HYDROGEOLOGY AND WATER QUALITY AT THE MANAGEMENT SYSTEMS EVALUATION AREA NEAR PIKETON, OHIO

By Martha L. Jagucki, Christopher D. Finton, Abraham E. Springer, and E. Scott Bair

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

Multiply	By	To obtain
inch (in.)	2.54	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
square foot (ft^2)	0.0929	square meter
square mile (mi^2)	2.589	square kilometer
acre	4,047	square meter
inch per hour (in/h)	2.54	centimeter per hour
inch per year (in/yr)	2.54	centimeter per year
foot per day (ft/d)	0.3048	meter per day
foot per mile (ft/mi)	0.1894	meter per kilometer
foot squared per day (ft^2/d)	0.0929	meter squared per day
cubic foot per second (ft^3/s)	0.02832	cubic meter per second
cubic foot per day (ft^3/d)	0.02832	cubic meter per day
gallon (gal)	3.758	liter
gallon per minute (gal/min)	0.06308	liter per second
gallon per day (gal/d)	3.785	liter per day
million gallons per day (Mgal/d)	3.785	million liters per day

Water and air temperatures in degrees Celsius (°C) can be converted to degrees Fahrenheit (°F) by the following equation:

$$^{\circ}F = 1.8\ (^{\circ}C) + 32$$

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 units used in this report: Concentrations of chemical constituents and temperatures of air and water are given in metric units. Concentration is given in milligrams per liter (mg/L); this unit expresses the concentration of chemical constituents in solution as weight (milligrams) of solute per unit volume (liter) of solution (water). For concentrations of dissolved solids in water less than 7,000 mg/L, the numerical values are, for practical purposes, the same as for concentrations in parts per million. In places, concentrations are given in millimoles per liter (mmol/L). Concentration of tritium, a radioisotope of hydrogen, is given in tritium units (TU).

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.

Pore sizes of membrane filters are given in micrometers (µm). Eh is expressed as millivolts (mv).

Farm at The Ohio State University, Columbus, Ohio.

Hydrogeology and Water Quality at the Management Systems Evaluation Area near Piketon, Ohio

By Martha L. Jagucki, Christopher D. Finton[1], Abraham E. Springer[1], *and* E. Scott Bair[1]

Abstract

This report presents the results of a study by the U.S. Geological Survey, The Ohio State University, and the U.S. Department of Agriculture to describe the hydrology, water quality, and geochemical factors controlling water quality at the Ohio Management Systems Evaluation Area (MSEA). The Ohio MSEA is located on a 650-acre farm in the Scioto River Valley in Pike County, south-central Ohio. The farm is underlain by an incised bedrock valley filled with about 70 feet of sand and gravel outwash deposits that are covered by a veneer of silty clay alluvium and silty loam and sandy loam soils. Outwash sediments are composed predominantly of dolomite, quartz, and calcite, and have a median organic carbon concentration of 0.39 weight percent. Horizontal hydraulic conductivity of the sediments based on results of multiple-well aquifer tests ranges from 400 to 560 feet per day.

Ground-water flow is from east to west-southwest at an average velocity of 1.5 to 2.5 feet per day. Ground water and surface water at the site are highly interconnected. Big Beaver Creek recharges the outwash aquifer along the eastern edge of the study area, and ground water discharges to the Scioto River at the western edge of the study area. High-stage events on the Scioto River cause frequent flow reversals in the aquifer that allow streamwater to travel a maximum observed distance of 190 feet inland. A zone of oxidizing waters (characterized by high dissolved oxygen concentration and Eh) is found in shallow ground water for several hundred feet adjacent to Big Beaver Creek and the Scioto River. This zone of oxidizing ground water is caused by the periodic inflow of surface waters to the aquifer.

A ground-water budget for the study area indicates that the aquifer received 17.7 inches of recharge during water year 1992; of this amount, 72 percent originated as infiltrating precipitation, 28 percent as infiltration of surface water from Big Beaver Creek, and 0.2 percent as leakage from bedrock.

Areal variation in water quality is caused by areal differences in the relative importance of these three recharge sources. The effects of bedrock leakage are evident only in the northeast corner of the study area. Here, deep outwash waters are transitional in composition between the calcium magnesium bicarbonate waters found elsewhere in the outwash aquifer and the calcium sodium chloride waters of the bedrock aquifer. Mixing calculations indicate that these deep outwash waters are composed of as much as 26 percent bedrock water. In the southern part of the MSEA, ground water is diluted by surface water from Big Beaver Creek as it recharges the aquifer through a sand and gravel streambed. At the northeast corner of the MSEA, however, Big Beaver Creek flows across a shale streambed through which no infiltration occurs.

Redox reactions in the outwash aquifer control variations in aquifer chemistry with depth. From the water table to about 40 feet below land surface, oxidizing conditions are characterized by

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the presence of dissolved oxygen and nitrates in ground water, Eh greater than 200 millivolts, ferrihydrite coatings on sediment grains, and the absence of dissolved iron and manganese. From about 40 feet below land surface to the base of the aquifer, reducing conditions are characterized by dissolved oxygen concentrations less than 2 mg/L, Eh less than 200 millivolts, and the presence of dissolved iron and manganese. Denitrification in the reducing zone naturally remediates anthropogenic nitrate contamination of the aquifer while oxidizing pyrite in the aquifer sediments.

INTRODUCTION

Recent reconnaissance studies of ground-water and surface-water quality have created national concern about agricultural nonpoint-source contamination of water resources. The U.S. Environmental Protection Agency's National Pesticide Survey (1990) concluded that nitrate is present at detectable concentrations in water from more than 50 percent of the Nation's drinking-water wells. In addition, the survey estimated that pesticides are present at detectable concentrations in water from 10 percent of all community water-well systems in the United States. The U.S. Geological Survey's Midcontinental Herbicide Initiative, one purpose of which was to investigate the effects of agricultural nonpoint-source contamination on surface water, found detectable concentrations of pesticides in streams throughout the Midwest (Goolsby and Battaglin, 1993).

The Management Systems Evaluation Area (MSEA) project was established as part of the Presidential Initiative on Enhancing Water Quality (Bush, 1989) to address the concern about agricultural nonpoint-source contamination. MSEA's were established in Iowa, Minnesota, Nebraska, Missouri, and Ohio because, combined, the Corn Belt States[2] are responsible for more than 59 percent of the Nation's total pesticide use (Gianessi and Puffer, 1988). The purpose of the MSEA studies is to investigate the effects of different agricultural management systems on ground-water quality in the Corn Belt and to investigate factors affecting the fate and transport of pesticides and nutrients.

Accurate evaluation of the effects of agricultural practices on ground-water quality requires quantification and description of the physical and chemical characteristics of the soil and geologic materials underlying each MSEA site. The hydrologic and water-quality characterization at the Ohio MSEA was part of a collaborative effort between the U.S. Geological Survey (USGS) and The Ohio State University (OSU), Department of Geological Sciences. The overall MSEA program in Ohio is a cooperative research and educational effort of the Ohio Agricultural Research and Development Center and the Ohio Cooperative Extension Service at OSU; the U.S. Department of Agriculture (USDA) Agricultural Research Service, Cooperative State Research Service, and Extension Service; the USGS; and other State and Federal agencies.

Purpose and Scope

The purpose of this report is to (1) describe the geologic and hydrologic setting, including the ground-water budget, at the Ohio MSEA, and (2) characterize ground-water and surface-water quality at the Ohio MSEA. The report also includes a discussion of the geologic and geochemical factors controlling water quality in the subsurface materials. Spatial variability of physical aquifer characteristics and spatial and temporal variability of water-quality characteristics are emphasized.

This report presents findings from data collected during January 1991-September 1992 by the USGS and by personnel of OSU, the Piketon Research and Extension Center (PREC), and the USDA.

Physical Setting

The Ohio MSEA is on the 650-acre Van Meter farm in Pike County, south-central Ohio, approximately 1.2 mi south of Piketon (fig. 1). The farm is in the relatively flat Scioto River Valley, which is approximately 1.5 mi wide in the study area. The valley is bounded on both sides by uplands that rise as much as 600 ft above the valley floor.

Geomorphic features on the valley floor include three terraces and numerous meander scars. The

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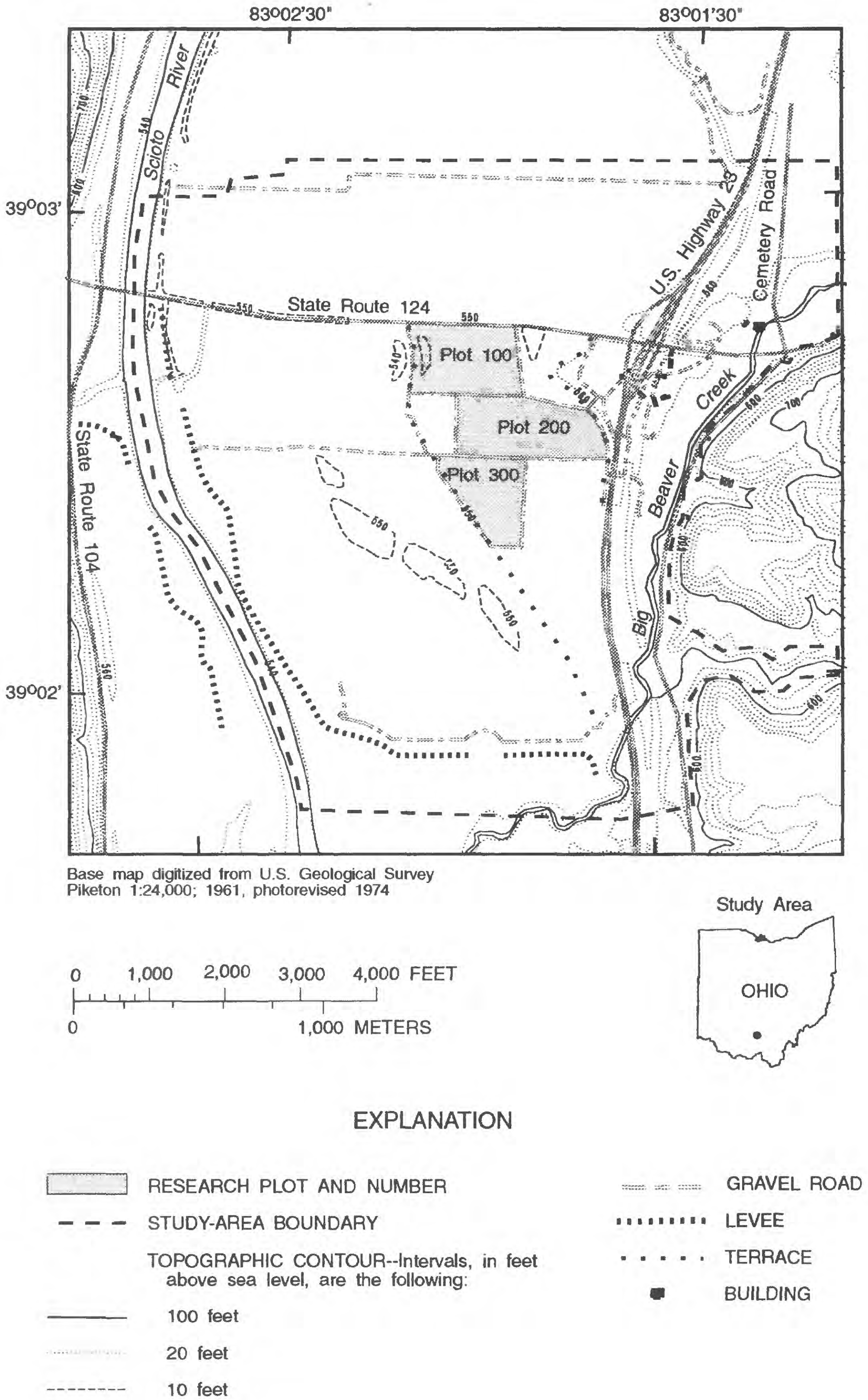
[2] The Corn Belt States are Illinois, Indiana, Iowa, Kansas, Michigan, Minnesota, Missouri, Nebraska, North Dakota, Ohio, South Dakota, and Wisconsin.



Base map digitized from U.S. Geological Survey Piketon 1:24,000; 1961, photorevised 1974

Figure 1. Location of the Ohio Management Systems Evaluation Area.

farmhouse and outbuildings are on the uppermost terrace. The middle and lower terraces are separated by a 5- to 8-ft-high bank. A particularly deep meander scar lies just west of the bank, on the north-central part of the farm. This historic meander is now partly filled with alluvial deposits. The Scioto River and its tributary, Big Beaver Creek, flow from north to south along the western and eastern edges of the valley, respectively. The valley slopes at an average of 6.4 ft/mi (0.12 percent) toward these streams. A wooded riparian wetland parallels the Scioto River and has a maximum width of 500 ft. The woods are bounded to the east by a low, earthen levee.

Approximately 134 acres on the middle terrace have been leased by the MSEA project to implement three different agricultural management practices. (See plots 100, 200, and 300 in fig. 1.) The hydrologic and water-quality data described in this report, however, represent the entire farm. The area between plot 100 and the uppermost terrace, as well as the land east of U.S. Highway 23, is used as pasture. The remainder of the site has been in agricultural production since 1801. Since the early 1960's, crops have consisted predominantly of corn, soybeans, and wheat. Only part of the farm has ever been irrigated, and that irrigation was done only during the drought of 1987-89.

Near the farm buildings are a private water-supply well and septic system. A well field at the southwest corner of the property (fig. 2) provides about 1.9 Mgal/d of sanitary and process water to the U.S. Department of Energy (DOE) facility, which is approximately 1.2 mi southeast of the Van Meter farm.

Soils at the MSEA consist of sandy loam and silt loam through which infiltration readily occurs. No subsurface drainage tile underlies the farm because waterlogging and soil erosion are not problems in this area (Ward and others, 1990). Site soils have been described in detail by Hendershot (1990) and the Soil Conservation Service (unpublished data on file at the Columbus, Ohio, office).

Mean annual precipitation for the study area was 40.14 in. for the period 1951-81, as recorded at the Waverly Waste Water Station of the National Weather Service (NWS) network (Hendershot, 1990, p. 154). Monthly precipitation in Pike County is typically greatest during March through August and least during October through February.

Previous Investigations

Descriptions of the five-state MSEA research project are given by Hatfield and others (1993) and Ward and others (1994). Objectives and methods of investigation specific to the Ohio MSEA are described by Ward and others (1990, 1993).

Geology and water-supply potential of the Scioto River Valley in Pike County have been extensively studied. Aquifer thickness and sediment characteristics west of Piketon and approximately 1.2 mi north of the MSEA site are described by Norris and Fidler (1969); also described are the hydraulic properties of the aquifer as determined by a 9-day multiple-well aquifer test. The logs of test borings and test wells for the above study and the drawdown and recovery data from the aquifer test were published separately by Norris and Fidler in 1967. Norris (1983a, 1983b) also published results of 12 additional multiple-well aquifer tests in Pike County along the Scioto River, 2 of which were done in the current study area. Norris (1991) later reinterpreted these data by use of time-drawdown methods.

Two reports describe water quality in addition to the hydrology of the area. A map titled "Ground-Water Resources of Pike County" (Raab, 1989) describes well yields and aquifer types in the county and includes chemical analyses of samples from five wells. Nortz (1991) simulated ground-water flow and characterized water quality in an 18-mi^2 area surrounding the Piketon well field. Nortz's study area included the northernmost edge of the MSEA.

Despite all the available hydrologic and water-quality data cited above, not enough detailed information existed about subsurface conditions at the Ohio MSEA to document the effects of agricultural practices on the aquifer. Consequently, MSEA researchers studied several hydrochemical processes in order to understand how the physical characteristics of the site affected the fate and transport of agrochemicals. Salchow and Lal (1993) described the physical and hydrologic properties of soils in the leased MSEA plots. Nitrogen cycling in soils was examined by Dick and Jacinthe (1993), Subler and Chammas (1993), and Subler and others (1993). Workman (1993) studied the persistence of agrochemicals in surface soils. To determine a conservative rate of chemical transport through the unsaturated zone, Springer and others (1993) did a tracer test by means of surface-applied bromide. Biodegradation rates of pesticides in the subsurface were examined by McMahon and

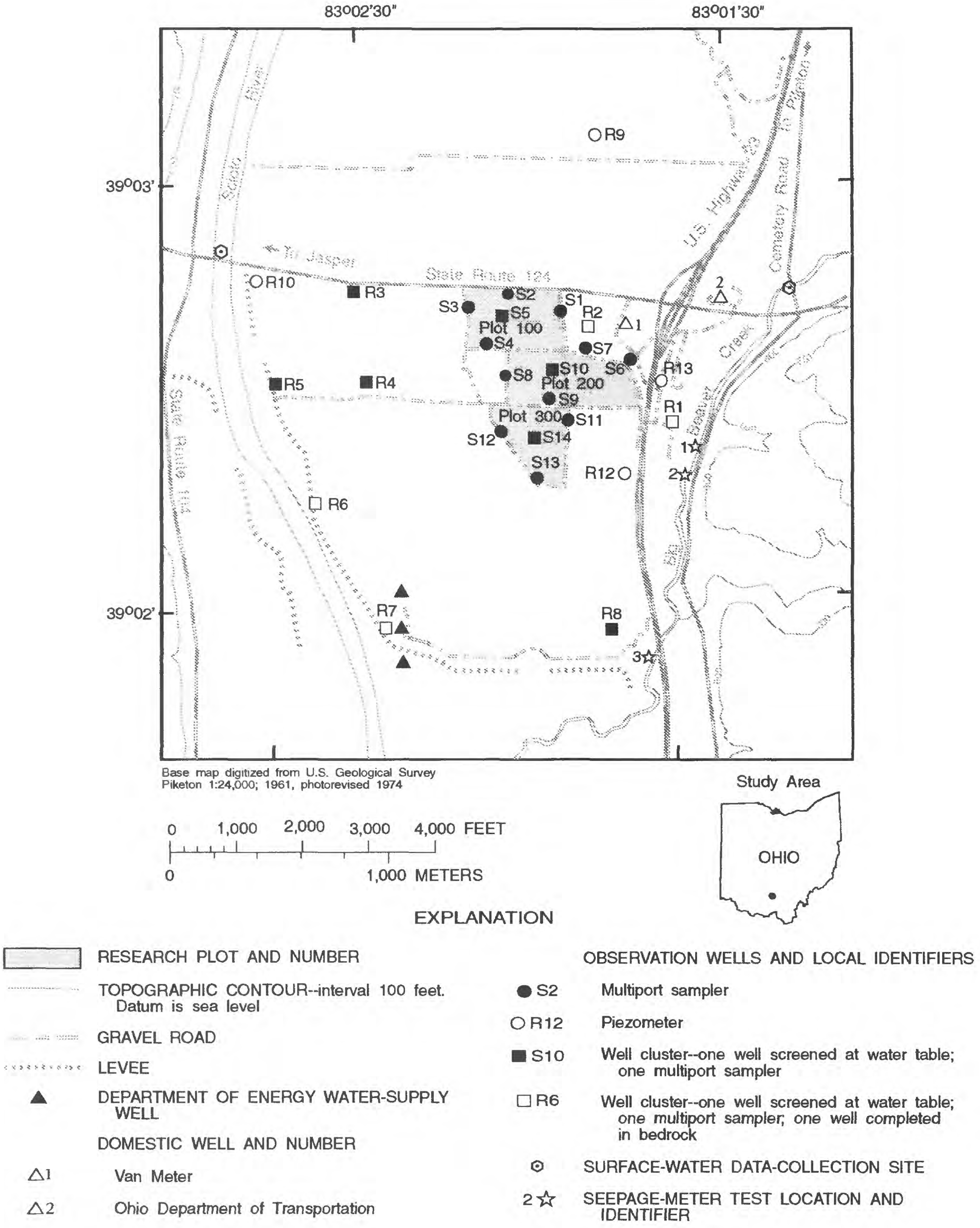


Figure 2. Data-collection network at the Ohio Management Systems Evaluation Area.

others (1992) and Radosevich and others (1993). O'Loughlin and others (1993a, b) also studied the spatial variability of atrazine and alachlor sorption in soils and aquifer sediments. Springer (1994) did a field-scale tracer test to measure transport parameters for reactive and nonreactive species in the outwash aquifer.

Acknowledgments

The authors express appreciation to John Van Meter, Madera Brown, and the Ohio Department of Transportation for allowing access to their wells for water-level measurement. The authors also gratefully acknowledge the assistance of Tom Kramer of Martin Marietta, for providing pumping-rate information for the DOE wells; and Bill Sheppard, for his drill-rig-towing services.

METHODS OF INVESTIGATION

Precipitation Measurement

Historic mean monthly precipitation and temperature for the area were obtained from the Waverly Waste Water Station, which is operated by the NWS and is about 6 mi north of the MSEA. Daily and monthly precipitation data were obtained from the NWS gage in Piketon, 1.2 mi north of the study site. The gage in Piketon has been measuring precipitation since 1942.

Discharge and Stage Measurement

Surface-water data were collected from two locations at the MSEA (fig. 2). Data on the Scioto River were collected from the SR 124 bridge over the Scioto River, 2.7 river mi southwest of Piketon and 0.1 mi east of Jasper. Data on Big Beaver Creek were collected from the bridge where Cemetery Road crosses Big Beaver Creek, approximately 0.3 mi east of the intersection of U.S. Highway 23 and SR 124, 1.4 mi south of Piketon and 2.9 mi upstream from the Scioto River.

Discharge data were collected three times (in April, June, and November 1991) at or near the above locations, according to methods described by Buchanan and Somers (1969). Beginning in April 1992, stream stage was measured daily on the Scioto River and Big Beaver Creek by PREC and USDA personnel. Stage measurements were made from surveyed reference marks on the bridges mentioned above.

Because stage data for the Scioto River measured by the NWS at its wire-weight gage in Piketon, Ohio, are more complete than stage data collected onsite, hydrographs of the Scioto River shown in this report are based on NWS data. Scioto River altitudes measured at the MSEA site were used to determine an average gradient correction factor of 3.71 ft between the NWS station in Piketon and the SR 124 bridge at the Ohio MSEA. Stream altitude adjacent to well cluster R6 was surveyed in December 1992 to obtain a gradient-correction factor of 1.39 ft between the surveyed location and the Scioto River at the SR 124 bridge. Gradient-correction factors adjacent to wells R5 and R7 were interpolated from the known stream-gradient data at the SR 124 bridge and adjacent to well R6.

Measurement of Streambed Hydraulic Conductivity

Hydraulic conductivity (K) of the Big Beaver Creek streambed was determined by seepage-meter tests at three locations (fig. 2) in July 1991 and August 1992. Seepage meters were designed according to specifications given by Lee and Cherry (1978), and details of the test methods used at the MSEA are described by Finton (1994). Adjacent to the seepage meter at each test location, a piezometer was driven approximately 3 ft into the streambed. Stream stage at each location and head in the piezometers were used to determine the vertical hydraulic gradient across the streambed.

The data from the seepage-meter tests were substituted into Darcy's equation,

$$Q = KA(dh/dl), \tag{1}$$

where Q is the volume of water discharged into or from the seepage meter, A is the area of streambed covered by the seepage meter, and dh/dl is the measured vertical hydraulic gradient (Darcy, 1856). The equation was solved for K to generate a vertical streambed K for each test location.

Well Installation

Figure 2 shows the location of the 2 domestic wells, 22 well clusters and (or) multiport samplers, and 4 piezometers used for data collection. The two domestic wells were installed in the 1950's for water supply; the other 37 wells were installed in clusters from January through March 1991. The four piezometers were installed after March 1991 to obtain additional water-level information. Well-construction details for each location are summarized in table 1 (at back of report).

One multiport well was completed in the outwash deposits at each of the 22 well-cluster locations to enable the collection of water-quality samples and water-level data at discrete depths. At 11 locations, a water-table well also was installed in the outwash, adjacent to the multiport well, to enable aquifer testing and instrumentation. Wells were completed in the bedrock at four of the clusters to allow investigation of bedrock-outwash relations. All boreholes in the cluster locations were drilled by cable-tool methods because of the noncohesive nature of the aquifer sediments. Water obtained from the Van Meter domestic well (fig. 2) was the only drilling fluid added to the boreholes. The piezometers were installed in boreholes drilled by means of a hollow-stem auger. All drilling tools were cleaned between cluster locations to avoid cross-contamination of potentially pesticide-containing sediments.

A sample well-construction log of each of the three well types (bedrock, multiport, and water table) is shown in figure 3. The annular space opposite the screened zones of all wells was packed with pure quartz sand, and a fine quartz sand of a smaller diameter than the surrounding aquifer material was added as a permeability boundary above and (or) between the screened zone(s). Permeability boundaries were used instead of bentonite seals because of the absence of confining units in the aquifer and the tendency for organic compounds to adsorb to bentonite and potentially alter the ground-water chemistry. Aquifer materials were allowed to collapse around the casing and, at a minimum, the uppermost 5 ft of each borehole was grouted with a mixture of cement and 2 percent bentonite. A cement seal was added at land surface. Polyvinyl chloride (PVC) casing, when used, was flush threaded.

Each multiport well was constructed of a telescoping 2.5- to 3.0-in.[3]-diameter PVC casing string with sampling and water-level-measurement ports at approximately 14, 18, 22, and 26 ft below land surface. (For specific port depths, see table 1.) Ten wells had additional ports at approximately 40 and 60 ft below land surface. Each port consisted of a 3/8-in.-diameter, 75-μm stainless-steel-mesh filter that is attached to a hollow stainless steel stem inside the PVC casing. Polyethylene tubing (3/8 in.) was clamped to the stainless steel stem and rises to land surface inside the sealed casing string. Each port screen was wrapped with filter cloth to keep silt from entering the opening. Sampling ports were equipped with a dedicated nitrogen-drive system consisting of two stainless steel, one-way check valves attached directly to the port stem.

The 11 water-table wells were constructed with 6-in. PVC casing and completed in the outwash aquifer. A 20-ft-long, PVC, 0.10-in. slotted well screen was positioned to bracket the highest and lowest expected water-table altitudes in each well.

The four bedrock wells were constructed by driving 8-in. steel casing through the outwash sediments and into competent bedrock and then drilling another 15 ft into bedrock. A 4-in., 10-ft-long, 0.10-in. slotted PVC screen was then placed in the bottom of the borehole (in the upper zone of the bedrock) and was sandpacked. A 4-in. PVC casing was installed inside the steel casing, and the annular space between the two casings was grouted to the surface with a mixture of cement and 2 percent bentonite.

The four piezometers were constructed with 0.5-in. PVC casing. Each piezometer had a 5-ft-long, 0.2-in. slotted screen. Screens were emplaced below the lowest expected altitude of the water table.

Water-Level Measurement and Well Instrumentation

The 11 water-table wells and 4 bedrock wells were instrumented with shaft encoders and electronic data loggers that recorded hourly water levels. Specific conductance and temperature also were recorded hourly in six of the water-table wells (table 1). Changes in water level, specific conductance, and temperature in the water-table wells were used to identify episodes of recharge to the outwash aquifer (hereafter called recharge events). These data were also

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[3] All pipe, tubing, and split-spoon sampler sizes mentioned in this report represent nominal inside diameters.

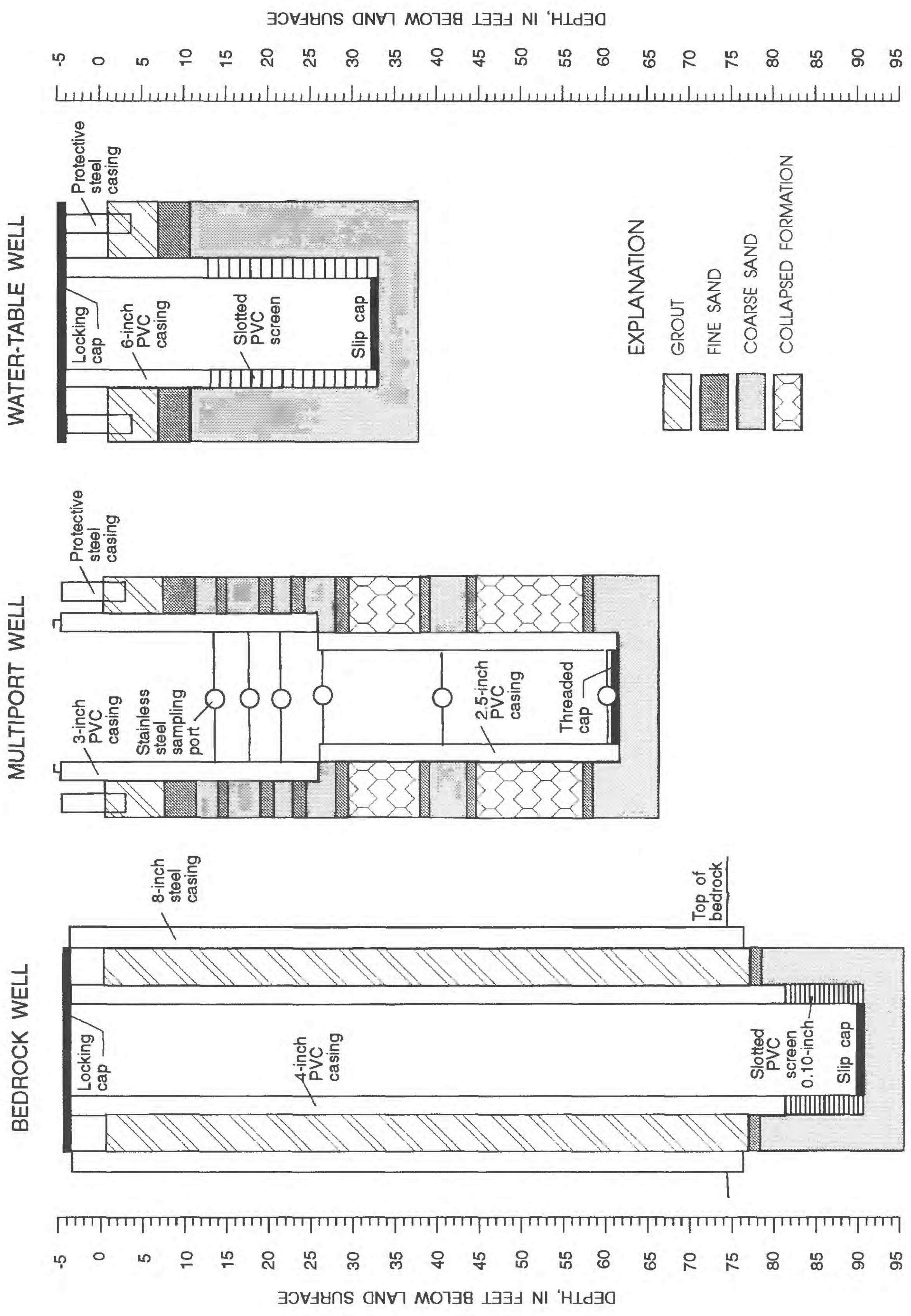

Figure 3. Diagram of generalized well construction, Ohio Management Systems Evaluation Area.

collected so that hydrographs from paired water-table and bedrock wells could be compared and that interaction of the bedrock and outwash aquifers could be qualitatively defined.

To define vertical- and horizontal-flow gradients, investigators measured water levels in the 22 multiport wells at least once a month beginning in April 1991. Water levels in the two domestic wells were measured monthly beginning in August 1991. Monthly measurement of water levels in the four piezometers began in June 1992. Measurements were made by use of an electric tape; measurement error was ±0.03 ft.

Sediment Sampling and Analysis

Subsurface sediment samples for evaluating the spatial variability of physical aquifer characteristics were obtained at the time of well installation by use of a split-spoon sampler. Depending on sediment grain size, a split-spoon sampler of appropriate diameter (4-in. diameter by 3-ft long or a 2-in. diameter by 2-ft long) was used for optimal sample recovery. Samples were collected at a maximum interval of every 5 ft in the deepest borehole at each well-cluster location and were logged by an onsite geologist. If no major stratigraphic changes were observed within the sample, the length of the sample was composited; approximately 500 g of the composited sediment was placed in a plastic bag for analysis of grain-size distribution, and an additional 50 g was collected in an opaque polyethylene jar for determination of total carbon and organic carbon concentration. The organic carbon samples were refrigerated until time of analysis. If two or more sediment types were collected within a split-spoon sample, separate grain-size distribution and organic carbon analyses were done for each sediment type.

Grain-size samples were analyzed by dry-sieve methods at the Columbus, Ohio, office of the USGS according to methods described by W.S. Tyler, Inc. (1976) and Guy (1969). Sieve sizes were 64, 32, 16, 8, 4, 2, 1.4, 1.0, 0.5, 0.25, 0.125, and 0.063 mm. Clay, clayey silt, and bedrock samples were not suitable for analysis by this method.

Vertical variations in K were estimated by means of grain-size-distribution data. Estimation methods were developed by Hazen (1893) and by Norris and Fidler (1965). The Hazen method and the Norris and Fidler method are empirical relations based on characteristics taken from grain-size distribution curves. The Hazen equation (eq. 2) is based on the measurement of effective grain size:

$$K = C(D_{10})^2, \qquad (2)$$

where K is hydraulic conductivity (vectorless, in centimeters per second); D_{10} is the effective grain size (in centimeters), defined as the particle size at which 10 percent of the sample, by weight, is finer; and C is a coefficient (with units of 1/(cm)(s)) based on observed grain-size distribution. According to Fetter (1988, p. 81-82), C for very fine to fine, poorly sorted sand is 40 to 80; C for medium, well- sorted sand is 80 to 120; C for coarse, poorly sorted sand also is 80 to 120; and C for coarse, well- sorted sand is 120 to 150. For purposes of comparison in this report, K's calculated by use of the Hazen method were converted to units of feet per day.

The Norris and Fidler equation (eq. 3) is based on the determination of the uniformity coefficient obtained from grain-size distribution data:

$$K = CU, \qquad (3)$$

where K is hydraulic conductivity (vectorless, in gallons per day per square foot); C is a constant ranging from 250 to 275 (250 was used for this study); and U is the uniformity coefficient, defined as D_{60}/D_{10}, or the ratio of the particle size (in millimeters) at which 60 percent of the sample (by weight) is finer to the effective grain size (in millimeters).

Results of equations 2 and 3 can be affected by sampling bias. Use of a split-spoon sampler that is smaller in diameter than gravel clasts in the formation will exclude gravel from the sample, and the Hazen method of calculation will give artificially low K's. Alternatively, loss of fines from the split-spoon sampler will produce artificially high K's if the Hazen method is used. The effects of sampling bias on K's calculated by use of the Norris and Fidler method vary with the degree of sorting of the sediment sample.

Carbon analyses were done at the Soil Characterization Laboratory, OSU Department of Agronomy, Columbus, Ohio. The total carbon concentration was determined by the dry combustion method, as specified by the U.S. Soil Conservation Service (1967) and by Post (1958). The calcium carbonate equivalent (CCE) of inorganic carbon was determined by gasometric evolution of carbon dioxide (CO_2), and organic carbon was estimated as the difference between the total carbon and the inorganic carbon by use of the following equation:

$$OC = TC - [(CCE \times 0.12) - 0.34], \quad (4)$$

where 0.12 is the gram atomic weight ratio of carbon to calcium carbonate and 0.34 is (CCE x 0.12) of an acid-washed sea-sand blank.

Water-Quality Sampling and Analysis

Water-quality data were used, in conjunction with other ground-water and surface-water data, to conceptualize the flow system, to identify anthropogenic inputs to the system, and to evaluate geochemical controls on the system. A total of 32 ground-water samples were collected by USGS personnel[4] from 10 well-cluster locations 3 times during the 1991 growing season (April, preplanting; June, postplanting and agrochemical application; November, post-harvest). Three zones within the outwash aquifer were sampled from the multiport wells. "Deep" samples were collected from approximately 60 ft below land surface (port 1), "intermediate" samples were obtained from approximately 40 ft below land surface (port 2), and "shallow" samples were collected from the uppermost saturated port (port 3, 4, 5, or 6). Bedrock wells also were sampled where present. All samples were analyzed for specific conductance, pH, temperature, dissolved oxygen, alkalinity, major ions, iron, manganese, nutrients, organic carbon, and selected herbicides. (Redox potential was measured by USGS personnel during the November sampling only, by use of a platinum electrode.) Approximately 20 percent of the samples were also analyzed for tritium.

Ground-water samples were collected from the multiport wells by use of a peristaltic pump, with dedicated C-flex[5] tubing attached to the polyethylene riser on each port to eliminate potential cross-contamination of samples. When the water table fell to more than 30 ft below land surface, a dedicated nitrogen-lift pump was used to collect samples. Ground-water samples were collected from bedrock wells by use of a submersible pump. Samples were not collected until at least three well volumes had been purged from each well and specific conductance, dissolved oxygen, water temperature, and pH had stabilized. Stability was reached when three successive measurements made at intervals of 5 minutes or more differed by less than the following amounts: specific conductance, 1 percent; pH, 0.1 unit; temperature, 0.1°C; dissolved oxygen, 0.1 mg/L; and redox potential (when measured), 10 mv. These ground-water characteristics were monitored in an inline flow chamber. Samples were collected from multiport wells at a flow rate of approximately 0.09 gal/min and from bedrock wells at a flow rate of less than 1 gal/min.

Samples to be analyzed for dissolved constituents were pumped through an inline, 0.45-µm filter before they were discharged into the sample bottle. Samples for analysis of metals were preserved with nitric acid to a pH less than 2 and were chilled to 4°C before being transported to the lab. Nutrient samples were preserved with mercuric chloride to prevent microbial degradation of the sample, and alkalinity titrations were done in the field to minimize evolution of CO_2 before sample analysis. Replicate samples and equipment blanks were obtained at a rate of 5 percent for quality control. Tritium analyses were done by the USGS low-level tritium laboratory in Reston, Va. All other samples were analyzed by the USGS National Water Quality Laboratory in Denver, Colo.

Surface-water samples were collected immediately after measurement of stream discharge by use of the equal discharge increment method described by Guy and Norman (1970, p. 31-32). The same analytes and sample-handling, preservation, and analytical methods that were used for ground-water samples also were used for surface-water samples.

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[4] The water-quality data presented in this report represent, for the most part, data collected and analyzed by USGS personnel. Additional ground-water-quality samples have been collected every 2 to 4 weeks since April 1991 by OSU, PREC, and USDA personnel and are handled and analyzed according to methods described by Beak and others (1993) and Springer (1993). These additional water-quality data are unpublished but are on file at the Environmental Quality Laboratory at the OSU School of Natural Resources in Columbus, Ohio. The unpublished data were used to supplement USGS-determined redox potentials for April and June and were also used as additional data points in figures showing contoured constituent concentrations that are presented in the "Water Quality" section of this report.

[5] The use of brand, trade, or firm names in this report is for identification purposes only and does not constitute endorsement by the U.S. Geological Survey.

HYDROGEOLOGY

The hydrogeology of the study area was characterized by means of (1) a literature review of the geologic materials and geomorphic history of the area and (2) an analysis of spatial variations in grain-size distribution, hydraulic conductivity, mineralogy, and organic carbon concentration of the unconsolidated sediments at the MSEA. A conceptual model of the ground-water-flow system was developed from analysis of ground-water levels and comparison of ground-water levels to surface-water stage. In addition, a ground-water budget was developed to help determine the sources of recharge to the unconsolidated sediments and the relative magnitudes of these sources.

Geologic Setting

Bedrock Hydrostratigraphy

The bedrock in the study area was deposited in a shallow, nearshore marine environment during the Devonian and Mississippian Periods. Bedrock units in southern Ohio typically dip and thicken to the southeast toward the Appalachian Basin. A generalized stratigraphic column is shown in table 2.

The lowermost unit of interest in this study is the Cleveland Member of the Ohio Shale. The Ohio Shale is of Devonian age and is approximately 300 ft thick in Pike County (Lamborn and others, 1938, p. 35); it thickens to the east by about 20 to 30 ft/mi (Stout and others, 1943, p. 131). The altitude of the top of the Ohio Shale along the east valley wall at the MSEA is about 520 ft. The Cleveland Member, the uppermost unit of the Ohio Shale, is typically black to dark bluish gray in the study area, is approximately 60 ft thick (Honeycutt, 1982), and is arenaceous, calcareous, and fissile. The Cleveland Member has a high organic-matter concentration, which was evident by an oily sheen on shale cuttings brought up during drilling at the Ohio MSEA. Pyrite also is a common accessory mineral in the formation (Lamborn and others, 1938, p. 36). The shale yields very little water except along joints and bedding planes (Stout and others, 1943, p. 131). Wells completed in the Cleveland Member at the MSEA yield, at best, only about 1 gal/min, probably from intersecting joints and fractures in the shale.

The Bedford Shale of Mississippian age (table 2) averages 100 ft in thickness and conformably overlies the Ohio Shale (Law Engineering Testing Company, 1978). The top of the Bedford Shale is at an altitude of about 620 ft on the east valley wall, or about 60 ft above Big Beaver Creek at the northeast corner of the MSEA. Previous studies (Geraghty and Miller, Inc., 1990; Stout and others, 1943; Lamborn

Table 2. Generalized stratigraphic column of bedrock near the Ohio Management Systems Evaluation Area

System	Formation	Member	Description
Mississippian	Cuyahoga Formation	Portsmouth Shale	Laminated, siliceous, blue-gray shales, interbedded with thin sandstone lenses. 70 to 120 feet thick.
		Buena Vista Sandstone	Hard, fine-grained, bluish-gray to yellow sandstone. Beds are 1 to 2 feet thick, interbedded with shales of varying thickness. 60 to 100 feet thick.
		Henley Shale	Gray shale with thin sandstone lenses. 30 to 50 feet thick.
	Sunbury Shale		Fissile, black shale, rich in organics. 0 to 20 feet thick. Not a source of water.
	Berea Sandstone		Hard, light-gray sandstone with siliceous, gray shale partings. 30 feet thick. Wells yield 1 gal/min.
	Bedford Shale		Hard, arenaceous and calcareous, gray to brown shale, interbedded with thin, ripple-marked, gray sandstone and siltstone. 100 feet thick. Wells yield 0.5 gal/min.
Devonian	Ohio Shale	Cleveland	Fissile, calcareous, black to dark bluish-gray shale, high in organic content. Pyrite is a common impurity. 60 feet thick. Wells yield 0.0006 gal/min to 0.8 gal/min.

and others, 1938) indicate that, in southern Ohio, the Bedford Shale consists of hard, gray to brown arenaceous and calcareous shale with thin interbeds of gray, ripple-marked sandstone and siltstone. Law Engineering Testing Company (1978) estimates that 33 to 50 percent of the formation is sandstone in the study area, although these sandstone units are so tight and dense that they supply little or no water (Stout and others, 1943). Sandstone interbeds increase in number upward in the stratigraphic section. Law Engineering Testing Company (1978) noted crude oil saturation of the sandstone units below an altitude of 570 ft in three borings near the study area. The Bedford Shale is used for private water supply by homeowners who live in the uplands surrounding the study area, away from outwash deposits. Typical yields for wells completed in the Bedford Shale are approximately 0.5 gal/min, as determined from drillers' logs.

The top of the Bedford Shale grades into the overlying Berea Sandstone (table 2), which averages 30 ft in thickness in the uplands. The Berea Sandstone is composed of hard, light-gray sandstone with gray, siliceous shale partings (Geraghty and Miller, Inc., 1990). Water yields from the Berea Sandstone are low (Stout and others, 1943).

Overlying the Berea Sandstone is the Sunbury Shale (table 2), which ranges from 0 to 20 ft in thickness in the uplands near the study area. The Sunbury Shale is black, fissile, and rich in organics. Pyrite is also abundant in the Sunbury Shale—especially in the basal 1 to 3 in. of the formation, where a thin zone of sulfide mineralization marks the contact between the Sunbury and the Bedford Shales (Geraghty and Miller, Inc., 1990). The Sunbury Shale is not a source of water for Pike County residents.

The Cuyahoga Formation (table 2), overlies the Sunbury Shale and is the uppermost bedrock unit in the uplands bordering the study area. It reaches a thickness of 300 ft between Chillicothe and Waverly (Lamborn and others, 1938). The Cuyahoga Formation consists of gray, thinly bedded shale with thin lenses of fine-grained sandstone at its base (Geraghty and Miller, Inc., 1990). The sandstone beds increase in abundance and thickness upward in the formation (Stout and others, 1943).

