ft	178	--
MW-102d, Altitude 528.13 ft		
Sand, medium, brown; some gravel	0	27
Sand, medium, gravelly	27	70
Sand, fine; some silt; little gravel	70	107
Silt, sandy, gray	107	110
Sand; gravel	110	125
Till, silty, gravelly, gray; end of hole 127 ft	125	--

(page 87 follows)

APPENDIX 4

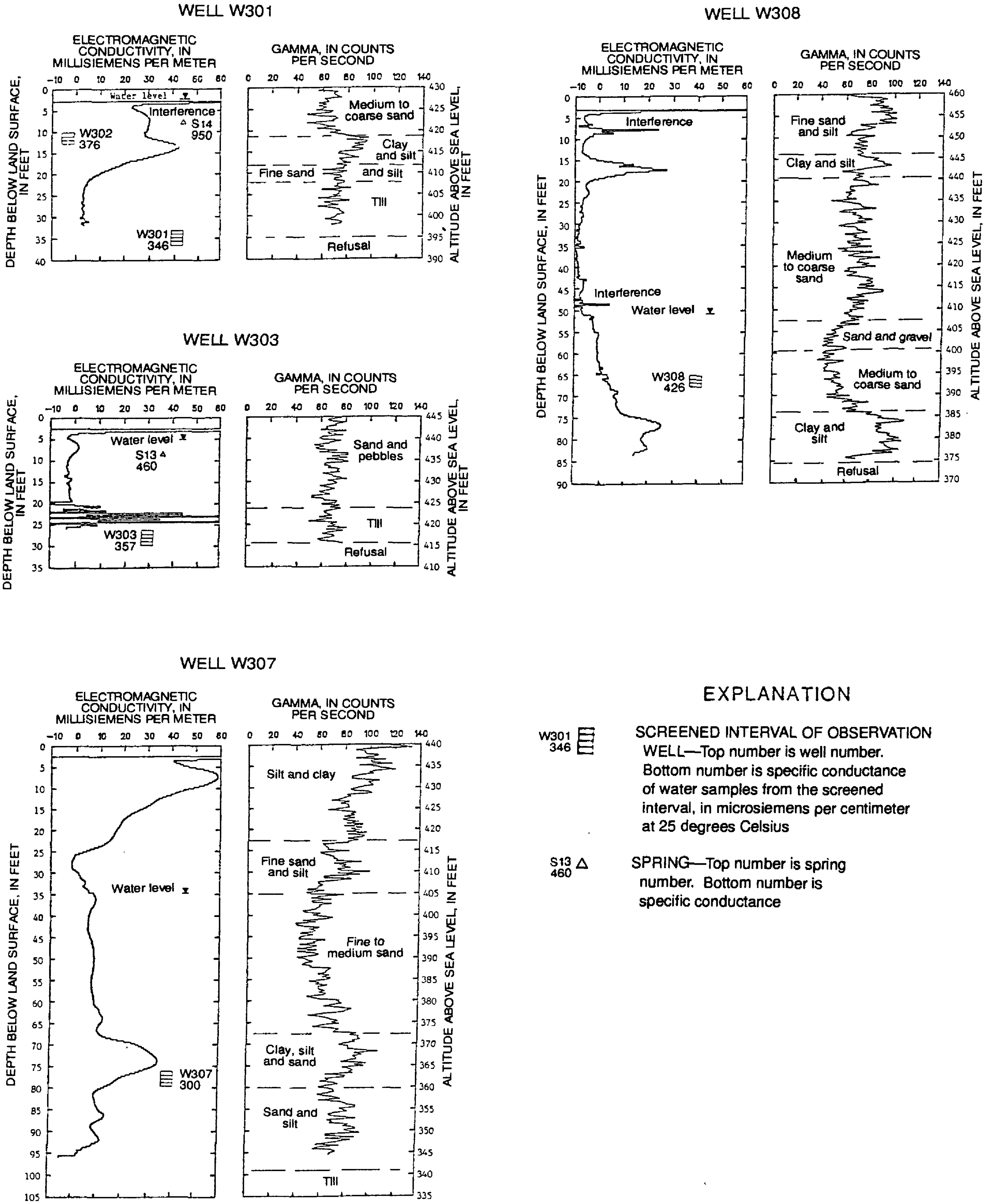
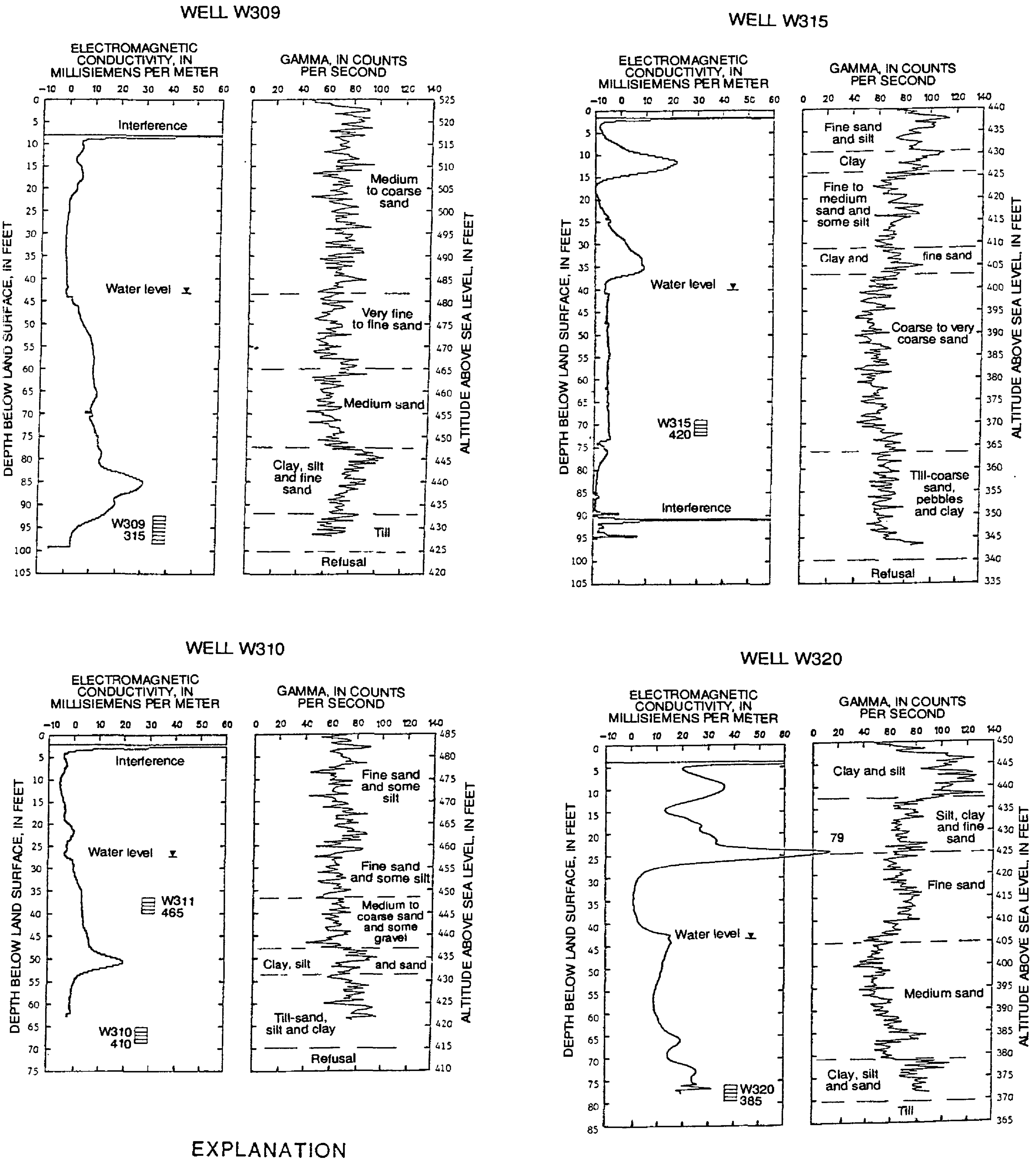


Figure 4-1. Borehole geophysical logs showing geologic section, screened interval, and associated specific conductance of ground water for observation wells W301, W303, W 307, and W308, and for spring S14, Bristol, Vermont (locations shown in figs. 5 and 8)



EXPLANATION

W309
315 — SCREENED INTERVAL OF OBSERVATION WELL—Top number is well number. Bottom number is specific conductance of water samples from the screened interval, in microsiemens per centimeter at 25 degrees Celsius.

Figure 4-2. Borehole geophysical logs showing geologic section, screened interval, and associated specific conductance of ground water for observation wells W309, W310, W315, and W320, Bristol, Vermont (locations shown in fig. 5).

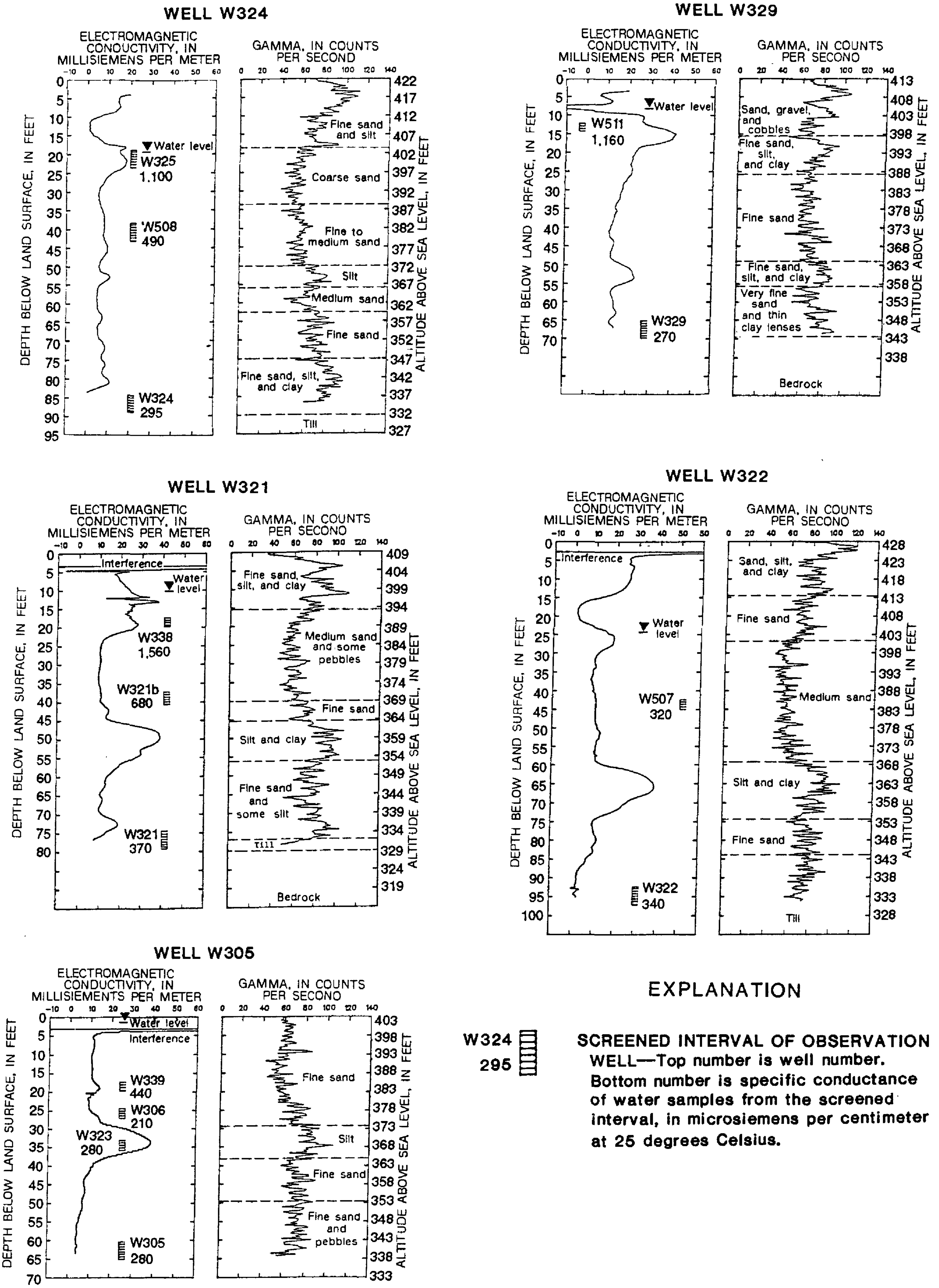


Figure 4-3. Borehole geophysical logs showing geologic section, screened interval, and associated specific conductance of ground water for observation wells W324, W321, W305, W329, and W322, Bristol, Vermont (locations shown in fig. 5).