Geomorphic History

Prior to glaciation, the ancestral Portsmouth River flowed northward along a path roughly parallel to that of the present-day Scioto River but at an altitude approximately 150 ft higher than the present-day river valley (Stout and others, 1943, p. 61-63) and along a course about 2 mi east (Geraghty and Miller, Inc., 1990). The Portsmouth River was a short tributary that emptied into the ancestral Teays River at a point 5 mi north of Piketon, near Waverly. When the Teays drainage was disrupted by the Kansan glacial advance, flow was reversed in the former Portsmouth River Valley, and the ancestral Newark River was formed (fig. 4). The Newark River was a major stream that drained a large area of central and southern Ohio and was a tributary to the ancestral Cincinnati River. It generally followed the course of the older stream but cut a new channel west of the Portsmouth River channel near Piketon (Stout and others, 1943, p. 81-82).

Regional uplift occurred after the Newark River was established, causing downcutting of the stream into a deep, narrow valley with steep walls (Stout and others, 1943, p. 78). Most of the bedrock features shown on the bedrock-surface contour map (fig. 5) probably reflect this stage of channel development. The Newark River cut through at least 55 ft of Ohio Shale to an altitude of approximately 465 ft in the study area. The ancestral river flowed near the west valley wall in the study area, as does the present-day Scioto River. The valley floor was broad and relatively flat floored, with very steep sides, especially along the west valley wall. Locally high erosional remnants were located under the current DOE well field and just north of U.S. Highway 124. Very steep channel walls near well R1 may have been caused by the entrance of a tributary to the valley at this point. Evidence of weathering of the valley floor includes areas of soft, gray shale found above competent shale during the drilling operations, as well as discontinuous pockets of bluish-gray to dusky yellowish-brown clay at the top of the shale.

The second (Illinoian) and third (Wisconsinan) glacial advances both halted about 20 mi north of Piketon. As the glaciers melted, outwash filled the Newark River Valley with 100 ft or more of sand, gravel, and silt. The modern Scioto River came into existence during the final retreat of the Wisconsinan glacier and has since removed 20 to 30 ft of the original outwash deposits (Norris and Fidler, 1969; Stout and others, 1943). Original outwash levels are marked by the uppermost terrace at the MSEA (fig. 1).

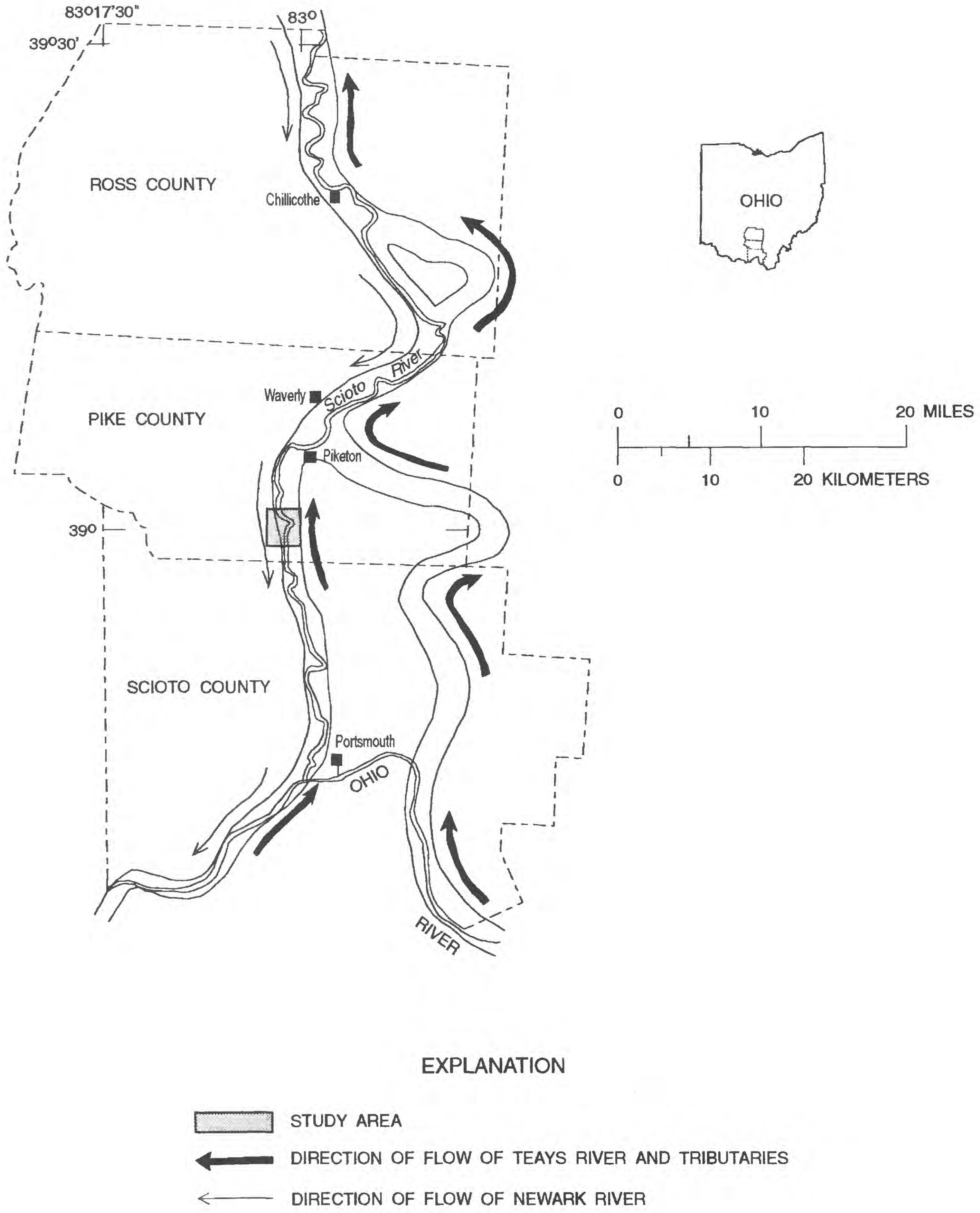
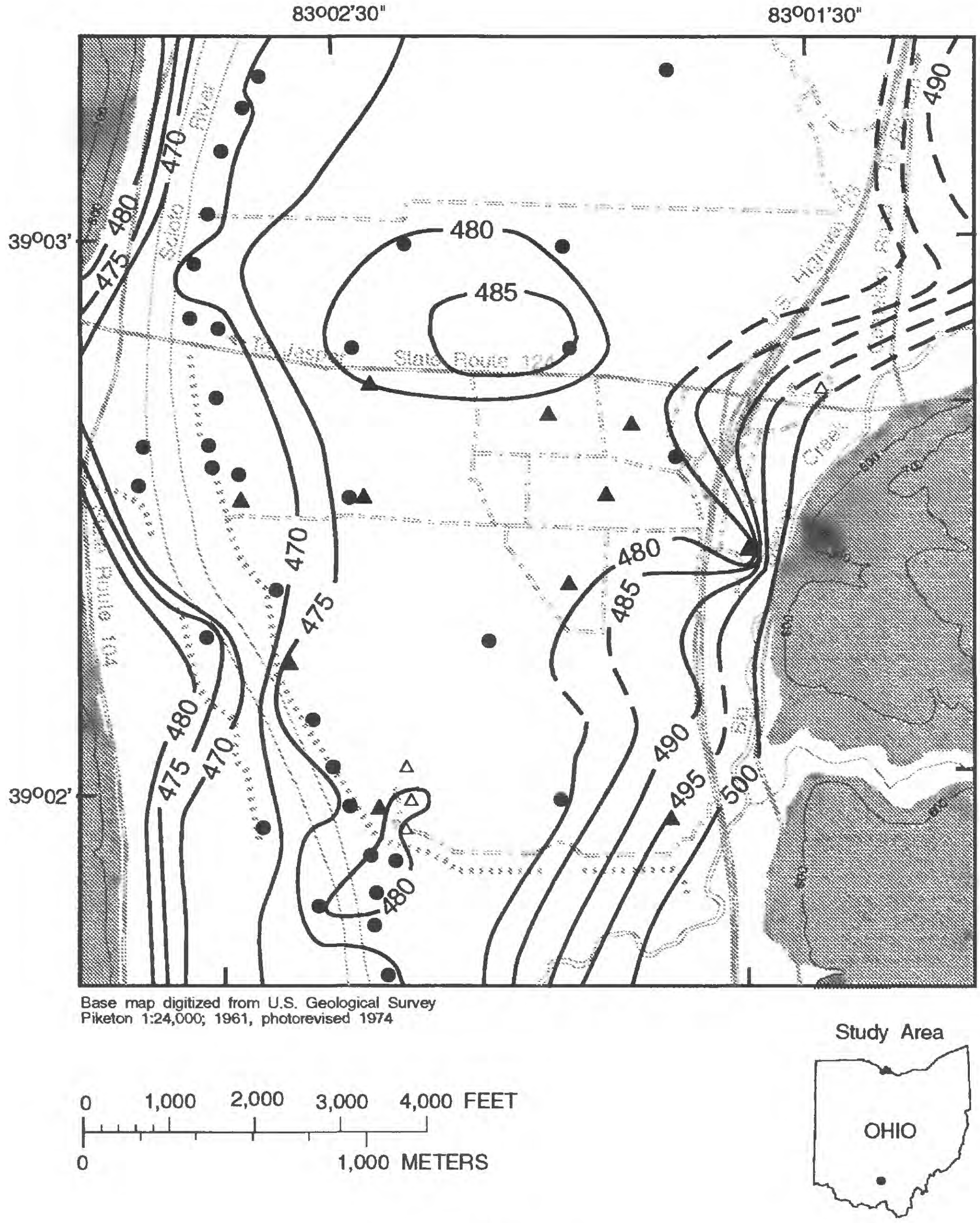


Figure 4. Ancestral Teays and Newark River Valleys in southern Ohio. (Modified from Norris and Fidler, 1969.)



Base map digitized from U.S. Geological Survey
Piketon 1:24,000; 1961, photorevised 1974

EXPLANATION

▨	BEDROCK OUTCROP
—480—	TOP-OF-BEDROCK CONTOUR--Shows altitude of the top of bedrock, in feet above sea level. Dashed where inferred. Contour interval 5 feet
———	TOPOGRAPHIC CONTOUR--Interval 100 feet. Datum is sea level
┄	GRAVEL ROAD
⸾⸾⸾⸾⸾	LEVEE
●	U.S. GEOLOGICAL SURVEY BOREHOLE (1976-77 unpublished data on file at the Columbus, Ohio office of the Water Resources Division)
▲	MANAGEMENT SYSTEMS EVALUATION AREA WELL
△	DOMESTIC OR SANITARY WATER-SUPPLY

Figure 5. Bedrock topography in the Ohio Managment Systems Evaluation Area.

Outwash Aquifer and Alluvial Deposits

The unconsolidated sediments at the site range in total thickness from 61.5 to 83.5 ft. Unconsolidated sediments are thickest at the northwest corner of the site, overlying a deeply incised section of the ancient Newark River Valley. Sediments generally thin to the north-central part of the site and also to the southwest, where bedrock highs underlie the MSEA (fig. 6).

The valley-fill stratigraphy is illustrated by generalized geologic sections (figs. 7-9). Outwash sediments consist predominantly of discontinuous interbeds of sand, gravelly sand, and sandy gravel, indicating deposition in a high-energy environment. Clasts are subrounded, as large as 5 in. in diameter, and composed predominantly of carbonates, siltstone, chert, tillite, granite, and gneiss. Sediments undergo a distinct color change from moderate yellowish-brown near land surface to gray at depth. In many places, the color change is abrupt. The color boundary ranges in depth from 12 to 15 ft below land surface near the east edge of the site to generally between 25 and 45 ft below land surface on the west edge, near the Scioto River.

With the exception of thin, discontinuous clay layers overlying the bedrock surface at the base of the outwash deposits, few clay or silt layers are present in the valley at depths greater than 26 ft below land surface. In fact, a discontinuous silty clay layer was found at depth only at well R2 (fig. 2) at 48-58 ft below land surface. Wood fragments were commonly found at depth in the outwash aquifer.

Recent flooding of the Scioto River and Big Beaver Creek has overlain the outwash deposits with a veneer of alluvium composed of clay and silty clay. This alluvium is of varied thickness, but generally interbeds with sand and gravelly sand to a depth less than 11 ft below land surface. Greater thicknesses of alluvium interbedded with coarse-grained deposits to depths as great as 17 ft below land surface are found near the Scioto River, in old meander scars, and between U.S. Highway 23 and Big Beaver Creek. Of the three leased MSEA test fields, plot 100 has the overall greatest thickness of alluvium, and plot 300 has the thinnest alluvial cover.

Because of its high yields, the outwash aquifer in the Scioto River Valley is widely used as a source of public and private water supply. One of the DOE sanitary supply wells at the southwest corner of the MSEA (fig. 2) yields as much as 1,300 gal/min. Wells in the Scioto River Valley also are used for rural water supply, industrial process water, and aquaculture.

Selected Aquifer Characteristics

Selected aquifer characteristics known to affect the fate and transport of water and agrochemicals were analyzed to determine their spatial variability in the subsurface. Analyses of grain-size distribution, hydraulic conductivity, sediment mineralogy, and organic-carbon concentration are described in the sections that follow.

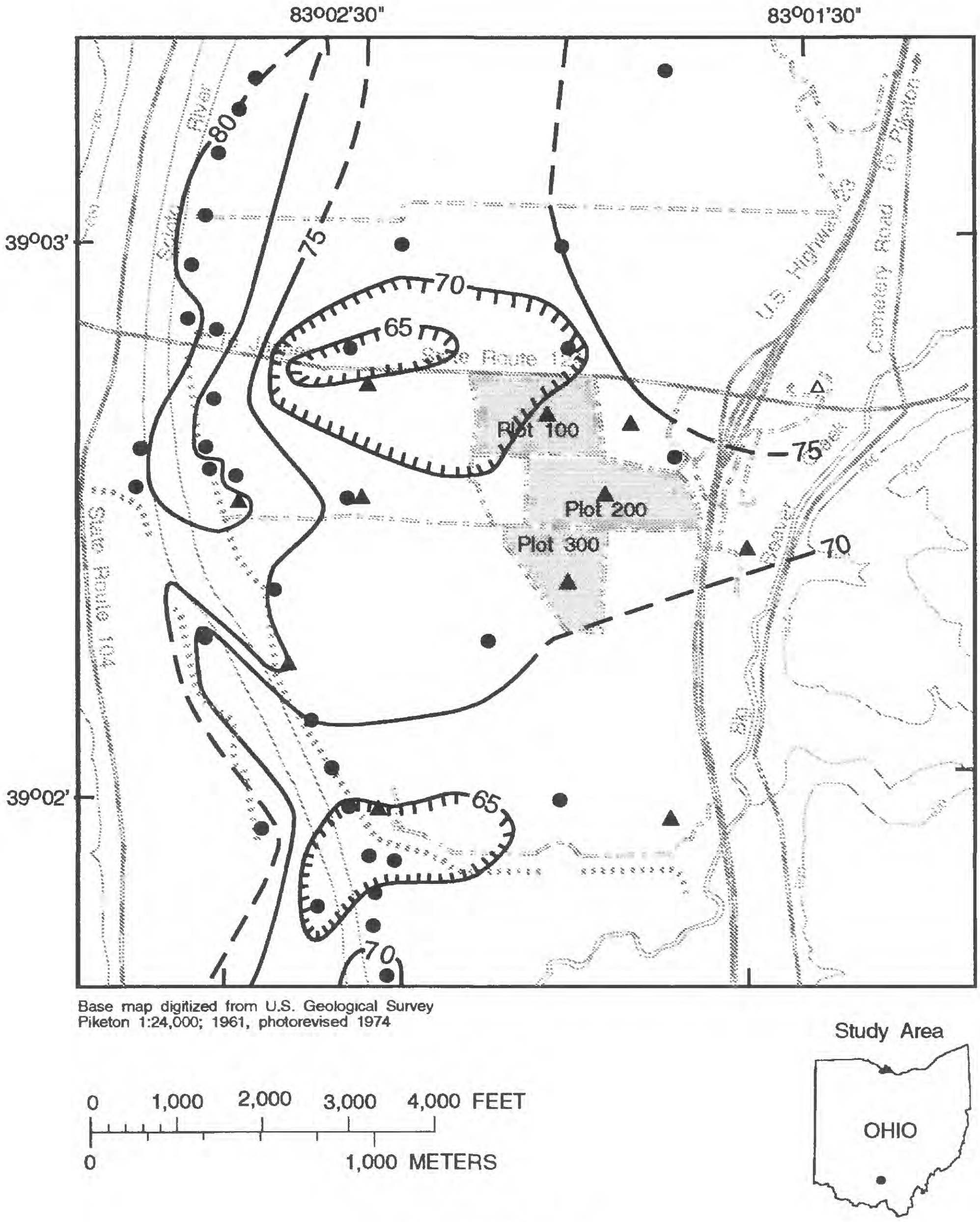
Grain-size Distribution

Knowledge of grain-size distribution enhances stratigraphic definition of the unconsolidated deposits and identifies areas where high content of silt and clay may cause sorption of dissolved organics and chemical reactions with sediments to occur more readily than elsewhere. A total of 98 sediment samples were analyzed for grain-size distribution. A complete listing of grain-size distribution results was published in Shindel and others (1993, p. 437-439). From these results, sediments at the Ohio MSEA were grouped into the following categories: poorly permeable materials (clay and silt); very fine to fine sand; fine to coarse sand with less than 10 percent gravel; gravelly sand (medium to coarse sand with 10 to 50 percent gravel); and sandy gravel (50 percent or more gravel with medium to coarse sand). These sediment groupings were used to delineate the hydrostratigraphic units shown in the geologic sections (figs. 8 and 9).

Clay and silt content of the outwash sediments did not vary greatly. Units identified on well logs and the cross sections as sands or sand and gravel units contained 0.4 to 9.1 percent clay by weight. Alluvium deposits and fine sands contained 8.5 to 20.5 percent silt and clay by weight. Grain-size distribution of very clayey samples could not be determined by use of the dry-sieve methods used in this study.

Hydraulic Conductivity

Hydraulic conductivity (K) is a coefficient that describes the rate at which water can move through a permeable medium. An understanding of spatial variability in K beneath the various farming plots at the MSEA is crucial to the interpretation of agrochemical impacts of farming practices. K affects ground-water-flow rates, as well as ground-water-flow paths and



Base map digitized from U.S. Geological Survey Piketon 1:24,000; 1961, photorevised 1974

EXPLANATION

▨ RESEARCH PLOT AND NUMBER

—70— ISOPACH OF VALLEY-FILL SEDIMENT--Shows thickness of valley-fill sediment. Dashed where inferred. Interval 5 feet. Hachures indicate enclosed area of decreasing thickness

┈┈┈ TOPOGRAPHIC CONTOUR--Interval 100 feet. Datum is sea level

┄ ┄ ┄ GRAVEL ROAD

⸗⸗⸗⸗ LEVEE

● U.S. GEOLOGICAL SURVEY BOREHOLE (1976-77 unpublished data on file at the Columbus, Ohio, office of the Water Resources Division)

▲ MANAGEMENT SYSTEMS EVALUATION AREA WELL

△ DOMESTIC OR SANITARY WATER-SUPPLY WELL

Figure 6. Thickness of valley-fill sediments at the Ohio Management Systems Evaluation Area.

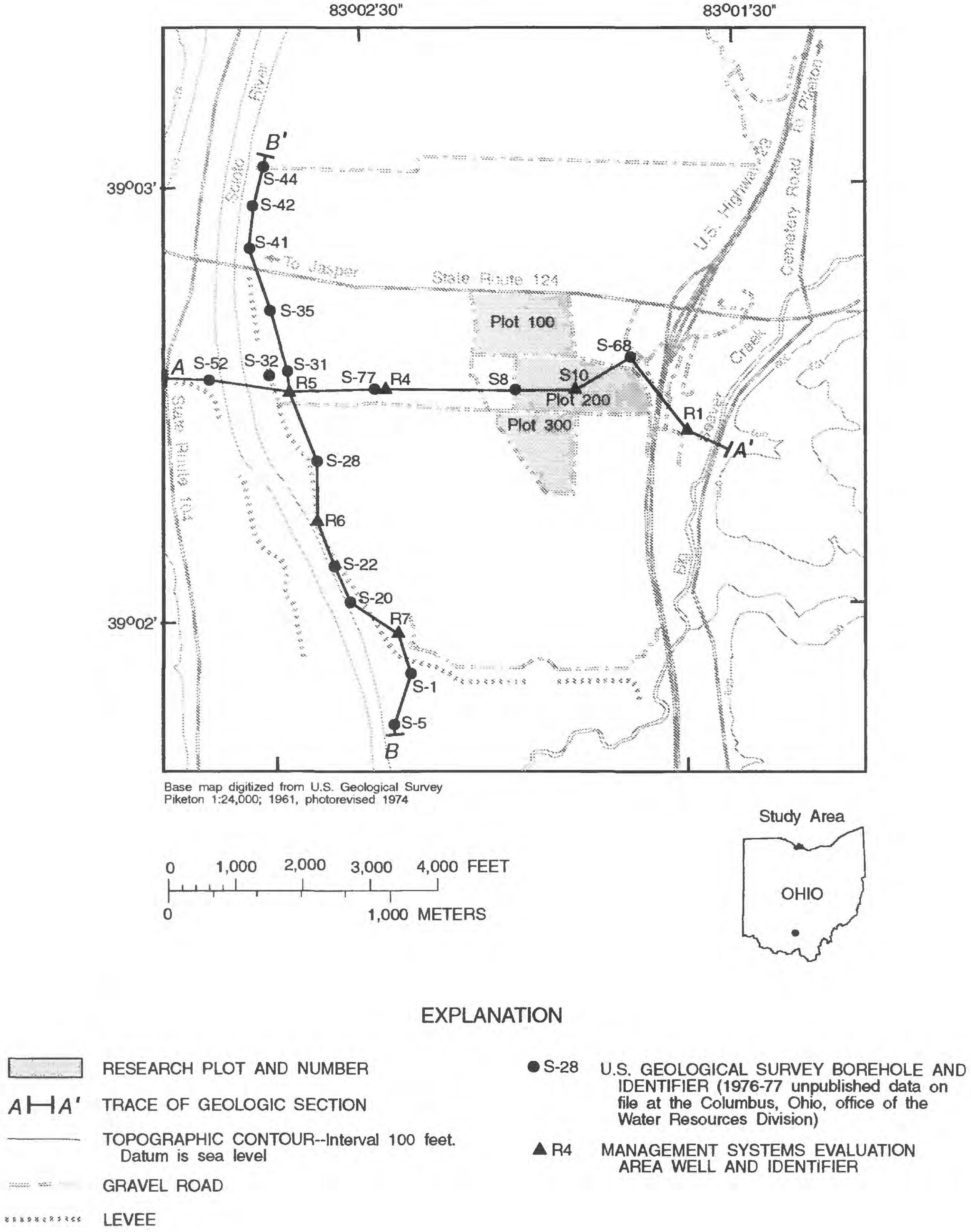
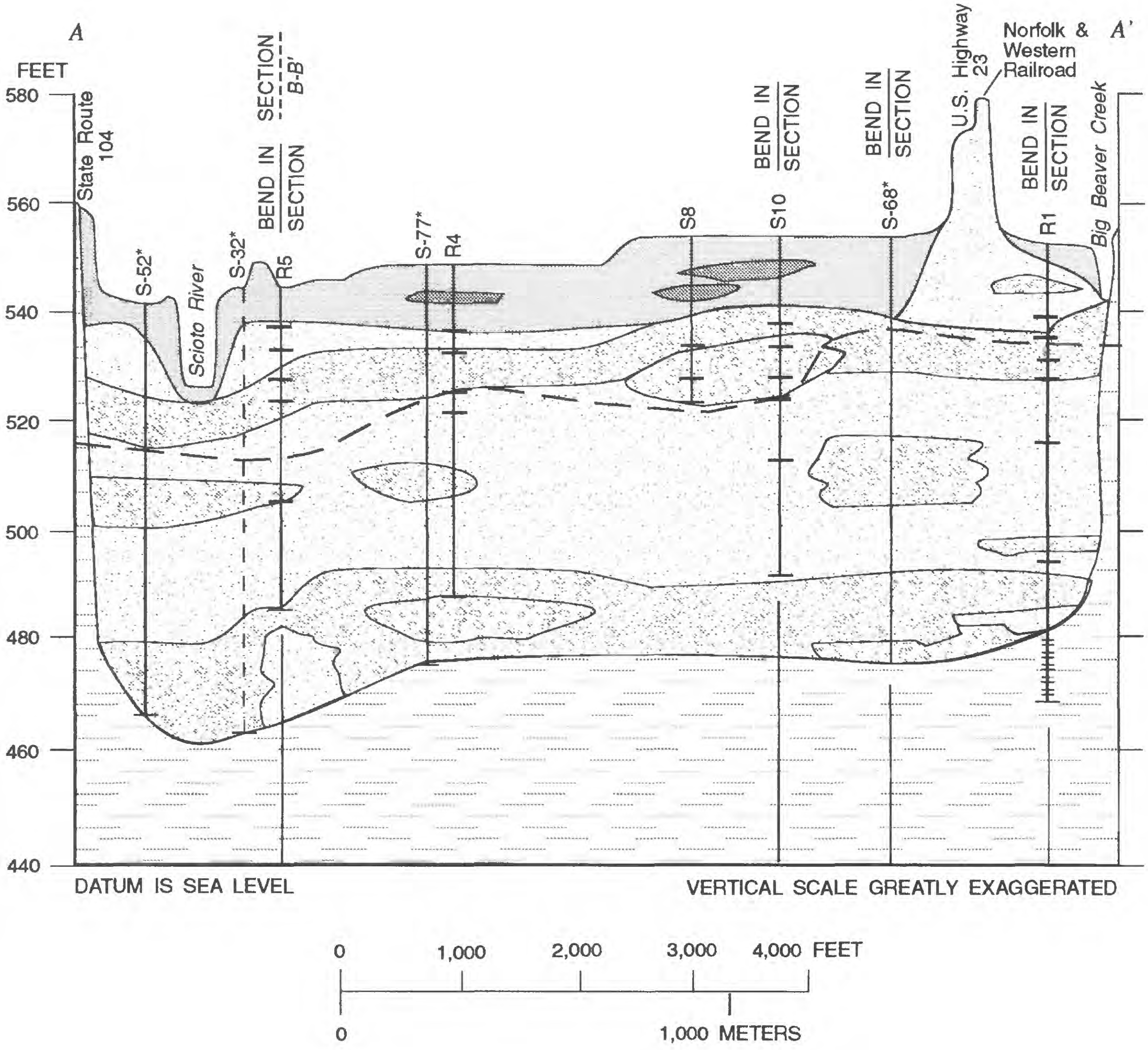


Figure 7. Location of geologic sections and borings, Ohio Management Systems Evaluation Area.



EXPLANATION

▒	POORLY PERMEABLE MATERIALS (CLAY, SILT, AND ALLUVIAL SOIL)
▓	VERY FINE TO FINE SAND
░	FINE TO COARSE SAND--less than 10 percent gravel
░	MEDIUM TO COARSE SAND--10 to 50 percent gravel
░	GRAVEL AND MEDIUM TO COARSE SAND--50 percent or more gravel
┄	SHALE BEDROCK
— —	SEDIMENT--Color change from moderate yellowish brown (above line) to gray (below line)
S8	WELL/BORING LOCATION AND IDENTIFIER--Asterisk indicates unpublished boring (data on file with the U.S. Geological Survey, Columbus, Ohio, office of the Water Resources Division). Dashed well has been projected onto section line
┼	Sampling port
╪	Well screen--For well clusters, screened interval shown is that of the deepest well

Figure 8. Geologic section *A-A'*, Ohio Management Systems Evaluation Area. (Section trace shown on fig. 7.)

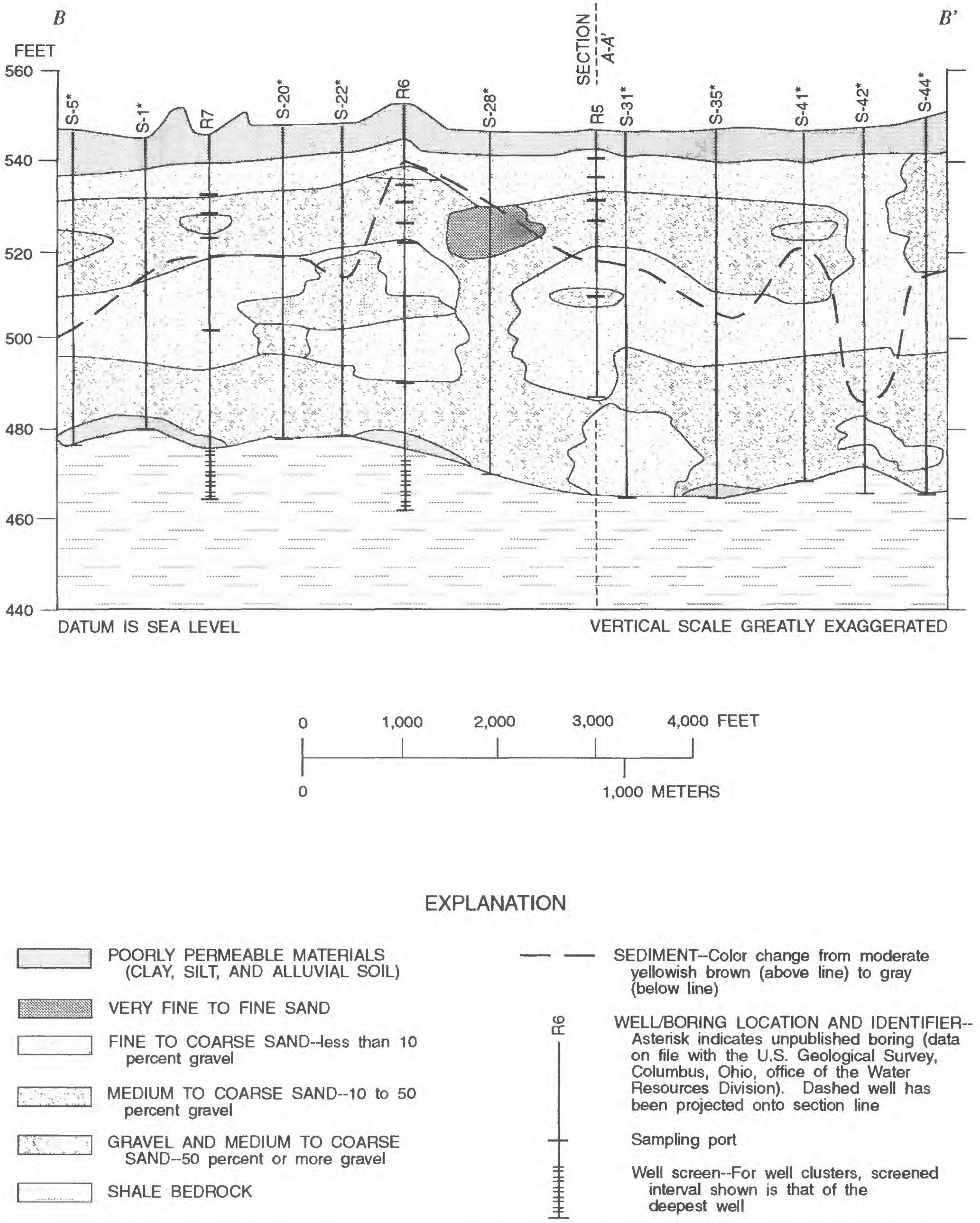


Figure 9. Geologic section *B-B'*, Ohio Management Systems Evaluation Area. (Section trace shown on fig. 7.)

recharge rates. K is generally one of the most sensitive input parameters in a ground-water-flow model (Finton, 1994; Dumouchelle and others, 1993; Nortz, 1991). Spatial variations in K at the Ohio MSEA were evaluated at several different scales by use of the following methods: grain-size distribution formulas, slug tests, specific-capacity tests, and multiple-well aquifer tests.

Grain-Size Distribution Formulas

The methods of Hazen and Norris and Fidler were used to estimate K's for 94 sediment samples. Results of these analyses are contained in Finton (1994). Variations in K with sediment depth and stratigraphic unit at three wells are shown in figure 10. Although the two methods of calculation result in similar vertical trends in K for many of the boreholes, actual K_{Hazen} and $K_{Norris\ and\ Fidler}$ values are poorly correlated (Spearman's rho = 0.292). The reason for this lack of correlation may be that the results derived from the Norris and Fidler equation are much more responsive to poor sorting and gravel content than results from the Hazen equation are.

Results from both methods show that K varies significantly with stratigraphic unit. A Mann-Whitney analysis of variance (ANOVA) was used to compare the median K's calculated for the five sediment types shown in figures 8 and 9. At 95-percent confidence, sand, gravelly sand, and sandy gravel sediments had significantly different K's that increased with gravel content. However, the median K of very fine to fine sand did not differ significantly from that of sand. Mann-Whitney test results were the same regardless of which method was used to derive K. Results for silts and clays could not be tested by statistical methods, owing to insufficient amounts of data.

K's calculated by grain-size-distribution methods are not accurate at a site scale because of sampling-bias effects and because structural features within the sediments that may affect K (such as bedding planes) are ignored by such methods. However, grain-size-distribution methods reveal valuable information about K variability with depth and between stratigraphic units. As figures 8, 9, and 10 all show, relatively high-K, sandy-gravel and gravelly-sand units present from approximately 5 to 20 ft below land surface and near the base of the outwash aquifer are separated by lower-K, gravel-poor sands in the center of the outwash aquifer. Areally, higher K's are expected near SR 124 and also near well S10, owing to the gravel-rich sediments at these locations.

Slug Tests

A total of 39 slug tests were done in the 11 water-table wells at the MSEA (fig. 2) to examine areal variations in horizontal K[6]. Slug tests provide information on the hydraulics of aquifer materials located within a few feet of the well screen. All of the tested wells were identical in construction and were completed at approximately the same depth. Two to five replicate tests were done at each well to ensure validity of the results and to enable an evaluation of variability within replicate measurements. A 4-in. diameter, 5-ft long PVC slug was used in the tests. Water-table response to the withdrawal of the slug was monitored by use of a pressure transducer and was recorded by use of a data logger. Complete descriptions of the slug tests can be found in Finton (1994). Data were interpreted by use of the Bouwer and Rice method (1976), accounting for the well-construction specifications and the partial penetration of the tested wells. The mean K's as determined from replicate slug tests at each well are listed in table 3. The lowest horizontal K was measured at well cluster R1, where silt and clay interbeds intersect the well screen. The highest horizontal K was measured at well S10, which intersects 15 ft of gravelly sand. Coefficients of variance for the replicate tests at a given well ranged from 1 to 13 percent (mean, 5 percent), indicating that precision of the slug-test method was good.

Specific-Capacity Tests

To examine areal variations in horizontal K at an even larger scale, the investigators did three 7-hour specific-capacity tests between December 1991 and July 1992. Tests were done in water-table wells S5, S10, and S14, each of which is near the center of one of the three management-practices plots (fig. 2). Pumping rates used in the tests ranged from approximately 45 to 60 gal/min. Step-drawdown tests were done prior to the specific-capacity tests to evaluate the efficiency of each well. Water-table response in the step-drawdown and the specific-capacity tests was

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[6] All subsequent references to K's determined by means of slug tests, specific-capacity tests, or multiple-well aquifer tests will be horizontal K's unless otherwise specified. K's estimated by grain-size-distribution methods are vectorless.

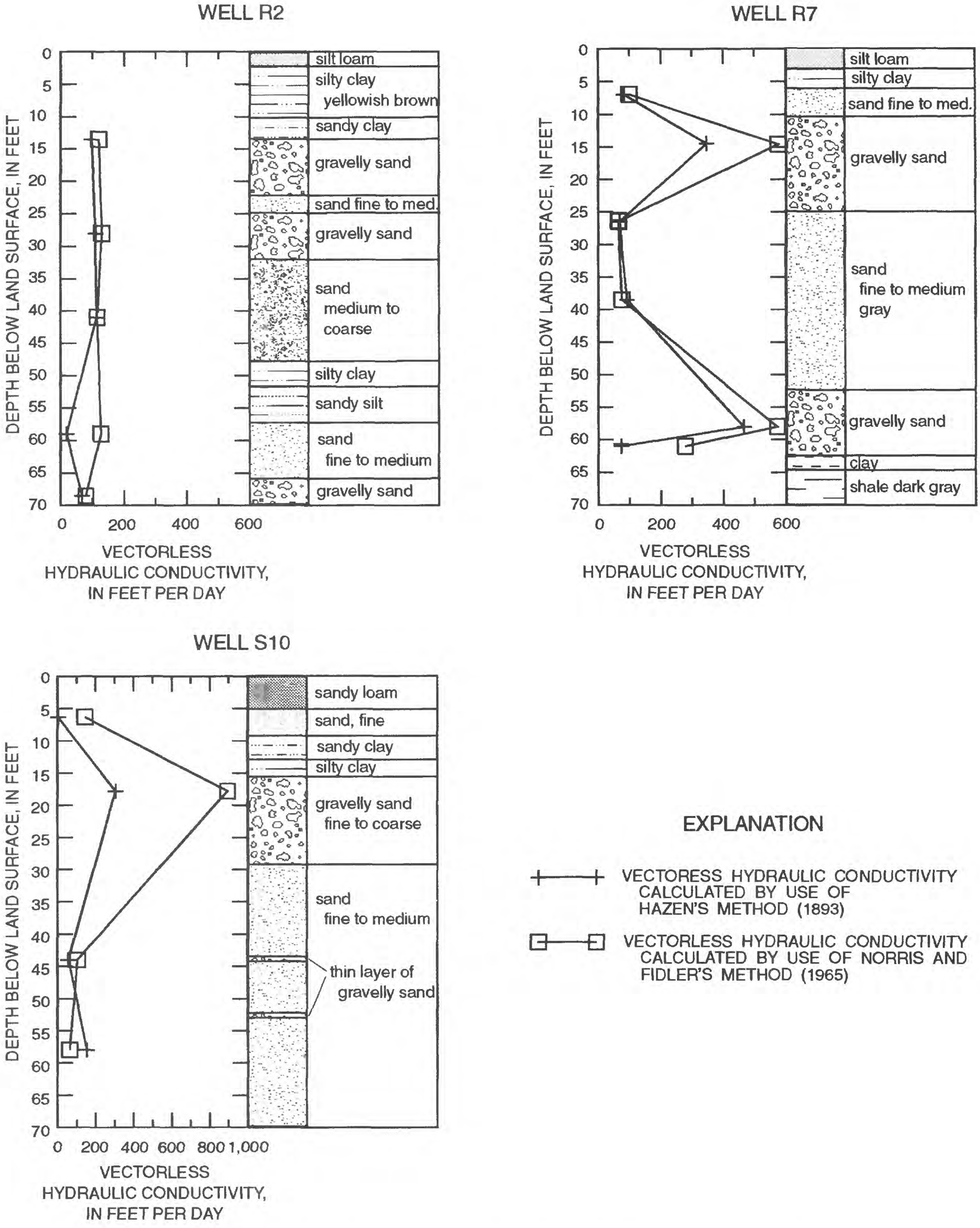

Figure 10. Hydraulic conductivities, Ohio Management Systems Evaluation Area, calculated for various sediment types.

monitored by use of a pressure transducer and recorded by use of a data logger. Complete descriptions of each of the specific-capacity tests can be found in Finton (1994).

Data from the step-drawdown tests were interpreted by means of methods described by Bear (1972). Data from the specific-capacity tests were interpreted by use of a PC-based computer code developed by Bradbury and Rothschild (1985). K's estimated from the specific-capacity-test data ranged from 320 to 815 ft/d and are listed in table 3. The relative ranking of K's determined by means of this method agree with the ranking obtained by use of slug-test methods ($K_{\text{well S10-WT}} > K_{\text{well S5-WT}} > K_{\text{well S14-WT}}$); however, the K's based on specific-capacity tests are approximately twice those obtained by slug-test analysis. Thus, the slug-test results can be used to determine relative areal differences in K across the study area, but the specific-capacity results are more useful for estimating K at the site scale.

Table 3. Horizontal hydraulic conductivity as determined from slug tests, specific-capacity tests, and multiple-well aquifer tests at the Ohio Management Systems Evaluation Area

[Well locations shown in figure 2. Slug-test data analyzed by use of the method described by Bouwer and Rice (1976). Horizontal hydraulic conductivity given for each well is the mean of two to five replicate slug tests. Data from specific-capacity tests analyzed by use of computer code developed by Bradbury and Rothschild (1985). Details of the multiple-well aquifer tests and analysis of the distance-drawdown data by use of the Rorabaugh method (1956) are described by Norris (1983a). Abbreviations: ft/d, feet per day; --, test not done at this well]

Well number	Horizontal hydraulic conductivity (ft/d)		
	Slug test	Specific-capacity test	Multiple-well aquifer test
R1-WT	11	--	--
R2-WT	69	--	--
R3-WT	216	--	--
R4-WT	36	--	--
R5-WT	210	--	401
R6-WT	42	--	--
R7-WT	176	--	555
R8-WT	172	--	--
S5-WT	247	451	--
S10-WT	443	815	--
S14-WT	113	320	--
Mean	158	529	478

Multiple-Well Aquifer Tests

Site-specific K's determined by means of slug-test and specific-capacity methods were compared to published information from multiple-well aquifer tests. Between 1953 and 1979, the USGS did 13 multiple-well aquifer tests at 11 sites in the Scioto River Valley. The tests were done along the east side of a 7-mi reach of the Scioto River, extending south from Piketon. Results of the aquifer tests were included in several published articles (Norris and Fidler, 1967, 1969; Norris 1983a, 1983b). A complete description of each aquifer test can be found in these references. Initial interpretation of the aquifer-test data was done by USGS personnel by plotting distance-drawdown data and by means of an interpretation method developed by Rorabaugh (1956). Later, time-drawdown data were plotted and analyzed by use of the Neuman method (1975) as part of a report on regional aquifer-test analyses prepared for the Ohio MSEA project (Norris, 1991).

The two methods of data analysis produced similar results. The mean horizontal K calculated for the 7-mi reach from distance-drawdown data is 466 ft/d, and the standard deviation is 118 ft/d. (In the time-drawdown analysis, the mean K is 489 ft/d, and the standard deviation is 100 ft/d.) Of the seven sites, only two were on the MSEA. One aquifer-test site was in the southwest corner of the study area, near MSEA well R7 and the sanitary well field presently operated by DOE for the Portsmouth Gaseous Diffusion Plant. Another site was on the west border of the study area, near MSEA well R5. Calculated K's from distance-drawdown data gathered from tests near wells R5 and R7 were 401 and 555 ft/day, respectively (table 3). K could not be calculated by time-drawdown methods for the site near well R7 because the data do not correspond to assumptions required by Neuman. However, a K of 474 ft/d was calculated from the time-drawdown data obtained from the aquifer test near well R5. A comparison of the mean K for the 7-mi reach (466 ft/d) with results of the two onsite aquifer tests (401 and 555 ft/d) indicates that K's at the MSEA are typical of the Scioto River Valley in Pike County.

Multiple-well aquifer tests near wells R5 and R7 yielded K's two to three times those determined by slug-test methods (table 3). K's derived from specific-capacity tests were comparable to K's derived from multiple-well aquifer tests; thus, K's derived from specific-capacity tests can be appropriately used to model flow at the site scale. One exception was the K of 815 ft/d determined from the specific-capacity test at well S10 (table 3), which seems to be somewhat higher than K's determined for the aquifer by the multiple-well aquifer tests. It should be noted that the specific-capacity test evaluated only the uppermost 15 ft of the aquifer (which contains much gravel at well S10), whereas the multiple-well aquifer tests evaluated the entire saturated thickness of the aquifer.

Mineralogy of Aquifer Materials

Aquifer mineralogy was analyzed to (1) identify reactive phases that may be controlling aqueous chemistry in the various stratigraphic units of the outwash aquifer and bedrock, and (2) determine if the color change observed in the outwash sediments is controlled by mineralogy. Considered to be the most chemically reactive size fraction, the silt and clay matrix (<0.063-mm size fraction) of sediment samples from selected boreholes was analyzed by X-ray diffraction (XRD) at the USGS laboratory in Sacramento, Calif., to determine aquifer mineralogy. Results are summarized in table 4, where minerals are listed in order of apparent relative abundance. Relative abundance of minerals is only approximate and was based on the intensity of the most prominent peaks on the diffractograph, as well as the completeness of the diffractograph pattern when compared to the standard patterns. The relative efficiency for diffraction by different classes of minerals was not considered. The detection limit of XRD analysis is 5 percent by volume.