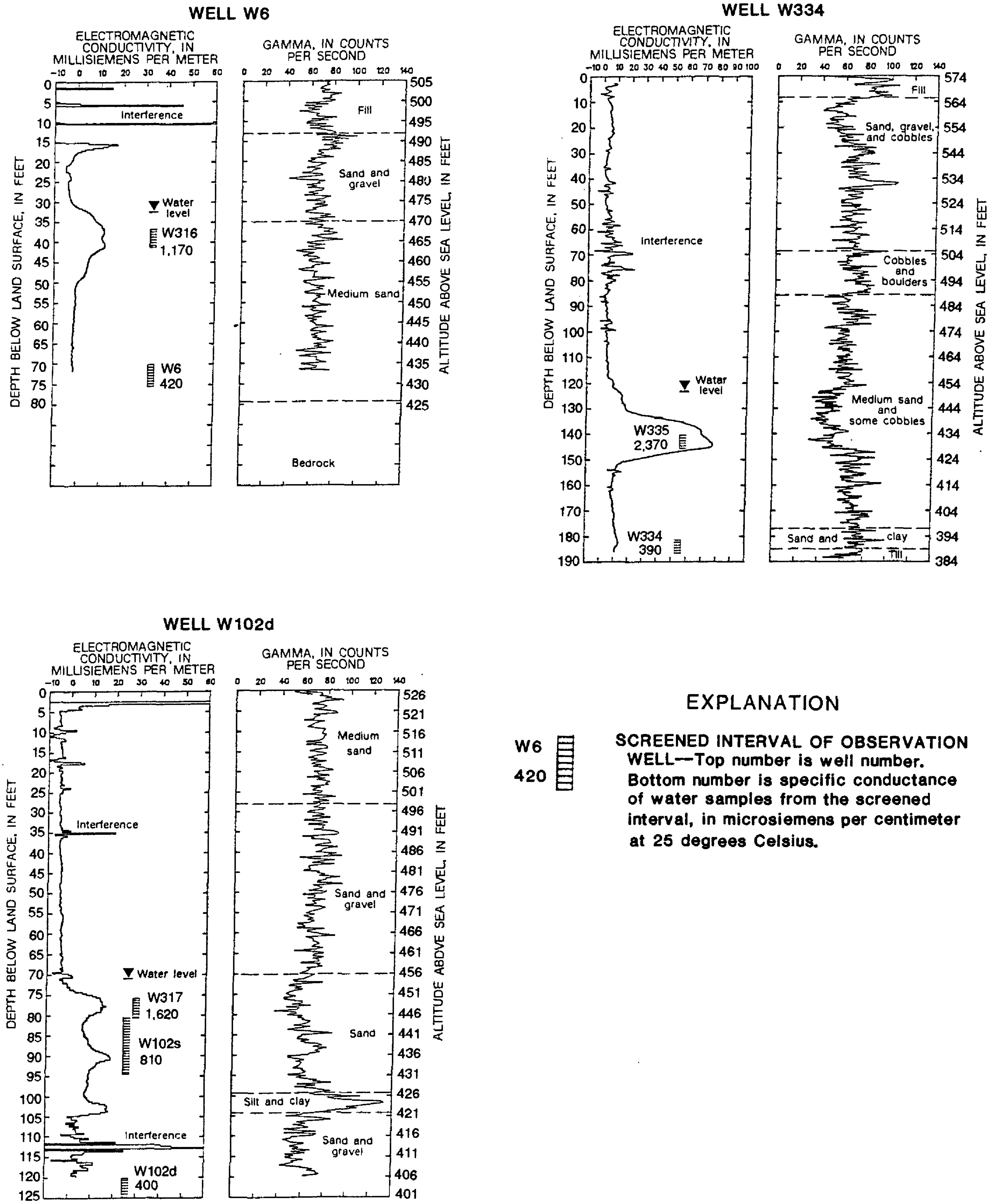


Figure 4-4. Borehole geophysical logs showing geologic section, screened interval, and associated specific conductance of ground water for observation wells W6, W102d, and W334, Bristol, Vermont (locations shown in fig. 5).

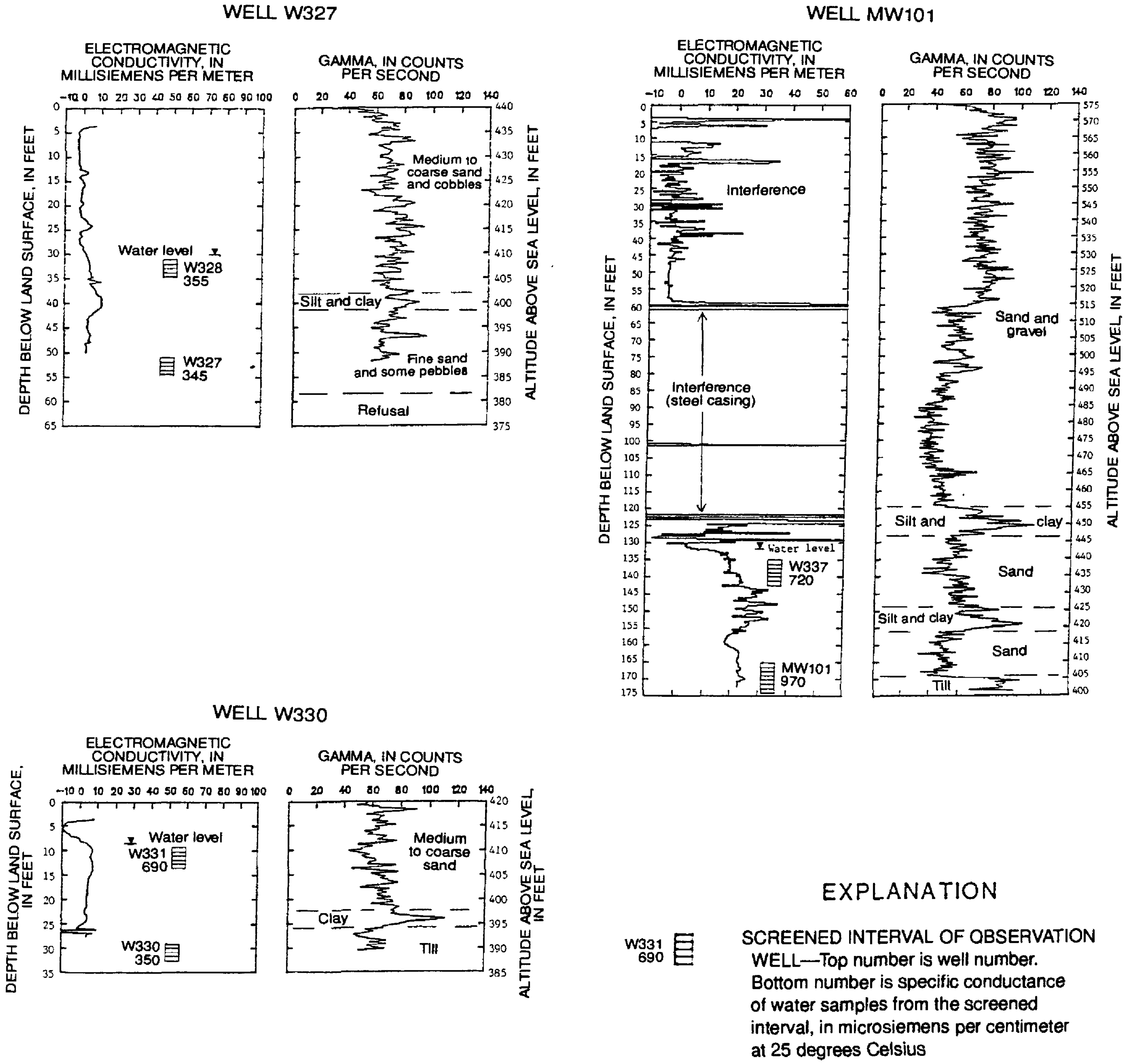


Figure 4-5. Borehole geophysical logs showing geologic section, screened interval, and associated specific conductance of ground water for observation wells W327, W330, and W101, Bristol, Vermont (locations shown in fig. 5)

(page 95 follows)

APPENDIX 5

Appendix 5. Hydraulic conductivity and lithologic descriptions of wells in Bristol, Vermont

[Type of aquifer test: Specific-capacity tests were limited by the capacity of the pump used. ft/d, foot per day. >, actual value is greater than value shown; <, actual value is less than value shown]

Well No.	Hydraulic conductivity (ft/d)	Description of material near well screen	Type of aquifer test
W-303	0.7	Till, clayey sandy	Slug
W-304	2	Sand, fine	Slug
W-305	3	Sand, poorly sorted (till)	Slug
W-306	10	Sand, very fine; silt	Slug
W-308	60	Sand, coarse; gravel	Slug
	>150		Specific capacity
W-309	4	Till, sand, clay pebbles	Slug
W-310	45	Sand, coarse; pebbles; clay	Slug
W-311	30	Sand, medium	Slug
W-312	55	Sand, fine to coarse	Slug
	>150		Slug
W-315	>150	Sand, fine to coarse	Specific capacity
W-320	10	Sand, fine	Slug
W-321	5	Sand; silt; clay	Slug
W-323	30	Silt; clay; fine sand and pebble lenses	Slug
W-325	12	Sand, fine to medium	Slug
W-327	5	Sand, fine	Slug
W-329	27	Sand, fine	Slug
W-330	3	Till, sand, silt	Slug
W-331	200	Sand, medium to coarse	Slug
W-335	>150	Sand and gravel	Specific capacity
W-337	>150	Sand and gravel	Specific capacity
W-338	30	Sand, fine to medium	Slug
W-339	27	Sand, fine	Slug
W-341	60	Sand, medium	Slug
W-356	250	Sand and gravel	Slug
BR-2	<1	Sand, fine	Slug
MW-4	30	Sand	Slug
W-507	100	Sand, coarse	Slug
W-509	120	Sand	Slug
W-508	>150	Sand	Specific capacity
W-510	180	Sand and gravel	Slug
W-511	15	Sand	Slug

(page 99 follows)

APPENDIX 6

(page 101 follows)

Appendix 6. Measured and calculated heads in Bristol, Vermont, April 1991

[--, no data]

Well No.	Model			Head			Hydraulic gradient		
	Layer	Row	Column	Measured (ft)	Calculated (ft)	Difference (ft)	Measured (ft)	Calculated (ft)	Difference (ft)
W-12	2	8	45	[1]480.0	479.6	-0.4	--	--	--
W-318	1	14	32	453.1	454.1	1.0	--	--	--
W-302	1	18	15	428.9	423.3	-5.6	--	--	--
W-301	2	18	15	[1]435.0	433.7	-1.3	6.0	10.4	4.4
W-303	2	14	14	444.4	443.9	-.5	--	--	--
W-311	1	11	15	455.8	454.9	-.9	--	--	--
W-310	2	11	15	458.0	461.7	3.7	2.2	6.8	4.4
BR-2	1	8	25	480.8	479.7	-1.1	--	--	--
W-319	1	7	25	484.1	485.5	1.4	--	--	--
MW-2	1	7	31	486.4	487.4	1.0	--	--	--
W-309b	1	6	38	484.1	483.2	-.9	--	--	--
W-309	2	6	38	488.4	488.3	-.1	4.3	5.1	.8
W-316	1	9	31	470.9	471.2	.3	--	--	--
BR-6	2	9	31	482.1	483.1	1.0	11.2	11.9	.7
MW-4	1	11	27	461.5	461.1	-.4	--	--	--
W-317	1	12	23	453.0	449.9	-3.1	--	--	--
MW-102s	1	13	23	452.9	446.8	-6.1	--	--	--
MW-102d	2	13	23	455.4	453.4	-2.0	2.5	6.6	4.1
W-328	1	26	13	[1]408.6	412.5	3.9	--	--	--
W-327	2	26	13	[1]411.9	425.5	3.6	3.3	3.0	-.3
W-331	1	23	13	[1]413.7	415.2	1.5	--	--	--
W-330	2	23	13	[1]415.3	419.0	3.7	1.6	3.7	2.1
W-335	1	14	25	[1]447.3	447.8	.5	--	--	--
W-334	2	14	26	[1]455.5	453.4	-2.1	8.2	5.6	-2.6
W-337	1	19	21	[1]446.9	437.4	-9.5	--	--	--
MW-101	2	19	20	445.4	439.1	-6.3	-1.5	1.7	-3.2
W-315	1	27	3	[1]400.4	399.1	-1.3	--	--	--
W-304	1	30	13	406.5	409.8	3.3	--	--	--
W-509	1	28	9	403.1	408.0	4.9	--	--	--
W-510	1	31	9	401.8	406.3	4.5	--	--	--
W-306	1	36	5	399.3	398.5	-.8	--	--	--
W-305	2	36	5	401.0	398.8	-2.2	1.7	.3	1.4
W-323	2	36	5	[1]399.8	398.8	-1.0	--	--	--
W-340	1	35	6	[1]401.2	400.5	-.7	--	--	--
W-321b	1	36	7	402.3	402.2	-.1	--	--	--
W-321	2	36	7	404.8	404.3	-.5	2.5	2.1	-.4
W-307	2	43	10	405.1	407.5	2.4	--	--	--
W-308	1	40	25	407.5	412.9	5.4	--	--	--
W-320	2	40	19	406.1	410.2	4.1	--	--	--
W-507	1	40	13	404.8	408.5	3.7	--	--	--
W-322	2	40	10	404.1	407.8	3.7	--	--	--
W-325	1	38	10	[1]403.2	406.2	3.0	--	--	--

Appendix 6. Measured and calculated heads in Bristol, Vermont, April 1991--*Continued*

Well No.	Model			Head			Hydraulic gradient		
	Layer	Row	Column	Measured (ft)	Calculated (ft)	Difference (ft)	Measured (ft)	Calculated (ft)	Difference (ft)
W-324	2	38	10	[1]404.8	408.0	3.1	1.6	1.8	.2
W-341	1	37	7	[1]402.7	402.1	-0.6	--	--	--
W-508	1	37	8	402.7	403.7	1.0	--	--	--
W-326	1	35	9	[1]404.9	405.1	.2	--	--	--
W-511	1	34	8	401.7	404.3	2.6	--	--	--
W-329	2	34	8	402.9	405.6	2.7	1.2	2.3	1.1
W-332	2	36	25	[1]414.1	416.8	2.7	--	--	--
W-312	1	46	32	408.0	411.2	3.2	--	--	--
W-233	2	56	42	[1]405.0	400.5	-1.5	--	--	--
W-551	1	42	47	[1]483.9	483.6	-.3	--	--	--
W-552	1	40	47	[1]485.3	485.3	0	--	--	--
W-553	1	37	46	[1]470.2	475.0	4.8	--	--	--
W-554	1	36	46	[1]465.5	474.9	9.4	--	--	--
W-555	1	38	46	[1]472.5	474.9	2.4	--	--	--
W-556	1	38	45	[1]463.8	461.4	-2.4	--	--	--
W-82	1	57	40	[1]398.0	396.7	-1.3	--	--	--
Standard mean				--	--	+.6	--	--	+1.0
Absolute mean				--	--	2.5	--	--	2.0

[1] Estimated head.

APPENDIX 7

Appendix 7. Summary of water-quality analyses in Bristol, Vermont

[mmddyy, month, day year; µS/cm, microsiemen per centimeter at 25 degress Celsius; mg/L, milligram per liter; µg/L, microgram per