Mineralogical analyses show that the Ohio Shale is composed predominantly of quartz but also contains moderate amounts of muscovite and illite and lesser amounts of kaolinite and albite. The mineralogical analyses of the sand and gravel samples reveal that outwash sediments are composed primarily of dolomite and quartz. Spatial variations in the outwash mineralogy are minimal, except that quartz is somewhat more abundant in the gray sediments of the intermediate and deep zones of the outwash aquifer, and dolomite is more abundant in the yellowish-brown sand and gravel deposits of the shallow zone of the outwash aquifer. Analysis of the sand samples by optical microscopy shows that most of the quartz grains in the gray sand and gravelly sand units are clear, whereas quartz grains in the shallow, yellowish-brown samples are iron stained. Thus, the color variation in the sediments with depth appears to be due to lack of iron staining at depth and likely is not due to mineralogical variations in the outwash caused by different depositional sources.

Several iron- and magnesium-bearing minerals are present in the very fine sand fraction (0.063 to 0.125-mm) of the outwash sediments. The very fine sand fraction from the four sand and gravel samples listed in table 4 was composited, and the strongly magnetic minerals in the composite sample were collected by means of a strong hand magnet. XRD analysis revealed these magnetic minerals to be magnetite (Fe_3O_4), orthopyroxene ($(Mg,Fe)SiO_3$), and amphibole ($((Ca,Na)_{2\text{-}3}(Mg,Fe,Al)_5(Si,Al)_2O_{22}(OH)_2)$).

Mineralogy of the surficial alluvium is similar to that of the outwash, except that quartz is the dominant component of the silty clays and dolomite is only moderately abundant. Despite the prevalence of clay-sized particles in the alluvium, only trace amounts of a clay mineral (kaolinite) were detected.

Forms of Carbon

Organic carbon has long been linked with the oxidizing capacity of sediments and the ability of sediments to adsorb organic compounds (Armstrong and others, 1967). Organic carbon also is used by microbes as the electron donor in many redox reactions such as denitrification and sulfate reduction. Owing to these factors, the presence or absence of organic carbon may exert a strong influence on ground-water chemistry and on the fate and transport of agrochemicals in the aquifer. Total carbon data were needed to evaluate organic carbon concentration and to help provide insights into carbonate equilibrium reactions occurring at the site. The total carbon and organic carbon concentrations of subsurface sediments at the Ohio MSEA are summarized in table 5.

Total Carbon

Total carbon concentrations range from 0.23 to 5.99 weight percent; the median concentration is 2.72 weight percent. A nonlinear, nonmonotonic relation of total carbon to depth is evident in figure 11. At depths of less than 10 ft below land surface, where alluvium consisting of silty clay is the predominant sediment type, total carbon concentration of sediment

Table 4. Mineralogy of aquifer sediments at the Ohio Management Systems Evaluation Area

[Mineralogy determined by X-ray diffraction of silt and clay matrix of sediment samples. Minerals listed in order of apparent decreasing abundance for each sediment type. Sample locations are shown on figure 2. Sample depth is given as feet below land surface]

Formation	Sediment type	Sample location(s) and depth	Minerals present	Chemical formula	Relative abundance
Alluvium	Silty clay	R4, 11-13 ft	Quartz	SiO_2	Dominant
			Dolomite	$CaMg(CO_3)_2$	Moderate
			Albite	$NaAlSi_3O_8$	Scant
			Muscovite	$KAl_2(AlSi_3O_{10})(OH, F)_2$	Scant
			Calcite	$CaCO_3$	Scant
			Kaolinite	$Al_2Si_2O_5(OH)_4$	Scant
Outwash (shallow zone)	Yellowish-brown sand	S14, 11-14 ft	Dolomite	$CaMg(CO_3)_2$	Dominant
		R7, 14 ft	Quartz	SiO_2	Dominant
			Calcite	$CaCO_3$	Moderate
			Albite	$NaAlSi_3O_8$	Scant
			Orthoclase	$KAlSi_3O_8$	Scant
			Muscovite	$KAl_2(AlSi_3O_{10})(OH, F)_2$	Scant
			Kaolinite	$Al_2Si_2O_5(OH)_4$	Scant
Outwash (intermediate and deep zones)	Gray sand	S14, 38-41 ft	Quartz	SiO_2	Dominant
		R7, 59-60.5 ft	Dolomite	$CaMg(CO_3)_2$	Dominant
			Calcite	$CaCO_3$	Moderate
			Albite	$NaAlSi_3O_8$	Moderate/ Scant
			Muscovite	$KAl_2(AlSi_3O_{10})(OH, F)_2$	Scant
			Kaolinite	$Al_2Si_2O_5(OH)_4$	Scant
Ohio Shale	Shale	R2, 70-78 ft	Quartz	SiO_2	Dominant
			Muscovite	$KAl_2(AlSi_3O_{10})(OH, F)_2$	Moderate
			Illite	$(Ca, Mg,K)Al_2(Si_4O_{10})(OH)_2$	Moderate
			Kaolinite	$Al_2Si_2O_5(OH)_4$	Scant
			Albite	$NaAlSi_3O_8$	Scant

Table 5. Carbon content of aquifer sediments at the Ohio Management Systems Evaluation Area

[Sample locations correspond to well locations shown in figure 2. Detection limit of analytical method is 0.01 percent of the pretreated and dried sample weight]

Sample location	Sample depth (feet below land surface)	Total carbon (percentage of dry weight)	Organic carbon (percentage of dry weight)
R1	5 - 7.5	0.83	0.65
	12 - 14.5	.46	.31
	26 - 29	4.43	.47
	38 - 41	4.13	.58
R2	8.5 - 10	.25	.19
	12 - 15	3.12	.35
	40 - 42	3.29	.55
	50 - 52	.48	.49
	58 - 60	.31	.35
R3	7 - 9	2.82	.33
	13 - 15	2.73	.25
	28 - 29.5	3.07	.24
	59 - 61	1.49	.42
R4	5 - 8	1.67	.52
	29 - 32	3.40	.46
	45.5 - 47	3.00	.50
	59 - 62	.91	.15
R5	5 - 8	2.16	.43
	11 - 14	2.34	.43
	20 - 23	3.39	.49
	38 - 41	3.15	.41
	59 - 62	1.75	.29
R6	5 - 7	2.39	1.21
	15 - 16	3.06	1.42
	39 - 40.5	3.14	.43
	59 - 60.5	3.28	.46
	61.5 - 62	2.87	.40
	73 - 74	1.59	.56
R7	5 - 5.5	1.25	.74
	5.5 - 8	1.51	.29
	14 - 15	2.95	.13
	26 - 27.5	3.46	.67
	44 - 46	3.27	.38
	61 - 62	.61	.33
R8	4 - 6	2.38	.19
	13 - 14	2.66	.08
	14 - 15	2.51	.01
	28 - 29.5	3.76	.39

Table 5. Carbon content of aquifer sediments at the Ohio Management Systems Evaluation Area—Continued

Sample location	Sample depth (feet below land surface)	Total carbon (percentage of dry weight)	Organic carbon (percentage of dry weight)
S1	5 - 7	0.78	0.74
	7 - 8	2.12	.32
	11 - 14	3.03	1.14
	25 - 26	3.08	.43
S2	5 - 7	1.10	.03
	12 - 13	4.08	<.01
	24 - 25.5	3.34	1.39
S3	5 - 8	1.70	.57
	6 - 6	1.99	1.83
	17.5 - 19	3.23	.27
S4	5 - 6	1.16	.98
	13 - 14	3.98	.30
S5	8.5 - 10	1.85	.31
	12 - 15	3.32	.13
	39 - 42	2.79	.74
	56 - 59	.23	.14
S7	5 - 6.5	.81	.60
	14 - 16	2.50	.14
	22.5 - 23.5	3.38	.38
S8	5 - 6	1.93	.20
	6 - 7.5	2.10	.01
	14 - 17	2.77	.05
	15.5 - 15.5	2.22	.36
	26 - 29	4.37	.59
S9	5 - 7.5	2.10	.38
	12.5 - 13	2.27	.27
	13.1 - 13.5	2.57	.06
	25 - 26	5.99	.22
S10	5 - 7.5	1.96	.30
	43 - 45	2.72	.61
	58 - 59.5	3.61	.63
S11	5 - 8	1.44	.06
	14 - 15	2.06	.38
S12	6.4 - 9	1.86	.24
	12.5 - 15	1.70	.10
	24 - 26	3.83	.61

Table 5. Carbon content of aquifer sediments at the Ohio Management Systems Evaluation Area—Continued

Sample location	Sample depth (feet below land surface)	Total carbon (percentage of dry weight)	Organic carbon (percentage of dry weight)
S13	5 - 7.5	2.07	0.32
	14 - 15.5	3.40	.16
	24 - 25.5	4.05	.50
S14	6.5 - 8	2.13	.33
	11 - 14	2.75	.03
	32 - 35	3.36	.44
	38 - 41	3.23	.65
	60.5 - 62	3.78	.39

is less than 2.82 percent. From 10 to about 26 ft below land surface (where yellowish-brown outwash sediments predominate), total carbon concentration of sediments increases with depth, and a maximum concentration of 5.99 weight percent is found in this depth range at well S9. At about 26 ft below land surface, where the color of outwash sediments changes from yellowish-brown to gray, the total carbon concentration of sediments generally begins to decrease with increasing depth. Minimum concentrations are found from 50 to 75 ft below land surface. An exception to this general trend is found at well locations R1, S10, and S14, where total carbon concentration does not decrease significantly with depth below 26 ft.

The observed changes in total carbon concentration with depth are consistent with the relative abundance of carbonate minerals found with increasing depth in the aquifer (table 4), in that dolomite is the predominant mineral in alluvium and shallow outwash samples but not in the intermediate to deep, gray outwash sediments. The assertion that total carbon concentration is dependent on depth rather than on grain size of the sediments is supported by the Mann-Whitney test, which indicated no statistically significant difference (at 95-percent confidence) in total carbon concentration of very fine to fine sands, medium to coarse sands, gravelly sands, and sandy gravels (as shown in geologic sections, figs. 8 and 9).

Organic Carbon

As indicated by geologic descriptions and by the oily sheen found on cuttings during drilling operations, the Ohio Shale is highly carbonaceous. The organic carbon concentration of the one shale sample analyzed is 3.43 weight percent.

Contrastingly, unconsolidated sediments at the site have a median organic carbon concentration of 0.39 weight percent; concentrations range from less than 0.01 to 1.83 percent. As shown in figure 11, organic carbon concentration is generally highest in shallow (less than 10 ft below land surface), fine-grained alluvium samples. The lowest organic carbon concentrations (less than 0.5 percent) are present at approximately 15 ft below land surface, near the water table in the outwash deposits. Below 15 ft, organic carbon concentration has a nonlinear relation to depth similar to that displayed by total carbon. Organic carbon concentration increases to about 26 ft below land surface (the approximate depth of the sediment color change) and then generally begins to decrease with depth in the outwash deposits.

As with total carbon, organic carbon concentrations appear to be dependent on depth rather than on the grain size of sediments. The Mann-Whitney test indicated no statistically significant difference (at 95-percent confidence) between the organic carbon concentration of very fine to fine sands, medium to coarse sands, gravelly sands, and sandy gravels. Owing to the paucity of carbon analyses of silt and clay samples, this sediment size was excluded from the statistical evaluations.

Relatively high organic carbon concentration (1.21 to 1.42 percent) in shallow samples from borehole R6 may be due to the proximity of this well to the Scioto River. The charcoal-containing sediment sample collected in borehole S3 at a depth of 6 ft below land surface does have an elevated organic carbon

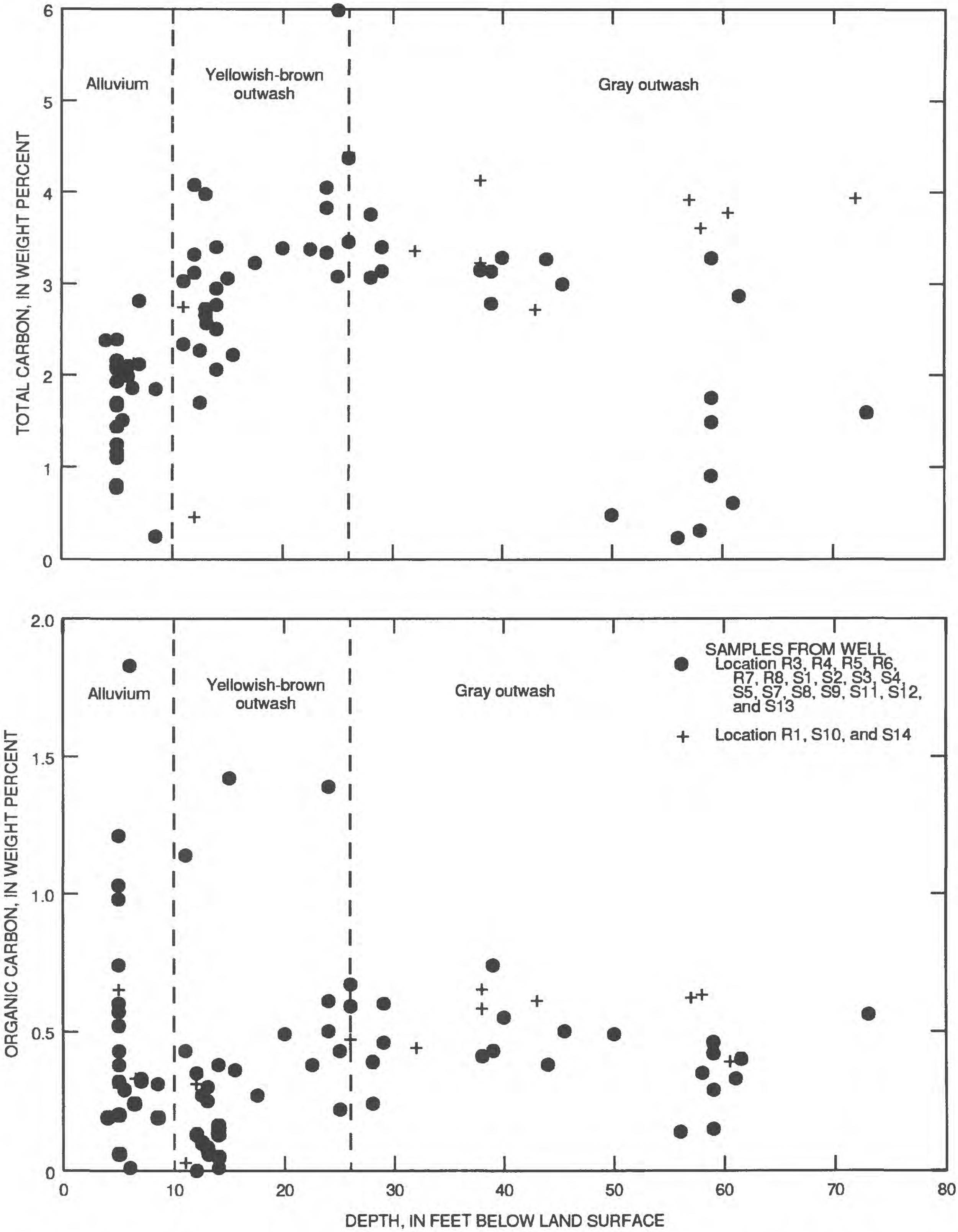

Figure 11. Relation of concentration of total carbon and organic carbon in alluvium and outwash sediments as a function of depth below land surface.

concentration of 1.83 percent, but not enough samples were collected near wood fragments to analyze the effects of these fragments on carbon concentration in nearby sediment.

Conceptual Model of Ground-Water-Flow System

Solute transport and fate, in addition to being affected by the distribution of physical aquifer properties, also are controlled by spatial and temporal variations in ground-water-flow directions and rates. Thus, a conceptual model of the ground-water-flow system at the Ohio MSEA was developed to aid in interpretation of agricultural effects on water quality and to guide the construction of the regional flow model that was developed by Finton (1994). A conceptual model is a working hypothesis of a process that cannot be observed directly or is difficult to observe directly. The conceptual model of ground-water flow at the Ohio MSEA was derived by collecting and synthesizing data on ground-water levels, stream-aquifer relations, and leakage from underlying geologic units.

Horizontal Flow

The direction of ground-water flow at the MSEA can be estimated from water-level contour maps, given that ground water flows from areas of high hydraulic head to areas of low hydraulic head. Under isotropic aquifer conditions, ground water will move along paths perpendicular to water-level contours (lines of equal hydraulic head).

Ground water in the outwash aquifer at the MSEA typically flows from east to west-southwest and perpendicular to the terrace ridge, as shown in figure 12. The water table typically is about 15 ft below land surface. Altitude of the water table fluctuates approximately 4 ft annually beneath the eastern half of the valley; fluctuations are greater closer to the Scioto River. The water-table altitude near the river varies as much as 12 ft annually. A cone of depression around the DOE wellfield at the southwestern corner of the property (figures 2 and 12) captures some of the ground water that would normally flow to the Scioto River.

Vectors of hydraulic gradient for each of the three MSEA farming plots are shown in figure 13. Vectors were constructed by use of water-level data measured monthly from April 1991 through October 1992 in the water-table wells and (or) the uppermost saturated port of the multiport wells. Vector length indicates the magnitude of the gradient at the water table, and vector orientation indicates flow direction as calculated by Cramer's Rule (Pinder and Abriola, 1982). Water-table gradients in the study area are a function of the altitude of ground water and the stage of the Scioto River. Periods of heavy rainfall recharge the outwash aquifer and, at the same time, produce high river stage. High river stage can result in low water-table gradients across the site. After a storm passes, river stage drops much faster than ground-water levels do, resulting in high water-table gradients near the river that then propagate upgradient. Direction of ground-water flow beneath the MSEA generally becomes more southwesterly as water-table gradients increase, especially beneath plot 300. Periods of little precipitation cause ground-water levels to decline gradually and river stage to remain low. During such periods, water-table gradients gradually decrease. At all times, ground-water gradients on the lower terrace are greater than those on the middle terrace, and gradients are greatest at the southwest end of the lower terrace because of the pumping of wells at the DOE well field. During the period of study, water-table gradients beneath the middle terrace varied temporally and spatially from approximately 4×10^{-4} (2 ft/mi) to 1.8×10^{-3} (9.5 ft/mi), and flow directions varied from approximately S. 47°W. to N. 72°W.

Advective ground-water-flow velocity (v) can be calculated by means of the following equation:

$$v = (K/n_e)(dh/dl), \qquad (5)$$

where K is hydraulic conductivity, n_e is the effective porosity of the aquifer sediments, and dh/dl is the water-table gradient. Advective flow velocities for shallow ground water beneath the three MSEA research plots were calculated by use of (1) K's obtained from the specific-capacity tests done at each of the three farming plots (table 3), (2) an assumed effective porosity of 0.25[7], and (3) the mean hydraulic gradient for each plot measured from April 1991

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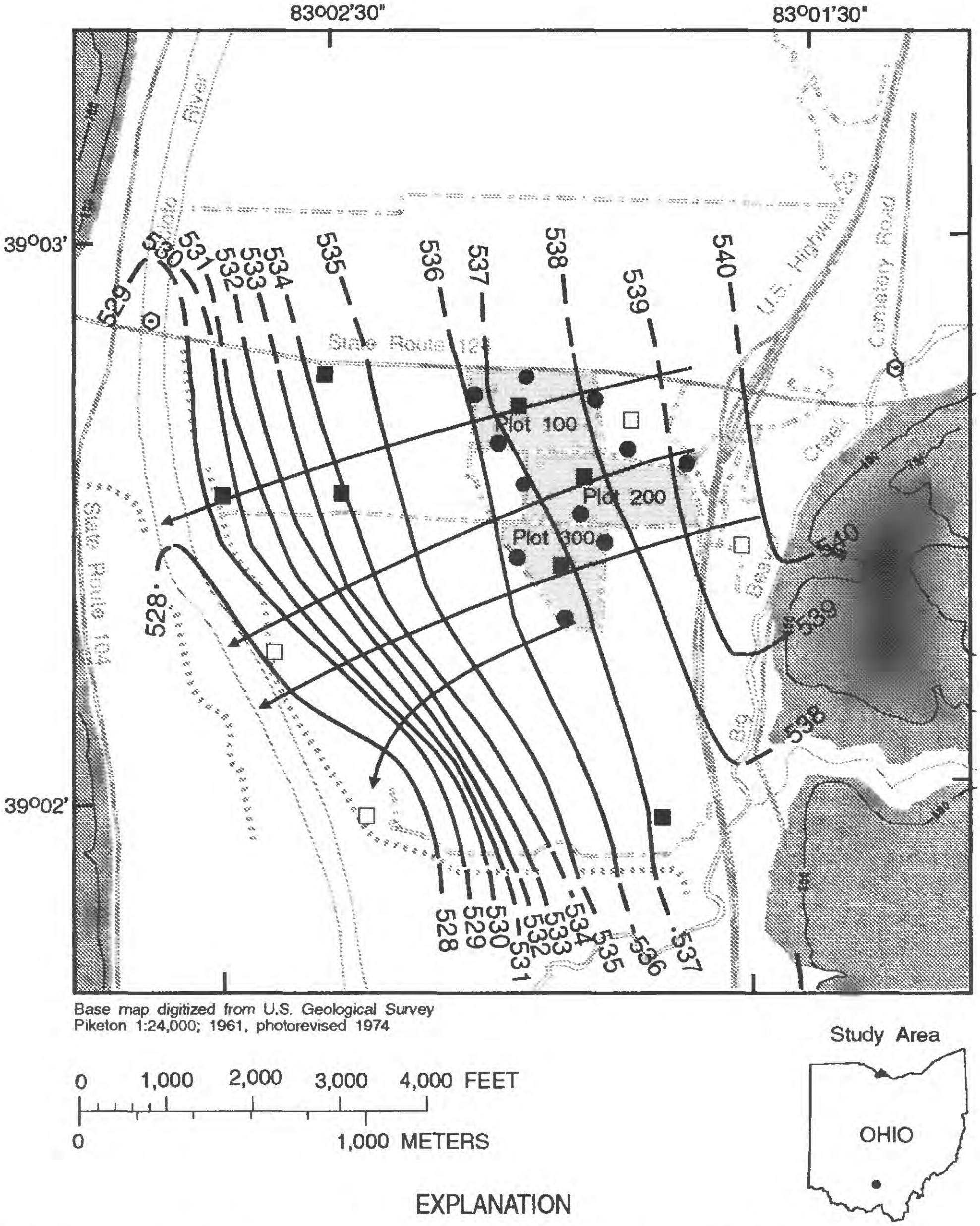
[7] In 1969, Norris and Fidler determined specific yields from time-drawdown data in 10 wells about 2 mi north of the MSEA. Specific yields ranged from 0.18 to 0.22 and averaged 0.20. Because specific yield underestimates formation porosity, the value of 0.25 is used in this report as a reasonable approximation of effective porosity in the outwash aquifer.



Base map digitized from U.S. Geological Survey
Piketon 1:24,000; 1961, photorevised 1974

EXPLANATION

▨	RESEARCH PLOT AND NUMBER
▨	BEDROCK OUTCROP
—536—	GROUND-WATER-LEVEL CONTOUR--Shows altitude of water table in the outwash aquifer. Dashed where inferred. Contour interval 1 foot. Datum is sea level
←	GENERAL DIRECTION OF GROUND-WATER FLOW
—	TOPOGRAPHIC CONTOUR--Interval 100 feet. Datum is sea level
┄	GRAVEL ROAD
⸗	LEVEE
	OBSERVATION WELL
●	Multiport sampler
■	Well cluster--one well screened at water-table; one multiport sampler
□	Well cluster--one well screened at water-table; one multiport sampler; one well completed in bedrock
⊙	SURFACE-WATER DATA-COLLECTION SITE

Figure 12. Ground-water levels and ground-water-flow paths at the Ohio Management Systems Evaluation Area, July 1-3, 1991.

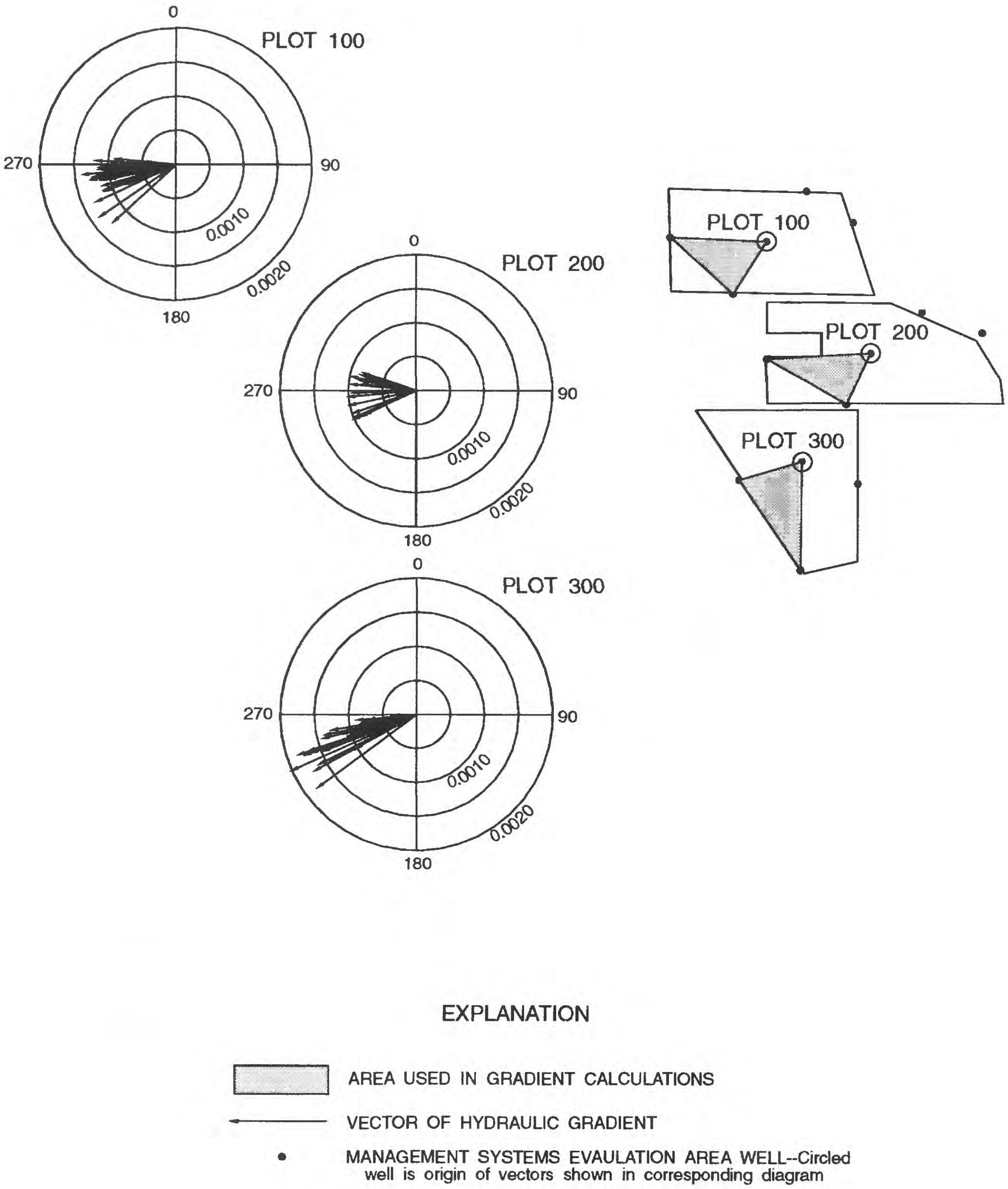


Figure 13. Vectors of hydraulic gradient at the water table for each farming plot, Ohio Management Systems Evaluation Area, April 1991-October 1992. (Locations of plots 100, 200, and 300 are shown on fig.1.)

through October 1992. On the basis of these calculations, the average linear flow velocity for shallow ground water beneath plot 200 (fig. 12) is about 3.3 ft/d, compared to an average ground-water-flow velocity of about 1.7 ft/d beneath plots 100 and 300. Through the use of a steady-state numerical ground-water-flow model, Finton (1994) estimated that 2.1 to 8.8 years would be required for water particles originating at the downgradient edges of the plots to travel to their discharge points at the Scioto River or the DOE well field.

Vertical Flow

The vertical component of ground-water flow was assessed by measuring the difference between river stage and water levels in the outwash aquifer, between water levels in paired wells completed in the bedrock and outwash aquifers, and in multilevel well ports completed at different depths throughout the outwash aquifer. Heads that decrease with depth indicate a downward component of flow in the system (downflow), whereas an upward flow component (upflow) is indicated by heads that increase with depth. Vertical-flow data were used to assess ground-water/surface-water interactions, bedrock-outwash relations, and flow paths within the outwash aquifer. These interpretations are corroborated by chemical data later in this report.

Ground-Water/Surface-Water Interactions

The drainage area for the Scioto River upgradient of the MSEA is about 5,800 mi^2 of farmed river valley and till plains, plus some forested bedrock upland areas. The maximum measured discharge of the Scioto River at the MSEA during the period of study was 5,750 ft^3/s in April 1991. Discharge decreased through the 1991 growing season to 676 ft^3/s in November 1991.

To determine the relation between the Scioto River and the ground-water-flow system, the investigators compared ground-water levels in the water-table wells near the river to stage of the Scioto River. Hydrographs from wells R4, R5, and R6 and the river (fig. 14) show that the water-table surface was generally higher than the Scioto River stage (but by less than 0.2 ft) from August through November 1991, indicating that water from the shallow zone of the outwash aquifer discharges to the river during low streamflow. Water-table altitude in the outwash aquifer at well R7 (fig. 14) follows a pattern similar to that for wells R5 and R6. However, during September 1991-March 1992, water levels at R7 were generally below those of the Scioto River, indicating that pumping of the nearby DOE sanitary supply wells was producing a cone of depression encompassing well R7 and creating the potential for induced infiltration from the river into the aquifer at this location.

All of the near-river hydrographs are characterized by rapid fluctuations in response to changing river stage (especially during December 1991-August 1992), an indication that the riverbed is relatively permeable and the outwash aquifer and the river are in close hydraulic connection. At high river stage, ground-water levels are lower than surface-water stage, and flow is temporarily reversed as water from the Scioto River infiltrates the surrounding aquifer and is temporarily stored in the streambanks. The effects of bank storage on the flow system can be seen in figure 15. During very high river stage, ground-water flow away from the river causes a potentiometric trough to develop in the center of the lower terrace, and ground-water flow from the upper terrace as well as from the river banks is temporarily directed toward this trough.

Bank-storage events occur frequently, as evidenced by records from electronic data loggers at the site. In water year 1991, approximately 19 flow reversals occurred (fig. 14). Events in which water levels in well R5 were less than those of the river and less than those in well R4 resulted in small, short-lived potentiometric troughs that formed near the river. Only when levels in well R5 surpassed those in well R4 did a major trough, like the one shown in figure 15, develop.

The maximum duration of a flow-reversal event during the study period was 17 days in July and August 1992. The difference in head between the river and well R5-WT during each day of this event is listed in table 6, as well as a computed linear gradient between the river and well R5-WT. A simple calculation by use of these gradient data, a K of 401 ft/d (table 3, multiple-well aquifer test data, well R5-WT), and an assumed porosity of 0.25 indicates that a particle of water could have migrated about 190 ft inland through the outwash aquifer during this flow-reversal event. This calculated migration distance is a maximum because the vertical K of the streambed is likely much lower than the horizontal K of the outwash aquifer; thus, the traveltime through the streambed would be correspondingly less.

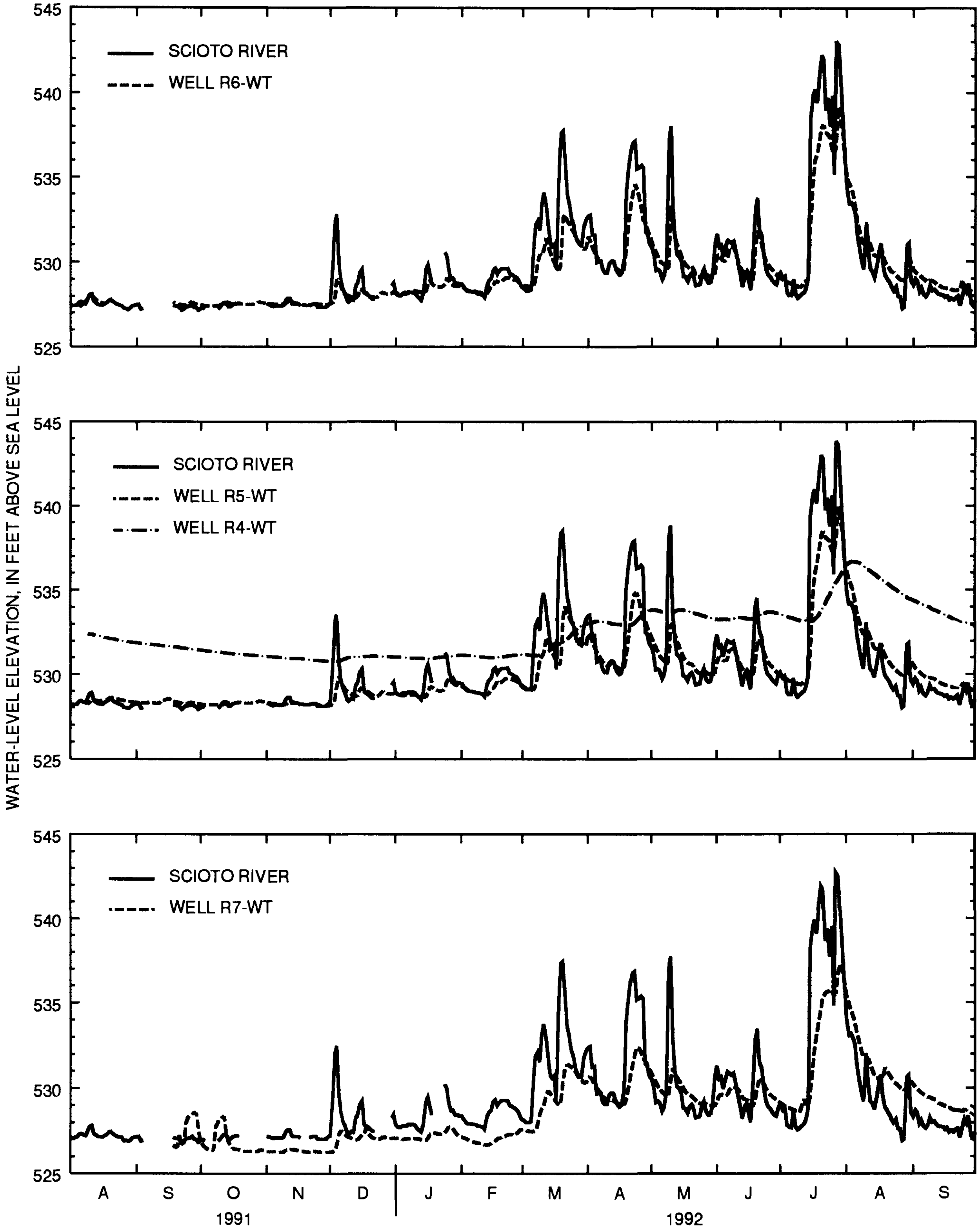
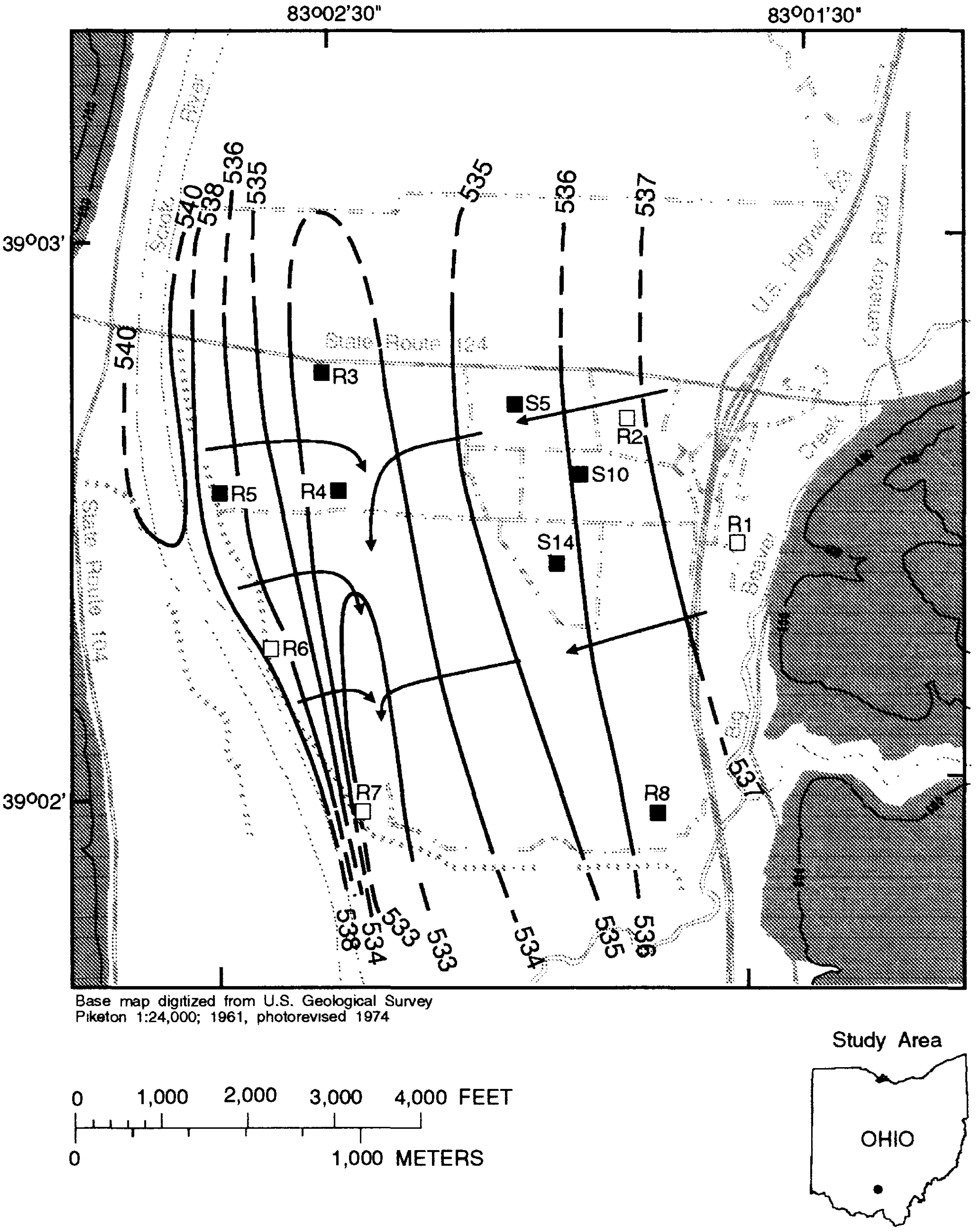


Figure 14. Hydrographs showing relation between stage of the Scioto River (observer data) and water levels in wells R6-WT, R5-WT and R4-WT, and R7-WT (continuous data logger), Ohio Management Systems Evaluation Area, August 1991-September 1992.



Base map digitized from U.S. Geological Survey Piketon 1:24,000; 1961, photorevised 1974

EXPLANATION

▨ BEDROCK OUTCROP

—536— GROUND-WATER-LEVEL CONTOUR--Shows altitude of water table in the outwash aquifer. Contour interval, in feet, is variable. Datum is sea level

← APPARENT GROUND-WATER-FLOW PATH AND DIRECTION

—— TOPOGRAPHIC CONTOUR--Interval 100 feet. Datum is sea level

GRAVEL ROAD

LEVEE

OBSERVATION WELLS AND LOCAL IDENTIFIERS

■ R4 Well cluster--one well screened at water table; one multiport sampler

□ R6 Well cluster--one well screened at water table; one multiport sampler; one well completed in bedrock

Figure 15. Ground-water levels and apparent ground-water-flow paths at the Ohio Management Systems Evaluation Area during a transient flow-reversal event, July 17, 1992.

Table 6. Daily linear gradient between the Scioto River and well R5-WT and cumulative distance traveled inland by a water particle during a flow-reversal event at the Ohio Management Systems Evaluation Area, July 14-31, 1992

[The location of well R5-WT in relation to the Scioto River is shown in figure 2. A negative head difference and gradient means that ground water was flowing toward the Scioto River. A positive head difference and gradient means that ground water was flowing away from the Scioto River, toward well R5-WT. Gradient was calculated from measured head differences between the Scioto River and well R5-WT, located 475 feet inland. Velocity was calculated from an assumed effective porosity of 0.25 and a hydraulic conductivity of 401 feet per day, as determined by Norris (1991) from analysis of aquifer-test data from near well cluster R5 (table 3). Cumulative distance traveled inland is based on the assumption that flow is directly toward well from river. Abbreviations: ft, feet; ft/d, feet per day]

Date	Head difference between river and well R5-WT (ft)	Hydraulic gradient	Velocity (ft/d)	Cumulative distance traveled inland (ft)
07/14/92	0.20	0.0004	0.64	0.64
07/15/92	7.94	.0167	27	28
07/16/92	6.68	.0141	23	51
07/17/92	5.26	.0111	18	69
07/18/92	3.74	.0079	13	82
07/19/92	4.64	.0098	16	98
07/20/92	4.80	.0101	16	110
07/21/92	4.25	.0089	14	130
07/22/92	1.61	.0034	5.4	[a]130
07/23/92	2.42	.0051	8.2	140
07/24/92	.88	.0019	3.0	[a]140
07/25/92	3.24	.0068	11	160
07/26/92	-1.28	-.0027	-4.3	150
07/27/92	6.03	.0127	20	170
07/28/92	3.61	.0076	12	180
07/29/92	2.09	.0044	7.0	190
07/30/92	.68	.0014	2.2	[a]190
07/31/92	.25	.0005	.80	[a]190

[a] Distance traveled was too small to be reflected in rounded number.

Although hydrographs from the water-table wells provide data on interactions of shallow ground water with the Scioto River, head data from multiport well R6, 150 ft from the river, provide further information about the effects of the river on ground-water flow in the deep and intermediate zones of the aquifer. These data show that ground-water flow only 150 ft from the river is predominantly horizontal throughout the depth of the aquifer, as evidenced by the slight differences in head measured at the different depths. Deep waters between ports 1 and 2 (40 and 60 ft below land surface) generally have an upward component of flow (a mean measured head difference of 0.04 ft and a mean vertical gradient of 0.002[8]). The vertical

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[8] Measured head differences between ports 1 and 2 ranged from less than 0.03 ft (within measurement error) to 0.09 ft. Thus, vertical hydraulic gradients between ports ranged from 0 to 5 x 10^{-3}.

component of flow between the uppermost saturated port and port 2 (40 ft below land surface) is highly variable in direction, but slight downflow generally occurs in this zone, likely caused by numerous, repeated bank-storage events. The maximum measured upward and downward gradients between the uppermost saturated port and port 2 were 0.009 (maximum head difference of 0.10 ft; distance between ports of 10.98 ft). (Even an assumed constant upward gradient of 0.009 in this zone multiplied by the distance between the river and well R6 (150 ft) would yield an estimate that a particle of water originating 30 ft below land surface at well R6 would travel only 1.35 ft upward by the time it reached the edge of the Scioto River.) Thus, given the absence of consistent, significant upflow throughout the outwash aquifer at well R6, and the fact that the Scioto River is the only discharge area for ground water in the valley, one might infer that dramatic upflow of ground water occurs directly beneath the Scioto River and (or) deep ground water in the sand and gravel aquifer turns abruptly south beneath the river and flows downvalley before eventually discharging to the Scioto River.

In addition to investigating discharge to the Scioto River at the west edge of the study area, the nature of ground-water interaction with Big Beaver Creek (on the east edge of the study area) was evaluated. Big Beaver Creek drains an area composed predominantly of the poorly permeable bedrock uplands from which runoff is rapid and bank storage is minimal. This set of conditions produces rapid stage fluctuations in the creek during storms and only intermittent flow during dry summer months.

Big Beaver Creek seems to be recharging the outwash aquifer on the basis of the following water-level evidence obtained from wells R1 and R8, stage data from Big Beaver Creek, and results of seepage-meter tests:

1. The altitude of the Big Beaver Creek streambed was, for the period of record, generally higher than water-table altitude in well R1-WT, located near the creek (fig. 16). This relation indicates that Big Beaver Creek cannot be fed by ground-water discharge from the outwash aquifer under typical ground-water conditions. The stage of Big Beaver Creek was also higher than the water-table altitude at well R8-WT (fig. 16) — an indication that water flows from the creek to the aquifer at this location also.