Well No.	Date (mmddyy)	pH	Specific conductance (µS/cm)	Calcium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Chloride, dissolved (mg/L)	Sodium, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Zinc, dissolved (mg/L)	Chromium, dissolved (mg/L)
W-12	091990	--	298	--	--	13.0	5.80	19	<10	--	308	<10
W-303	090690	7.6	230	--	--	1.4	3.24	1,020	326	--	<40	<10
	052091	7.45	355	42.8	1.48	1.5	--	21	87	25.4	<40	<10
	101091	--	357	39.2	1.20	1.7	3.12	18	153	23.1	<40	<10
W-304	090590	7.6	230	--	--	2.2	7.29	20	85	--	<40	<10
	050891	--	293	65.8	3.80	1.5	6.98	2,890	583	34.7	<40	<10
	052291	--	302	--	--	--	--	--	--	--	--	--
	100891	--	326	28.8	2.26	3.2	4.52	18	90	21.8	<40	<10
W-305	090590	7.6	280	--	--	1.3	2.91	<10	80	--	<40	<10
	050891	--	--	42.6	1.35	1.2	2.33	147	109	21.9	<40	<10
	100891	--	414	44.7	1.48	2.6	3.82	14	95	24.5	<40	<10
W-306	090590	7.6	210	--	--	12.4	12.3	<10	245	--	<40	<10
	050891	--	--	43.0	1.28	14.9	10.9	42	155	17.5	<40	<10
	100891	--	385	39.6	1.18	16.0	11.2	<10	13	16.8	<40	<10
W-309	090590	--	180	--	--	8.0	3.79	10	215	--	<40	<10
	050891	--	303	33.2	1.11	9.6	5.65	147	265	16.8	<40	<10
	052291	--	318	--	--	--	--	--	--	--	--	--
	100991	--	315	36.9	1.11	9.0	3.44	<10	127	17.0	<40	<10
W-310	090590	7.4	260	--	--	4.8	4.06	984	439	--	<40	<10
	052091	7.08	412	45.4	1.20	5.0	--	117	13	26.9	<40	<10
	101091	--	416	43.4	1.10	4.9	4.40	43	149	26.3	<40	<10
W-311	090690	--	330	--	--	32.4	7.35	469	1,100	--	51	<10
	052091	6.58	386	--	--	6.7	--	54	26	--	<40	<10
	101091	--	465	61.1	1.30	19.7	10.1	42	16	17.1	<40	<10
W-312	091990	--	376	--	--	29.3	25.2	1,030	240	--	<40	<10
	050891	--	365	30.6	0.76	40.9	25.8	408	171	13.9	<40	<10
	052291	--	364	--	--	--	--	--	--	--	--	--
	101091	--	--	32.6	1.48	31.0	23.9	244	10	12.9	<40	<10
W-315	091990	--	417	--	--	11.8	7.12	131	62	--	<40	<10
W-318	021391	--	335	--	--	27.6	16.9	60	<10	--	<40	<10
	052091	7.18	458	30.5	1.66	30.5	--	14	<10	17.5	<40	<10
	101891	--	--	42.2	1.02	23.8	19.7	<10	<10	17.6	<40	<10
W-307	090690	--	--	--	--	5.1	9.15	293	96	--	<40	<10
	050891	--	320	18.3	0.23	1.3	18.3	1,030	191	13.9	<40	<10
	052291	--	356	28.1	1.34	--	15.1	17	96	13.7	<40	<10
	100891	--	300	26.8	1.31	1.9	17.8	21	94	13.1	<40	<10
W-308	090690	--	--	--	--	6.7	--	438	187	--	112	<10
	050891	--	426	45.4	1.33	16.7	9.09	342	94	20.8	72	<10
	100891	--	459	51.0	1.30	18.0	8.96	<10	<10	22.0	<40	<10
W-320	021391	--	291	--	--	2.5	9.81	76	541	--	<40	<10
	050891	--	368	36.1	1.41	1.6	9.43	252	551	21.7	168	<10

liter; <, actual value is less than value shown. --, no data]

Well No.	Date (mmddyy)	Cadmium, dissolved (mg/L)	Copper, dissolved (mg/L)	Lead, dissolved (mg/L)	Cobalt, dissolved (mg/L)	Arsenic, dissolved (mg/L)	Selenium, dissolved (mg/L)	Barium, dissolved (mg/L)	Nickel, dissolved (mg/L)	Silver, dissolved (mg/L)	Sulfate, dissolved (mg/L)
W-12	091990	<2	--	<10	--	--	--	--	--	--	13.2
W-303	090690	<2	--	<10	--	--	--	--	--	--	19.1
	052091	<2	<10	<10	<10	<5	<5	30	<10	<1	19.6
	101091	<2	<10	<10	--	<5	<5	39	<10	<1	20.0
W-304	090590	<2	--	<10	--	--	--	--	--	--	17.3
	050891	<2	<10	<10	<10	<5	<5	52	<10	<1	8.6
	052291	--	--	--	--	--	--	--	--	--	--
	100891	<2	<10	<10	--	<5	<5	111	<10	<1	17.0
W-305	090590	<2	--	<10	--	--	--	--	--	--	24.8
	050891	<2	<10	<10	<10	<5	<5	38	<10	<1	22.7
	100891	<2	12	<10	--	<5	<5	75	<10	<1	22.9
W-306	090590	<2	--	<10	--	--	--	--	--	--	15.8
	050891	<2	<10	<10	<10	<5	<5	27	<10	<1	16.5
	100891	<2	<10	<10	--	<5	<5	76	<10	<1	17.3
W-309	090590	<2	--	<10	--	--	--	--	--	--	42.2
	050891	<2	<10	<10	<10	<5	<5	36	<10	<1	43.0
	052291	--	--	--	--	--	--	--	--	--	--
	100991	<2	<10	<10	<10	<5	<5	40	<10	<1	41.2
W-310	090590	<2	--	<10	--	--	--	--	--	--	23.5
	052091	4	<10	<10	<10	<5	<5	42	<10	<1	23.6
	101091	<2	<10	<10	--	<5	<5	21	<10	<1	23.4
W-311	090690	<2	--	15	--	--	--	--	--	--	15.7
	052091	<2	27	<10	<10	<5	<5	132	<10	<1	11.3
	101091	<2	<10	<10	--	<5	<5	31	<10	<1	11.2
W-312	091990	<2	--	<10	--	--	--	--	--	--	6.0
	050891	<2	<10	10	<10	<5	<5	24	<10	<1	5.6
	052291	--	--	--	--	--	--	--	--	--	--
	101091	<2	12	<10	--	<5	<5	14	<10	<1	6.9
W-315	091990	<2	--	<10	--	--	--	--	--	--	15.2
W-318	021391	<2	<10	<10	--	--	--	--	--	--	18.2
	052091	<2	11	<10	<10	<5	<5	72	<10	<1	19.6
	101891	<2	13	<10	<10	<5	<5	18	<10	<1	19.6
W-307	090690	<2	--	<10	--	--	--	--	--	--	8.8
	050891	2	158	21	<10	<5	<5	24	<10	<1	11.9
	052291	<2	<10	<10	<10	<5	<5	50	<10	<1	--
	100891	<2	<10	<10	--	<5	<5	28	<10	<1	11.6
W-308	090690	<2	--	<10	--	--	--	--	--	--	11.5
	050891	<2	21	73	<10	<5	<5	24	<10	<1	14.2
	100891	<2	17	<10	<10	<5	<5	31	<10	<1	15.5
W-320	021391	<2	12	<10	--	--	--	--	--	--	14.6
	050891	<2	<10	11	<10	<5	<5	38	<10	<1	17.9