2. Hydrographs for the water-table wells at clusters R1 and R8 (fig. 16) show a greater number of small water-level rises (representing recharge to the water table) than hydrographs for wells not located near a surface-water body (for example, well R4, figure 14). This finding indicates that the area near Big Beaver Creek is a recharge zone. Stage data for Big Beaver are not complete enough to correlate to ground-water-level hydrographs for wells R1 and R8.

3. Head measurements in the multiport well at cluster R1 show that flow, although predominantly horizontal, has a consistent component of downflow from the water table to 40 ft below land surface. Vertical gradients range from a minimum of 9×10^{-4} (4.75 ft/mi) during September and November of 1991 (when Big Beaver Creek was dry) to a maximum of 0.021 (110.9 ft/mi) on August 4, 1992, following heavy rains that resulted in high creek stage. Ground-water flow deeper than 40 ft below land surface has no vertical flow component that exceeds the water-level measurement error. Vertical flow in well R8 is rare.

4. Seepage-meter tests were done in July 1991 and August 1992 by OSU personnel to measure vertical K of the Big Beaver Creek streambed at three locations. Test locations are shown in figure 2, and data from the tests are summarized in table 7. These tests showed that, on the dates tested, the stream was losing water through the streambed to the underlying aquifer at the test locations. Streambed permeability as estimated by nine tests ranged from 0.89 to 19.9 ft/d; the mean vertical K for the three locations was 6.10 ft/d. Seepage meters could not be installed between S.R. 124 and test location 1 because the streambed between the two points is composed of shale. Elsewhere in the study area, Big Beaver Creek has a gravel streambed.

In summary, analysis of ground-water/surface-water interactions indicates that Big Beaver Creek is losing water to the outwash aquifer and that the Scioto River is gaining water from the outwash aquifer.

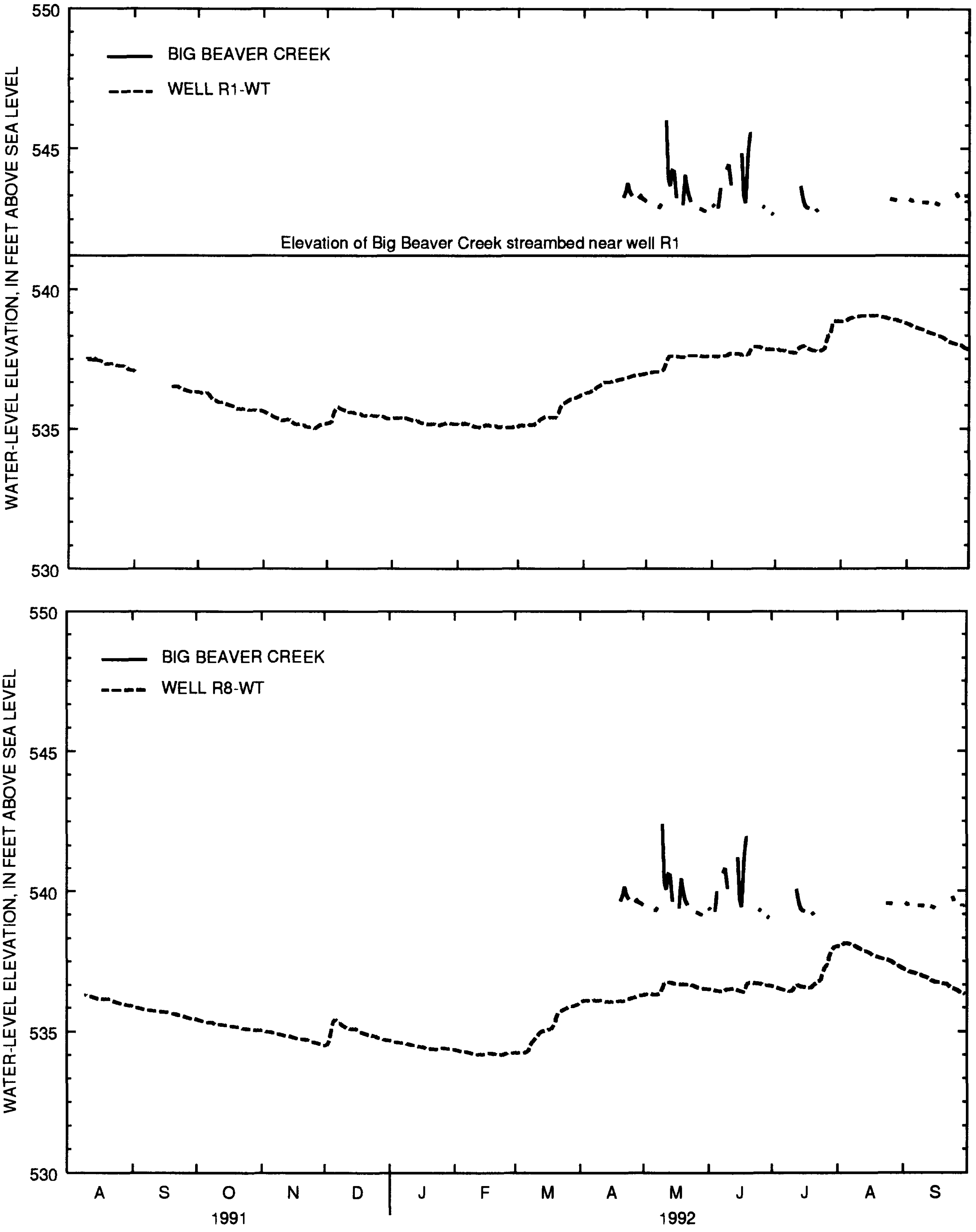

Figure 16. Hydrographs showing relation between stage of Big Beaver Creek (observer data) and water levels in the outwash aquifer at wells R1-WT and R8-WT (continuous data logger), August 1991-September 1992.

Table 7. Vertical hydraulic conductivity of streambed of Big Beaver Creek at the Ohio Management Systems Evaluation Area

[Vertical hydraulic conductivities were estimated from seepage-meter tests and water-level measurements in adjacent piezometers. Test locations are shown in figure 2. Abbreviations: ft., feet; ft/d, feet per day; SW., southwest]

Test location	Location description	Date of test	Number of replicate tests	Range of vertical hydraulic gradients observed	Mean vertical hydraulic conductivity (ft/d)	Standard deviation of hydraulic conductivity (ft/d)
1	Downstream from shale streambed (east of well R1)	7/11/91	2	0.088 - 0.088	1.55	0.69
2	East of Shepherd House	8/20/92	3	0.0033 - 0.0067	16.6	4.1
3	30 ft SW. of SR 23 bridge	8/20/92	4	0.011 - 0.038	.147	.050

Bedrock-Outwash Interaction

Using techniques similar to those employed to examine ground-water/surface-water interactions, the investigators evaluated the relation of the bedrock and outwash aquifers. A comparison of hydrographs from paired wells completed in the bedrock and outwash aquifers at locations R1, R2, and R6[9] (fig. 17) shows that vertical flow between the two aquifers varies in direction and magnitude across the study area. At well cluster R1, potentiometric altitudes in the outwash aquifer are higher than those in the bedrock aquifer (fig. 17) and indicate that downflow from the outwash to the bedrock aquifer occurs consistently. Exceptions were observed during periods of rapidly rising water levels, when no flow between the two aquifers was measured. The maximum downward gradient from the water table in the outwash aquifer to the upper zone of the bedrock aquifer was 0.005 (26.4 ft/mi).

In contrast, higher head in the bedrock than in the outwash aquifer at well cluster R2 shows that consistent upflow from the bedrock to the outwash deposits occurs at this location beneath the middle terrace (fig. 17). In addition to the difference in vertical flow direction, gradients are much greater at R2 than at well R1; the maximum observed gradient at R2 was 0.033 (174 ft/mi) on August 18, 1991, and was the largest vertical flow gradient observed at the MSEA.

Paired bedrock and outwash hydrographs from well R6 (fig. 17) show that upflow from the bedrock to the outwash aquifer generally occurs at this location; the maximum vertical gradient of 0.014 (73.9 ft/mi) was observed during base-flow conditions on October 1, 1991. During periods of bank storage in the outwash aquifer, however, vertical flow between the two formations is reversed, and gradients of as much as 0.023 are briefly observed as ground water flows from the outwash into the bedrock aquifer.

Vertical Flow Gradients within the Outwash Deposits

Data defining interactions of the outwash aquifer with streams and the bedrock aquifer were combined with head data obtained from multiport wells throughout outwash deposits to create an overview of vertical flow beneath the study area. The resulting potentiometric section through the study area (figs. 18 and 19) clearly illustrates that recharge to the aquifer occurs near Big Beaver Creek and the east edge of the study area. From this area, ground water flows beneath the terraces and shallow waters discharge to the Scioto River. Although horizontal flow dominates beneath the terraces, slight but temporally consistent vertical gradients are present near the base of the outwash aquifer at well R6. The upflow of deep outwash waters at well R6 is likely driven by upward leakage from the underlying bedrock. Moreover, the

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[9] Data from well R7 were not used for this analysis because of the slow recovery time of the bedrock well after sampling; the hydrograph from well R7-BR does not represent undisturbed head conditions.

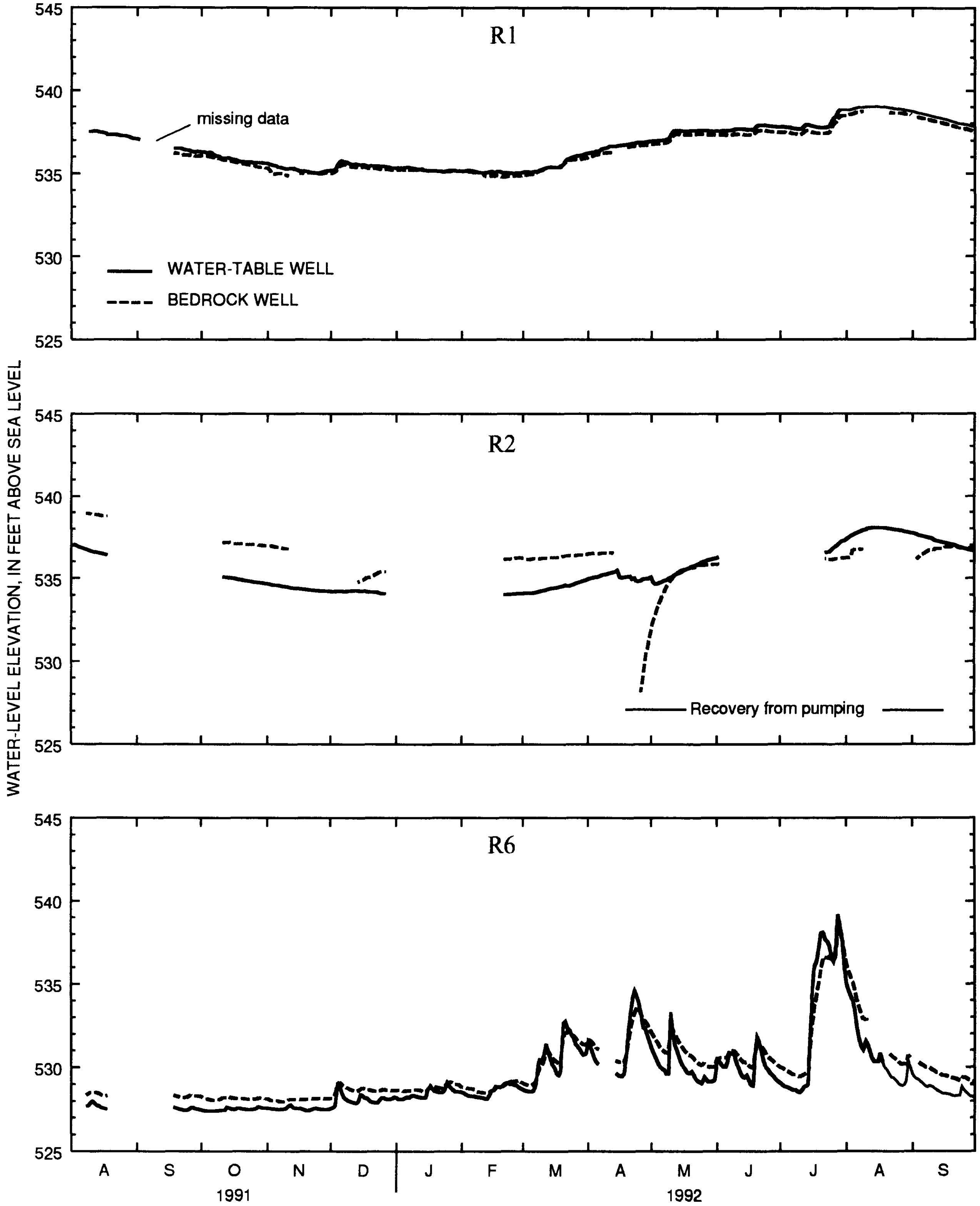

Figure 17. Hydrographs showing relation between water levels in the outwash aquifer and the bedrock aquifer at well clusters R1, R2, and R6, Ohio Management Systems Evaluation Area.

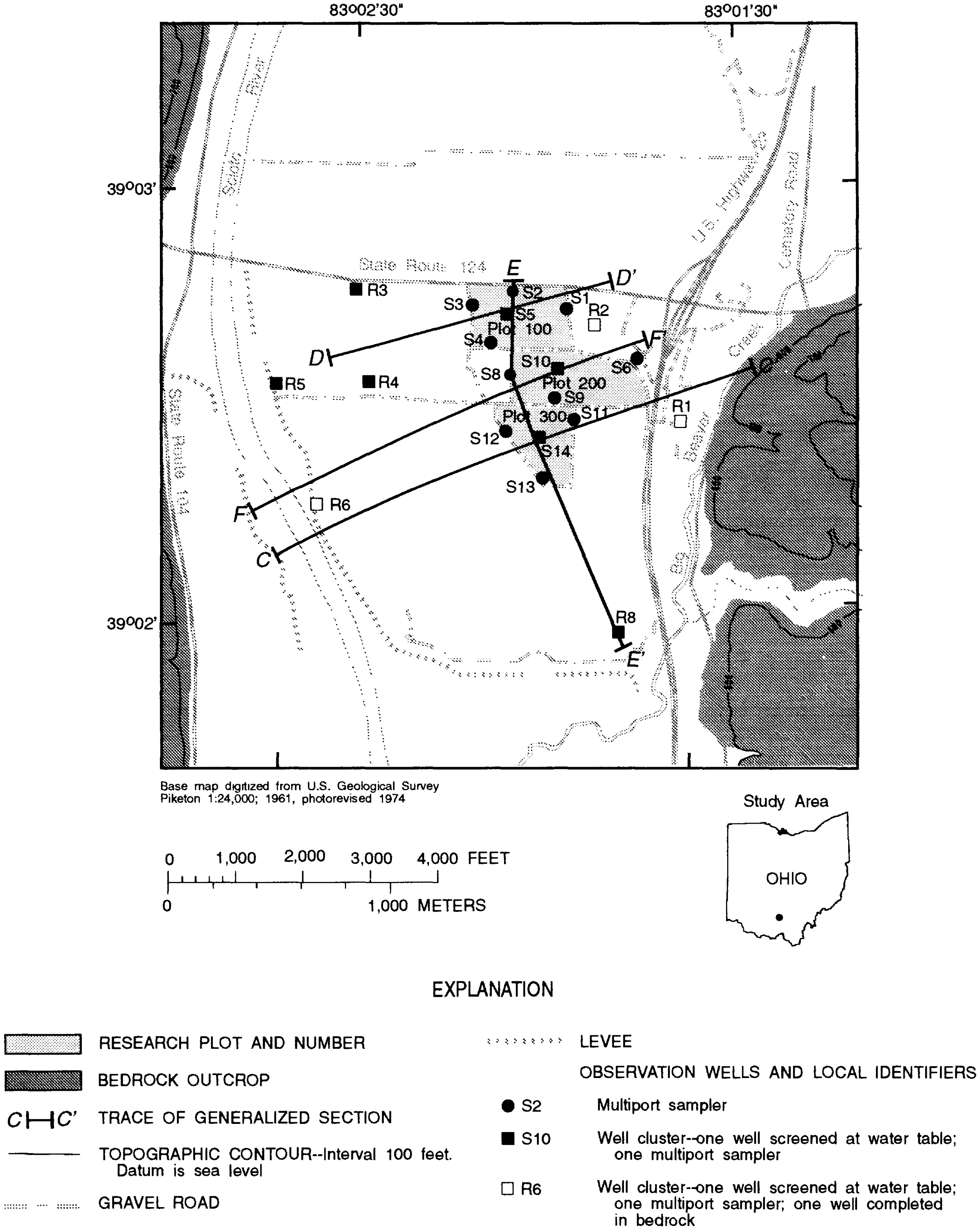

Figure 18. Location of generalized potentiometric sections and sections showing vertical variation in constituent concentrations, Ohio Management Systems Evaluation Area.

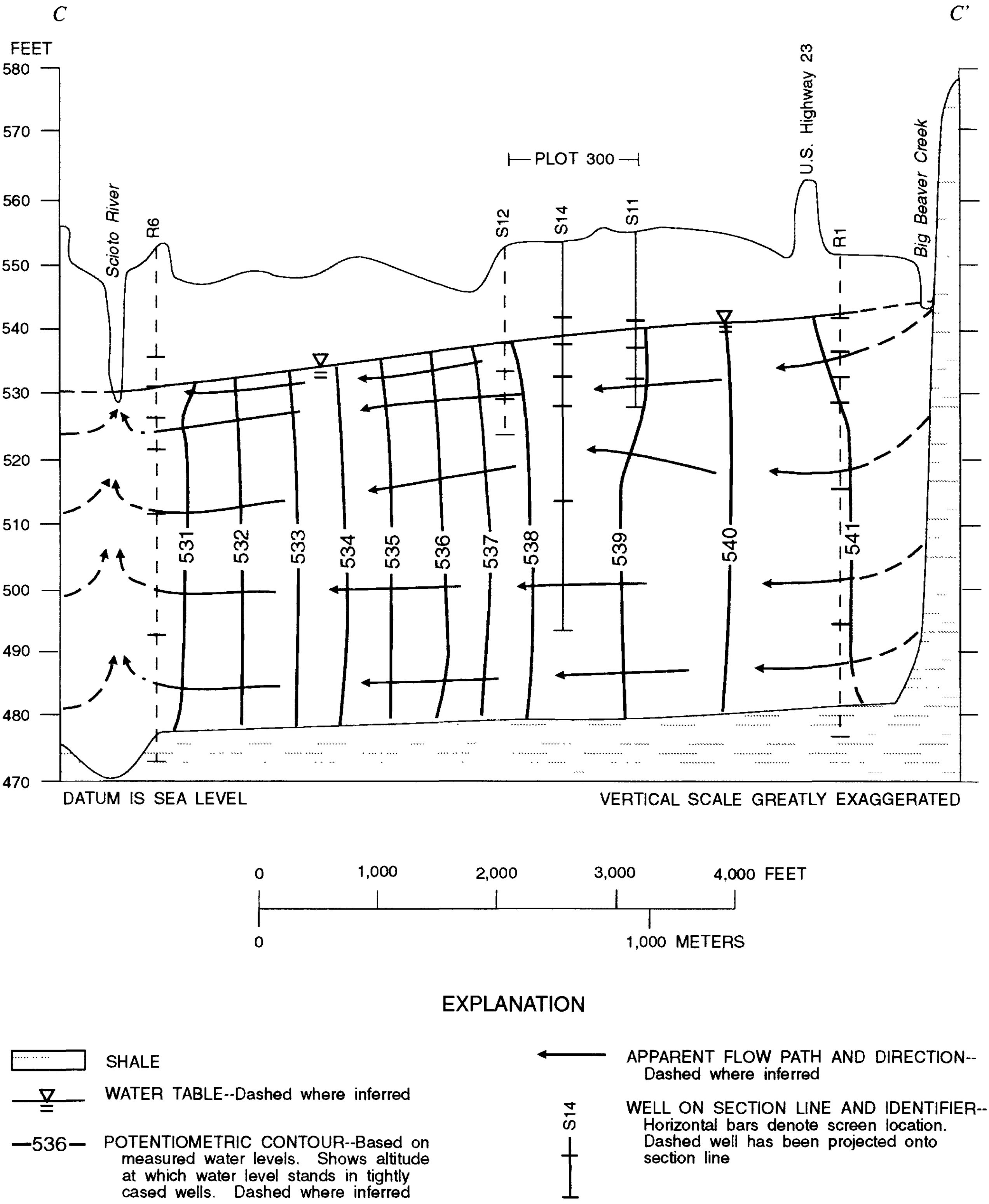

Figure 19. Potentiometric section along flow path through plot 300, Ohio Management Systems Evaluation Area, June 3-5, 1991. (Section trace shown on fig. 18.)

confluence of ground water flowing toward the Scioto from the west side of the valley provides an additional driving force for upward flow of water.

Near the water table are ephemeral, shallow flow cells that are affected by spatially variable infiltration of precipitation (likely caused by runoff from hillslopes and ponding of rainfall on the land surface). The existence of these flow cells and the influences on them are evidenced by temporally variable flow gradients observed in shallow multiport wells. For example, head measurements in wells R2, S2, and S6, in the northeast part of the middle terrace, all showed significant downflow on July 2, 1991. These wells are all at the base of a hillslope, so the recharge measured at these wells on July 2 may have been the result of runoff from a heavy rainfall on June 18. The only area where downflow is temporally consistent is the east edge of the valley, near Big Beaver Creek.

Intersection of the water table with land-surface topography also affects the shallow flow system. During April-June 1991 (when the water table was approximately 7 to 12 ft below land surface), two separate shallow flow cells were present beneath the middle and lower terraces. During this period, shallow ground water flowed horizontally beneath the middle terrace and discharged into the east side of the meander scar, at the base of the terrace bank (fig. 20). Water then discharged from the west side of the meander scar and flowed horizontally beneath the lower terrace until it discharged to the Scioto River. When the meander scar dried up at the end of June because of declining ground-water levels, vertical flow in wells S3 and R3 reversed direction as a deeper flow cell formed (fig. 20). The deeper flow cell was characterized by recharge on the middle terrace and slight upflow beneath the lower terrace that resulted in discharge to the Scioto River.

Ground-Water Budget

A ground-water budget is an accounting of all waters entering or leaving a ground-water-flow system. The water budget helps to refine the conceptual flow model and ensures that it is viable. The water-budget equation for a flow system is

$$\text{Inflow} = \text{Outflow} \pm \text{Storage}. \qquad (6)$$

For the water-budget calculations relevant to the Ohio MSEA, an area of 6.92×10^7 ft^2 was considered, as outlined by the study area boundary in figure 1. The east boundary is the edge of the bedrock valley, and the west boundary of the area of interest is the center of the Scioto River. The area is bounded by a flow path north of the site and by a flow path south of the cone of depression that was created by pumping of the DOE wells (figs. 2, 12). Flow-path boundaries are used so that regional underflow is a possible equation parameter only beneath the Scioto River. Any underflow entering the study area beneath the Scioto River is assumed to equal the volume of underflow leaving the study area beneath the river; therefore, underflow is not included in the water-budget equation. Surface runoff is considered negligible at the MSEA; infiltration of precipitation is rapid over the unconsolidated deposits, and runoff from the uplands is channeled to Big Beaver Creek on the east and to the Scioto River on the west. Thus, runoff is not considered to be a separate source of inflow to the ground-water system. Finally, the water-budget equation relevant to the MSEA is based on the assumption that inflow to the aquifer from the Scioto River during periods of high stage equals outflow from bank storage to the Scioto River when river stage falls, so that net bank storage is zero.

Given the preceding assumptions, the major components of ground-water inflow and outflow to the outwash aquifer at the Ohio MSEA can be further subdivided into the following components:

Inflow

Infiltration of precipitation
Leakage from bedrock
Infiltration from Big Beaver Creek

Outflow

Discharge to Scioto River
Withdrawals from the DOE sanitary-supply wells[10]

On the basis of these water sources and sinks, a water budget for water year 1992 was calculated. Where data permitted, component values were compared to

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[10] In calculations of well withdrawals from the outwash aquifer, induced infiltration from the Scioto River is subtracted from the total measured well withdrawals. Therefore, induced infiltration is not included as a separate inflow term.

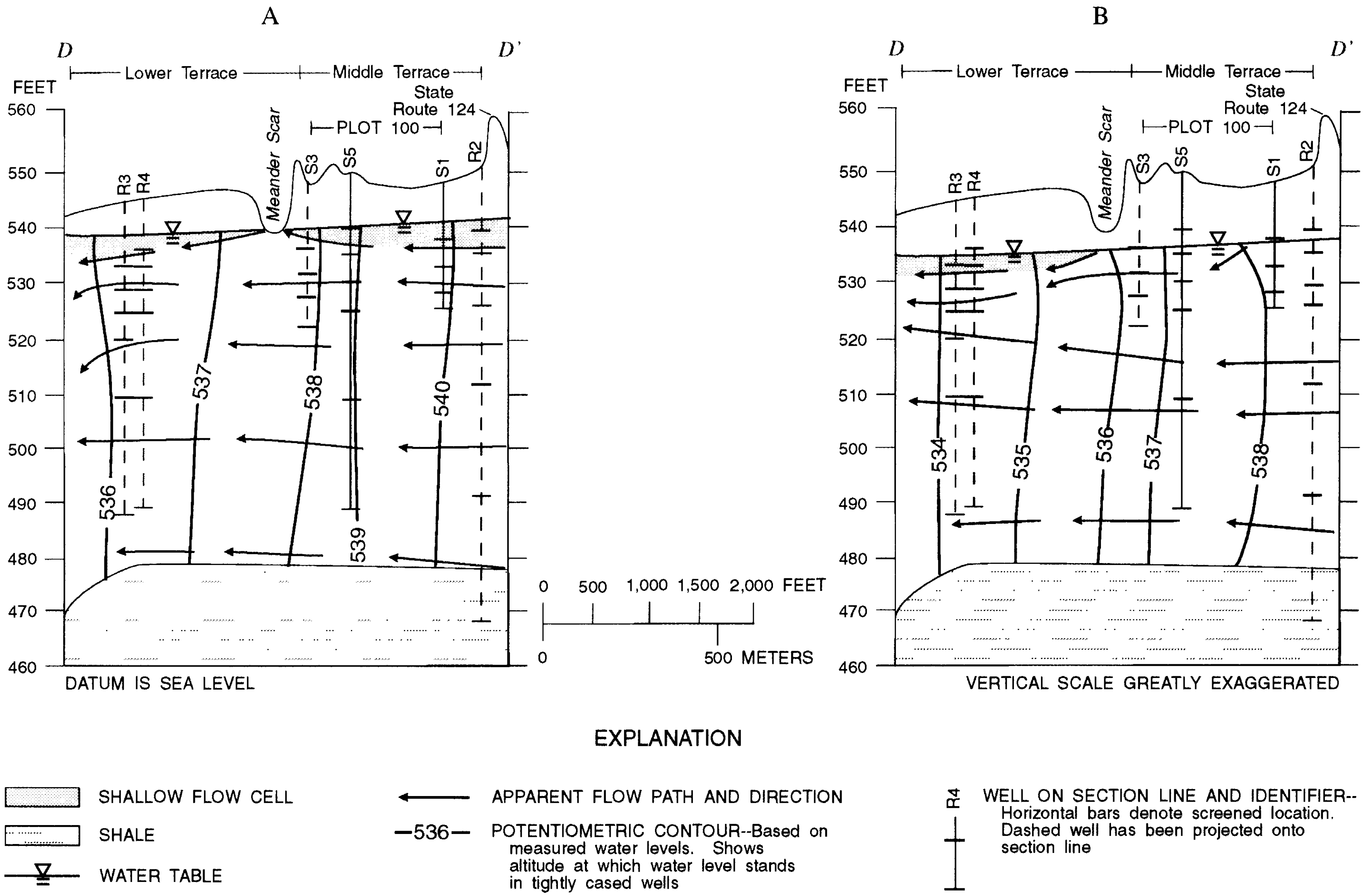


Figure 20. Potentiometric section along flow path through plot 100 (as shown on fig. 18), Ohio Management Systems Evaluation Area, during (A) June 3-5, 1991, and (B) July 1-3, 1991.

those for water year 1991. Ground-water withdrawal from DOE wells (fig. 2) was the only budget component that could be measured directly. Estimation of the other formula parameters is described in the following sections. Estimation of the parameters was done by use of more than one method, when possible, to corroborate results.

Infiltration of Precipitation

Heavy rains and flooding occurred throughout much of Ohio in December 1990 and January 1991, and above-normal monthly rainfall continued through April 1991. Thus, ground water was at near-record-high levels in January 1991, at the beginning of the study period (Shindel and others, 1992). Weather-system changes brought drought conditions in May of 1991 that continued through 1992. Thus, the data presented in this report represent one wet and one dry year in south-central Ohio. Total precipitation at Piketon in water years 1991 and 1992 was 43.32 and 38.33 in., respectively. The mean annual precipitation for south-central Ohio is 40.14 in. (Hendershot, 1990, p. 154).

The relation between precipitation and water levels beneath the middle terrace in the outwash aquifer is illustrated in figure 21. A muted water-level rise is apparent immediately after some of the precipitation events. The maximum immediate observed response to a precipitation event occurred on July 10, 1992, when water levels rose 1.2 in. in wells S5 and S10 in response to 1.55-in. of rain. If an effective porosity of 0.25 is assumed for the outwash aquifer and used as a multiplier for the observed water-level rise, then 0.30 in. of water reached the water table to cause the water-level rise. This calculation indicates that a maximum of 19 percent of the precipitation in the specified event reached the water table within 2.5 days.

Rapid infiltration of part of the precipitation at the site was also noted by Springer and others (1993) and Springer (1994), who applied a conservative bromide tracer to the surface of one-quarter of plot 100 (fig. 1) in spring 1991. They observed that low concentrations of bromide were present at the water table beneath the treatment area after significant precipitation events. However, mass-balance calculations show that a maximum of 4 percent of the total applied mass of bromide was detected at the water table after any one precipitation event and that less than 10 percent of the applied mass of bromide was detected in ground-water samples collected between application and September 1992. Instead, analysis of bromide in soil cores has shown that the bulk of the bromide is moving slowly through the soil matrix and that the center of the bromide mass was only 6 to 7.5 ft below land surface in February 1994, roughly 3.8 years after bromide application (unpublished data on file in the Environmental Chemistry Water Quality Analytical Laboratory at The Ohio State University, School of Natural Resources, Columbus, Ohio). Thus, the tracer tests and hydrograph analyses indirectly indicate that rapid infiltration of water and conservative solutes to the water table partially occurs along pathways of preferential flow through the unsaturated zone. The rest of the rainfall either infiltrates the unsaturated zone as matrix flow at the average rate of about 2 ft/yr or is lost as evapotranspiration during summer.

The amount of precipitation that infiltrates to the water table at the study area was estimated as the difference between the volume of precipitation and the volume of water lost as evapotranspiration. As previously stated, precipitation for water year 1992, as measured at the NWS station in Piketon, was 38.33 in. Monthly estimates of potential evapotranspiration (ET) were calculated by means of the modified Blaney-Criddle equation, as described in James (1988):

$$ET = K_{scs}K_tNP(T/K1+K2), \qquad (7)$$

where K_{scs} is the crop coefficient for corn (James, 1988, table C2, p. 432); K_t is equal to $K_3T + K_4$, where K_3 is a constant equal to 0.0173 (James, 1988, table 1.14, p. 34), K_4 is a constant equal to -0.314 (James, 1988, table 1.14, p. 34), and T is the mean temperature for the month, in degrees Fahrenheit, as obtained from the NWS weather station in Waverly, Ohio; N is the number of days in the month; P is the mean daily percentage of annual daytime hours for the month; K_1 is a constant equal to 100 (James, 1988, table 1.14, p. 34); and K_2 is a constant equal to 0 (James, 1988, table 1.14, p. 34).

Monthly temperature, precipitation, and ET for 1991 and 1992 are shown in table 8. ET was highest during April through September each year. Total ET for water year 1992 was 25.63 in. Subtraction of estimated ET from the amount of total precipitation for water year 1992 (38.33 in.) yields 12.70 in. of

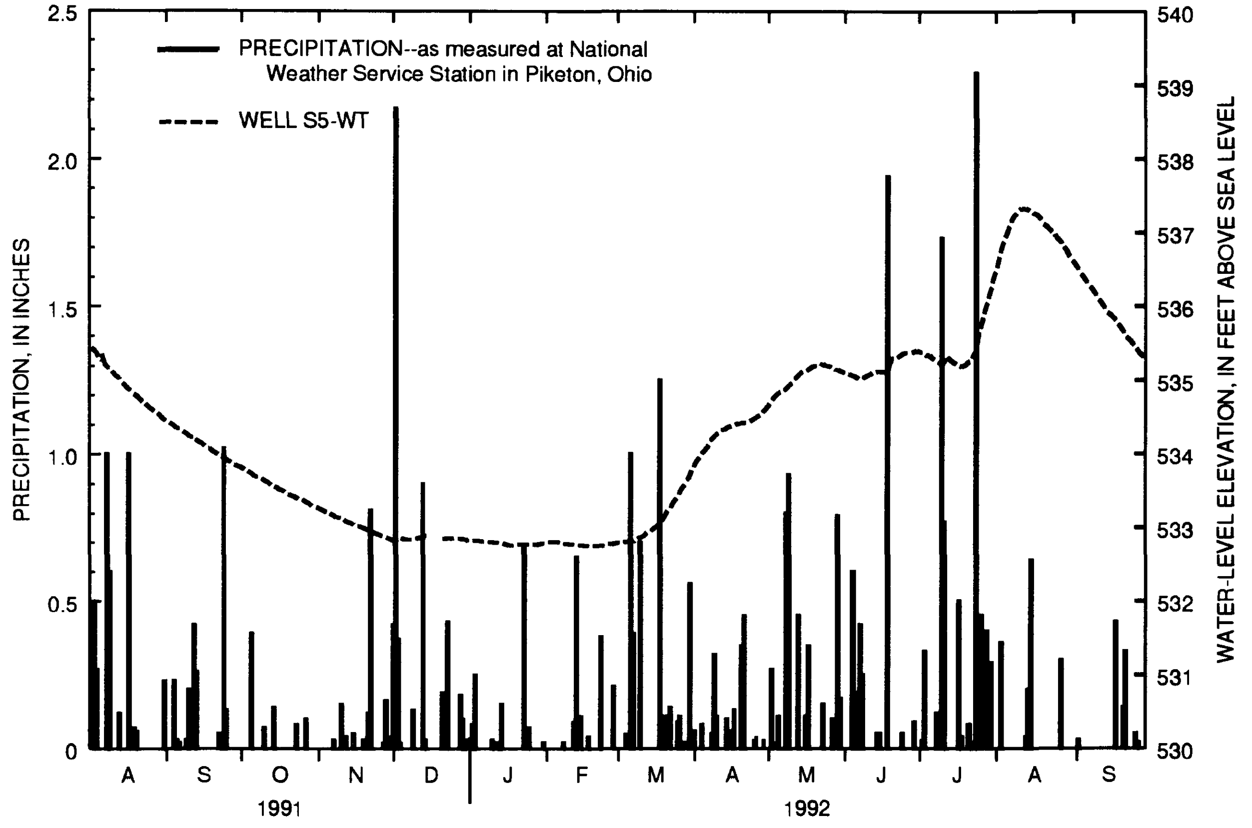


Figure 21. Hydrograph showing relation between precipitation and water level at well S5-WT, Ohio Management Systems Evaluation Area, August 1991-September 1992.

water (33 percent of total precipitation) that was available to infiltrate to the water table.

An infiltration amount of 12.70 in. is reasonable for water year 1992 when compared to recharge rates estimated for outwash deposits elsewhere in Ohio, as listed in table 9. One would expect lower recharge rates in the Scioto River Valley of Franklin County than in Pike County because of the greater amount of urbanization in Franklin County and the presence of overlying till layers in southern Franklin County.

Leakage from Bedrock

To calculate leakage of ground water from the bedrock along the sides and bottom of the buried valley, the investigators determined K of the bedrock, hydraulic gradient across the bedrock-outwash contact, and area of the bedrock valley walls and valley floor. K of the bedrock was estimated by means of slug-test analysis at well R6-BR. Analysis of recovery of water levels in this well after pumping yielded a horizontal K (K_h) of 3.0 x 10^{-3} ft/d (Finton, 1994), a reasonable value to use for determining leakage from the valley walls. To calculate leakage from the valley floor, the investigators used a vertical K (K_v) estimated to be one-tenth of the K_h, or 3.0 x 10^{-4} ft/d.

Data for hydraulic gradient across the bedrock-outwash contact was available only for along the valley floor. The maximum gradient measured between the bedrock and the deep outwash aquifer was 0.033, observed at well cluster R2 in August 1991. This maximum gradient was used in the calculation of bedrock leakage to produce a maximum rate of bedrock discharge.

The area of the east valley wall (A_w) in contact with the saturated part of the outwash aquifer was estimated as being equal to the length of the east valley wall within the study boundaries (1 x 10^4 ft) multiplied by the saturated thickness of the aquifer (approximately 55 ft), or a total of 5.5 x 10^5 ft^2. The area of the bedrock valley floor (A_f) was calculated as being 6.52 x 10^7 ft^2, equal to the surface area of the study area (6.92 x 10^7 ft^2) minus the area adjacent to Big Beaver Creek where downflow occurs. The area

Table 8. Estimation of evapotranspiration and infiltration of precipitation at the Ohio Management Systems Evaluation Area

[Monthly evapotranspiration = (crop coefficient)(Kt)(number of days in month)(mean daily percentage of annual daytime hours)(mean temperature)(1/100 + 0), as estimated by the modified Blaney-Criddle method (James, 1988). Crop coefficient from James (1988, p. 432, table C2); Kt = 0.0173(mean temperature) + -0.314; mean temperature derived from the National Weather Service data from station in Waverly, Ohio; constants derived from James (1988, p. 34, table 1.14)]

Water year	Month	Crop coefficient	Kt	Number of days	Mean daily percentage of annual daytime hours	Monthly mean temperature (degrees Fahrenheit)	Evapotranspiration (inches)	Precipitation (inches)	Infiltration (inches)
1991	January	0.05	0.1998	31	0.22	29.7	0.020	3.09	3.07
	February	.05	.2742	28	.24	34.0	.031	3.16	3.13
	March	.25	.4316	31	.27	43.1	.389	6.20	5.81
	April	.44	.6410	30	.30	55.2	1.401	3.81	2.41
	May	.58	.8866	31	.32	69.4	3.540	1.76	-1.78
	June	.92	.9281	30	.34	71.8	6.254	2.69	-3.56
	July	1.08	.9818	31	.33	74.9	8.124	2.44	-5.68
	August	1.00	.9333	31	.31	72.1	6.467	4.29	-2.18
	September	.85	.8001	30	.28	64.4	3.679	2.74	-.94
1992	October	.25	.6064	31	.25	53.2	.625	1.23	.60
	November	.05	.3607	30	.22	39.0	.046	1.58	1.53
	December	.05	.2880	31	.21	34.8	.033	5.99	5.96
	January	.05	.1912	31	.22	29.2	.019	1.27	1.25
	February	.05	.3036	28	.24	35.7	.038	1.62	1.58
	March	.25	.3728	31	.27	39.7	.310	4.64	4.33
	April	.44	.5458	30	.30	49.7	1.074	2.37	1.30
	May	.58	.6825	31	.32	57.6	2.262	4.66	2.40
	June	.92	.8070	30	.34	64.8	4.907	3.93	-.98
	July	1.08	.9333	31	.33	72.1	7.435	7.76	.33
	August	1.00	.8486	31	.31	67.2	5.480	2.00	-3.48
	September	.85	.7638	30	.28	62.3	3.398	1.28	-2.12
						Water year sum	25.627	38.33	12.70

of downflow is assumed to be approximately 400 ft wide by 1 x 10^4 ft long (4.0 x 10^6 ft^2).

Estimates of ground-water leakage from the bedrock valley wall (Q_w) and valley floor (Q_f) were calculated by means of Darcy's equation (eq. 1),

$$Q_w = K_h A_w dh/dl$$

and

$$Q_f = K_v A_f dh/dl,$$

resulting in annual leakage amounts of 2.0 x 10^4 ft^3 and 2.4 x 10^5 ft^3, respectively. Summing of valley-wall and valley-floor leakage amounts produces a total bedrock leakage amount that can then be divided by the surface area within the study boundaries (6.92 x 10^7 ft^2), resulting in an annual bedrock contribution to the outwash aquifer of approximately 0.045 in.

Table 9. Recharge rates for selected areas of Ohio and estimation methods used

Recharge rate (inches per year)	Area	Estimation method	Reference
12	Dayton	Ground-water budget calculations, ground-water recession method, flow-model calibration	Dumouchelle and others (1993)
9.4	Scioto River Valley, southern Franklin County	Base-flow separation of stream hydrographs	Stowe (1979)
9	Scioto River Valley, southern Franklin County	Digital flow model	Sedam and others (1988)

Infiltration from Big Beaver Creek

By measurement of streambed K, vertical hydraulic gradients in the streambed, and stream area of Big Beaver Creek, it is possible to use Darcy's equation (eq. 1) to estimate reasonable minimum and maximum volumes of water infiltrating from the creek into the outwash aquifer. K_v of the Big Beaver Creek streambed was determined by seepage-meter tests at three locations (fig. 2) in July 1991 and August 1992. A total of nine tests at these locations yielded K_v's ranging from 0.089 to 19.9 ft/d; the mean for the three locations was 6.10 ft/d. Seepage meters could not be driven into the streambed north of well R1, where the creek flows directly over shale. This reach of the creek was assumed to be impermeable relative to the reaches where the streambed is composed of gravel. The reach of the creek between wells R1 and R8 had a much greater K_v than the streambed adjacent to either well location (table 7). Vertical hydraulic gradients in the streambed, as measured during the seepage-meter tests, ranged from 3.3×10^{-3} to 0.088. Although the zone below the gravel reach of the streambed was saturated at the time of the seepage-meter tests (as indicated by flow of water into piezometers screened 3 ft below the streambed), this zone likely becomes variably saturated as ground-water levels decline (therefore, the moisture content of sediments beneath the stream varies areally and with time).

Minimum infiltration estimates were obtained by means of the minimum measured streambed K_v (0.089 ft/d) and the minimum vertical hydraulic gradient (3.3×10^{-3}). Maximum estimates of infiltration from Big Beaver Creek were made by means of the maximum measured K_v (19.9 ft/d) and the maximum measured vertical gradient (0.088). Stream width and stage data from three time points in 1991, as given in table 10, were used to determine the average stream area to be substituted into Darcy's equation. These data, which were obtained 50 to 200 ft from the Cemetery Road bridge, were then visually compared to a discontinuous stage hydrograph of Big Beaver Creek (fig. 16) (measured at the Cemetery Road bridge over a large pool that contains water even during periods of no flow) to determine that stream stage conditions in June 1991 can be considered average for the area. (August 1991-March 1992 data are missing from the Big Beaver hydrograph; however, it is reasonable to assume that flows in Big Beaver Creek would generally be low at this time, as were flows observed in the Scioto River (fig. 14) at this time.) Thus, an average stream width of 10 ft (similar to the width observed in June 1991) was assumed. Stream length in the study area is 12,000 ft; if 3,250 ft is subtracted (the reach length that flows over impermeable shale) and the resulting length (10,250 ft) is multiplied by the average stream width, then a stream area of 88,000 ft^2 is obtained.

Substituting the above values for K_v, area, and hydraulic gradient into Darcy's equation (eq. 1) and solving for discharge (Q) results in minimum and maximum infiltration rates of 0.0016 in/yr and 9.9 in/yr, respectively. The mean of these two values is 5.0 in/yr. Even the maximum infiltration value is only a small percentage (0.08 percent) of the total volume of water flowing in the stream under average conditions as exhibited in June 1991.

Stream-infiltration ranges calculated by use of Darcy's equation were compared to observed volumes of recharge to the aquifer determined by means of a ground-water hydrograph separation technique (modified from Johansson, 1987, p. 183-198; Lipinski, 1985; Weiss and Razem, 1980, p. 9-10). This method was used to calculate total recharge to the outwash aquifer from all sources. Water-level rises above a recession curve on each of the water-table well

Table 10. Stage, width, and discharge of Big Beaver Creek near the Ohio Management Systems Evaluation Area in April, June, and November 1991

Date	Stage of Big Beaver Creek (feet above streambed)	Width of Big Beaver Creek (feet)	Discharge (cubic feet per second)
April 1991	1.3	44	45
June 1991	.52	6.1	.67
November 1991	0	0	0

hydrographs were summed for the year, then multiplied by an assumed effective porosity of 0.25 to estimate the recharge by infiltration for water year 1992 (fig. 22). Because the extension of recession curves is subjective, no recession curve was drawn beyond that described by actual water-level data. Instead, a horizontal line was drawn from the base of each documented recession, and water-level rises were measured between the horizontal line and the water-level peak. This procedure had the effect of artificially minimizing the resulting water-level rises and subsequent recharge rates because, in reality, ground water would continue to flow out of the area in the absence of recharge, causing further declines in ground-water levels with time.

Results of the ground-water hydrograph separation are shown in table 11. Annual recharge was not calculated for wells R3 through R7 because of bank-storage effects in these wells caused by their proximity to the Scioto River. Annual recharge rates range from 14.82 to 17.97 in. and reflect all water recharging the outwash aquifer, including infiltrating precipitation, leakage from bedrock, and infiltration from Big Beaver Creek. Recharge rates are highest at wells R1 and R8, adjacent to Big Beaver Creek. Recharge rates are lowest at well S10, downgradient from the reach of Big Beaver Creek that has a shale streambed. Subtraction of infiltrating precipitation (12.70 in.) and the mean estimate of bedrock leakage (0.045 in.), where appropriate, from the total recharge rates determined by ground-water hydrograph separation reveals that 2.08 to 5.27 in. (or 14 to 29 percent of the total annual recharge for water year 1992) is from infiltration from Big Beaver Creek. Unfortunately, no wells are adjacent to the most permeable streambed reaches to ascertain recharge from Big Beaver Creek in these areas. For this reason, and because the use of horizontal recession curves for the ground-water hydrograph separation minimizes the total recharge rates calculated by hydrograph separation, the recharge from Big Beaver shown in table 11 should be considered minimum rates. These rates are between the minimum (0.0016 in/yr) and maximum (9.9 in/yr) infiltration estimates calculated by use of Darcy's equation. The mean estimate of infiltration from hydrograph separation is 3.64 in/yr, compared to the mean estimate of 5.0 in/yr calculated from Darcy's equation. The latter value was used in the 1992 water budget summary presented at the end of this section because of the known conservative nature of the mean estimate derived from hydrograph separation.

Discharge to the Scioto River

Discharge from the outwash aquifer to the Scioto River was calculated by means of two methods: stream-hydrograph separation and a flux calculation. For the stream-hydrograph separation method, volumes of water discharged as base flow from the outwash aquifer into the Scioto River were calculated by means of historical discharge data (water years 1931-92) from the Higby gaging station (03234500) approximately 14 mi upstream from the MSEA. Base-flow separation was done by use of HYSEP software (Sloto, 1991), a USGS adaptation of programs originally developed by Pettyjohn and Henning (1979). HYSEP calculates percentage of discharge from the aquifer to the stream by means of fixed-interval, sliding-interval, and local-minima methods. The median annual mean base flow at Higby is 5.58 in. (0.266 Mgal/d/mi^2), or 47 percent of streamflow. The annual mean base flow calculated for water year 1992, 4.77 in. (0.227 Mgal/d/mi2), was less than the median annual mean. In fact, 59 percent of the calculated annual mean base flows during water years 1931-92 at

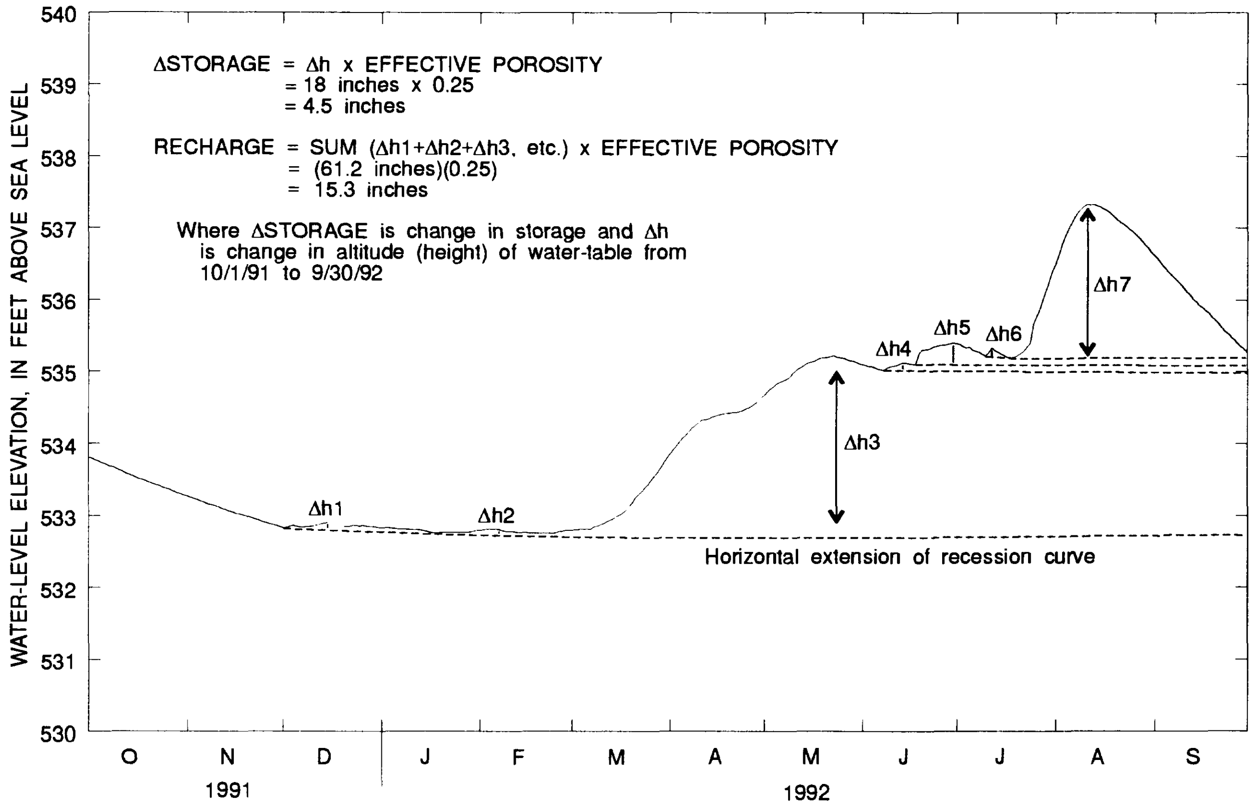


Figure 22. Estimation of infiltration of precipitation and ground-water storage by use of hydrograph-separation methods, well S5-WT, Ohio Management Systems Evaluation Area, October 1, 1991-September 30, 1992.

Higby were greater than the 1992 annual mean base flow, an indication that 1992 was indeed a mild drought year.

The base flow to the Scioto River calculated from data at Higby is likely much lower than the discharge to the Scioto River at the MSEA because of the absence of surficial till deposits in the Piketon area and the presence of tills in much of the basin upgradient from Higby. Base flow is influenced by recharge, which is approximately 2 to 5 in. less in till-covered parts of the Scioto River Basin than in outwash-covered areas (Sedam and others, 1988; Stowe, 1979). Adding a correction factor of 2 to 5 in/yr to the annual mean base flow determined at Higby for water year 1992 yields an estimated discharge to the Scioto River at the MSEA of 6.8 to 9.8 in/yr.

The flux method of calculating ground-water discharge to the Scioto River uses Darcy's equation (eq. 1) and is based on the assumption that all water present in the study area discharges to the Scioto River. However, in this case, the cross-sectional area (A) is measured parallel to the Scioto River (9,000 ft in length) and does not include the stream length affected by the cone of depression created by the DOE well field. In addition, a saturated thickness of 50 ft is assumed. A horizontal hydraulic gradient of 1.1×10^{-3} is used, as measured in April 1992, and K is assumed to be equal to 401 ft/d (table 3, multiple-well aquifer test near well R5-WT). Flux calculations based on these parameters indicate that 7.3×10^7 ft^3 of water was discharged from the unconsolidated aquifer beneath the study area to the Scioto River in water year 1992. This is equivalent to 12.6 in. of discharge per year.

For the 1992 water budget summary presented at the end of this section, the mean of all three values (6.8 in/yr, 9.8 in/yr and 12.6 in/yr) was used as the estimate of discharge of ground water to the Scioto River. This mean value is 9.7 in/yr.

Withdrawals From Wells

The only significant ground-water withdrawals from the site are at the DOE wells at the southwest corner of the farm. As determined from pumping records supplied by the DOE (unpublished data on file at the Columbus, Ohio, office of the USGS), the DOE withdraws approximately 9.22 x 10^7 ft^3 (6.9 x 10^8 gal) of water per year from three wells (fig. 2) for sanitary supply and process water. Divided by the area of the MSEA, discharge due to well withdrawals is 16.0 in/yr. In addition to water withdrawn from the aquifer, this amount includes induced infiltration of water from the Scioto River. Nortz (1991) used the numerical model, MODFLOW (McDonald and Harbaugh, 1988), to determine percentage of induced infiltration in discharge to pumped wells approximately 2.2 mi north of the DOE well field, along the Scioto River. Nortz concluded that if a well of construction similar to that of the DOE wells and located 200 ft to 1,000 ft from the Scioto River was pumped at a rate of 1.0 x 10^6 ft^3/d, then 50 to 70 percent of the water pumped from the well would be induced infiltration. If 60 percent induced infiltration is assumed for the DOE wells at the MSEA, only 6.4 in. of the 16.0 in. of water pumped from the DOE wells is actually discharge of ground water from the outwash aquifer.

Storage

For a long-term water budget under steady-state aquifer conditions, the storage term is assumed to cancel from the budget equation. However, for an annual budget, storage is calculated by use of well-hydrograph separation methods. A storage value for water year 1992 was estimated from each well hydrograph by subtracting the ground-water altitude at the beginning of the water year (October 1, 1991) from the ground-water altitude at the end of the water year (September 30, 1992) and multiplying the difference by an assumed effective porosity of 0.25. Storage values for the wells are summarized in table 11. Precipitation late in 1992 produced partial recovery from the drought, causing a positive change in storage of 2.7 to 4.6 in. in water year 1992. The mean of the storage values for the six wells presented in table 11 (4.2 in.)

Table 11. Areal variation in recharge and ground-water storage at Ohio Management Systems Evaluation Area, October 1, 1991-September 30,1992

[Well locations are shown in figure 2. Wells are listed in order of increasing distance from Big Beaver Creek. Total annual recharge was determined by separation of ground-water hydrograph as demonstrated in figure 22; infiltration of precipitation was estimated by subtraction of evapotranspiration from precipitation; bedrock leakage was estimated to be 0.045 inches by use of the Darcy equation; recharge from Big Beaver Creek was calculated by subtraction of infiltrating precipitation and bedrock leakage from total annual recharge; annual change in storage was estimated as difference between water level at beginning of water year and water level at end of water year multiplied by effective porosity]

Well number	Total annual recharge (inches)	Annual infiltration of precipitation and bedrock leakage (inches)	Annual recharge from Big Beaver Creek (inches)	Annual change in storage (inches)
R1-WT	17.63	[a]12.70	4.93	4.6
R8-WT	17.97	[b]12.70	5.27	2.7
R2-WT	16.71	12.74	3.97	4.6
S10-WT	14.82	12.74	2.08	4.1
S14-WT	15.75	12.74	3.01	4.5
S5-WT	15.30	12.74	2.56	4.5

[a] A consistent downward component of flow is measured from the outwash aquifer to the bedrock aquifer at well R1; therefore, bedrock leakage is assumed to be nonexistent at this location.

[b] Although no bedrock well exists at this location, the flow regime is assumed to be similar to that at well R1 and bedrock leakage is estimated to be nonexistent.

was used as the estimate of the change in storage for the 1992 water-budget summary presented below.

Summary of Ground-Water Budget

Ground-water-budget components needed to solve equation 6 are summarized in table 12. When summed, the percent difference between water-budget inputs and outputs plus storage is 14 percent, which is within the range of error on individual component estimates.

According to the calculated budget, 72 percent of recharge to the outwash aquifer at the Ohio MSEA is derived from infiltrating precipitation, 0.2 percent is from bedrock leakage, and 28 percent is from infiltration of surface water from Big Beaver Creek. Twenty-four percent of the water recharged to the aquifer in water year 1992 (17.7 in.) was stored in the ground-water system at the end of the water year. Of the 16.1 in. of water discharged from the aquifer, 60 percent exited as base flow to the Scioto River, and 40 percent was withdrawn at the DOE well field.

WATER QUALITY

The goals of the water-quality data-collection program were to (1) characterize spatial and temporal variability in major-ion chemistry of ground water, with emphasis on constituents that could potentially affect the fate and transport of agrochemicals; (2) identify factors controlling ground-water chemistry; and (3) use the chemical data to help corroborate the conceptual flow model. To achieve this end, the investigators collected data on ground-water and surface-water quality.

Results of analyses of water-quality samples collected by USGS personnel at the MSEA are listed in table 13 (at end of report). As a preliminary overview of major-ion geochemistry in the study area,

Table 12. Summary of ground-water budget for the Ohio Management Systems Evaluation Area, October 1, 1991-September 30, 1992

[Infiltration of precipitation was calculated by subtracting evapotranspiration from total precipitation; leakage from bedrock and infiltration from Big Beaver Creek were calculated by use of Darcy's equation (1856); discharge to the Scioto River was estimated as the mean value of stream-hydrograph-separation calculations and a flux calculation based on Darcy's equation; withdrawals from Department of Energy (DOE) well field were summed from pumping records; ground-water storage was estimated as mean of annual storage values presented in table 11]

	Water source or destination	Inches per unit study area
Inflow:	Infiltration of precipitation	12.7
	Leakage from bedrock	.045
	Infiltration from Big Beaver Creek	5.0
	Sum	**17.7**
Outflow:	Discharge to Scioto River	9.7
	Withdrawals from DOE well field[a]	6.4
	Sum	**16.1**
Storage:	–	**4.2**

[a] Corrected for induced infiltration from the Scioto River.

relative percentages of major ions in each sample were plotted on Piper diagrams (fig. 23). To examine variation in chemical composition of ground water with depth, the investigators constructed boxplots for each property and constituent of interest (fig. 24). The boxplots allow visual comparison of median concentration, interquartile range, and total range of measured values for analyses of the following waters: (1) surface water (samples collected from Big Beaver Creek and the Scioto River), (2) shallow water from the outwash aquifer (samples collected from the multilevel wells at ports 3, 4, 5, or 6—whichever was the uppermost saturated port at the time of sampling), (3) water from intermediate zones of the outwash aquifer (collected from approximately 40 ft below land surface at port 2 of the multilevel wells), (4) deep water from the outwash aquifer (samples collected from port 1 of the multilevel wells, at approximately 60 ft below land surface, and (5) bedrock water collected from wells completed in the Ohio Shale. Grouping of samples by depth is valid for this aquifer because flow paths are primarily horizontal. Boxplots for each depth summarize data from all three sampling rounds.

Surface Water

Properties and constituents of surface water were determined to evaluate the effect of ground-water/surface-water interactions on quality of water in the outwash aquifer and in the streams bounding the study area. Results of analyses of samples collected at the two surface-water sites are included in table 13. No water-quality data are presented for Big Beaver Creek in November 1991 because the stream was not flowing at that time. The Piper diagram (fig. 23) shows that Big Beaver Creek waters are calcium magnesium sulfate bicarbonate type and that Scioto River waters are generally calcium magnesium bicarbonate type. In general, major ions in both streams became more concentrated from April to November throughout the 1991 growing season, as discharge decreased (table 13). The exception was silica, the concentration of which decreased with time. Dissolved oxygen and total iron concentrations also decreased with time.

Big Beaver Creek had much lower alkalinity and specific conductance, as well as lower concentrations of calcium, magnesium, sodium, sulfate, chloride, dissolved solids, nutrients, pesticides, and organic carbon than the Scioto River. Dissolved solids and nutrient concentrations in Big Beaver Creek also were less than those in the shallow outwash aquifer. The maximum concentration of nitrates in Big Beaver Creek was detected in April, at 0.72 mg/L of nitrate-nitrogen. The only herbicide or herbicide-degradation product found in Big Beaver Creek (desethyl-atrazine) was also detected in April but was present near the detection limit at a concentration of 0.08 µg/L (detection limit = 0.05 µg/L).

In contrast, the maximum nutrient concentrations detected in the Scioto River were found in the fall (nitrate, 3.6 mg/L as N; dissolved phosphorous, 0.32 mg/L), and the greatest herbicide concentrations were generally detected in June (atrazine, 2.7 µg/L; alachlor, = 0.3 µg/L). Atrazine was detected during all three sampling rounds, as was metolachlor. Metolachlor concentrations were greatest in November, at 1.9 µg/L. (It should be noted that metolachlor was not applied at the MSEA.)

Ground Water

Distinguishing characteristics, as concluded from the Piper and boxplot diagrams, are discussed for each aquifer in the paragraphs that follow. Water-quality trends with depth within the outwash aquifer also are discussed, as are areal and temporal variations in ground-water quality.

Bedrock Aquifer

Distinguishing Characteristics

Several differences in the aqueous chemistry of bedrock and outwash waters can be discerned by use of the Piper diagram (fig. 23) and boxplots (fig. 24). Bedrock waters are classified as calcium sodium chloride waters in contrast to the calcium magnesium bicarbonate waters from the shallow part of the outwash aquifer. In addition, bedrock waters had specific conductances and concentrations of sodium, potassium, chloride, and dissolved solids that were more than double those measured in outwash-aquifer waters. The bedrock waters also had somewhat higher median calcium and magnesium concentrations than the outwash waters, somewhat lower alkalinity, and much lower sulfate concentrations. The median sulfate concentration in bedrock waters was 1.8 mg/L, compared to a median sulfate concentration of approximately 72 mg/L in the deep outwash. Nitrate was not

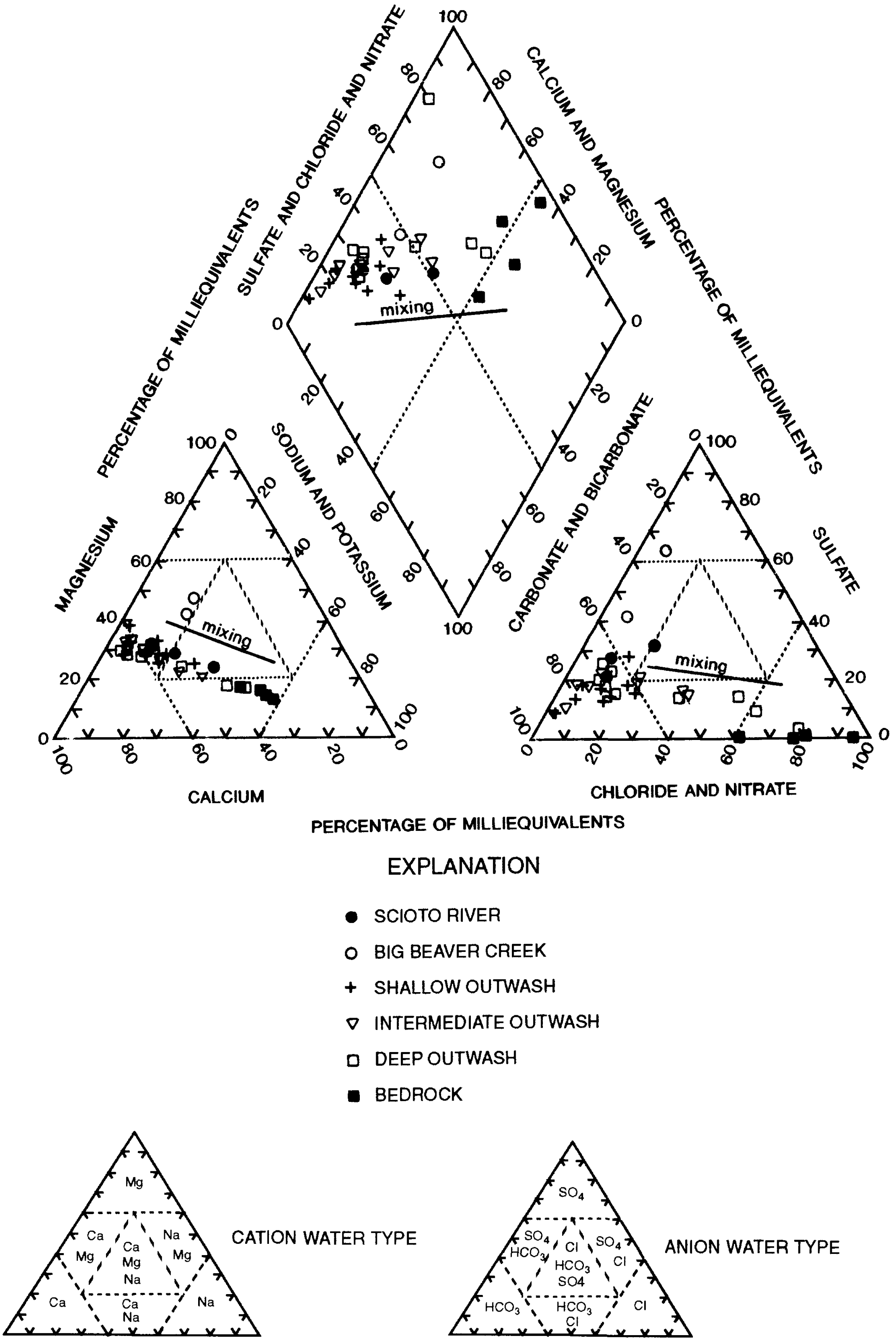
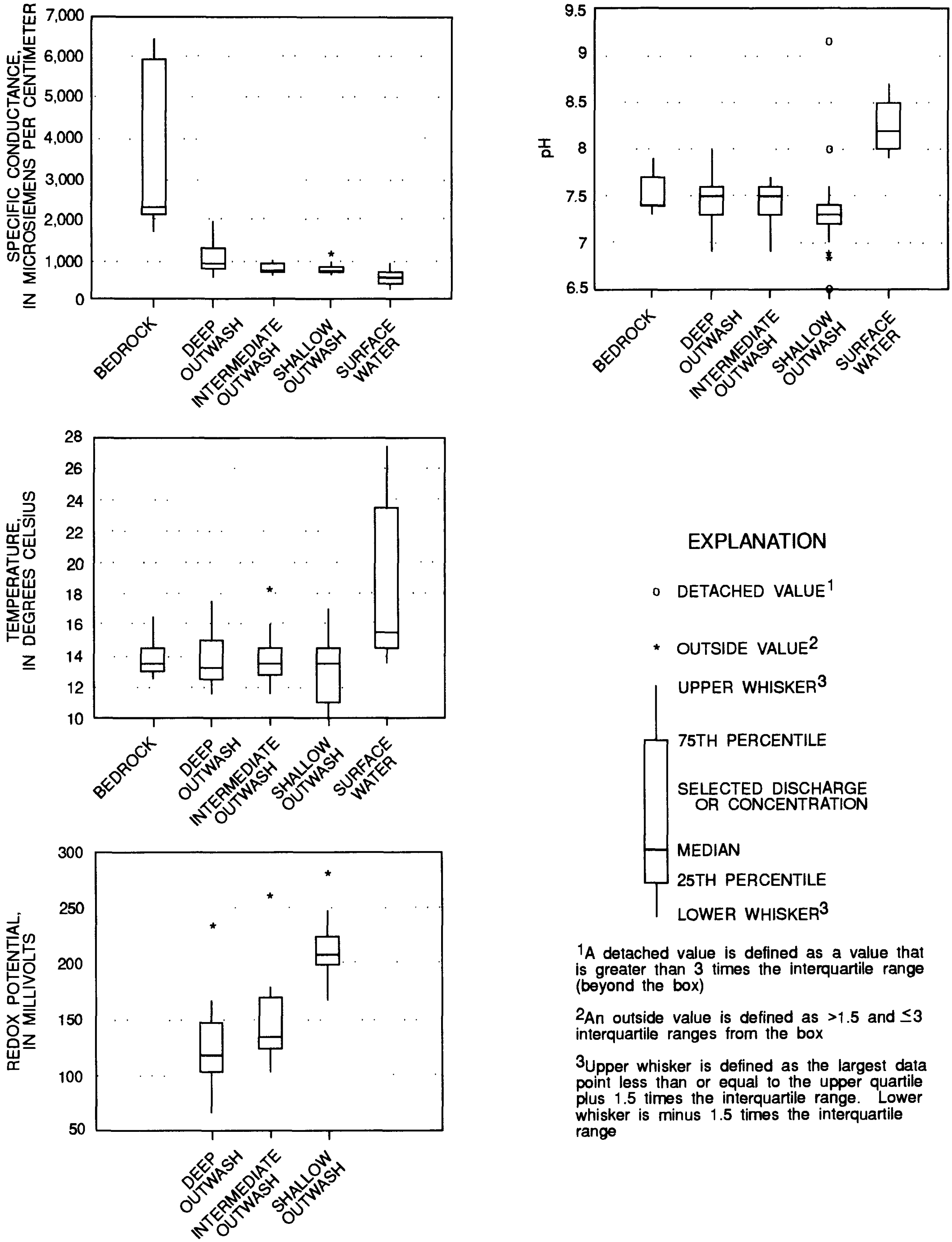


Figure 23. Piper diagram showing major cation and anion percentages of ground-water samples (November 1991) and surface-water samples (April - November 1991) collected at the Ohio Management Systems Evaluation Area.



[1] A detached value is defined as a value that is greater than 3 times the interquartile range (beyond the box)

[2] An outside value is defined as >1.5 and ≤3 interquartile ranges from the box

[3] Upper whisker is defined as the largest data point less than or equal to the upper quartile plus 1.5 times the interquartile range. Lower whisker is minus 1.5 times the interquartile range

Figure 24. Water quality as a function of depth at the Ohio Management Systems Evaluation Area, October 1, 1991-September 30, 1992.

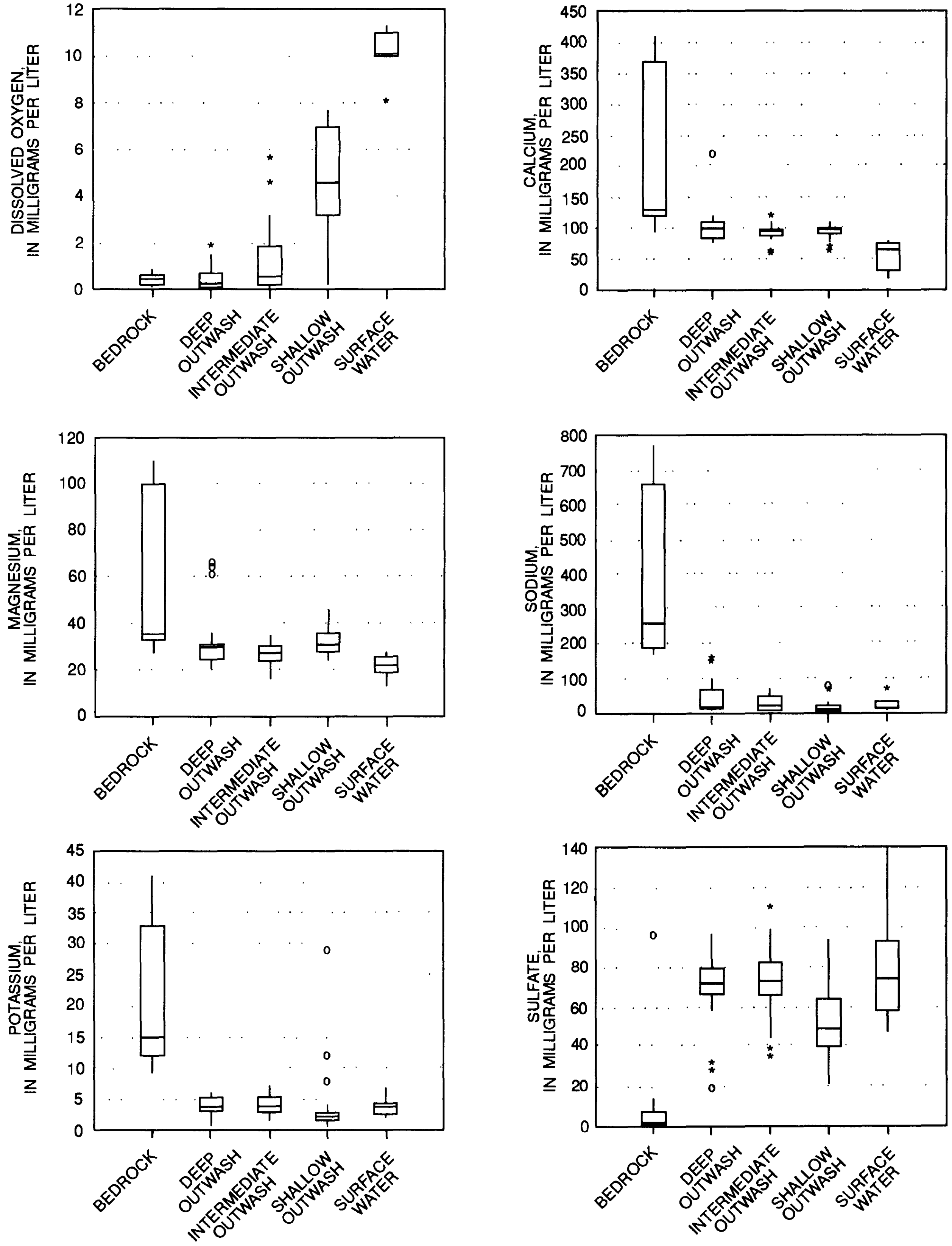


Figure 24. Water quality as a function of depth at the Ohio Management Systems Evaluation Area, October 1, 1991-September 30, 1992--Continued.

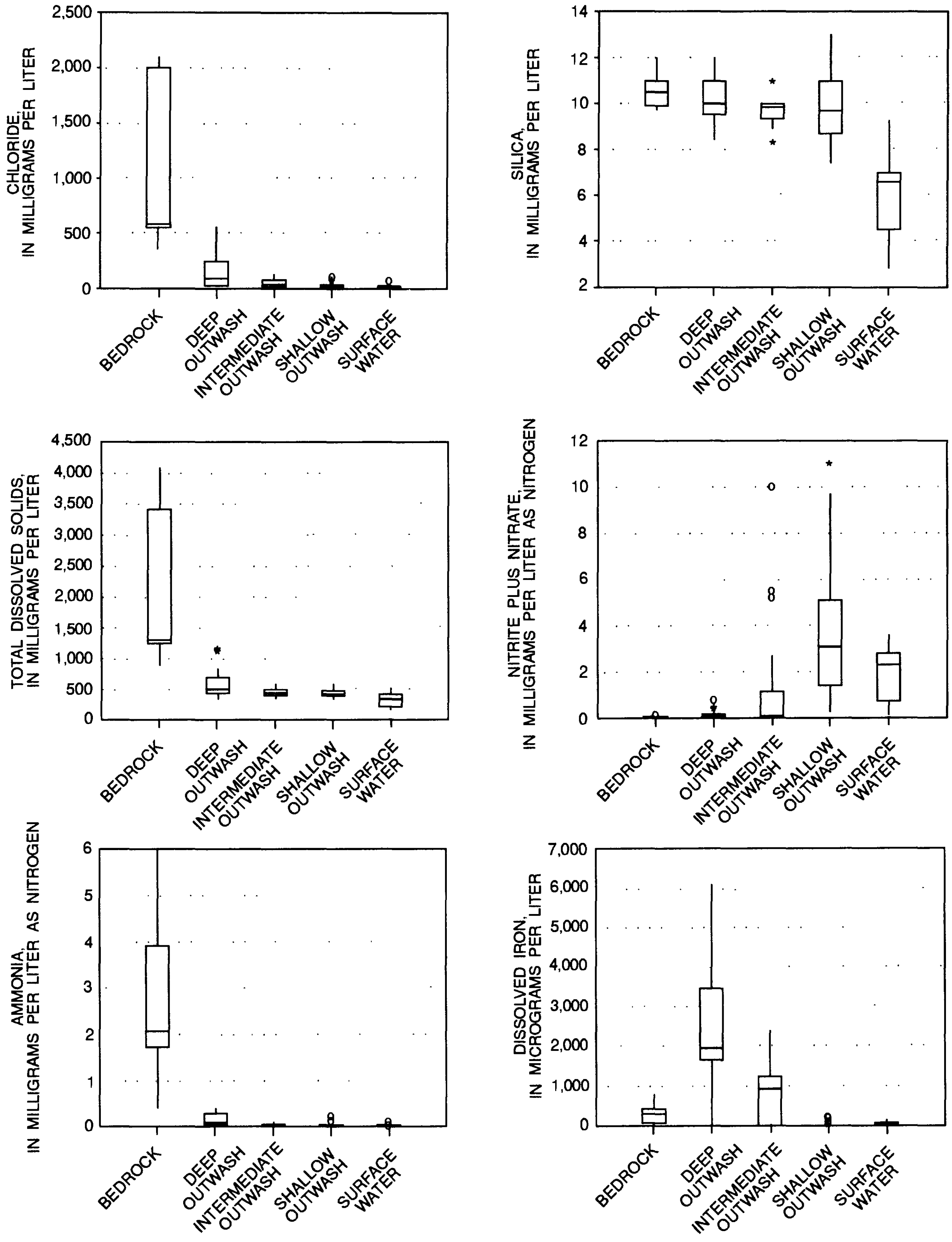


Figure 24. Water quality as a function of depth at the Ohio Management Systems Evaluation Area, October 1, 1991-September 30, 1992--Continued.

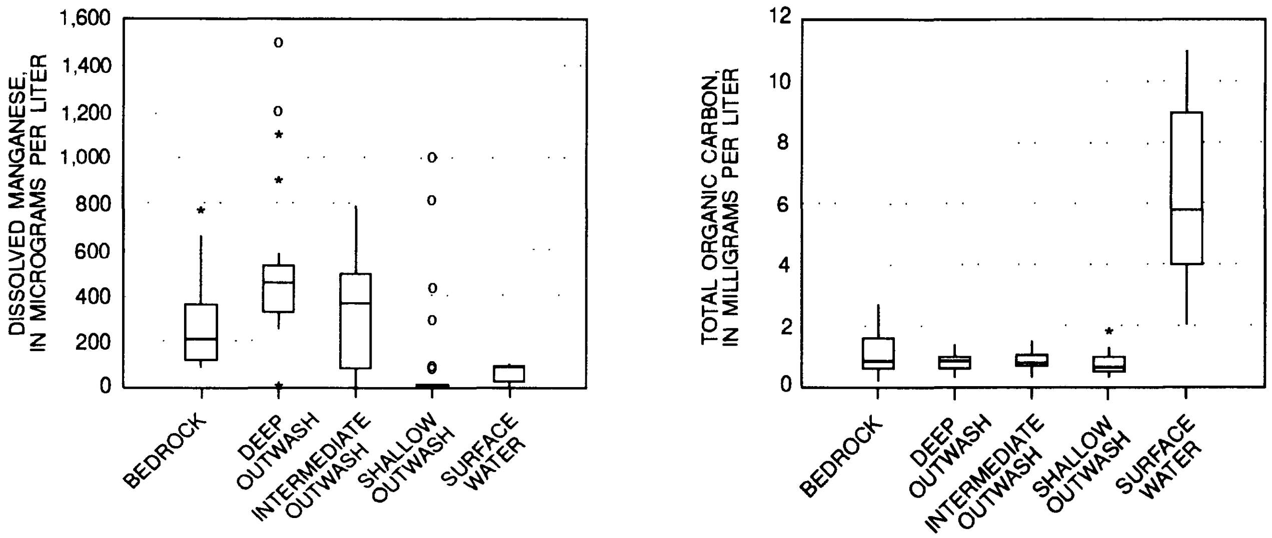


Figure 24. Water quality as a function of depth at the Ohio Management Systems Evaluation Area, October 1, 1991-September 30, 1992--Continued.

detected in the bedrock aquifer; however, ammonia, a reduced form of nitrogen, was detected at a maximum concentration of 6.0 mg/L. In contrast, the maximum concentration of ammonia found in the outwash aquifer was 0.21 mg/L.

Areal Differences and Temporal Variations

The large interquartile range on some of the boxplots of bedrock data (fig. 24) can be attributed largely to areal differences in constituent concentrations within the bedrock aquifer. Bedrock water from well R2-BR consistently had greater specific conductance and greater concentrations of dissolved solids, chloride, and major cations than did water from other bedrock wells (table 13). Well R1-BR consistently produced water with the least dissolved solids relative to the other bedrock wells, and water samples from wells R6-BR and R7-BR were intermediate in dissolved solids concentration. Areal differences in bedrock water quality were much more prevalent than temporal variations in constituent concentrations. For example, the largest absolute difference in concentration due to temporal variation was a 22-percent difference in potassium concentration at well R2-BR, whereas the percentage difference in potassium concentration due to areal differences (calculated between wells R2-BR and R1-BR) was 127 percent.

Outwash Aquifer

Distinguishing Characteristics

Shallow outwash waters generally are classified as calcium magnesium bicarbonate type. Intermediate and deep outwash waters in the northeastern part of the study area have a composition that is transitional between calcium magnesium bicarbonate and calcium sodium chloride (or bedrock) waters (fig. 23). Deep outwash waters in this area closely approach the composition of bedrock water. Intermediate and deep waters beneath the southern part of the study area are similar in composition to the calcium magnesium bicarbonate waters of the shallow outwash aquifer.

A third distinct water type, calcium magnesium chloride, was found in the deep zone of the outwash aquifer at well S5. The relative percentage of chloride ions in this water (76 to 88 percent of total anions) was comparable to that of bedrock waters; however, the dominant cations in samples from well S5-MP1 were calcium and magnesium, in contrast to calcium and sodium (the dominant cations in bedrock waters). Water from well S5-MP1 was also characterized by anomalously low concentrations of sodium, potassium, and sulfate, and anomalously elevated concentrations of magnesium, calcium, iron, and manganese compared to the rest of the deep outwash aquifer. Magnesium, calcium, and sulfate concentrations at this location were intermediate between typical ion concentrations of the outwash and bedrock aquifers. Even though calcium magnesium chloride waters were found at only one location, results of chemical analyses from well S5-MP1 were consistent during all three sampling rounds, indicating that this is an actual anomaly and not a result of sampling error. In addition, cation/anion balances were within ±10 percent as calculated by Hem's method (1989, p. 164), indicating that no significant errors were associated with the analytical results.

Spatial Differences and Temporal Variations

As indicated by the variation in dominant ions discussed above, water quality of the outwash aquifer is characterized by substantial spatial differences. These differences can be grouped into two types: (1) properties and constituents found at much different levels in the northeastern part of the study area than elsewhere at the MSEA (includes specific conductance, dissolved solids, chloride, sodium, potassium, and alkalinity), and (2) properties and constituents found at different levels near the streams than elsewhere in the aquifer (includes Eh, dissolved oxygen, iron, and manganese in the intermediate and deep parts of the aquifer).

Differences in water quality with depth also were noted, as is illustrated by the boxplots (fig. 24). Differences with depth were of two types: (1) levels of properties and constituents that increased with depth in the northeastern part of the study area; and (2) levels of properties and constituents that showed a trend with depth that was observed across most of the site. Temporal variation was generally greatest near the water table and was most pronounced for indicators of redox reactions, such as Eh, dissolved oxygen, and nitrate.

Properties and constituents having the most pronounced spatial differences and temporal variations are described in the paragraphs that follow. Herbicides and phosphorous were rarely detected in the outwash aquifer and therefore are not included in the following discussion. Organic carbon is also not included because concentrations were apparently unaffected by location, depth, or time.

Specific Conductance, Dissolved Solids, and Chloride

Specific conductance and dissolved solids are both proportional to the ionic concentration of a solution; hence, they were highly correlated among samples from the MSEA (r = 0.99). Because chloride concentrations also were highly correlated with specific conductance (r = 0.99), spatial differences in outwash concentrations of dissolved solids and chloride follow the same pattern of areal differences found for specific conductance (fig. 25). These three characteristics all indicate a distinct zone of mineralized waters encompassing wells R2, S10, R4, and (in the deep zone of the outwash aquifer) well S5. Levels of these characteristics within the mineralized zone were more than twice those at the south end of the site in the deep outwash waters. The mineralized zone is approximately 1,500 ft wide and extends across the valley almost to the river. Two-dimensional spatial differences in specific conductance, dissolved solids, and chloride are typified by contoured specific conductances in a hydrogeologic section cutting north-south through the middle terrace (fig. 26). In general, specific conductance and dissolved solids and chloride concentration all increased with depth within the mineralized zone. Outside of the mineralized zone, however (from well S14 south), specific conductance and dissolved solids and chloride concentration decreased with depth. Levels of these characteristics in the deep outwash aquifer showed little temporal variation.

Sodium and Potassium

Maps showing distribution of the cations sodium and potassium (fig. 27) show elevated concentrations of both constituents in the northeastern part of the site; however, potassium concentrations were elevated in the deep zone of the aquifer only. The shape of the sodium and potassium contours is somewhat altered relative to the zone of elevated specific conductance, dissolved solids, and chloride, owing to the anomalous paucity of sodium and potassium ions in water from well S5-MP1.

Alkalinity and pH

The areal trend in alkalinity, in contrast to that for dissolved solids, chloride, sodium, and potassium, was inversely correlated with specific conductance. Instead of increasing to the northeast, alkalinity of shallow ground water decreased distinctly in the northeast corner of the MSEA. Alkalinity concentrations generally decreased with depth, whereas pH increased. Values for both properties showed little temporal variation during the period of study.