Appendix 7. Summary of water-quality analyses in Bristol, Vermont--*Continued*

Well No.	Date (mmddyy)	pH	Specific conductance (µS/cm)	Calcium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Chloride, dissolved (mg/L)	Sodium, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Zinc, dissolved (mg/L)	Chromium, dissolved (mg/L)
W-320	052291	--	365	37.0	1.44	--	10.0	28	444	21.3	<40	<10
	100891	--	386	36.9	1.13	0.9	8.31	<10	226	22.2	<40	<10
W-321	021391	--	304	--	--	7.2	13.3	26	535	--	<40	<10
	050891	--	372	41.1	1.35	3.8	11.0	139	150	20.3	<40	<10
	052291	--	369	--	--	--	--	--	--	--	--	--
	100891	--	383	39.7	1.71	3.1	6.03	<10	18	21.3	<40	<10
W-322	050891	--	340	37.6	1.12	1.4	14.3	261	149	16.6	41	<10
	052291	--	338	--	--	--	--	--	--	--	--	--
W-324	100891	--	354	41.4	1.47	3.1	8.13	14	117	23.6	<40	<10
W-323	100891	--	359	37.9	1.25	12.0	11.3	<10	114	16.1	<40	<10
W-329	100891	--	354	37.3	1.35	2.8	4.58	<10	56	20.8	<40	<10
W-334	081491	--	398	--	--	--	--	--	--	--	--	--
	082291	8.24	390	--	--	4.5	48.2	87	--	12.2	<40	<10
	101791	--	--	37.7	1.89	3.8	15.5	29	341	21.8	<40	<10
W-509	092090	--	379	--	--	7.6	4.63	291	386	--	<40	<10
	050891	--	318	118	1.45	9.1	6.93	733	944	43.8	<40	<10
	052291	--	336	--	--	--	--	--	--	--	--	--
	100891	--	350	42.4	.96	7.2	6.03	14	39	21.3	<40	<10
S-13	052091	--	--	50.2	.90	15.6	--	167	<10	19.3	<40	<10
	101091	--	464	61.0	.60	15.1	9.15	16	84	18.5	<40	<10
S-15	090590	7.1	520	--	--	68.4	34.1	70	58.0	--	<40	<10
	050991	--	367	39.4	1.60	22.1	13.8	66	11.0	15.7	<40	<10
	101091	--	--	37.0	3.94	50.1	24.8	144	83.0	15.5	<40	<10
S-12	070690	6.9	310	--	--	18.7	8.82	8,180	249	--	23	<5
	090590	7.3	340	--	--	22.9	9.47	98	49.0	--	<40	<10
	050991	--	291	33.4	1.42	10.2	6.75	39	41.0	14.2	<40	<10
S-11	070690	8.0	200	--	--	24.7	14.4	1,150	76.0	--	<20	<5
	090590	7.1	190	--	--	24.7	14.8	182	135	--	<40	<10
W-301	090690	7.6	240	--	--	1.5	3.36	331	194	--	<40	<10
	052091	7.28	358	37.7	.91	1.5	--	40	117	22.8	<40	<10
	052091	--	--	38.8	1.00	--	3.31	380	151	23.0	<40	<10
	101091	--	346	37.8	.95	1.7	3.28	39	131	22.1	<40	<10
W-302	091990	7.6	376	--	--	2.9	4.88	591	708	--	<40	<10
S-14	070690	7.1	470	--	--	11.7	6.17	888	108	--	<20	<5
	090590	7.4	520	--	--	13.3	6.1	135	<10	--	<40	<10
	052091	6.60	810	100	6.31	31.4	--	<10	--	38.3	<40	<10
	082291	7.12	1,000	--	--	44.2	37.6	53	--	44.0	<40	<10
	101091	--	952	95.5	19.9	41.5	30.5	12	16	41.3	<40	<10
W-317	021391	--	955	--	--	23.8	22.4	539	207	--	<40	<10
	050991	--	1,618	--	--	96.8	88.0	167	624	--	<40	<10
	100991	--	1,135	164	1.62	16.4	16.1	39	924	55.3	<40	<10
MW-102s	091990	--	1,260	--	--	26.4	7.23	839	4,790	--	<40	<10
	052091	5.96	812	--	--	36.6	--	<10	137	--	<40	<10

Well No.	Date (mmddyy)	Cadmium, dissolved (mg/L)	Copper, dissolved (mg/L)	Lead, dissolved (mg/L)	Cobalt, dissolved (mg/L)	Arsenic, dissolved (mg/L)	Selenium, dissolved (mg/L)	Barium, dissolved (mg/L)	Nickel, dissolved (mg/L)	Silver, dissolved (mg/L)	Sulfate, dissolved (mg/L)
W-320	052291	<2	12	<10	--	<5	<5	50	<10	<1	--
	100891	<2	<10	<10	--	<5	<5	98	<10	<1	19.0
W-321	021391	<2	<10	<10	--	--	--	--	--	--	14.0
	050891	<2	<10	<10	<10	<5	<5	48	<10	<1	17.1
	052291	--	--	--	--	--	--	--	--	--	--
	100891	<2	<10	<10	--	<5	<5	59	20	<1	19.8
W-322	050891	2	18	<10	<10	<5	<5	35	<10	<1	12.8
	052291	--	--	--	--	--	--	--	--	--	--
W-324	100891	<2	<10	<10	--	<5	<5	203	<10	<1	20.7
W-323	100891	<2	12	<10	--	<5	<5	125	<10	<1	15.3
W-329	100891	<2	<10	<10	--	<5	<5	93	<10	<1	14.8
W-334	081491	--	--	--	--	--	--	--	--	--	--
	082291	<2	76	<10	<10	<5	<5	20	23	<1	27.5
	101791	<2	<10	<10	<10	<5	<5	21	13	<1	23.6
W-509	092090	<2	--	<10	--	--	--	--	--	--	13.3
	050891	<2	10	<10	<10	<5	<5	170	<10	<1	11.6
	052291	--	--	--	--	--	--	--	--	--	--
	100891	<2	<10	<10	--	<5	<5	99	<10	<1	12.9
S-13	052091	<2	11	<10	<10	<5	<5	34	<10	<1	13.9
	101091	<2	<10	<10	<10	<5	<5	21	<10	<1	14.2
S-15	090590	<2	--	<10	--	--	--	--	--	--	10.6
	050991	<2	<10	<10	<10	<5	<5	14	<10	<1	6.8
	101091	<2	<10	<10	--	<5	<5	28	<10	<1	16.7
S-12	070690	<1	--	9	--	--	--	--	--	--	7.4
	090590	<2	--	<10	--	--	--	--	--	--	7.1
	050991	<2	13	<10	<10	<5	<5	10	<10	<1	9.5
S-11	070690	<1	--	<5	--	--	--	--	--	--	12.3
	090590	<2	--	10	--	--	--	--	--	--	9.8
W-301	090690	<2	--	--	--	--	--	--	--	--	19.7
	052091	<2	18	<10	--	<5	<5	38	<10	<1	19.0
	052091	<2	<10	<10	--	<5	<5	20	<10	<1	--
	101091	<2	<10	<10	--	<5	<5	18	<10	<1	19.8
W-302	091990	<2	--	<10	--	--	--	--	--	--	20.3
S-14	070690	<1	--	<5	--	--	--	--	--	--	13.8
	090590	<2	--	<5	--	--	--	--	--	--	15.9
	052091	<2	<10	<10	--	<5	<5	66	<10	<1	23.8
	082291	<2	<10	<10	<10	<5	<5	75	10	<1	27.4
	101091	<2	<10	<10	<10	<5	<5	98	<10	<1	21.9
W-317	021391	<2	32	<10	--	--	--	--	--	--	64.7
	050991	<2	18	<10	<10	<5	<5	190	29	<1	42.6
	100991	<2	<10	<10	<10	<5	<5	135	11	<1	30.8
MW-102s	091990	<2	--	--	--	--	--	--	--	--	17.0
	052091	<2	14	<10	<10	<5	<5	112	29	<1	14.4