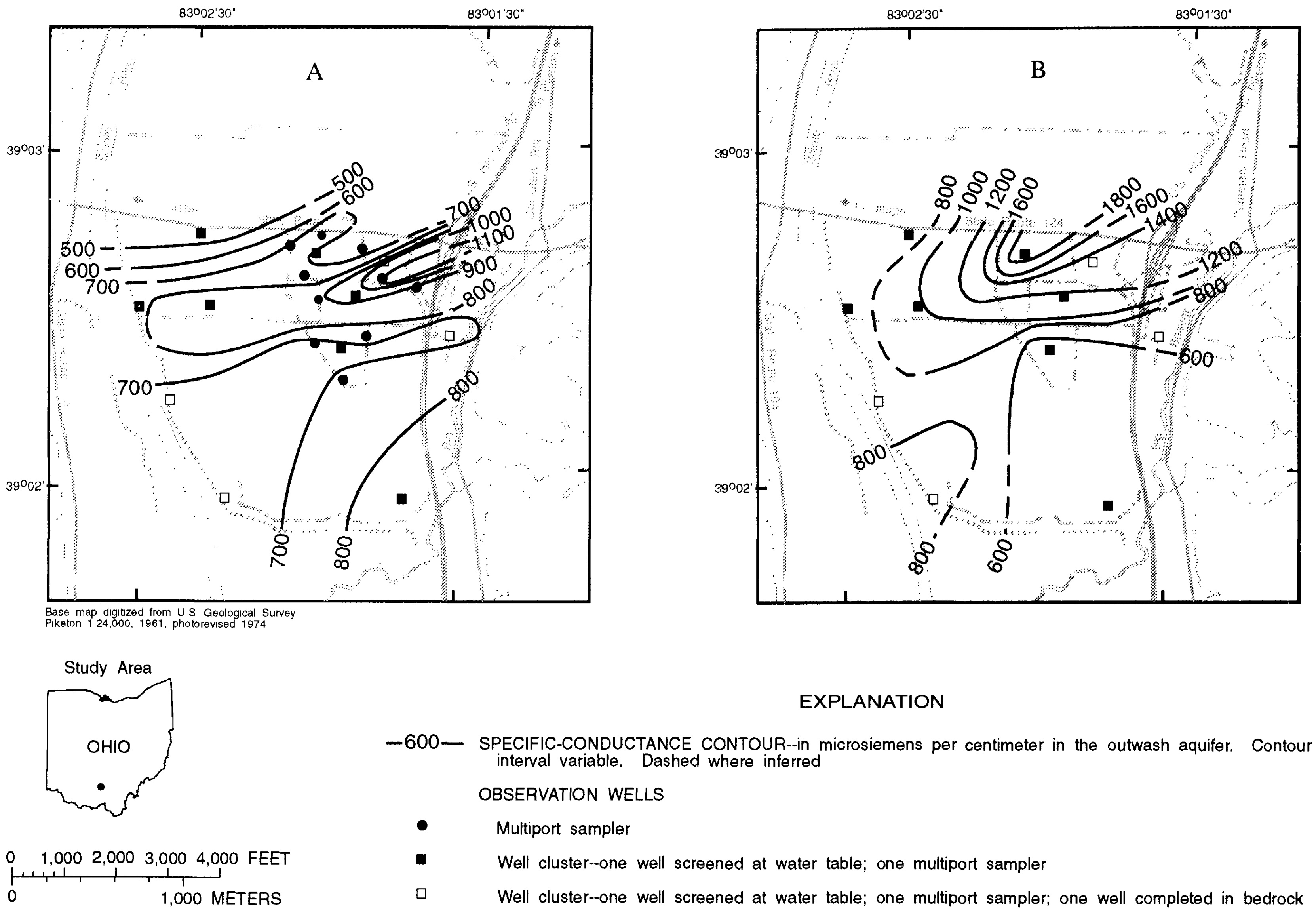
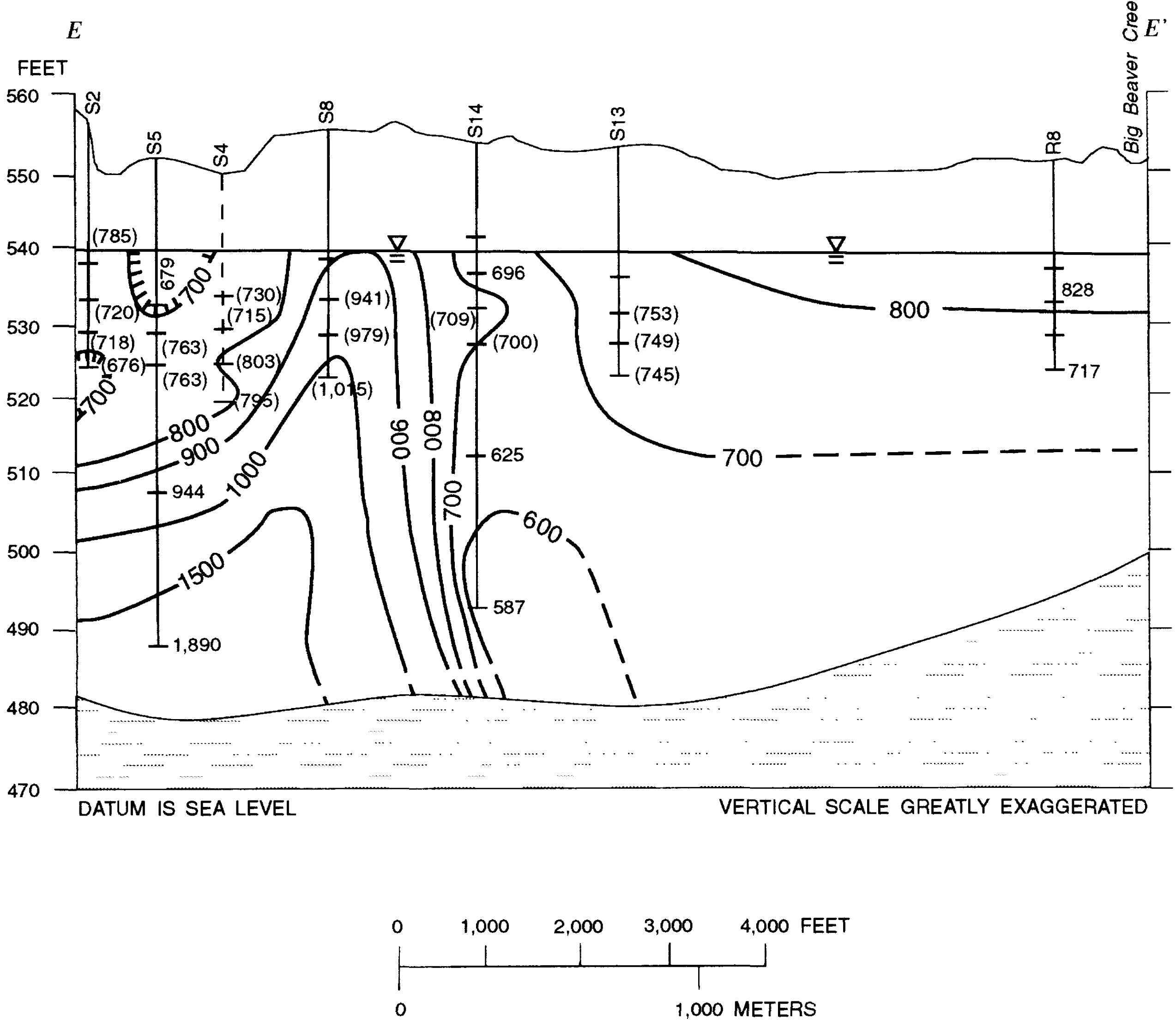


Figure 25. Areal variation in specific conductance in the outwash aquifer (A) near the water table, and (B) at approximately 60 feet below land surface, June 1991.



EXPLANATION

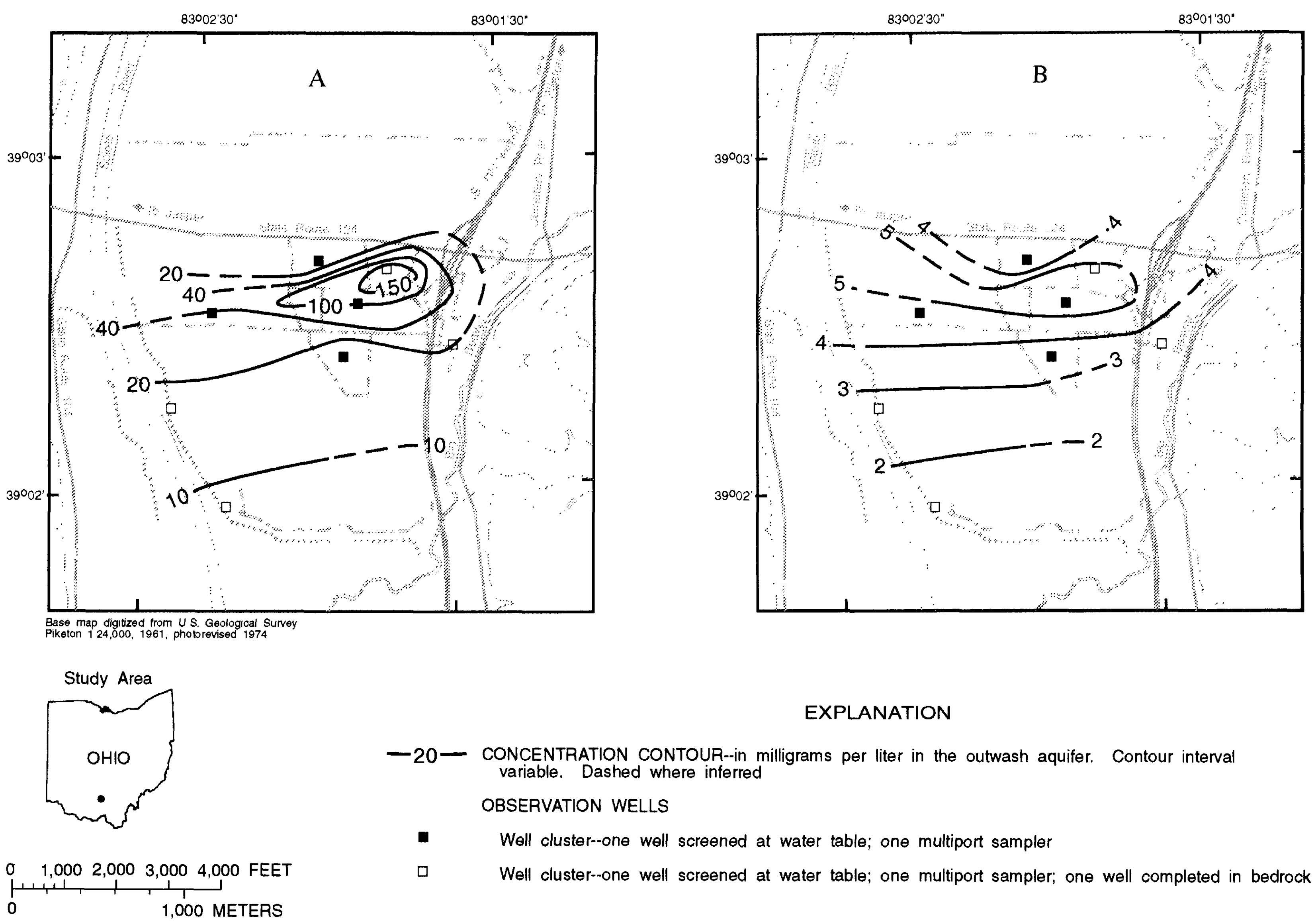
▭ SHALE

▽ WATER TABLE

—800— SPECIFIC-CONDUCTANCE CONTOUR--Shows specific conductance, in microsiemens per centimeter at 25 degrees Celsius, in the outwash aquifer. Dashed where inferred. Hachures indicate area of decreasing specific conductance. Contour interval is variable

S14 696 (700) WELL ON SECTION LINE AND IDENTIFIER--Horizontal bars denote screen location. Value is specific conductance (values in parentheses are unpublished data on file at the School of Natural Resources, The Ohio State University, Columbus, Ohio). Dashed well has been projected onto section line

Figure 26. Diagramatic section showing vertical variation in specific conductance approximately perpendicular to flow in the outwash aquifer, June 1991. (Section trace shown on fig. 18.)



EXPLANATION

—20— CONCENTRATION CONTOUR--in milligrams per liter in the outwash aquifer. Contour interval variable. Dashed where inferred

OBSERVATION WELLS

■ Well cluster--one well screened at water table; one multiport sampler

□ Well cluster--one well screened at water table; one multiport sampler; one well completed in bedrock

Figure 27. Areal variation in (A) sodium concentration and (B) potassium concentration in the outwash aquifer at approximately 60 feet below land surface, June 1991.

Eh

The redox potential, or Eh, of ground water measures the relative intensity of the oxidizing or reducing conditions within the system. A positive Eh indicates relatively oxidizing conditions, and a negative Eh indicates relatively reducing conditions. The interpretation of an Eh measurement in ground water is problematic because the Eh measured by the platinum electrode is really a mixed potential that is affected by all redox pairs present in the aquifer. Still, Eh measurements can provide useful information about relative variations in redox conditions spatially within the aquifer. All Eh's in the outwash aquifer were positive, although Eh generally decreased abruptly with depth between the water table and about 40 ft below land surface (figs. 24 and 28). Significant areal differences in Eh also were found, but these differences are not related to the mineralized zone of water in the northeastern corner of the site. Instead, ground water in the northwestern part of the site (wells R3, R5, and R4) had consistently lower Eh at all depths than anywhere on the rest of the site. Shallow waters at R2 and S14 were consistently reduced relative to Eh at the same depth elsewhere on the middle terrace, whereas S6 and S4 waters were relatively oxidizing. On the lower terrace, Eh decreased along the ground-water flow path to the west. To the east, waters at well R1 (near Big Beaver Creek) were consistently more oxidizing at greater depths than waters from other wells at the site. Temporally, redox potential at all depths across the site dropped by about 100 to 200 mv between April and November 1991 (unpublished data on file at the Environmental Chemistry Water Quality Analytical Laboratory of The Ohio State University, School of Natural Resources, Columbus, Ohio).

Dissolved Oxygen

The maximum concentration of dissolved oxygen near the water table was 7.7 mg/L, and the mean concentration was 4.5 mg/L for the three time points sampled (fig. 24). Concentrations decreased to less than 1.0 mg/L at a depth of approximately 40 ft below land surface (fig. 29). The depth at which dissolved oxygen concentrations equal 1.0 mg/L (0.03 mmol/L) is known, for the purposes of this report, as the oxic/anoxic boundary. This boundary increased in depth to about 70 ft below land surface to the west (near the Scioto River) and to about 50 ft below land surface to the east, near well R1 and Big Beaver Creek.

Areal differences in concentrations of dissolved oxygen in shallow waters were substantial. Concentrations were generally high at wells S6 and S13, relative to the rest of the site, and generally low at wells R2 and S4. Concentrations also varied temporally, as is expected of a constituent correlated with recharge. In shallow waters across the entire site, dissolved oxygen decreased by about 4 mg/L from April to November, consistent with the marked decline in Eh noted over the same time interval. However, decrease of dissolved oxygen in shallow waters did not affect the depth of the oxic/anoxic boundary, which remained fairly stable with time (except at well locations S6, S10, and R4).

Sulfate

Sulfate concentrations increased gradually with depth, from a median concentration of approximately 48 mg/L in the shallow zone of the outwash aquifer to a median concentration of approximately 72 mg/L in the intermediate and deep zones of the outwash aquifer (fig. 24). Anomalously low concentrations (19 to 32 mg/L) were found at well S5-MP1 (table 13). In general, sulfate concentrations did not vary much temporally.

Nitrate

Elevated concentrations of nitrate have been linked to methemoglobinemia (blue baby syndrome) in infants and young children. Because of this health risk, the USEPA has set a maximum contaminant level (MCL) of 10 mg/L of nitrate as N for public drinking-water supplies.

Nitrate was present in shallow outwash waters during the three sampling periods, as shown by plots of temporal changes in nitrate concentration along a flow path through plot 200 (fig. 30). In April and November, nitrate concentrations were below the MCL of 10 mg/L; however, in June, elevated nitrate concentrations were noted at wells S10 and S6. The highest concentration of nitrate was measured near the water table at S6, where 16.8 mg/L was detected (unpublished data on file at the Environmental Chemistry Water Quality Analytical Laboratory, School of Natural Resources, The Ohio State University, Columbus, Ohio).

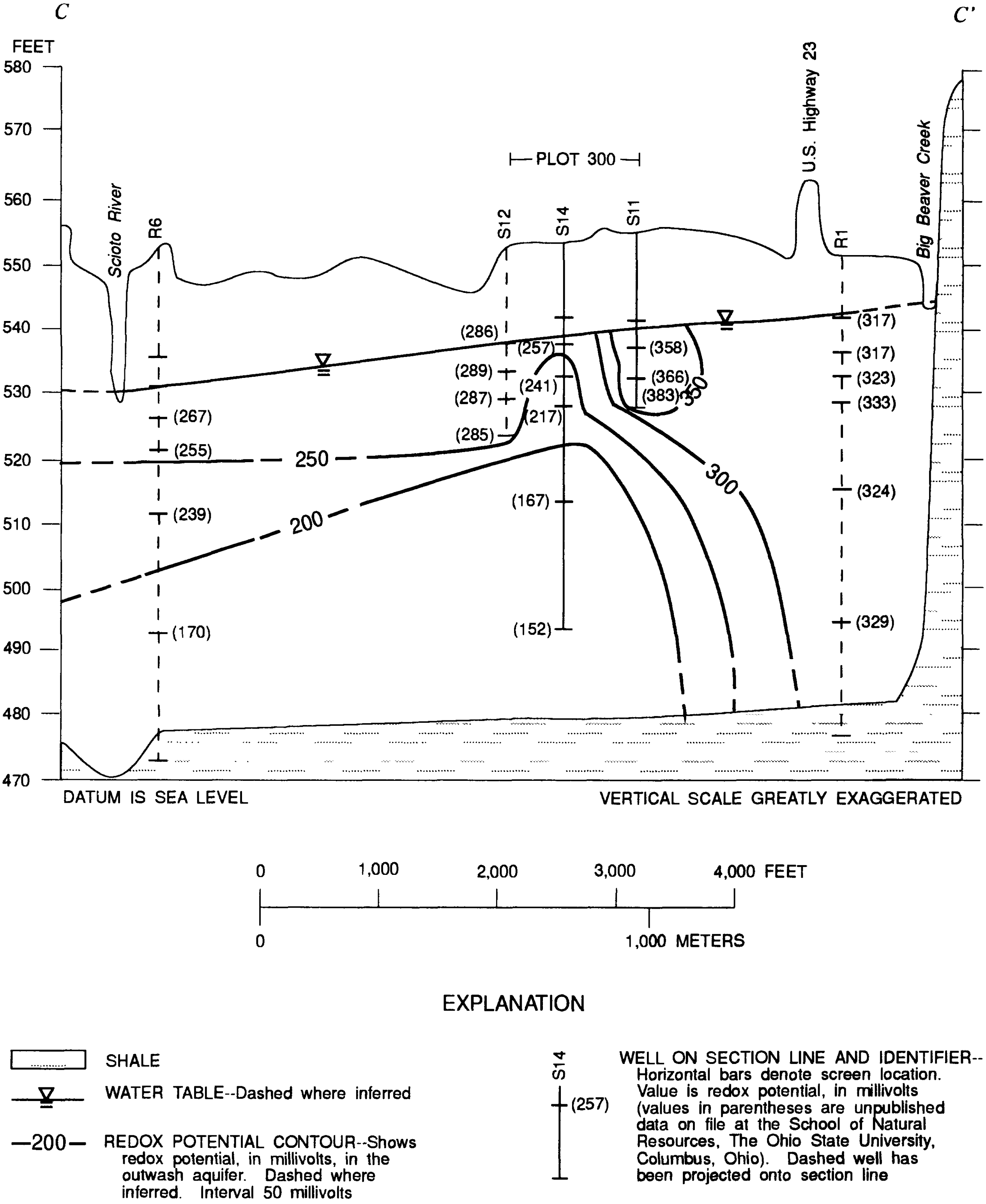
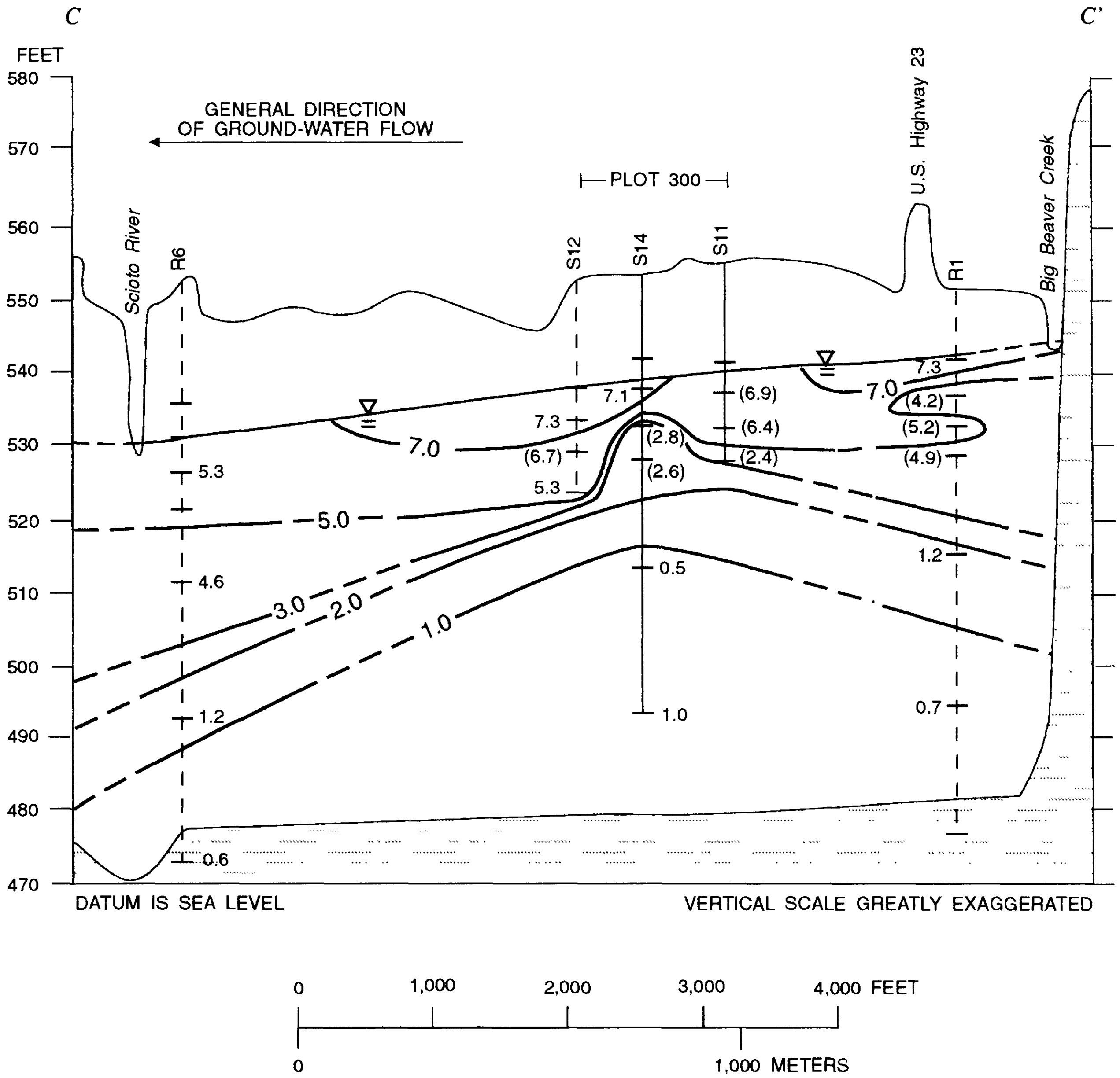


Figure 28. Generalized section showing vertical variation in redox potential, parrallel to flow in the outwash aquifer, Ohio Management Systems Evaluation Area, June 1991. (Section trace shown on fig. 18.)



EXPLANATION

▨	SHALE
▽	WATER TABLE--Dashed where inferred
—2.0—	CONCENTRATION CONTOUR--Shows concentration of dissolved oxygen, in milligrams per liter, in the outwash aquifer. Dashed where inferred. Contour interval is variable
S14 0.5 (2.6)	WELL ON SECTION LINE AND IDENTIFIER--Horizontal bars denote screen location. Value is constituent concentration, in milligrams per liter (value is parentheses are unpublished data on file at the School of Natural Resources, The Ohio State University, Columbus, Ohio). Dashed well has been projected onto section line

Figure 29. Generalized section showing vertical variation in dissolved-oxygen concentration, parallel to flow in the outwash aquifer, Ohio Management Systems Evaluation Area, June 1991. (Section trace shown on fig. 18.)

The June nitrate data indicate a source of nitrogen beneath plot 200 that is available only in late spring or summer. This source may be 1.35 x 10^5 gal of dairy-cattle manure that was applied to plot 200 after the first sampling round in April 1991. The fact that well S6 is just upgradient from plot 200 indicates a second nitrate source—perhaps from fertilizer-transfer activities either at the liquid manure tank near well S6 or at the barn upgradient from well S6.

Of particular note is the fact that the vertical hydraulic gradient indicated by the nitrate plume encompassing wells S6 and S10 is steeper than that indicated by monthly water-level measurements. Because vertical hydraulic gradients were measured only once per month at the site wells, chemical gradients likely are a more accurate, temporally averaged representation of the hydraulic gradients at the site than the widely spaced water-level measurements.

Vertically, nitrate concentrations decreased with depth from a median of 3.1 mg/L near the water table to below detection limits at approximately 40 ft below land surface (fig. 24). Detectable nitrates were present at depths greater than 40 ft below land surface at well S10 (fig. 30), at well R1 near Big Beaver Creek, and at wells R5 and R6 near the Scioto River (fig. 31).

Iron and Manganese

Iron and manganese concentrations in the outwash aquifer increased with depth in the outwash aquifer, as Eh and dissolved oxygen decreased (fig. 24). Generally, dissolved iron and manganese concentrations were near or below detection limits near the water table, except at well R2. (Here, dissolved iron concentrations of 100 to 200 µg/L and dissolved manganese concentrations of 430 to 1,000 µg/L were consistently measured in shallow waters.) Beneath the middle terrace, dissolved iron concentrations abruptly increased to 1,000 µg/L at about 40 ft below land surface (at the oxic/anoxic boundary; fig. 32), and manganese concentrations increased to approximately 40 µg/L. As with Eh and dissolved oxygen, iron concentrations at well R1 were anomalous; dissolved iron concentrations at well R1 were consistently near detection limits at all depths—even though downgradient wells contained several thousand micrograms per liter of dissolved iron near the base of the aquifer. Manganese concentrations were also relatively low at all depths at well R1. In addition, the depth to detectable iron concentrations increased to the west, near the Scioto River. Iron and manganese concentrations were fairly consistent temporally.

Age of Outwash and Bedrock Waters

Tritium concentrations were used to assess the relative ages of ground water in different parts of the aquifer, thus providing further information on ground-water flow in the aquifer. Tritium is a radioactive isotope of hydrogen with a half-life of 12.4 years. Low levels of tritium are produced naturally in the Earth's atmosphere by interaction of cosmic-ray neutrons with ^{14}N (Faure, 1977, p. 328). However, tests of large thermonuclear devices during late 1954 - 1963 released large quantities of manmade tritium to the atmosphere. Tritium in the atmosphere enters the hydrologic cycle in precipitation.

It is estimated that the natural, pre-bomb tritium content of precipitation was about 2 to 10 tritium units (TU) (Thatcher, 1962). One tritium unit is equal to one tritium atom in 10^{18} atoms of hydrogen, or 3.24 picocuries per liter. After 1953, tritium concentrations in precipitation increased two to three orders of magnitude, peaking in 1962 and 1963 in response to large-scale testing (Michel, 1989). Then, with the cessation of nuclear atmospheric testing in the mid-1960's, tritium concentrations in precipitation began declining and are now approaching pre-Bomb levels.

Tritium concentrations in precipitation differ with latitude. Estimates of concentration of tritium in precipitation in central Ohio from 1953 to 1983 (Michel, 1989) were corrected for radioactive decay through June 1991 and are shown in figure 33. Other than radioactive decay, tritium is not significantly affected by geochemical reactions; thus, it is useful for age-dating ground water. Considering the tritium half-life of 12.4 years (International Atomic Energy Agency, 1981), ground water that recharged the water table before 1953 should have had a tritium concentration of less than or equal to approximately 1 TU in 1991. Therefore, if samples of ground water from south-central Ohio contain tritium concentrations at less than or equal to 1 TU, the water either originally entered the saturated zone some time before 1953 or mixed extensively with pre-1953 ground water. Similarly, ground water containing tritium at concentrations greater than 1 TU either initially recharged the water table some time after 1953 or mixed with a large fraction of post-1953 water. In ground-water-flow systems where no mixing of waters has occurred, the

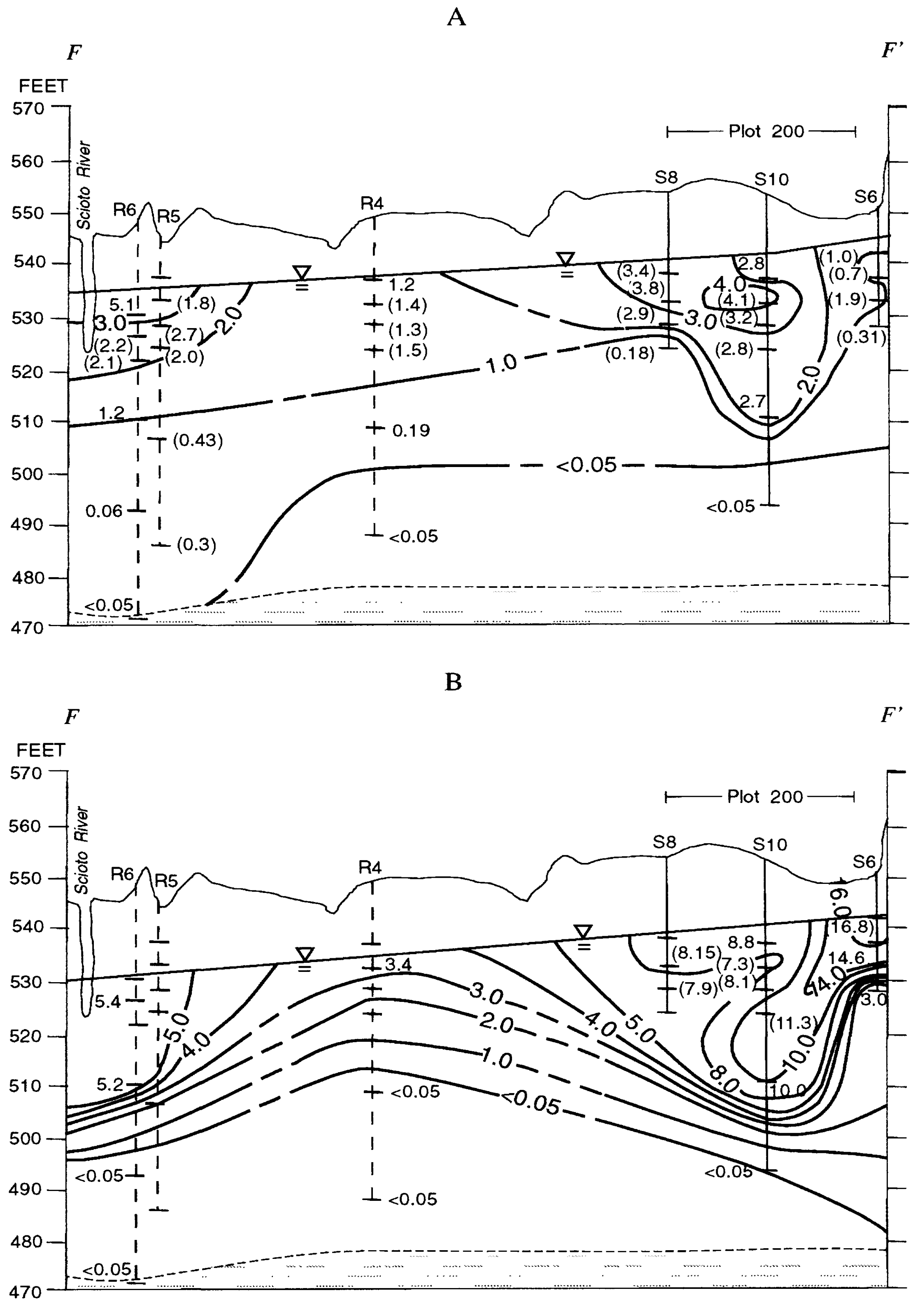
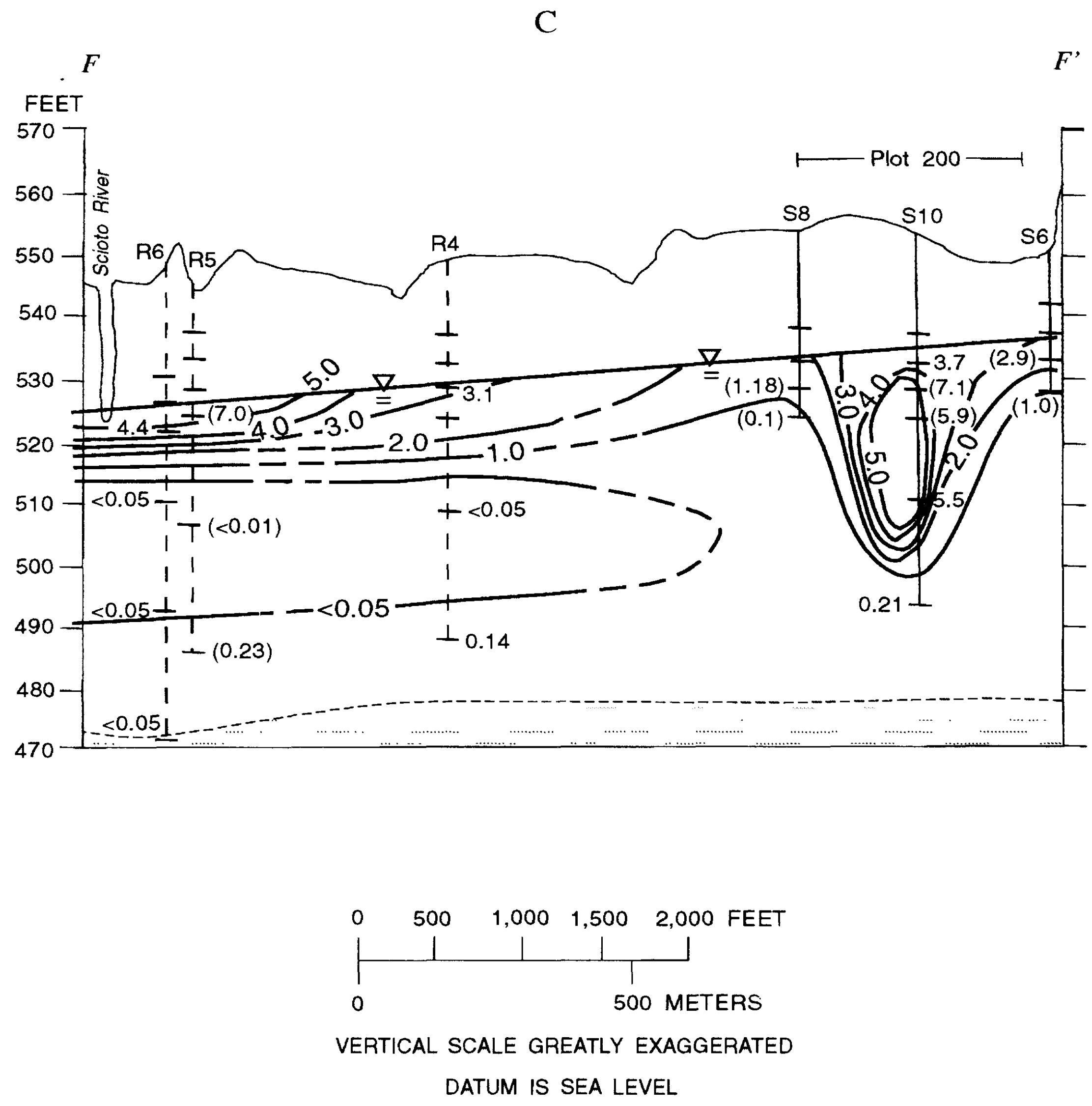


Figure 30. Diagramatic section showing vertical variation in nitrate concentration in the outwash aquifer in (A) April, (B) June, and (C) November 1991, parallel to flow, Ohio Management Systems Evaluation Area. (Section trace shown on fig. 18.)



EXPLANATION

▽ WATER TABLE

—2.0— CONCENTRATION CONTOUR--Shows concentration, in milligrams per liter, of nitrite plus nitrate as N in the outwash aquifer. Contours are interpolated between values in adjacent ports of projected wells. Contour interval is variable. Dashed where inferred

S6 1.2 (1.0) WELL ON SECTION LINE AND IDENTIFIER--Horizontal bars denote screen location. Value is constituent concentration, in milligrams per liter (values in parentheses are unpublished data on file at the School of Natural Resources, The Ohio State University, Columbus, Ohio). Dashed well has been projected onto section line

Figure 30. Diagramatic section showing vertical variation in nitrate concentration in the outwash aquifer in (A) April, (B) June, and (C) November 1991, parallel to flow, Ohio Management Systems Evaluation Area. (Section trace shown on fig. 18.)--Continued.

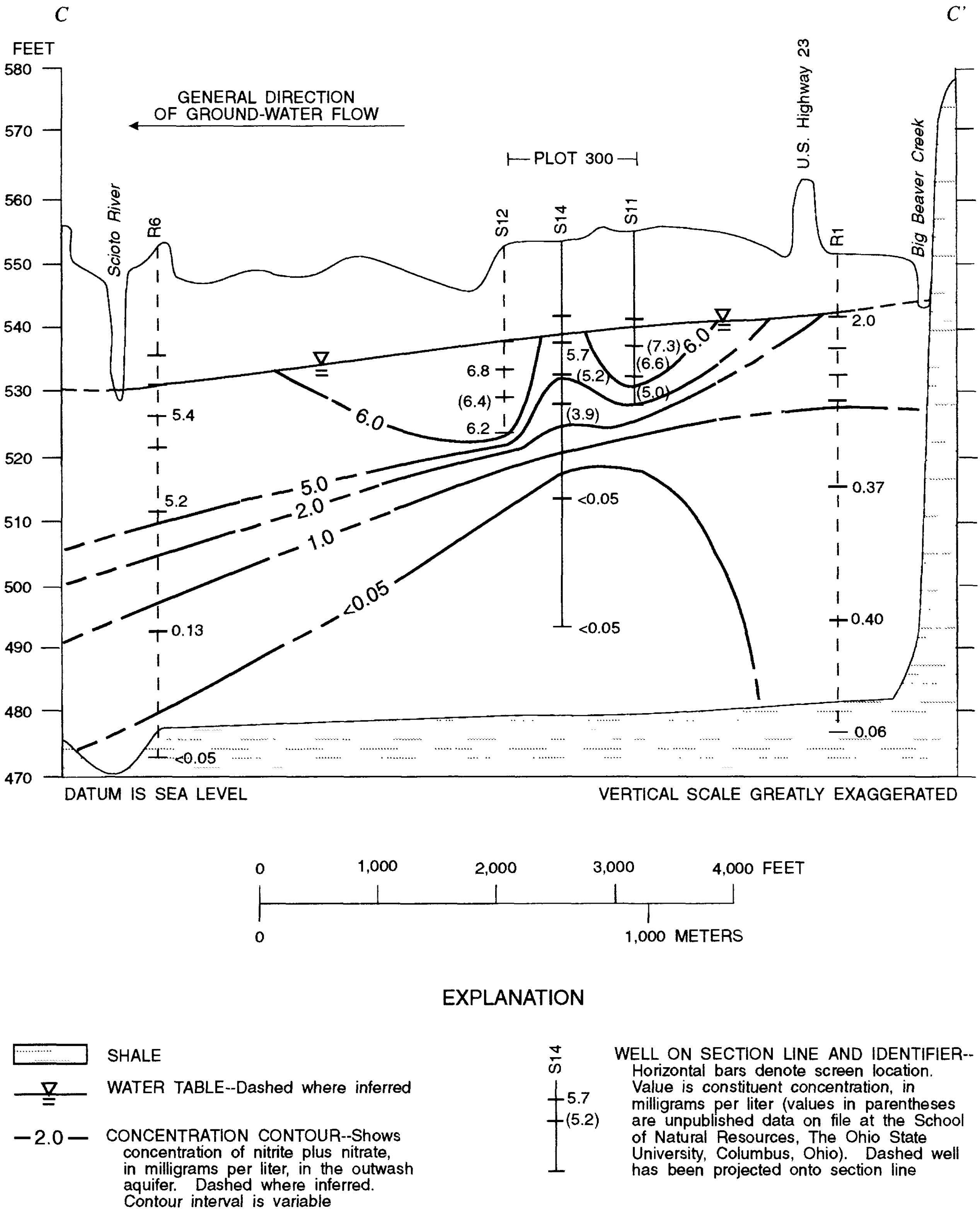


Figure 31. Generalized section showing vertical variation in nitrate concentration, parallel to flow in the outwash aquifer, Ohio Management Systems Evaluation Area, June 1991. (Section trace shown on fig. 18.)

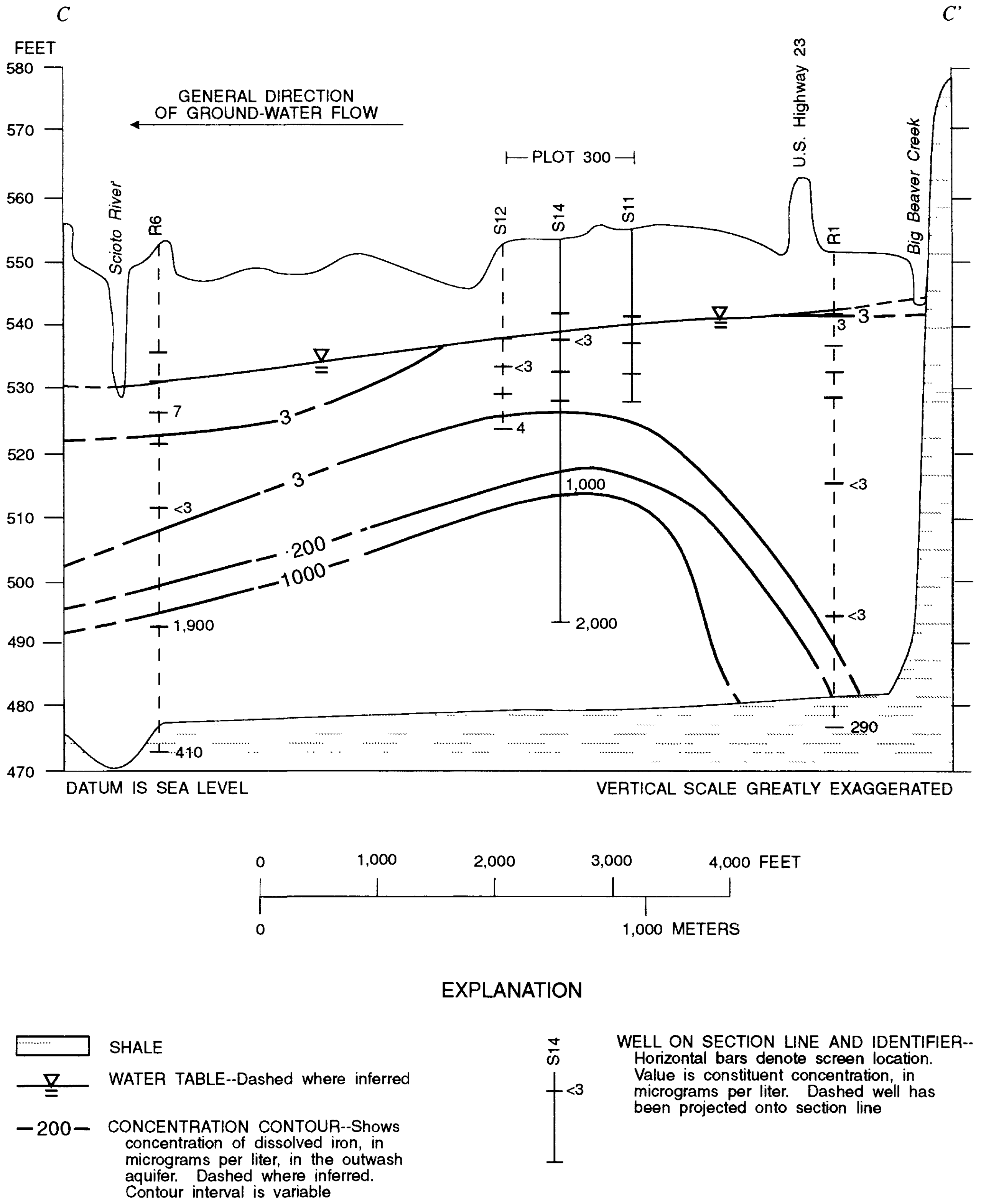


Figure 32. Generalized section showing vertical variation in dissolved-iron concentration, parallel to flow in the outwash aquifer, Ohio Management Systems Evaluation Area, June 1991. (Section trace shown on fig. 18.)

tritium-input curve (fig. 33) can be used to estimate the recharge age of waters. For example, figure 33 shows that water containing more than 32 TU would have recharged the ground-water system some time between 1958 and 1972.

Results of tritium analyses of MSEA ground-water samples are summarized in table 14. MSEA samples fell into two categories based on tritium concentration: tritiated and untritiated. Ground waters with tritium concentrations below the detection limit of 1.8 TU included the bedrock wells at clusters R1, R2, and R6. The untritiated nature of these bedrock wells indicates that shallow bedrock water at these locations entered the ground-water-flow system as recharge at an upgradient location before 1954.

The tritiation of well R7-BR (4.35 TU) likely was caused by inadvertent contamination of the well during development; it is unlikely bedrock water at well R7 would be naturally tritiated because upgradient bedrock wells are not tritiated and mixing with outwash waters is unlikely (based on the slow recovery of this well—greater than 2 months—after pumping). Unlike other bedrock wells at the MSEA, well R7 was developed by jetting with water from a different well (one that is screened in the deep outwash aquifer); other (non-tritiated) bedrock wells were developed by use of a submersible pump.

The tritium content of samples collected from the outwash aquifer ranged from 8.6 to 33.1 ±2.8 TU, indicating a component of post-1953 recharge. Because all of the concentrations were less than 32 TU (when considered with the precision estimate), recharge dates could not be further specified. No trends in tritium concentration with depth are apparent (fig. 34). These data indicate that significant amounts of recharge water reach even the deepest and most downgradient zones of the outwash aquifer in less than 38 years. This finding is in agreement with the high recharge rates estimated previously in this report and supports the chemical evidence (demonstrated by nitrate data, fig. 30) that vertical hydraulic gradients are steeper than indicated by monthly water-level data.