Appendix 7. Summary of water-quality analyses in Bristol, Vermont--*Continued*

Well No.	Date (mmddyy)	pH	Specific conductance (µS/cm)	Calcium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Chloride, dissolved (mg/L)	Sodium, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Zinc, dissolved (mg/L)	Chromium, dissolved (mg/L)
MW-102s	100991	--	805	76.7	7.29	39.2	36.5	26	3,790	37.8	<40	<10
MW-102d	091990	--	342	--	--	18.7	10.0	837	182	--	<40	<10
	052091	7.12	401	--	--	18.3	--	57	70	--	<40	<10
	101091	--	395	39.0	.84	19.8	11.7	<10	73	22.1	<40	<10
BR2	091990	--	268	--	--	4.3	3.07	5,350	2,770	--	118	<10
	050891	--	230	--	--	10.8	7.02	4,070	2,270	--	<40	<10
	052291	--	219	--	--	--	--	--	--	--	--	--
	101091	--	255	27.7	.88	4.8	3.64	2,270	2,300	11.2	<40	<10
BR6	091990	--	433	--	--	22.7	9.16	3,140	1,980	--	198	<10
	050991	--	451	--	--	23.6	8.55	153	344	--	<40	<10
	100991	--	472	88.1	4.49	30.3	21.0	15	39	32.8	<40	<10
W-316	021391	--	819	--	--	44.0	40.0	4,250	3,090	--	49	<10
	050991	--	636	--	--	18.2	13.9	1,470	2,340	--	<40	<10
	100991	--	366	54.6	3.32	37.6	14.0	551	2,960	13.9	<40	<10
W2	091990	--	490	--	--	26.5	11.3	1,060	862	--	<40	<10
	050891	--	200	44.1	1.25	43.4	19.2	973	840	31.2	<40	<10
	052291	--	409	--	--	--	--	--	--	--	--	--
	101091	--	492	34.7	1.02	31.6	16.2	13	<10	16.0	<40	<10
W4	091990	--	524	--	--	28.7	16.6	973	561	--	<40	<10
	050991	--	454	--	--	19.9	16.5	132	120	--	<40	<10
	101091	--	368	47.0	2.26	27.2	14.8	29	15	21.3	<40	<10
W-327	081592	--	345	--	--	--	--	--	--	--	--	--
	100991	--	511	34.7	.90	8.9	4.12	24	230	23.5	<40	<10
W-328	081592	--	444	--	--	--	--	--	--	--	--	--
	082291	7.55	506	--	--	33.0	11.4	26	--	25.0	<40	<10
	100991	--	355	61.9	.97	12.8	9.33	35	<10	24.1	<40	<10
W-330	100991	--	354	38.6	1.18	1.7	3.33	59	352	22.9	<40	<10
W-331	082291	7.25	694	--	--	33.2	20.3	131	--	59.0	<40	<10
	100991	--	762	51.4	.97	34.2	9.50	64	<10	25.6	<40	<10
MW-101	091990	--	1,260	--	--	86.2	56.9	952	1,020	--	--	<10
	052091	6.33	970	--	--	48.8	--	21	392	--	<40	<10
	081591	--	820	--	--	--	--	--	--	--	--	--
	101791	--	--	107	9.98	41.2	25.4	24	548	36.0	<40	<10
W-335	090591	--	2,920	--	--	820	432	139	--	44.0	<40	<10
	101791	--	--	79.5	2.28	536	269	11	273	45.8	<40	<10
W-337	081491	--	720	--	--	--	--	--	--	--	--	--
W-337	082291	--	421	--	--	46.8	25.6	29	--	34.3	<40	<10
W-337	101791	--	--	106	12.1	65.2	28.3	<10	556	34.3	<40	<10
W-507	092090	--	386	--	--	18.4	9.12	1,220	536	--	<40	<10
W-507	050891	--	323	45.3	1.36	17.5	10.6	553	66.0	66.0	<40	103
W-507	100991	--	276	40.2	1.48	18.2	10.3	19	93	14.9	<40	<10
W-508	092090	--	544	--	--	27.9	18.2	133	96.0	--	<40	<10

Well No.	Date (mmddyy)	Cadmium, dissolved (mg/L)	Copper, dissolved (mg/L)	Lead, dissolved (mg/L)	Cobalt, dissolved (mg/L)	Arsenic, dissolved (mg/L)	Selenium, dissolved (mg/L)	Barium, dissolved (mg/L)	Nickel, dissolved (mg/L)	Silver, dissolved (mg/L)	Sulfate, dissolved (mg/L)
MW-102s	100991	<2	12	<10	--	<5	<5	197	24	<1	13.8
MW-102d	091990	<2	--	--	--	--	--	--	--	--	10.2
	052091	<2	<10	<10	--	<5	<5	11	<10	<1	11.4
	101091	<2	<10	<10	--	<5	<5	14	<10	<1	12.2
BR2	091990	<2	--	<10	--	--	--	--	--	--	12.6
	050891	<2	15	16	19	5	<5	16	18	<1	14.6
	052291	--	--	--	--	--	--	--	--	--	--
	101091	<2	<10	<10	<10	<5	<5	21	20	<1	14.9
BR6	091990	<2	--	<10	--	--	--	--	--	--	22.2
	050991	<2	<10	<10	<10	<5	<5	42	<10	<1	31.0
	100991	<2	<10	<10	<10	<5	<5	146	<10	<1	24.8
W-316	021391	<2	--	<10	--	--	--	--	--	--	102.0
	050991	<2	<10	<10	<10	<5	<5	90	<10	<1	50.0
	100991	<2	30	<10	<10	<5	<5	137	<10	<1	20.6
W2	091990	<2	--	15	--	--	--	--	--	--	26.0
	050891	<2	25	<10	<10	<5	<5	43	<10	6.0	15.5
	052291	--	--	--	--	--	--	--	--	--	--
	101091	<2	<10	<10	--	<5	<5	26	<10	<1	22.1
W4	091990	<2	--	<10	--	--	--	--	--	--	54.6
	050991	<2	<10	<10	<10	<5	<5	78	<10	<1	39.6
	101091	<2	<10	<10	--	<5	<5	31	11	<1	28.4
W-327	081592	--	--	--	--	--	--	--	--	--	--
	100991	<2	27	<10	--	<5	<5	127	<10	<1	19.3
W-328	081592	--	--	--	--	--	--	--	--	--	--
	082291	<2	<10	<10	<10	<5	<5	58	<10	<1	21.2
	100991	<2	<10	<10	--	<5	<5	95	<10	<1	17.8
W-330	100991	<2	20	<10	--	<5	<5	138	<10	<1	19.9
W-331	082291	<2	12	<10	<10	<5	<5	102	<10	<1	30.4
	100991	<2	12	<10	--	<5	<5	133	<10	<1	27.3
MW-101	091990	<2	--	<10	--	--	--	--	--	--	71.8
	052091	<2	<10	<10	<10	<5	<5	168	<10	<1	68.1
	081591	--	--	--	--	--	--	--	--	--	--
	101791	<2	<10	<10	<10	<5	<5	89	<10	<1	46.2
W-335	090591	<2	<10	<10	<10	<5	<5	110	15	<1	35.2
	101791	<2	<10	<10	<10	<5	<5	103	<10	<1	31.7
W-337	081491	--	--	--	--	--	--	--	--	--	--
	082291	<2	23	<10	<10	<5	<5	85	12	<1	44.4
	101791	<2	<10	<10	<10	<5	<5	91	<10	<1	34.1
W-507	092090	<2	--	<10	--	--	--	--	--	--	10.0
	050891	<2	<10	<10	<10	<5	<5	8	51	<1	9.9
	100991	<2	<10	<10	<10	<5	<5	22	<10	<1	11.0
W-508	092090	<2	--	<10	--	--	--	--	--	--	12.7