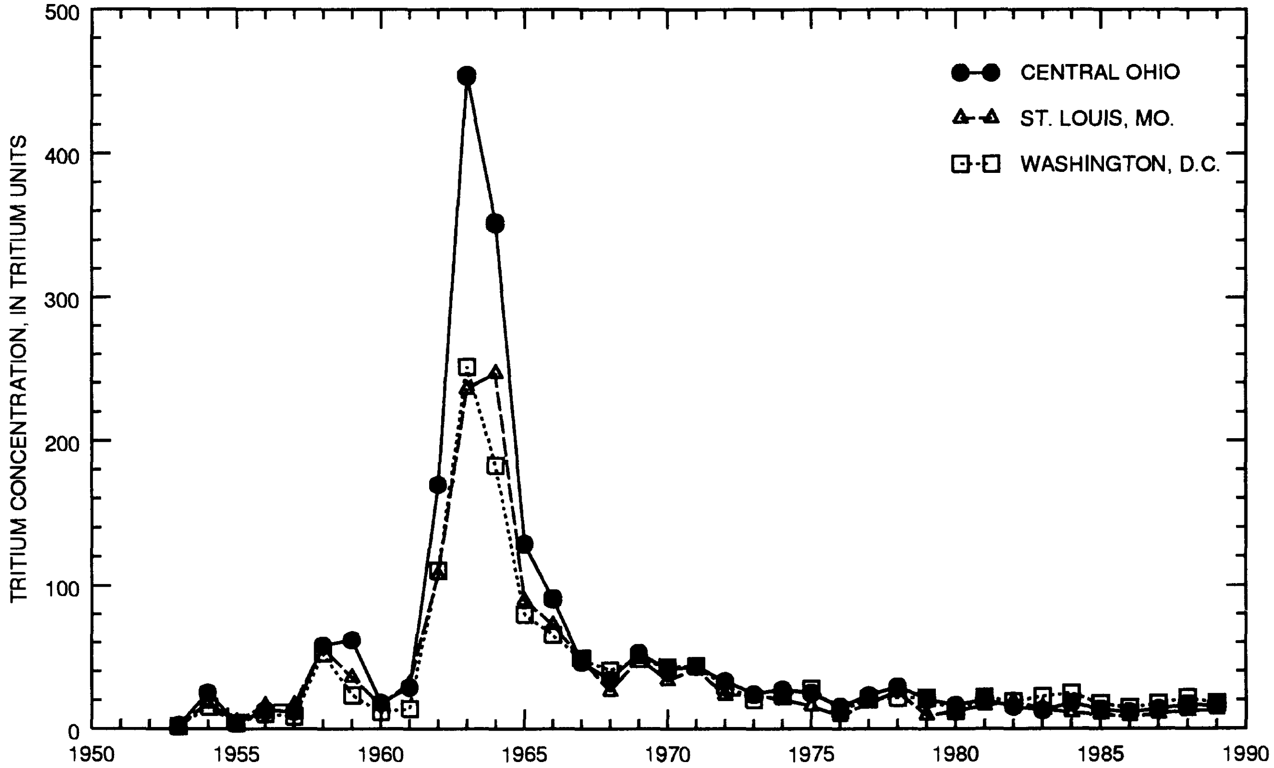


Figure 33. Estimates of concentration of tritium in precipitation for central Ohio, St. Louis, Mo. and Washington, D.C.,1953-89. (From Dumouchelle and others, 1993.)

Table 14. Tritium content of ground water at the Ohio Management Systems Evaluation Area

[Well locations are shown in fig. 2. Samples analyzed at laboratory of R.L. Michel, U.S. Geological Survey, Reston, Va. TU, tritium unit; one TU equals 3.24 picocuries per liter. Analytical detection limit is 1.8 TU]

Well number	Date sample collected	Total tritium content (TU)	Precision estimate (TU)
R1-BR	9/10/92	<1.8	1.2
R1-MP1	8/12/92	10	1.6
R1-MP3	8/12/92	8.6	1.4
R2-BR	6/17/91	0.0	.2
R2-MP1	6/17/91	25.3	1.3
R2-MP2	6/17/91	25.5	1.1
R2-MP5	6/17/91	17.6	.8
R6-BR	9/01/92	<1.8	1.6
R6-MP1	8/05/92	16	1.6
R6-MP3	8/05/92	13	1.6
R7-BR	9/02/92	4.35	1.6
R7-MP1	9/02/92	13.9	1.6
R7-MP3	9/02/92	33.1	2.8
S10-MP2	4/02/91	21.4	.8
S10-MP6	4/02/91	15.0	.7
S12-MP1	4/03/91	15.6	.7
S12-MP4	4/03/91	13.5	.8

Tritiated outwash waters also indicate that human activities at land surface have considerable potential for affecting ground-water quality beneath the study area. Ground water currently present beneath the study area recharged the aquifer since farming began at the site and also since agrochemical use became widespread in the 1950's. Atrazine application at the farm began in approximately 1964 (John Van Meter, farm owner, oral commun., 1992).

CONTROLS ON WATER QUALITY

Accurate interpretation of water-quality data requires a knowledge of the processes controlling water composition. Factors influencing surface-water and ground-water quality are discussed in the paragraphs that follow. In addition, instances where water-quality data support or contradict hydraulic data are noted.

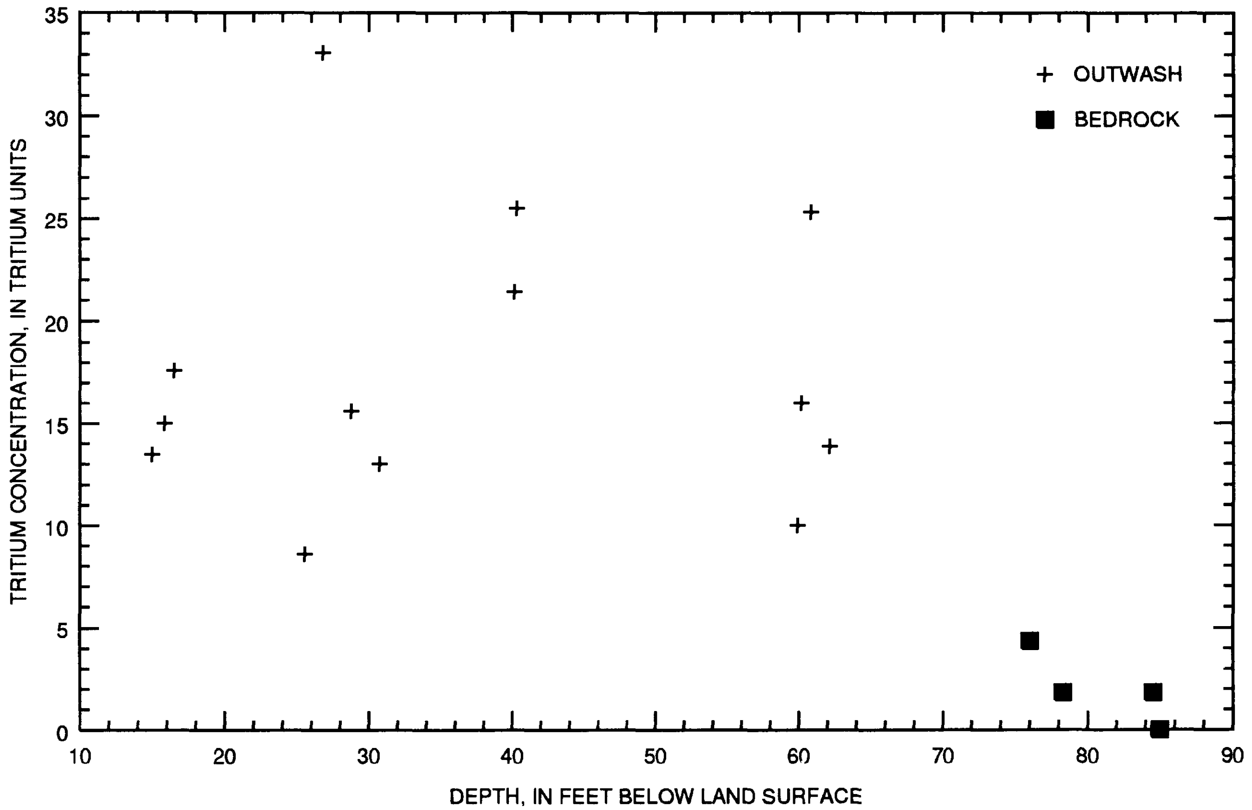


Figure 34. Tritium concentration in ground water as a function of depth below land surface at the Ohio Management Systems Evaluation Area, April 1991-September 1992.

Surface Water

The water quality of the Scioto River and Big Beaver Creek is greatly affected by the contrasting drainage basins of the two streams. Major-ion chemistry of the streams is controlled largely by basin geology. For example, Big Beaver Creek is fed by rapid runoff from the virtually impermeable shales and siltstones that make up its drainage basin. The runoff has a short contact time with the rocks of the basin; therefore, only a small amount of dissolution of minerals can occur, and streamwater is characterized by low specific conductance and low concentrations of dissolved solids, calcium, magnesium, bicarbonate, chloride, and other major ions. These findings are consistent with the hydrograph and seepage-meter data presented earlier, which indicate that Big Beaver Creek receives no recharge from ground water.

In contrast to Big Beaver Creek, streamwater in the Scioto River consists of 48 percent ground-water discharge (median estimated for historic record at Higby by use of the fixed-interval method of hydrograph separation). The long contact time of ground water with aquifer sediments allows dissolution of aquifer minerals to occur prior to discharge of ground water to the Scioto River as base flow. This dissolution, coupled with the greater amount of carbonate sediment in the Scioto Basin than in the Big Beaver Basin, contributes to the greater calcium, magnesium, and bicarbonate concentrations in the Scioto River.

Although major-ion concentrations of the two streams are controlled largely by basin geology, nutrient and herbicide concentrations of the two streams are controlled by land use in the drainage basins. Greater concentrations of pesticides, nutrients, and organic carbon in the Scioto River than in Big Beaver Creek reflect the greater amounts of farmland and industry in the Scioto River Basin. Temporal variations in agrochemical levels of the Scioto River are consistent with observations of other Midwestern streams, where nitrate concentrations commonly increase in the fall after plant uptake of nitrogen decreases and where herbicide concentrations increase in the spring and early summer because of runoff from fields after spring application of herbicides (Goolsby and Battaglin, 1993). The increase in the desethyl atrazine-to-atrazine ratio (DAR) from 0.05 to 0.25 between June and November 1991 indicates that biodegradation of atrazine has occurred somewhere in the system during this time period (Thurman and others, 1992). If the MSEA site is representative of the entire drainage basin (in that ground water is free of atrazine and its metabolite desethylatrazine), atrazine degradation must occur during the transient storage of field-runoff waters in the hyporheic sediments upstream from the site rather than being attibutable to an increase in the base-flow component of streamflow in November.

The observed decrease in dissolved oxygen concentration of surface water during the growing season was likely due to exsolution of oxygen as water temperatures rose throughout the summer and, in the Scioto River, to an increasing percentage of ground-water contribution as streamflow decreased throughout the summer.

Ground Water

Ground-water chemistry is a result of many concurrent influences, including mixing of waters from different sources, rock-water chemical interactions, redox reactions, anthropogenic inputs, and biological activity in the aquifer.

Mixing of Bedrock and Outwash Aquifer Waters

The fact that some intermediate and deep outwash waters have chemical characteristics of intermediate composition between those of bedrock water (calcium sodium chloride type) and shallow outwash water (calcium bicarbonate type), as evident in the Piper diagram (fig. 23), indicates that mixing of bedrock and outwash waters has an important effect on ground-water quality. (Mixing trends are denoted by lines on the Piper diagram.)

Chloride is the most conservative ion identified in the MSEA aquifer system, and its main source in the outwash aquifer beneath the MSEA seems to be an influx of bedrock waters. Sodium and chloride are the main ions in bedrock water at the site. Road salt is a common source of chloride and sodium in shallow aquifers; however, sodium and chloride concentrations at the MSEA site are higher at the base of the aquifer than near the water table, indicating that the ion source is likely at the base of the aquifer and not at land surface. Therefore, the authors attributed sodium and chloride concentrations to influx of bedrock waters and not to contamination of waters by road salt. A plot of the relation between sodium and chloride was used to indicate the proportions of bedrock water in the outwash aquifer (fig. 35). The dashed line on the plot represents the changing Na:Cl ratios that would be observed in ground water affected by conservative mixing between the two end-members, bedrock and outwash water. Most of the outwash and bedrock samples plot along this line; however, samples from well S5-MP1 in the outwash aquifer plot significantly below the mixing line. Waters from well S5-MP1 have a considerably lower Na:Cl ratio than can be explained by the two-endpoint mixing model, an indication of a major sodium sink at this location. Although mixing of bedrock and outwash waters may control chloride concentrations at S5-MP1, extensive exchange of sodium ions for calcium and magnesium ions would be required to create the unusual chemistry at this location. No stratigraphic data are available near the bedrock surface at location S5 to determine if an unusually thick clay layer overlies the bedrock that could provide a substrate for the ion exchange.

Mixing endpoints were chosen on the basis of field data and plotting location on figure 35. Because consistent upflow of waters from the bedrock to the outwash aquifer was measured at well cluster R2 and because waters from well R2-BR contain the highest sodium, chloride, magnesium, calcium, potassium, and dissolved solids concentrations, water samples from R2-BR are assumed to be representative of bedrock water that has not been significantly diluted by outwash water. Similarly, because well cluster R1 is in a recharge area and because R1-MP6 waters contain the lowest sodium and chloride concentrations, samples from R1-MP6 are assumed to be representative of outwash water unaffected by mixing with bedrock water. Proportions of bedrock water in each well sampled were estimated by means of these two endpoints and the following formula:

$$\text{Estimated percentage of bedrock water in well (by volume)} = (Cl_W - Cl_{OW})/(Cl_{BR} - Cl_{OW}), \qquad (8)$$

where Cl_W is the chloride concentration in the well of interest (in millimoles per liter), Cl_{OW} is the chloride

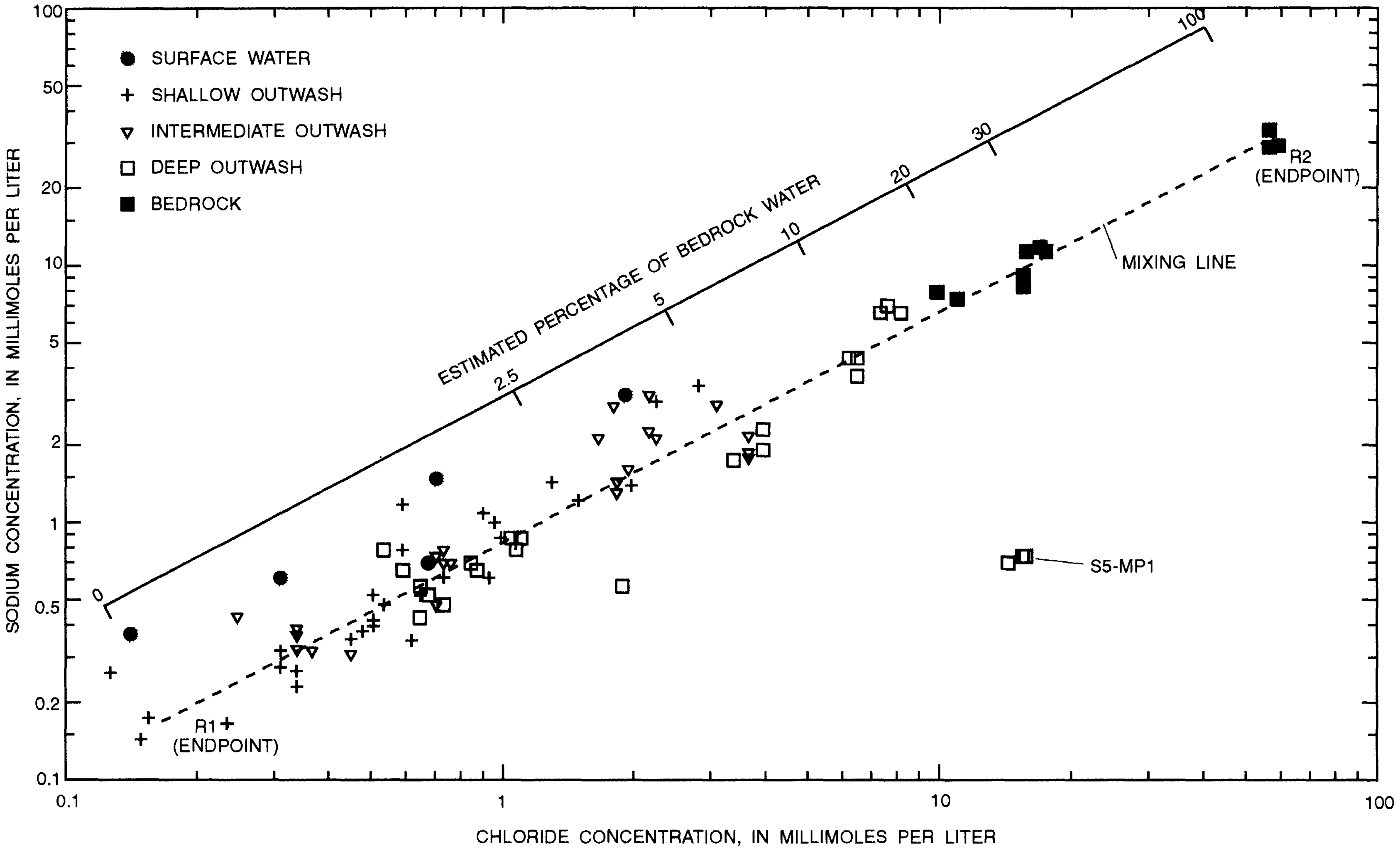


Figure 35. Concentration of sodium as a function of chloride concentration and calculated percentage of bedrock water in the outwash aquifer at the Ohio Management Systems Evaluation Area, April-November 1991.

concentration in pure outwash water (in millimoles per liter), and Cl_{BR} is the chloride concentration in pure bedrock water (in millimoles per liter). For each of the endpoint concentrations, the average of the three temporal samples was used. Results of the calculation for all wells are summarized in table 15. A range of values calculated by use of the chloride concentrations from three time points per well is shown.

Results of the mixing calculations agree with measured vertical hydraulic gradients between the bedrock and outwash aquifers. These results show that the bedrock aquifer has been diluted by water from the outwash aquifer in areas where downflow from the outwash to the bedrock was measured. Consistent downflow has created a mixture of 18 percent bedrock water and 82 percent outwash water at well R1-BR (the most dilute location), whereas periodic flow reversals near the Scioto River have resulted in 30 percent bedrock water and 70 percent outwash water at well R6-BR.

Of the four bedrock samples analyzed, the fact that the diluted bedrock wells are not tritiated indicates that flow of water between the bedrock and outwash units is very slow. A maximum flow velocity can be calculated for bedrock waters if a completely vertical, downward gradient in the bedrock aquifer is assumed and if the difference between the top of the bedrock and the midpoint of the well screen is used as the travel distance. The travel distance divided by a travel period of 38 years (the time period between 1991, when MSEA ground-water samples were collected for tritium analysis, and 1954, the most recent possible year that bedrock waters could have recharged without being tritiated) yields a maximum flow velocity of 0.2 ft/yr.

Mixing calculations also indicate that upward leakage of bedrock water into the outwash aquifer occurs at some locations. The effects of upward leakage seem to be most pronounced at the northeastern part of the site, where water levels in well cluster R2 indicated consistent upflow from bedrock to the outwash aquifer. Waters at approximately 60 ft below land surface in the northeast corner consist of 5 to 28 percent bedrock waters (fig. 36). Deep outwash waters in this area also have elevated specific conductance; elevated dissolved solids, sodium, and potassium concentrations; and low alkalinity (figs. 25 and 27)—all distinguishing features of the bedrock water. Elsewhere beneath the MSEA, deep outwash waters consist of less than 5 percent bedrock water. Percentage of bedrock-water contribution to ground-water chemistry decreases with decreasing depth in the aquifer (fig. 37).

The greater percentage of bedrock water in the outwash at the northeast part of the site than elsewhere at the MSEA could be caused by a local bedrock discharge area between wells S5 and R2 that is affected by the bedrock high immediately north of SR 124 (fig. 5). It seems more likely, however, that a local zone of dilution is present at the south end of the site (from well S14 south), caused by increased leakage from Big Beaver Creek where the streambed changes from shale to gravel. (This change in streambed is at about the same latitude as well R1.) This interpretation is consistent with data from seepage-meter tests and ground-water hydrograph separations presented previously in the report, which indicate that Big Beaver Creek is a losing stream along the southern half of the study site.

Rock-Water Interactions

Rock-water interactions include dissolution, precipitation, sorption, cation exchange, and redox reactions. Dissolution-precipitation reactions can be examined quantitatively by means of the USGS computer program WATEQ4F (Ball and Nordstrom, 1991), which uses field measurements of temperature, pH, Eh, dissolved oxygen, and alkalinity and the chemical composition of water to compute the distribution of aqueous species, the activity of dissolved constituents, and, most importantly, mineral saturation indices.

The saturation index for a mineral phase is defined as

$$SI = \log (IAP/K_t), \qquad (9)$$

where IAP is the ion-activity product of the reactants and K_t is the thermodynamic equilibrium constant of the reaction determined at the temperature of the water sample. The saturation indices (SI's) for a water sample indicate the tendency of the analyzed water to dissolve or precipitate specific minerals. An SI less than zero indicates that the mineral is undersaturated in the aqueous phase and that dissolution of the mineral is possible. An SI greater than zero means that the water is supersaturated with respect to the mineral and that precipitation of the mineral is possible. An SI of zero indicates that the ground water and the mineral are in equilibrium and that this equilibrium reaction may be

Table 15. Percentage of ground water contributed by bedrock to wells at the Ohio Management Systems Evaluation Area

[Well locations are shown in figure 2. Chloride concentrations presented are the range determined during three sampling rounds in April, June, and November 1991. Exceptions are wells R3 and R5; because the U.S. Geological Survey did not sample these wells, chloride concentrations were obtained from The Ohio State University for November 1991 only (unpublished data on file at the Environmental Chemistry Water Quality Analytical Laboratory, The Ohio State University, School of Natural Resources, Columbus, Ohio). Percentage of bedrock contribution was calculated by use of the average R2-BR concentration of chloride as the pure bedrock end-member, and the average R1-MP4-6 concentration as the pure outwash end-member]

Well Number[a]	Range of chloride concentration (millimoles per liter)	Range of percentage of ground water contributed by bedrock
R1-BR	9.9 - 11.0	17 - 19
R1-MP1	.5 - 1.1	1 - 2
R1-MP2	.2 - .3	0 - 1
R1-MP4-6	.1 - .2	0
R2-BR	56.4 - 59.2	99 - 104
R2-MP1	7.3 - 8.2	13 - 14
R2-MP2	1.8 - 3.1	3 - 5
R2-MP4-6	.1 - 2.8	0 - 5
R3-MP1	1.0	2
R3-MP2	1.1	2
R3-MP4-6	1.4	2
R4-MP1	3.4 - 4.0	6 - 7
R4-MP2	1.7 - 2.3	3 - 4
R4-MP4-6	.5 - 1.5	1 - 3
R5-MP1	.9	2
R5-MP2	1.1	2
R5-MP3-6	.4	1
R6-BR	15.8 - 17.5	28 - 31
R6-MP1	.8 - 1.1	2
R6-MP2	.3 - .5	1
R6-MP3-6	.3 - .5	1

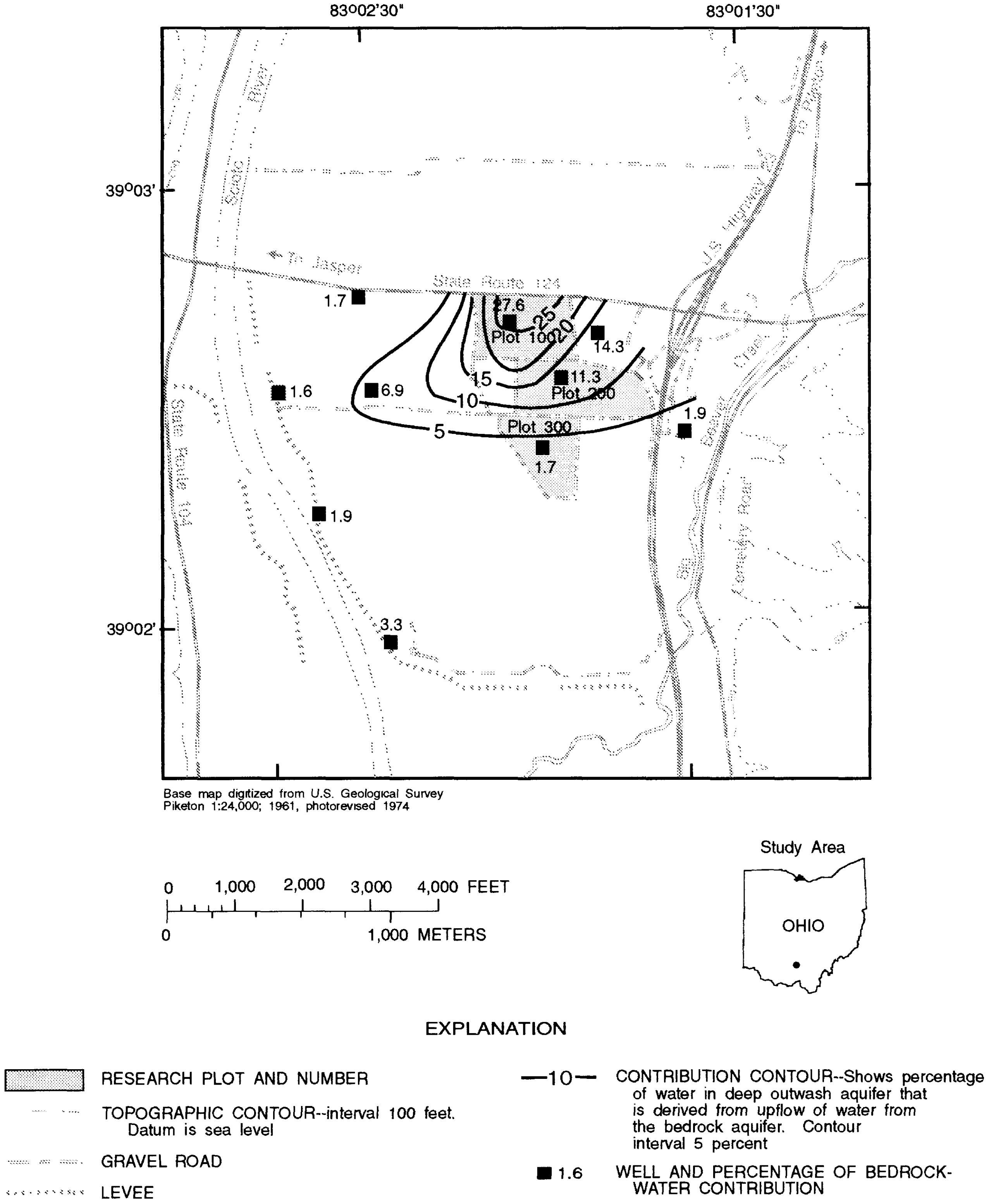
Table 15. Percentage of ground water contributed by bedrock to wells at the Ohio Management Systems Evaluation Area—Continued

Well Number[a]	Range of chloride concentration (millimoles per liter)	Range of percentage of ground water contributed by bedrock
R7-BR	15.5	27
R7-MP1	.6 - 1.9	1 - 3
R7-MP2	.7 - .8	1
R7-MP4-6	.3 - .5	1
S5-MP1	14.4 - 15.8	25 - 28
S5-MP2	3.7	6
S5-MP4-6	.6 - 1.3	1 - 2
S10-MP1	6.2 - 6.5	11
S10-MP2	1.8 - 1.9	3
S10-MP4-6	.6 - 2.0	1 - 4
S14-MP1	.6 - .7	1 - 2
S14-MP2	.7 - .8	1
S14-MP4-6	.3 - 1.0	1 - 2

[a] Abbreviations: BR, bedrock well; MP, multiport well completed in the outwash aquifer. Well numbers ending in "MP1" indicate that the well port is completed at a depth of approximately 60 feet below land surface; "MP2", completion depth is approximately 40 feet below land surface; "MP3-6", uppermost saturated port (exact completion depth is listed in table 1 at back of report).

controlling solute concentrations in the ground water. Complete equilibrium between ground water and aquifer minerals is the exception—not the rule. Total equilibrium is uncommon because the reaction rates for some minerals are slow compared to the ground-water residence time. Minerals most likely to be in equilibrium with ground water are ones that form at temperatures and pressures found at the Earth's surface.

Results of SI calculations for the minerals dolomite, quartz, and calcite (the most commonly detected minerals in the outwash aquifer, as is evident from table 4) are listed in table 16, in addition to SI's for the minerals chalcedony, ferrihydrite, and siderite. Because ferrihydrite is amorphous, its presence cannot be confirmed by X-ray diffraction (XRD), however, the yellowish-brown stain observed on quartz grains from shallow sediment samples is likely ferrihydrite or other oxyhydroxide minerals. Siderite and chalcedony are common, reactive minerals formed in low-temperature aqueous environments, and they may be present in the outwash aquifer below the XRD detection limit of 5 percent by volume. Similarly, pyrite is likely present below the XRD detection limit in the study area because pyrite is widely distributed in Paleozoic sedimentary rocks throughout Ohio (Botoman and Stieglitz, 1978); these Paleozoic rocks are the source for 96 percent of the mineral material in glacial deposits within the State (Strobel, 1990). The SI of pyrite could not be determined because of insufficient sulfide data. Lastly, partial pressure of carbon dioxide (pCO_2) was calculated from pH and alkalinity measurements and is listed in table 16. The pCO_2 in ground water is used to evaluate reactions involving carbonate minerals and the oxidation of organic matter. Similarly, results of the SI calculations are used to evaluate the equilibrium reactions proposed in the paragraphs that follow.



Base map digitized from U.S. Geological Survey
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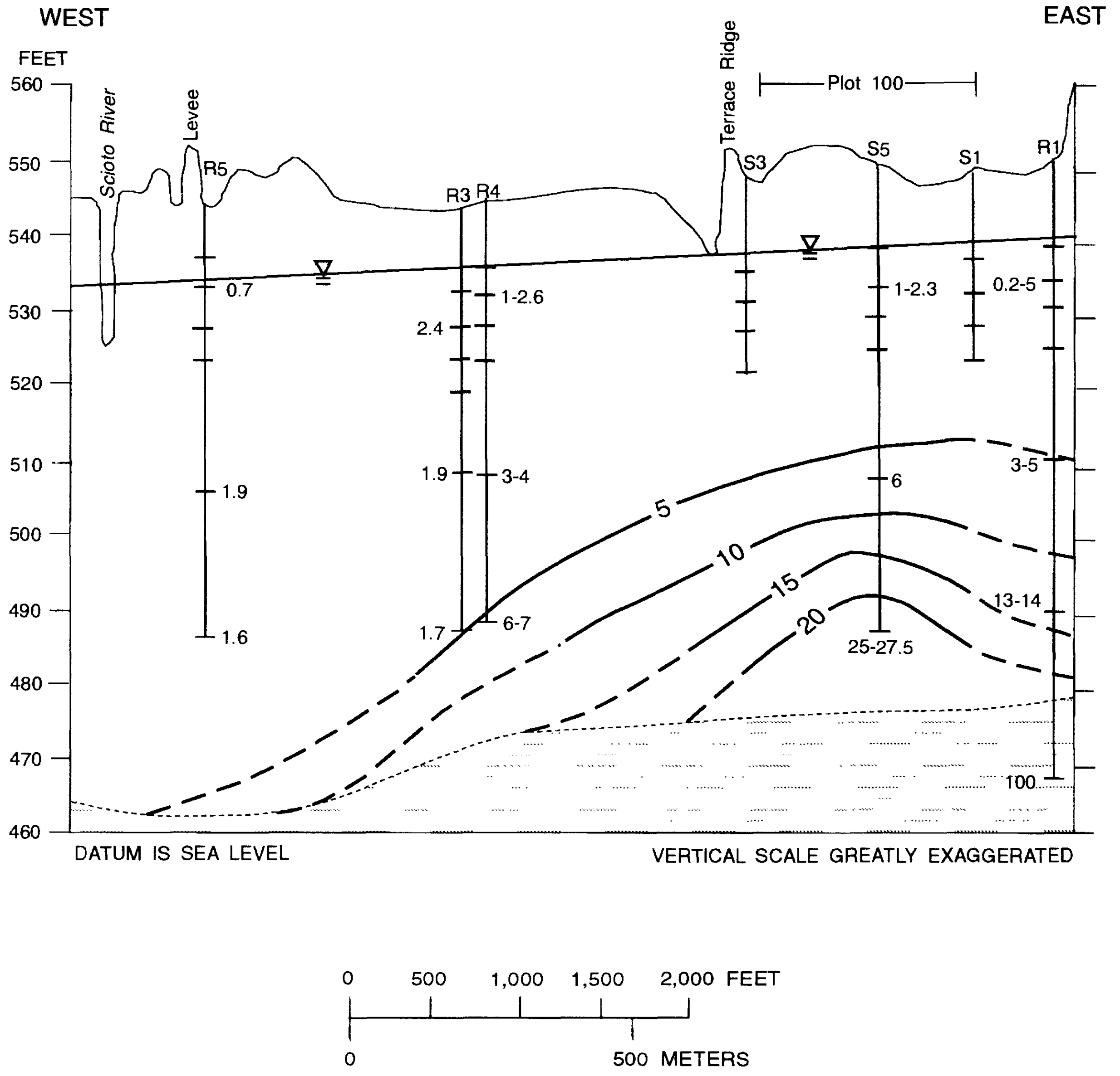
Figure 36. Areal variation in percentage contribution of bedrock water to the outwash aquifer at approximately 60 feet below land surface, April-November 1991.



EXPLANATION

☐ SHALE

$\overline{\underline{\nabla}}$ WATER TABLE

—10— CONCENTRATION CONTOUR--Shows percentage of water in the outwash aquifer derived from upflow of water from the bedrock aquifer. Contour interval 5 percent. Dashed where inferred

R4 3.4 6-7 WELL ON SECTION LINE AND IDENTIFIER--Horizontal bars denote screen location. Value is percentage of water at sampling location derived from bedrock contribution. May have a range of contribution values for three sampling rounds in April, June, and November 1991

Figure 37. Generalized section showing vertical variation in contribution of bedrock water to the outwash aquifer.

Table 16. Mineral saturation indices for ground water at the Ohio Management Systems Evaluation Area, November 1991

[Well locations are shown in figure 2. Abbreviations: pCO_2, partial pressure of carbon dioxide; atm, atmospheres; NA, saturation index could not be calculated because one or more mineral component was not detected]

Well number[a]	Log pCO_2 (atm)	Calcite	Dolomite (crystalline)	Chalcedony	Quartz	Ferrihydrite	Siderite (disordered)
R1-MP1	-2.04	0.29	0.28	-0.15	0.32	-1.06	-2.13
R1-MP2	-2.02	.34	.38	-.16	.30	-.20	-2.12
R1-MP5	-1.30	-.11	-.44	-.17	.30	-.56	-2.70
R2-MP1	-2.34	.23	.09	-.08	.39	2.06	.41
R2-MP2	-2.21	.11	-.18	-.09	.39	1.47	.04
R2-MP5	-1.33	-.38	-1.12	-.12	.35	-.42	-1.61
R4-MP1	-1.90	.08	-.22	-.07	.40	.15	.26
R4-MP2	-1.41	-.24	-.83	-.10	.37	-1.00	-.65
R4-MP4	-1.35	-.15	-.64	-.01	.46	-1.75	-2.49
R6-MP1	-1.86	.10	-.15	-.11	.36	1.75	.11
R6-MP2	-1.64	-.08	-.51	-.12	.35	-1.23	-2.68
R6-MP3	-2.53	.80	1.24	-.10	.38	1.77	-1.87
R7-MP2	-1.79	.25	.14	-.12	.35	1.14	.05
R7-MP4	-1.92	.29	.21	-.14	.33	NA	NA
R8-MP3	-2.08	-.02	-.27	-.16	.31	.57	-2.42
S5-MP1	-2.18	.32	.25	-.35	.38	1.87	.49
S5-MP2	-2.27	.29	.16	-.08	.39	2.01	.38
S5-MP4	-1.94	.22	.15	-.06	.40	NA	NA
S10-MP1	-2.23	.10	-.20	-.10	.37	3.34	.00
S10-MP2	-2.12	.24	.11	-.10	.37	NA	NA
S10-MP5	-1.71	.14	-.08	-.06	.40	NA	NA
S14-MP1	-2.07	.03	-.32	-.09	.44	1.09	.24
S14-MP2	-1.91	.00	-.38	-.07	.41	.95	-.11
S14-MP4	-1.83	.17	.01	-.05	.41	.26	-2.28
Error[b]	.20	.10	.20	.20	.20	.50	.50

[a] Well numbers ending in "MP1" indicate that the sampling port is screened approximately 60 ft below land surface; "MP2", screened zone is approximately 40 ft below land surface; "MP3", "MP4", and (or) "MP5", uppermost saturated port.

[b] Combined uncertainty of the mineral equilibrium constant and analytical error (from Dumouchelle and others, 1993, table 16, p. 118-119).

Sources of Silica

Waters are generally saturated with respect to chalcedony (within the range of error on the SI calculations), and oversaturated with respect to quartz (table 16). Although oversaturation with respect to quartz is common in natural waters, the kinetics of quartz make it unlikely to precipitate at surface temperatures (Hem, 1989, p. 71). Instead, SI calculations indicate that chalcedony may be controlling silica concentrations in ground water at the site.

Carbonate Equilibria Reactions

Many of the reactions controlling the aqueous chemistry at the site are caused by recharge across the face of the unconfined aquifer. As carbon dioxide from the atmosphere and produced by plants and microbial activity is leached from the soil zone by infiltrating precipitation, carbonic acid is produced:

$$H_2O + CO_2 = H_2CO_3 \text{ (carbonic acid)} = HCO_3^- + H^+. \qquad (10)$$

The acid dissociates into bicarbonate and hydrogen, thereby decreasing pH of ground water with respect to surface water.

One consequence of the presence of carbonic acid in waters recharging the aquifer is the dissolution of carbonate minerals. The dissolution reaction for dolomite (the dominant carbonate mineral in the outwash aquifer, as identified by XRD) can be written as follows:

$$CaMg(CO_3)_2(s) + 2H_2CO_3 = Ca^{2+} + Mg^{2+} + 4HCO_3^-. \qquad (11)$$

This reaction increases the bicarbonate concentration in the aquifer. Bicarbonate then acts to neutralize pH with depth, as shown by the boxplot data (fig. 24). The neutralization reaction also causes alkalinity to decrease with depth. To test whether the dissolution of dolomite is controlling concentrations of calcium, magnesium, and bicarbonate in the outwash aquifer at the MSEA, the investigators plotted molar concentrations of cations for dolomite as a function of the anion bicarbonate (fig. 38) for the shallow, intermediate, and deep zones of the aquifer. As indicated by the stoichiometry of equation 11, dissolution of dolomite would yield a molar ratio of calcium plus magnesium to bicarbonate of 1:2. Therefore, the ions in figure 38, if affected by carbonate dissolution, should plot along a line with a slope of 0.5.

Linear-regression analysis of calcium plus magnesium concentration as a function of bicarbonate concentration for the shallow aquifer data (fig. 38) yields a slope of 0.38 and a correlation coefficient of 0.62. Departure from the predicted slope of 0.5 indicates that other, concurrent reactions may be affecting calcium, magnesium, or bicarbonate concentrations in the aquifer. Several possible reactions involving bicarbonate are discussed later in this section. Departure from the predicted slope increases with depth below land surface; data from the deep zone of the outwash aquifer yield a slope of only 0.25 and a correlation coefficient of only 0.34.

Oxidation-Reduction Reactions

Oxidation of Organic Carbon

Like carbon dioxide, oxygen enters the ground-water system through infiltrating recharge and by movement of air through unsaturated material above the water table, so it is most concentrated in recharge areas. Atmospheric oxygen in contact with water becomes dissolved in the water at an equilibrium concentration that is a function of temperature and pressure. Areas where dissolved oxygen concentration in ground water are similar to atmospheric equilibrium concentrations indicate areas of rapid recharge.

Once dissolved oxygen enters the ground-water system, it may be depleted by biologically mediated reactions involving the oxidation of organic materials present in subsurface sediments. This process can be expressed as

$$CH_2O \text{ (organic matter)} + O_2(aq) = CO_2(g) + H_2O = HCO_3^- + H^+ \qquad (12)$$

The reaction depicted in equation 12 consumes organic carbon and dissolved oxygen in ground water and produces carbon dioxide in the form of carbonic acid, which can then react with available carbonate minerals to produce bicarbonate. Naturally, the magnitude of the reaction is greatest in areas where the reactants (organic carbon and dissolved oxygen) are most abundant. The occurrence of this reaction in the shallow aquifer is consistent with dissolved oxygen, organic carbon, and pCO_2 data. Dissolved oxygen profiles in the aquifer (fig. 29) show that dissolved

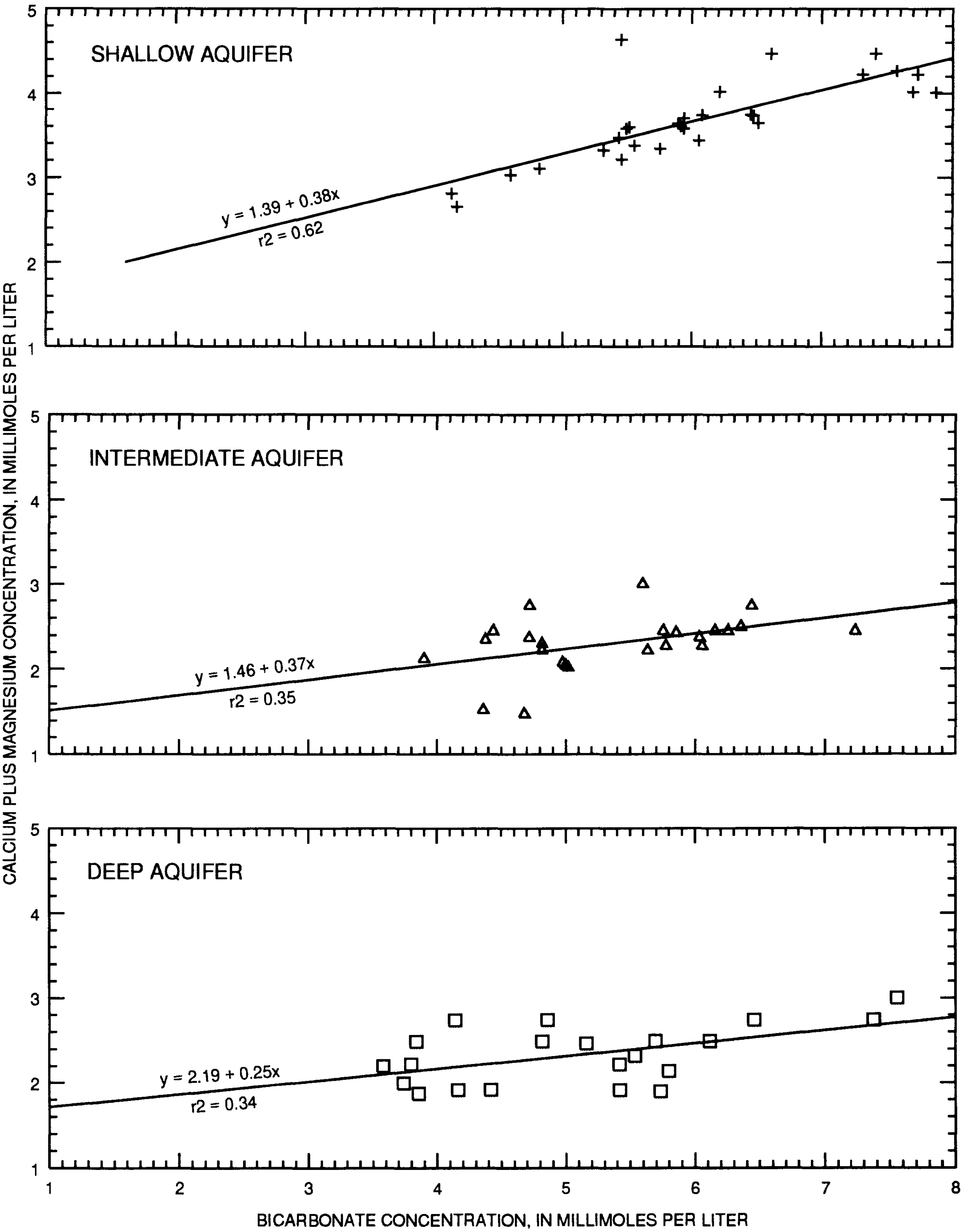


Figure 38. Concentration of calcium plus magnesium as a function of concentration of bicarbonate in shallow, intermediate, and deep zones of the outwash aquifer, November 1991.

oxygen becomes depleted with depth. Similarly, organic carbon profiles (fig. 11) show that organic carbon concentration is highest in the unsaturated zone, near the source of decaying vegetative matter. At the water table, where moisture facilitates chemical reactions and dissolved oxygen is abundant, organic carbon concentrations are near zero, likely because of the rapid oxidation of organic carbon to carbon dioxide. As the oxidation of organic carbon slows between the water table and about 40 ft below land surface as a result of the depletion of dissolved oxygen, organic carbon concentrations in the sediments increase as more of the originally deposited organic carbon in the sediments is left unoxidized.