Appendix 7. Summary of water-quality analyses in Bristol, Vermont--*Continued*

Well No.	Date (mmddyy)	pH	Specific conductance (µS/cm)	Calcium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Chloride, dissolved (mg/L)	Sodium, dissolved (mg/L)	Iron, dissolved (mg/L)	Manganese, dissolved (mg/L)	Magnesium, dissolved (mg/L)	Zinc, dissolved (mg/L)	Chromium, dissolved (mg/L)
W-508	100891	--	494	57.7	1.32	24.1	15.0	1,870	580	32.2	<40	<10
	050891	--	618	63.2	1.12	34.5	21.8	38	1,880	38.8	<40	<10
W-510	092090	--	807	--	--	107	59.4	159	40	--	<40	<10
	050991	--	420	90.7	2.97	94.8	18.8	64	<10	24.9	<40	<10
	100891	--	1,052	80.2	4.78	148	76.8	11	<10	31.5	<40	<10
W-511	092090	--	637	--	--	20.3	16.9	20,200	2,020	--	81	<10
	092090	--	656	--	--	20.6	17.3	19,900	201	--	58	<10
	050991	--	770	75.1	6.87	24.8	55.6	6,490	1,640	--	<40	<10
	100891	--	791	87.9	7.78	27.9	19.3	8,400	1,800	24.7	49	<10
W-325	082291	--	1,097	130	2.07	104	63.9	69	--	37.0	<40	<10
	100891	--	1,183	133	2.01	111	59.0	389	1,760	42.3	<40	<10
W-326	082291	--	--	--	--	--	--	141	10,100	--	<40	<10
	100891	--	1,050	120	23.9	37.3	24.8	1,180	9,320	33.3	<40	<10
W-332	081591	--	570	--	--	--	--	--	--	--	--	--
	082291	6.87	655	16.8	7.73	16.8	117	31	--	7.60	<40	<10
	081591	--	--	22.0	5.62	6.2	86.0	17	15	15.9	<40	<10
W-333	082291	--	452	11.5	22.9	26.9	213	62	--	3.59	<40	<10
W-339	081491	--	860	--	--	--	--	--	--	--	--	--
W-338	082291	6.40	951	69.9	2.06	153	78.6	152	--	37.6	<40	<10
	100891	--	1,250	72.3	2.18	228	104	11	911	45.8	<40	<10
W-339	081491	--	440	--	--	--	--	--	--	--	--	--
	082291	7.24	409	58.4	1.42	14.0	10.1	96	--	15.1	<40	<10
	100891	--	425	57.2	1.04	13.4	10.0	39	<10	17.4	<40	<10
W-340	081491	--	785	--	--	--	--	--	--	--	--	--
	082291	7.10	943	97.4	1.64	62.0	42.8	71	--	37.3	<40	<10
	100891	--	900	95	1.62	65.2	44.1	<10	75	35.0	50	<10
W-341	081491	--	1,980	--	--	--	--	--	--	--	--	--
	082291	7.05	1,407	156	1.74	144	97.7	20	--	45.8	<40	<10
	100891	--	927	--	--	--	--	--	--	--	--	--
W-342	081491	--	870	--	--	--	--	--	--	--	--	--
	082291	6.86	944	93.8	9.14	92.5	40.7	77	--	27.3	<40	<10
	100891	--	--	127	7.93	25.2	24.4	17	36	27.7	<40	<10
W-343	082291	6.74	1,160	81.7	5.80	112	67.4	22,600	--	38.9	<40	<10
Pond	070690	7.1	560	--	--	33.2	18.6	1,030	1,720	--	<20	<5
	090590	7.8	540	--	--	25.0	17.2	20	318	--	<40	<10
	050891	--	439	46.4	7.23	18.6	16.3	286	16.3	16.3	71	<10
	052291	--	439	--	--	--	--	--	--	--	--	--
S-16	090590	7.3	270	--	--	16.7	13.2	14	45.0	--	<40	<10
	101091	--	--	47.6	1.40	27.4	17.3	96	54.0	19.5	<40	<10
	050991	--	422	47.3	1.18	27.0	16.7	45	34.0	17.8	<40	<10
S-17	090590	6.6	560	--	--	27.6	21.3	1,620	1,710	--	87	<10
	100891	--	--	101	8.97	50.3	36.0	2,600	126	36.8	<40	<10

Well No.	Date (mmddyy)	Cadmium, dissolved (mg/L)	Copper, dissolved (mg/L)	Lead, dissolved (mg/L)	Cobalt, dissolved (mg/L)	Arsenic, dissolved (mg/L)	Selenium, dissolved (mg/L)	Barium, dissolved (mg/L)	Nickel, dissolved (mg/L)	Silver, dissolved (mg/L)	Sulfate, dissolved (mg/L)
W-508	100891	<2	<10	<10	<10	<5	<5	22	25	<1	12.2
	050891	<2	<10	<10	--	<5	<5	310	25	<1	14.9
W-510	092090	<2	--	<10	--	--	--	--	--	--	19.4
	050991	<2	<10	<10	<10	<5	<5	33	<10	<1	24.6
	100891	<2	19	<10	--	<5	<5	136	<10	<1	26.8
W-511	092090	<2	--	<10	--	--	--	--	--	--	<.2
	092090	<2	--	<10	--	--	--	--	--	--	1.3
	050991	<2	<10	<10	<10	<5	<5	105	<10	<1	2.7
	100891	<2	<10	<10	--	<5	<5	164	<10	<1	4.0
W-325	082291	<2	14	<10	24	<5	<5	58	11	<1	20.2
	100891	<2	11	<10	--	<5	<5	58	<10	<1	18.5
W-326	082291	<2	<10	<10	<10	--	--	--	<10	--	--
	100891	<2	<10	<10	--	<5	<5	310	<10	<1	2.9
W-332	081591	--	--	--	--	--	--	--	--	--	--
	082291	<2	23	<10	<10	<5	<5	24	<10	<1	97.7
	081591	<2	<10	<10	--	<5	<5	49	<10	<1	49.1
W-333	082291	<2	38	<10	<10	<5	<5	7	11	<1	248
W-339	081491	--	--	--	--	--	--	--	--	--	--
W-338	082291	<2	<10	<10	<10	<5	<5	45	14	<1	15.8
	100891	<2	<10	<10	<10	<5	<5	93	21	<1	15.0
W-339	081491	--	--	--	--	--	--	--	--	--	--
	082291	<2	17	<10	<10	<5	<5	36	<10	<1	23.1
	100891	<2	<10	<10	<10	<5	<5	40	11	<1	12.3
W-340	081491	--	--	--	--	--	--	--	--	--	--
	082291	<2	<10	<10	<10	<5	<5	68	<10	<1	15.7
	100891	<2	<10	<10	<10	<5	<5	46	<10	<1	15.2
W-341	081491	--	--	--	--	--	--	--	--	--	--
	082291	<2	<10	<10	<10	<5	<5	46	<10	<1	21.6
	100891	--	--	--	--	--	--	--	--	--	--
W-342	081491	--	--	--	--	--	--	--	--	--	--
	082291	<2	<10	<10	<10	<5	<5	58	<10	<1	3.9
	100891	<2	<10	<10	<10	<5	<5	189	<10	<1	18.1
W-343	082291	<2	<10	<10	<10	<5	<5	96	26	<1	5.8
Pond	070690	<1	--	<5	--	--	--	--	--	--	17.3
	090590	<2	--	<10	--	--	--	--	--	--	18.3
	050891	<2	<10	<10	<10	<5	<5	650	<10	<1	8.0
	052291	--	--	--	--	--	--	--	--	--	--
S-16	090590	<2	--	<10	--	--	--	--	--	--	12.5
	101091	<2	<10	<10	--	<5	<5	19	<10	<1	12.7
	050991	<2	<10	<10	<10	<5	<5	11	<10	<1	12.4
S-17	090590	<2	--	<10	--	--	--	--	--	--	9.8
	100891	<2	<10	<10	<10	<5	<5	179	10	<1	9.5