As data in table 16 indicate, outwash waters have abundant carbon dioxide, with partial pressures of carbon dioxide one to two orders of magnitude greater than water in equilibrium with the atmosphere (P_{CO2atm} = -3.5 at 25 °C) (Breen, 1988). Thus, a source of carbon dioxide other than atmospheric, such as that described by equation 12, must be adding carbon dioxide to the system. P_{CO2} is highest near the water table and generally decreases with depth (table 16).

The reaction described by equation 12 also explains the decrease in dissolved oxygen concentrations observed from April through November in the shallow outwash aquifer. Dissolved oxygen introduced in April by recharge to the aquifer was depleted by oxidation of organic matter. The subsequent onset of drought reduced the replenishment of dissolved oxygen to the aquifer. The decrease in dissolved oxygen concentrations also reduced the oxidizing potential of the aquifer, causing the observed decrease in Eh throughout the 1991 growing season.

Oxidation-Reduction of Iron and Manganese

In addition to the oxidation of organic carbon discussed above, several other redox reactions consume dissolved oxygen, including the oxidation of iron and manganese. Dissolved iron in ground water can have many sources, including (1) dissolution of iron-bearing calcite or dolomite, (2) dissolution of iron-bearing igneous and metamorphic minerals in outwash sediments, (3) dissolution of siderite, or (4) oxidation of pyrite.

After ferrous iron enters the ground-water system, it is oxidized according to reactions 13 and 14:

$$Fe^{2+} + 0.5\ O_2(aq) + 2H^+ = Fe^{3+} + H_2O; \qquad (13)$$

$$2Fe^{3+} + 6H_2O = 2Fe(OH)_3(\text{ferrihydrite}) + 6H^+. \quad (14)$$

Iron oxidation consumes dissolved oxygen, removes dissolved iron from the system by precipitating amorphous ferrihydrite, and results in a net decrease in pH. Evidence of iron oxidation at the MSEA can be obtained from a visual inspection of geochemical cross sections (figs. 28, 29, and 32). These plots show that dissolved iron generally is not present in the shallow outwash where oxygenated conditions (dissolved oxygen greater than 1 mg/L) and Eh greater than 200 mv prevail. Moreover, the transitional zone of the aquifer where dissolved iron concentrations go from below detection limits to greater than 1,000 µg/L corresponds fairly well to the observed color change from yellowish brown to gray in the aquifer sediments (and the associated change from stained to clear quartz grains). The yellowish-brown grain coatings are likely ferrihydrite or other oxyhydroxides. (A notable exception to the correlation of detectable dissolved iron concentrations and the presence of yellowish-brown sediments was drill cuttings from well R1, which were gray at altitudes well above those at which iron was detected in ground water. One explanation is that easily oxidized iron was already oxidized and flushed from the system in this recharge area.) Ferrihydrite coatings in the unsaturated zone and the shallow outwash aquifer are significant because they are highly absorptive (Hem, 1989) and may limit the movement of triazine herbicides in the unsaturated zone and in the shallow outwash aquifer (Armstrong and others, 1967).

In most intermediate and deep waters in the outwash aquifer, where dissolved iron is present, SI's for siderite (fig. 39) show that the iron carbonate is in equilibrium and, therefore, could be controlling the iron concentrations in this zone of the aquifer. In contrast, siderite is undersaturated (and therefore favored to dissolve) in shallow outwash waters, as well as in deep and intermediate levels, respectively, of wells R1 (near Big Beaver Creek) and R6 (near the Scioto River). The zones of siderite undersaturation in the aquifer correspond with oxidizing areas within the aquifer, where the ferrous iron in siderite would be thermodynamically unstable.

Dissolved manganese in the aquifer has a similar distribution to that of dissolved iron, but it is present at somewhat lower concentrations. Sources of manganese in the aquifer may include substitution of small amounts of manganese for calcium in dolomite and limestone, or weathering of pyroxene and amphibole sand grains (Hem, 1989, p. 85). Dissolved manganese, which is not as easily oxidized as ferrous iron, will generally precipitate as a coating of manganese dioxide in the presence of elevated dissolved oxygen (Hem, 1989, p. 86). Manganese is one of the metals that commonly coprecipitates with iron.

Oxidation-Reduction of Nitrogen

Nitrogen, like iron, manganese, dissolved oxygen, and organic carbon, is influenced by redox reactions. Nitrogen fixation by legumes and application of synthetic fertilizer and manure are likely sources of nitrogen at the MSEA. In the oxidizing conditions found in the unsaturated zone and the shallow outwash aquifer, nitrogen entering the system in reduced organic forms can be converted by soil bacteria into the anionic forms, nitrite and nitrate, through a process known as nitrification. Nitrite is metastable in oxygenated water and therefore is rarely detected. Nitrate is the form most commonly found in natural waters. Its presence in the outwash aquifer at the Ohio MSEA is correlated with oxic conditions.

As previously stated, flow of ground water across the oxic-anoxic boundary is indicated by tritium data (which indicate that the aquifer is well mixed) and by the vertical hydraulic gradients of chemical plumes (which are assumed to be more accurate in the long term than those indicated by monthly calculations of the gradient based on measured water levels). Because this site has been farmed since 1801, nitrates should be found in all tritiated waters if nitrate concentrations are being conserved in ground water. The fact that ground-water flow must be occurring across the oxic-anoxic boundary, and that nitrate is present in the oxic zone and absent in the anoxic zone

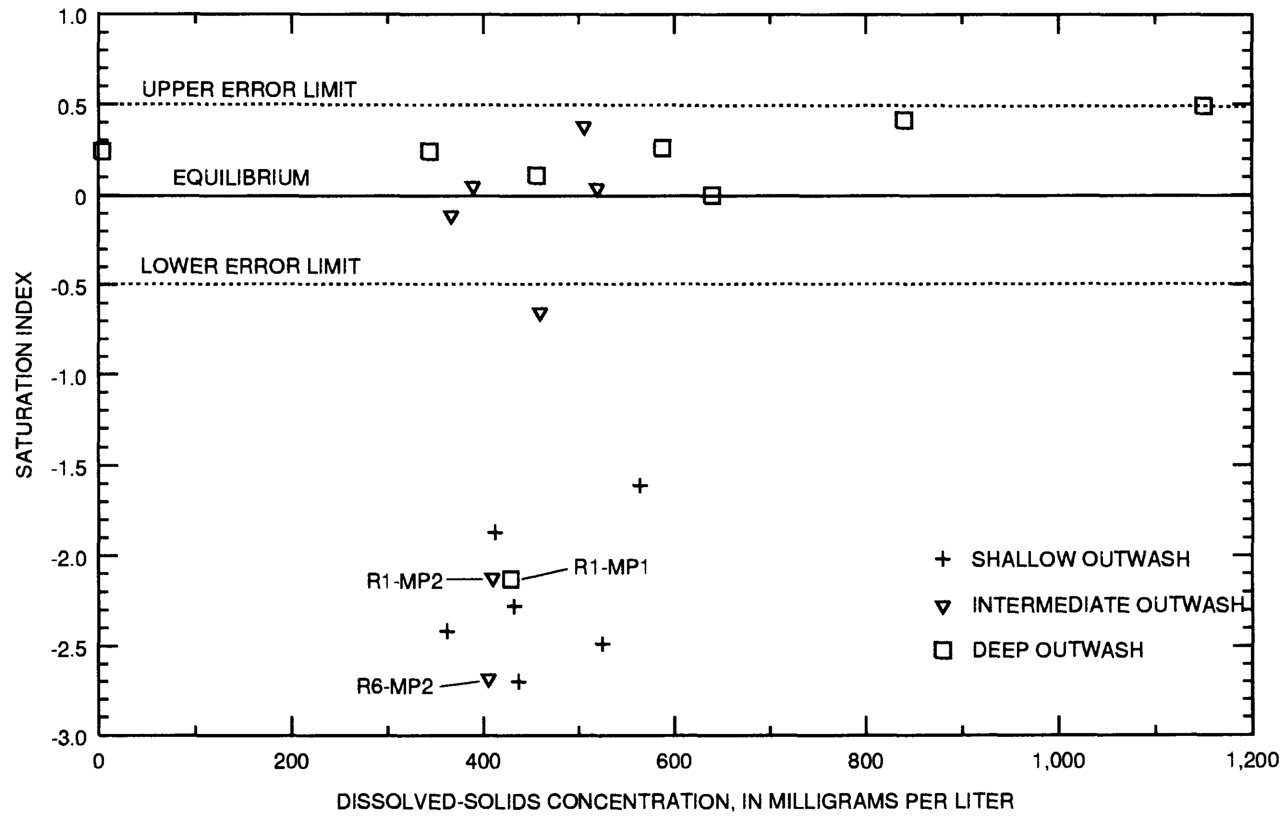


Figure 39. Saturation index of siderite as a function of dissolved-solids concentration in ground water at the Ohio Management Systems Evaluation Area, November 1991.

(figs. 30 and 31), indicates that reduction of nitrate (denitrification) must be occurring at or below the oxic-anoxic boundary.

To prove unequivocally that denitrification is occurring at the MSEA site would require more data concerning the nitrogen chemistry of the MSEA ground water, such as dissolved-gas data (to indicate the presence of excess nitrogen gas in the anoxic zone) or nitrogen isotope data. Although such data are unavailable, the following necessary conditions for denitrification do exist at the site:

1. Anoxic conditions exist at depth. Dissolved oxygen is thermodynamically favored to be reduced by microbes before nitrates are, therefore, anoxic conditions are necessary for denitrification to occur. Gilham and Cherry (1978) found that in an unconfined sandy aquifer, denitrification occurred at dissolved oxygen concentrations less than or equal to 2.0 mg/L.

2. Suitable electron donors are present in ground water and (or) aquifer sediments to enable the reduction of nitrates; such donors include organic carbon and reduced manganese, iron, and sulfides. One unit mass of organic carbon typically can reduce less than one unit mass of nitrate-nitrogen (Korom, 1992). Because ground water in the outwash aquifer typically contains about 0.7 mg/L of dissolved organic carbon and the median concentration of nitrate in shallow outwash water is 3.1 mg/L (with a maximum of 16 mg/L), less than one-fourth of the nitrate ions can be reduced by oxidation of organic carbon dissolved in ground water. Abundant organic carbon is present in the aquifer sediments (median concentration, 3.9 g/kg). Although little is known about the availability of organic matter in aquifers for use by bacteria (Postma and others, 1991), the decrease in organic carbon concentration of the aquifer sediments at depths greater than about 40 ft below land surface corresponds with the zone of iron and manganese reduction and the absence of nitrates in the ground water. This combination of factors indicates that organic matter in the sediments may be bioavailable and could be disappearing because of use as the electron donor for reduction of nitrate, iron, and (or) manganese oxides.

Another necessary condition for denitrification is the presence of bacteria with the metabolic capability of reducing nitrates. Although isolation of such bacteria was not attempted at the MSEA, it is reasonable to assume that denitrifying bacteria are widespread in aquifers (Starr and Gillham, 1993; Korom, 1991, 1992).

Denitrification at the MSEA was qualitatively tested according to two scenarios for donation of electrons to the nitrate ion: (1) oxidation of organic carbon and (2) oxidation of pyrite. The denitrification reaction using organic matter as the main electron donor can be written

$$5C + 4NO_3^- + 2H_2O = 2N_2(g) + 4HCO_3^- + CO_2, \quad (15)$$

where C is an organic compound with an oxidation state of zero (Korom, 1992, p. 1,660).

Oxidation of pyrite is microbially catalyzed (Kölle and others, 1987) and likely occurs in the shallow alluvial aquifer according to equation 16,

$$FeS_2(\text{pyrite}) + 3.5\,O_2(aq) + H_2O = Fe^{2+} + 2SO_4^{2-} + 2H^+, \quad (16)$$

providing a source of iron to the system as well as sulfur. Pyrite is a likely source of sulfur in the outwash aquifer, as indicated by sulfur isotope data from glacial deposits and underlying Silurian and Devonian carbonate bedrock from western Ohio. Isotope samples from parts of Ohio not proximate to Lake Erie were generally negative (-10.4 to -5.8 permil), indicating that a major source of sulfate in the aquifers is oxidation of pyrite (Lori L. Lesney, U.S. Geological Survey, oral commun., 1994; unpublished data on file at the Columbus, Ohio, office of the U.S. Geological Survey). Oxidation of pyrite consumes dissolved oxygen in the aquifer while lowering pH. Although the reaction specified in equation 16 may be occurring in the shallow outwash aquifer, increasing sulfate concentrations with depth in the outwash aquifer indicate that the oxidation of sulfides occurs even in the anoxic zone of the aquifer.

Autotrophic oxidation of pyrite concurrent with denitrification in the anoxic zone of the outwash aquifer can be written as

$$5FeS_2 + 14NO_3^- + 4H^+ = 7N_2 + 10SO_4^{2-} + 5Fe^{2+} + 2H_2O. \quad (17)$$

This reaction has been reported to occur in the Fuhrberger Feld aquifer in northern Germany (Kölle and others, 1985). Böttcher and others (1989) and Frind and others (1990) determined that, in the Fuhrberger Feld aquifer, this reaction has a half-life of 1.2 to 2.1 years, proving that it is a kinetically feasible reaction. Korom (1992) also verified the ability of some bacteria to reduce nitrates, using sulfides as electron donors. In the reaction specified by equation 17, sulfate is produced and nitrate is the electron acceptor. Chemical profiles in the outwash aquifer are consistent with the reaction proposed in equation 17 in that pH and concentrations of sulfate and dissolved iron increase with depth in the aquifer as nitrate concentrations decrease.

Denitrification (according to eqs. 15 and 17) was evaluated by use of the geochemical modeling program, NETPATH (Plummer and others, 1991). NETPATH helps quantify the net masses of minerals required to dissolve or precipitate along a hydrologic flow path to account for the observed differences in constituent concentrations between an initial and final water. The chemical evolution of water along the flow path, as calculated by NETPATH, is constrained by chemical mass balances involving element or electron balances. Input to the model includes initial and final water composition, sets of elemental constraints, and a set of phases considered to be appropriate for the reactions under consideration. Because water quality data were not available for wells directly on a flow path (see fig. 12), samples from wells S5-MP5 and R4-MP2 (June 1991) (fig. 2) were chosen as being representative of the chemical trends observed for ground water flowing from oxic to anoxic zones in the glacial aquifer. Elemental and electron mass-balances were used to constrain the mass-transfer models; phases selected were minerals and gases known to be present in the glacial aquifer, including phases hypothesized to be involved in denitrification reactions (eqs. 15 and 17) and phases involved in carbonate equilibria reactions and iron and manganese redox reactions proposed previously in this report. Results of previous SI calculations were used to constrain selected minerals phases to either dissolve or precipitate so that models that were not favored thermodynamically did not result (for example, precipitation of goethite[11] in the anoxic zone).

The mass-transfer equations resulting from NETPATH represent the net millimoles of each phase entering (left side of the equation) or leaving (right side of the equation) a kilogram of water by dissolution or precipitation as the kilogram of water moves from the upgradient well to the downgradient well. Two NETPATH models were generated, one using CH_2O as a reacting phase to simulate organic matter with a carbon valence of zero (eq. 18) and one with no organic matter to act as an electron donor (eq. 19). Thus, a kilogram of water moving along the flow path from well S5-MP5 to well R4-MP2 would evolve according to the following net reactions:

$$\begin{gathered}\text{S5-MP5 water} + 0.33\ \text{pyrite} + 0.041\ \text{dolomite} + \\ 0.56\ \text{calcite} + 1.2\ CH_2O + 0.0086\ \text{MnOOH} + \\ 1.2\ O_2(g) = \text{R4-MP2 water} + 0.39\ N_2(g) + \\ 0.31\ \text{siderite}\end{gathered} \tag{18}$$

or

$$\begin{gathered}\text{S5-MP5 water} + 0.14\ \text{goethite} + 0.33\ \text{pyrite} + \\ 0.041\ \text{dolomite} + 0.56\ \text{calcite} + 1.3\ CO_2(g) + \\ 0.0086\ \text{MnOOH} = \text{R4-MP2 water} + \\ 0.39\ N_2(g) + 0.45\ \text{siderite.}\end{gathered} \tag{19}$$

Equation 18, which uses organic matter as the predominant electron donor for redox reactions, is not realistic as written because it requires an unreasonable amount of O_2 to be added to the system (1.2 mmol, or 39 ppm dissolved oxygen). The highest concentration of dissolved oxygen measured in water recharging the water table at the MSEA was 0.24 mmol/L, or 7.7 ppm.

The mass-transfer modeling indicates that denitrification at the MSEA site may be largely driven by the autotrophic oxidation of pyrite present in aquifer sediments (eqs. 17 and 19). It also indicates that pyrite reserves in the sand and gravel aquifer may be critical to the remediation of anthropogenic nitrate contamination at the MSEA. Postma and others (1991) studied an unconfined sandy aquifer in western Denmark and found pyrite to be by far the dominant electron donor for nitrate reduction, even though organic carbon was abundant in the aquifer. It should be noted that no models could be generated by NETPATH (electrons could not be balanced) if nitrogen was not included as a reacting phase and a constraint.

Denitrification at depth creates a natural zone of remediation, where deleterious nitrates from

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[11] For the NETPATH modeling, goethite (also an oxihydroxide mineral) was substituted for ferrihydrite.

anthropogenic and natural sources are removed from the ground-water system. Generally, drinking water in this area of the Scioto River Valley is free of nitrates because most drinking-water wells in the valley are screened in the deep zone of the outwash aquifer. In the shallow aquifer, the presence of nitrates increases the oxidation potential of the aquifer, because nitrate is an electron acceptor.

In addition to reduction of nitrates to nitrogen gas, nitrates are sometimes reduced to ammonia (Korom, 1992). Such a process could be the source of ammonia that is found in bedrock waters at the MSEA (table 13, fig. 24); however the ammonia could also be formed by the microbial mineralization of organic nitrogen in the carbonaceous Ohio Shale to inorganic nitrogen. The latter process is the more likely, because connate water has very low nitrate concentrations to begin with. However, analysis of nitrogen isotopes would be required to verify the dominant process in the bedrock aquifer.

Oxidation-Reduction of Sulfur

Ground-water samples were not analyzed for hydrogen sulfide, which contains a reduced form of sulfur; however, this compound has a distinctive rotten-egg odor that is detectable to most humans at concentrations of a few tenths of a milligram per liter (Hem, 1989, p. 117). A hydrogen sulfide odor was noted only once in the outwash aquifer, at well S5-MP1. Sulfate reduction probably does occur in the bedrock aquifer, however. This interpretation is based on the odor of hydrogen sulfide detected in water from bedrock wells, the absence of sulfates above concentrations found in precipitation, and measured Eh values in bedrock water that fall within the prescribed range for sulfate reduction.

Effect of Surface Water on Ground-Water Chemistry

The chemistry of the outwash aquifer, in addition to being affected by rock-water interactions and bedrock mixing, is also affected by surface-water contributions from the streams bounding the study area. The presence of elevated dissolved oxygen concentrations and Eh and relatively low concentrations of dissolved iron and manganese at unusual depths near Big Beaver Creek and the Scioto River indicates that a zone of oxidized ground water surrounds the streams. The reducing zone that is observed beneath the middle terrace at about 40 ft below land surface is absent at well cluster R1, near Big Beaver Creek, and is present only at depths greater than approximately 60 ft below land surface near the Scioto River. The oxidizing zone around the streams indicates one of the following scenarios:

1. Downflow of oxygenated recharge occurs at these locations, at least periodically. At R1, advection and dispersion transport oxygen to the base of the aquifer.

2. Organic carbon concentration in R1, R5, and R6 sediments is less than that observed at other locations in the aquifer. Therefore, less oxidation of organic carbon occurs, and dissolved oxygen concentrations in the ground water are depleted less rapidly at these locations. Data in table 5 show that this is not the case; organic carbon concentration near the streams is greater than or equal to that observed at other aquifer locations.

3. The most reactive components of organic carbon have already been removed in these recharge areas. The remaining carbon is not easily oxidized and therefore does not deplete dissolved oxygen in the ground water.

A combination of scenarios 1 and 3 is the most likely explanation. Thus, the chemical data independently support the hydraulic evidence that recharge is occurring beneath Big Beaver Creek and that bank storage periodically recharges the alluvial aquifer adjacent to the Scioto River. However, differences in conservative ion concentrations in surface water and shallow outwash waters are not great enough to determine mixing percentages of surface water and ground water.

SUMMARY AND CONCLUSIONS

Five Management Systems Evaluation Areas (MSEA's) were established in the Midwestern United States in 1990 to address growing public concern about agricultural nonpoint-source contamination of ground-water resources. The purpose of the MSEA program is to investigate the effects of various agricultural management systems on ground-water quality and to identify factors affecting the fate and transport of pesticides and nutrients.

The Ohio MSEA is located on a 650-acre farm in the Scioto River Valley in Pike County, south-central Ohio. Three farming plots as well as small replicate plots were established at the site in 1991.

The farm is underlain by an incised bedrock valley composed of Ohio Shale. When melting of the Wisconsinian glaciers began, the valley was filled with about 70 ft of outwash sediments. The outwash is composed predominantly of discontinuous interbeds of sand, gravelly sand, and sandy gravel with few intervening silt or clay layers. The outwash deposits are overlain by a veneer of recent alluvium that is generally less than 11 ft thick and is composed of silty clay and some sand interbeds. Soils overlying the alluvium are silt loam and sandy loam. Wells in the Scioto River Valley yield as much as 1,300 gal/min and are used for public and private water supply, industrial process water, and aquaculture.

The hydraulic conductivity of sediments in the study area, as determined from equations based on grain-size distribution, increases with gravel content and, therefore, is greatest in shallow and deep zones of the outwash aquifer and lowest at intermediate depths. Hydraulic conductivities (K's) at the site were also evaluated by slug, specific-capacity, and regional aquifer tests. Although K's measured at a given location can vary with the scale of the test method, relative variation in K's determined by specific-capacity tests and slug tests is the same areally beneath the three research plots. Specifically, materials beneath plot 200 have the greatest K, followed by those at plot 100 and plot 300. Distance-drawdown data from USGS aquifer tests show K to range from 401 to 555 ft/d at the scale of the entire site.

Outwash sediments are composed predominantly of dolomite, quartz, and calcite. The median concentration of organic carbon in subsurface sediments was 0.39 weight percent, with concentrations ranging from less than 0.01 to 1.83 percent. No statistically significant correlation was found between organic carbon concentration and grain-size distribution. Instead, subsurface distribution of organic carbon differed with depth, and changes in concentration could generally be explained by redox reactions in the aquifer that consume organic carbon.

Ground water beneath the MSEA flows from east to west-southwest, except during periods of high stage on the Scioto River. Ground-water-flow velocities range from about 3.3 ft/d beneath plot 200 to an average flow velocity of about 1.7 ft/d beneath plots 100 and 300. Horizontal hydraulic gradients beneath the plots range from 5×10^{-4} to 0.002. A nitrate plume observed at the site indicates a steeper component of downflow in the aquifer than is indicated by monthly water-level measurements. It is assumed that the chemical data more accurately represent the long-term, average vertical hydraulic gradient than do the computations of gradient obtained from monthly water-level data. Tritium data also support the existence of a vertical component of flow by indicating that the aquifer is well mixed. Even the deepest and most downgradient parts of the aquifer contain water that has recharged the system since 1953.

Ground water and surface water at the site are highly interconnected. Comparison of ground-water and surface-water hydrographs shows that shallow ground water at the west edge of the site discharges to the Scioto River, except during frequent times of high river stage. At these times, flow reverses and water moves away from the stream into the riverbanks. The maximum observed duration of a flow reversal was 17 days, during which time a particle of water could have migrated about 190 ft inland.

Big Beaver Creek recharges the outwash aquifer on the southeastern edge of the study area, as indicated by seepage-meter tests, surface-water-stage and ground-water-head data, and water-quality data. A zone of oxidizing waters (high in dissolved oxygen and Eh) surrounding Big Beaver Creek and the Scioto River is caused by the periodic inflow of surface waters to the outwash aquifer.

Direction and magnitude of vertical ground-water flow between the bedrock and outwash aquifers differs areally across the site. Consistent downward gradients between the outwash and bedrock aquifers were measured at well R1 near Big Beaver Creek; consistent upward gradients from bedrock to outwash were measured at well R2. Direction of vertical flow between aquifers was variable near the Scioto River, owing to frequent episodes of high river stage. Areas in the bedrock unit where periodic or consistent downflow occurs are indicated by spatial differences in chloride concentration where bedrock waters have been diluted by outwash waters. This finding indicates not only that pressure gradients exist between the aquifers but also that flow between the two aquifers occurs. The untritiated nature of ground water in the bedrock aquifer, however, indicates that flow rates are very slow and may approach a maximum of only 0.2 ft/yr.

A ground-water budget for water year 1992 indicates that only 0.02 percent of recharge to the outwash aquifer was from leakage of bedrock waters. Infiltrating precipitation accounted for 72 percent (12.70 in.) of recharge, and infiltration of surface water from Big Beaver Creek accounted for 28 percent (5.0 in.). Twenty-four percent of water recharged to the aquifer in water year 1992 was stored in the ground-water system at the end of the year. Of the 16.2 in. of water discharged from the aquifer, 60 percent exited as base flow to the Scioto River, and 40 percent was withdrawn from the DOE well field.

Areal variation in water quality at the site is caused by areal differences in the relative importance of the 3 recharge sources. Most outwash waters are calcium magnesium bicarbonate type, whereas deep outwash waters in the northeast corner of the study area are transitional between calcium magnesium bicarbonate waters and the calcium sodium chloride waters of the bedrock aquifer. These deep waters are also high in specific conductance, dissolved solids, and potassium, and low in alkalinity—all distinguishing characteristics of water from the bedrock aquifer. Mixing calculations by use of the conservative constituent chloride indicate that deep waters in the northeast corner of the MSEA consist of as much as 26 percent bedrock water. The effect of bedrock water on the deep outwash waters in this part of the study area is likely caused by the lack of dilution by waters from Big Beaver Creek. In the central and southern parts of the study area, Big Beaver Creek flows over a sand and gravel streambed through which infiltration readily occurs, thereby quickly flushing from the outwash aquifer any evidence of bedrock upflow. However, in the northeastern part of the study area, the streambed of Big Beaver Creek is composed of shale; no infiltration occurs in this area.

Although mixing of bedrock and outwash waters appears to be the dominant process in the northeast corner of the study area, the distribution of aqueous constituents throughout the outwash aquifer is also affected by rock-water interactions. Chalcedony is in equilibrium with ground water at the site, an indication that dissolution and precipitation of chalcedony likely control silica concentrations in ground water. Calcium, magnesium, and bicarbonate concentrations in ground water are controlled by dissolution of dolomite, and concurrent processes such as bedrock mixing also affect concentrations of these constituents. Siderite is also in equilibrium with deep and intermediate ground water and, therefore, likely controls concentrations of iron in these zones of the aquifer.

Redox reactions in the outwash aquifer seem to be largely responsible for the observed variations in water quality with depth. The dissolved oxygen that enters the ground-water system as recharge is consumed by oxidation of organic carbon, iron, and manganese. Many of these reactions release H^+ ions, thereby lowering the pH of shallow ground water. (The pH of shallow ground water is also lowered by the leaching of carbonic acid from the soil zone to the water table. Neutralization of pH at depth is a result of the dissolution of dolomite.) Oxidized iron is precipitated as amorphous ferrihydrite, resulting in the iron staining observed on quartz grains in the shallow aquifer sediments. Also present in the shallow, oxidized zone of the aquifer are nitrates, formed by the oxidation of fertilizers and the bacterially fixed atmospheric nitrogen. The median concentration of nitrates in 1991 was 3.1 mg/L; the maximum concentration was16 mg/L during the three sampling rounds.

At approximately 40 ft below land surface, most of the dissolved oxygen in ground water has been consumed; here, bacteria switch from using dissolved oxygen to using nitrate as the electron acceptor for their metabolic reactions. At depths greater than 40 ft below land surface, the reduction of nitrates (denitrification) is linked with the oxidation of pyrite. The oxidation of pyrite releases sulfates to the deep aquifer waters. Geochemical modeling indicates that organic carbon is not the predominant electron donor in this zone of the aquifer. Once nitrates are consumed, iron and manganese are reduced by microbially mediated redox reactions.

Surface-water quality reflects the drainage-basin geology and residence time of waters in the subsurface materials before being discharged to the stream. Thus, waters of the Scioto River are similar in chemical composition to waters of the outwash aquifer, whereas waters from Big Beaver Creek are much more dilute. Land use in the drainage basin also affects water quality of the streams. The abundance of farming in the Scioto River Valley is evidenced by the year-round presence of the herbicides atrazine and metolachlor in the Scioto River, with peak concentrations occurring in June. Herbicides are not detected in ground water at the Ohio MSEA.

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Table 1. Summary of well construction at Ohio Management Systems Evaluation Area

[Well locations are shown in figure 2. Abbreviations: BR, bedrock well; MP, multiport well; WT, water-table well; P, piezometer, USGS, U.S. Geological Survey; ft, feet; QW, water quality; WL, water level SPC, specific conductance; T, water temperature. Screened interval begins at base of well. Instrumentation refers to data recorded hourly by electronic data loggers; K determination refers to slug and (or) specific-capacity tests to estimate hydraulic conductivity off the screened interval]

Local well number	USGS identification number	Land surface elevation (ft)	Total depth (ft below land surface)	Screen length (ft)	Well use and instrumentation
R1-BR	390227083013330	550.93	83.25	10	QW; instrumented for WL
R1-MP1	390227083013311	551.03	59.94	.03	QW, WL
R1-MP2	390227083013312	551.03	39.34	.03	QW, WL
R1-MP3	390227083013313	551.03	25.57	.03	QW, WL
R1-MP4	390227083013314	551.03	22.06	.03	QW, WL
R1-MP5	390227083013315	551.03	17.51	.03	QW, WL
R1-MP6	390227083013316	551.03	12.97	.03	QW, WL
R1-WT	390227083013320	551.07	28	20	Instrumented for WL, SPC, T; K determination
R2-BR	390239083015130	552.09	90	10	QW; instrumented for WL
R2-MP1	390239083015111	551.22	60.83	.03	QW, WL
R2-MP2	390239083015112	551.22	40.30	.03	QW, WL
R2-MP3	390239083015113	551.22	25.50	.03	QW, WL
R2-MP4	390239083015114	551.22	20.95	.03	QW, WL
R2-MP5	390239083015115	551.22	16.49	.03	QW, WL
R2-MP6	390239083015116	551.22	11.65	.03	QW, WL
R2-WT	390239083015120	549.90	34.8	20	Instrumented for WL, SPC, T; K determination
R3-MP1	390244083023011	548.77	60.75	.03	QW, WL
R3-MP2	390244083023012	548.77	40.08	.03	QW, WL
R3-MP3	390244083023013	548.77	29.37	.03	QW, WL
R3-MP4	390244083023014	548.77	24.72	.03	QW, WL
R3-MP5	390244083023015	548.77	20.28	.03	QW, WL
R3-MP6	390244083023016	548.77	15.44	.03	QW, WL
R3-WT	390244083023020	548.56	31	20	Instrumented for WL; K determination
R4-MP1	390229083022911	549.18	60.91	.03	QW, WL
R4-MP2	390229083022912	549.18	40.41	.03	QW, WL
R4-MP3	390229083022913	549.18	25.56	.03	QW, WL
R4-MP4	390229083022914	549.18	21.05	.03	QW, WL
R4-MP5	390229083022915	549.18	16.48	.03	QW, WL
R4-MP6	390229083022916	549.18	11.62	.03	QW, WL
R4-WT	390229083022920	549.22	32	20	Instrumented for WL;K determination
R5-MP1	390229083025011	545.80	59.5	.03	QW, WL
R5-MP2	390229083025012	545.80	39	.03	QW, WL
R5-MP3	390229083025013	545.80	21.5	.03	QW, WL
R5-MP4	390229083025014	545.80	17	.03	QW, WL
R5-MP5	390229083025015	545.80	12.50	.03	QW, WL
R5-MP6	390229083025016	545.80	8	.03	QW, WL
R5-WT	390229083025020	546.21	28	20	Instrumented for WL; K determination

Table 1. Summary of well construction at Ohio Management Systems Evaluation Area—Continued

Local well number	USGS identification number	Land surface elevation (ft)	Total depth (ft below land surface)	Screen length (ft)	Well use and Instrumentation
R6-BR	390207083024030	552.79	89.5	10	QW; instrumented for WL; K determination
R6-MP1	390207083024011	552.68	60.16	.03	QW, WL
R6-MP2	390207083024012	552.68	41.72	.03	QW, WL
R6-MP3	390207083024013	552.68	30.74	.03	QW, WL
R6-MP4	390207083024014	552.68	26.25	.03	QW, WL
R6-MP5	390207083024015	552.68	21.83	.03	QW, WL
R6-MP6	390207083024016	552.68	17.45	.03	QW, WL
R6-WT	390207083024020	552.35	31.40	20	Instrumented for WL, SPC, T; K determination
R7-BR	390156083022830	544.80	81	10	QW; instrumented for WL
R7-MP1	390156083022811	544.74	62.14	.03	QW, WL
R7-MP2	390156083022812	544.74	41.48	.03	QW, WL
R7-MP3	390156083022813	544.74	26.78	.03	QW, WL
R7-MP4	390156083022814	544.74	22.19	.03	QW, WL
R7-MP5	390156083022815	544.74	17.72	.03	QW, WL
R7-MP6	390156083022816	544.74	12.89	.03	QW, WL
R7-WT	390156083022820	544.75	32	20	Instrumented for WL, SPC, T; K determination
R8-MP1	390156083014311	553.02	28.56	.03	QW, WL
R8-MP2	390156083014312	553.02	24.06	.03	QW, WL
R8-MP3	390156083014313	553.02	19.52	.03	QW, WL
R8-MP4	390156083014314	553.02	14.70	.03	QW, WL
R8-WT	390156083014320	552.74	32	20	Instrumented for WL; K determination
R9-P	--	556.19	25	5	WL
R10-P	--	546.08	25	5	WL
R12-P	--	551.86	25	5	WL
R13-P	--	568.69	35	5	WL
S1-MP1	390242083015611	550.32	26.28	.03	QW, WL
S1-MP2	390242083015612	550.32	21.79	.03	QW, WL
S1-MP3	390242083015613	550.32	17.31	.03	QW, WL
S1-MP4	390242083015614	550.32	12.47	.03	QW, WL
S2-MP1	390245083020211	550.14	26.46	.03	QW, WL
S2-MP2	390245083020212	550.14	21.92	.03	QW, WL
S2-MP3	390245083020213	550.14	17.38	.03	QW, WL
S2-MP4	390245083020214	550.14	12.62	.03	QW, WL
S3-MP1	390241083021411	548.76	26.86	.03	QW, WL
S3-MP2	390241083021412	548.76	21.36	.03	QW, WL
S3-MP3	390241083021413	548.76	17.82	.03	QW, WL
S3-MP4	390241083021414	548.76	13.02	.03	QW, WL

Table 1. Summary of well construction at Ohio Management Systems Evaluation Area—Continued

Local well number	USGS identification number	Land surface elevation (ft)	Total depth (ft below land surface)	Screen length (ft)	Well use and instrumentation
S4-MP1	390237083020611	547.23	26.47	.03	QW, WL
S4-MP2	390237083020612	547.23	22.02	.03	QW, WL
S4-MP3	390237083020613	547.23	17.55	.03	QW, WL
S4-MP4	390237083020614	547.23	12.83	.03	QW, WL
S5-MP1	390241083020511	549.88	62.01	.03	QW[a]
S5-MP2	390241083020512	549.88	41.51	.03	QW[a]
S5-MP3	390241083020513	549.88	24.68	.03	QW[a]
S5-MP4	390241083020514	549.88	20.11	.03	QW[a]
S5-MP5	390241083020515	549.88	15.59	.03	QW[a]
S5-MP6	390241083020516	549.88	10.86	.03	QW[a]
S5-WT	390241083020520	550.03	36	20	Instrumented for WL; K determination
S6-MP1	390234083014611	553.33	24.57	.03	QW, WL
S6-MP2	390234083014612	553.33	20.01	.03	QW, WL
S6-MP3	390234083014613	553.33	15.55	.03	QW, WL
S6-MP4	390234083014614	553.33	10.74	.03	QW, WL
S7-MP1	390236083015011	548.27	26.57	.03	QW, WL
S7-MP2	390236083015012	548.27	22.06	.03	QW, WL
S7-MP3	390236083015013	548.27	17.49	.03	QW, WL
S7-MP4	390236083015014	548.27	12.73	.03	QW, WL
S8-MP1	390232083020411	553.56	29.44	.03	QW, WL
S8-MP2	390232083020412	553.56	24.92	.03	QW, WL
S8-MP3	390232083020413	553.56	20.37	.03	QW, WL
S8-MP4	390232083020414	553.56	15.52	.03	QW, WL
S9-MP1	390229083015811	555.55	28.33	.03	QW, WL
S9-MP2	390229083015812	555.55	23.95	.03	QW, WL
S9-MP3	390229083015813	555.55	19.35	.03	QW, WL
S9-MP4	390229083015814	555.55	15.23	.03	QW, WL
S10-MP1	390233083015511	553.66	60.34	.03	QW[a]
S10-MP2	390233083015512	553.66	40.16	.03	QW, WL
S10-MP3	390233083015513	553.66	29.70	.03	QW, WL
S10-MP4	390233083015514	553.66	25.29	.03	QW, WL
S10-MP5	390233083015515	553.66	20.74	.03	QW, WL
S10-MP6	390233083015516	553.66	15.81	.03	QW, WL
S10-WT	390233083015520	553.68	35	20	Instrumented for WL, SPC, T; K determination
S11-MP1	390224083015511	553.49	27.17	.03	QW, WL
S11-MP2	390224083015512	553.49	22.68	.03	QW, WL
S11-MP3	390224083015513	553.49	18.17	.03	QW, WL
S11-MP4	390224083015514	553.49	13.38	.03	QW, WL

Table 1. Summary of well construction at Ohio Management Systems Evaluation Area—Continued

Local well number	USGS identification number	Land surface elevation (ft)	Total depth (ft below land surface)	Screen length (ft)	Well use and instrumentation
S12-MP1	390223083020611	552.12	28.81	.03	QW, WL
S12-MP2	390223083020612	552.12	24.29	.03	QW, WL
S12-MP3	390223083020613	552.12	19.80	.03	QW, WL
S12-MP4	390223083020614	552.12	14.96	.03	QW, WL
S13-MP1	390216083020011	551.70	28.29	.03	QW, WL
S13-MP2	390216083020012	551.70	23.82	.03	QW, WL
S13-MP3	390216083020013	551.70	19.35	.03	QW, WL
S13-MP4	390216083020014	551.70	14.44	.03	QW, WL
S14-MP1	390224083020111	553.75	61.37	.03	QW, WL
S14-MP2	390224083020112	553.75	40.83	.03	QW, WL
S14-MP3	390224083020113	553.75	25.92	.03	QW, WL
S14-MP4	390224083020114	553.75	21.30	.03	QW, WL
S14-MP5	390224083020115	553.75	16.75	.03	QW, WL
S14-MP6	390224083020116	553.75	11.89	.03	QW, WL
S14-WT	390224083020120	553.44	29	20	Instrumented for WL, SPC, T; K determination
S15-WT	--	553.95	60	40	K determination
Van Meter	390238083014100	571.18	72	10	WL; drilling fluid; domestic water-supply
ODOT	--	577.76	78	--	WL; water-supply
Brown	--	594.27	85	5	WL; domestic water-supply

[a] Twisted tubing inside multiport well prevented measurement of water levels at this port.

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area

[Well locations and surface-water sampling sites are shown in figure 2; bls, below land surface; ft^3/s, cubic feet per second; ms/cm, microsiemens per centimeter; deg C, degrees Celsius; v, volts; mg/L milligrams per liter; mg/L, microgram per liter; --, not analyzed for]

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Water level (feet bls) or discharge (ft^3/s)	Specific conductance (ms/cm)	pH, field (standard units)	Temperature (deg C)	Eh (v)	Oxygen, dissolved (mg/L)	Calcium, dissolved (mg/L as Ca)	Magnesium, dissolved (mg/L as Mg)
R1-BR	390227083013330	06-19-91	11.04	1,650	7.4	13.5	--	0.1	93	27
		11-12-91	16.05	1,680	7.3	12.5	--	--	94	28
R1-MP1	390227083013311	04-08-91	7.54	610	7.5	12.0	--	1.1	76	27
		06-19-91	--	762	7.5	14.5	--	.7	89	31
		11-14-91	15.80	709	7.5	12.5	0.113	.5	86	29
R1-MP2	390227083013312	04-08-91	7.53	660	7.5	14.0	--	1.5	89	30
		06-19-91	--	688	7.4	13.0	--	1.2	98	33
		11-14-91	15.80	677	7.5	13.0	.162	1.5	91	30
R1-MP5	390227083013315	11-15-91	15.80	767	6.9	13.5	.249	1.0	98	38
R1-MP6	390227083013316	04-08-91	7.44	798	6.5	10.0	--	3.2	110	42
		06-19-91	11.40	637	7.4	17.0	--	7.3	90	34
R2-BR	390239083015130	04-01-91	10.19	6,100	7.4	13.5	--	.9	410	100
		06-18-91	10.52	5,910	7.8	16.5	--	.5	370	100
		11-13-91	15.46	6,400	7.4	13.0	--	--	390	110
R2-MP1	390239083015111	04-01-91	7.73	1,340	7.9	11.5	--	.0	75	20
		06-17-91	11.69	1,370	7.7	14.5	--	.1	80	21
		11-04-91	16.64	1,460	7.6	12.0	.124	.1	100	30
R2-MP2	390239083015112	04-01-91	7.84	772	7.6	11.5	--	.0	61	16
		06-17-91	11.78	712	7.7	14.0	--	.2	59	16
		11-04-91	16.64	869	7.5	12.0	.124	.2	85	23
R2-MP5	390239083015115	06-17-91	11.74	1,030	7.3	13.0	--	1.4	99	30
		11-04-91	16.64	940	6.8	10.5	.200	7.1	97	32
R2-MP6	390239083015116	04-01-91	7.79	586	7.3	11.0	--	.5	85	24
R4-MP1	390229083022911	04-09-91	9.57	934	7.6	15.0	--	.7	100	30
		06-12-91	13.87	939	7.4	15.0	--	.1	110	30
		11-07-91	17.97	979	7.3	12.0	.066	.1	99	29

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Water level (feet bls) or discharge (ft^3/s)	Specific conductance (ms/cm)	pH, field (standard units)	Temperature (deg C)	Eh (v)	Oxygen, dissolved (mg/L)	Calcium, dissolved (mg/L as Ca)	Magnesium, dissolved (mg/L as Mg)
R4-MP2	390229083022912	04-09-91	--	927	7.0	14.5	--	5.7	100	33
		06-12-91	13.83	889	7.3	14.5	--	1.4	98	32
		11-07-91	17.97	914	6.9	12.0	.103	1.1	98	31
R4-MP4	390229083022914	11-07-91	17.97	893	6.9	11.0	.168	4.3	110	36
R4-MP5	390229083022915	06-12-91	13.80	860	7.2	14.0	--	6.2	110	37
R4-MP6	390229083022916	04-09-91	9.51	775	7.1	12.0	--	6.4	110	36
R6-BR	390207083024030	04-04-91	17.26	2,260	7.7	14.0	--	.4	120	33
		06-19-91	22.88	2,270	7.4	14.5	--	.6	120	33
		11-13-91	24.66	2,280	7.4	13.5	--	--	120	33
R6-MP1	390207083024011	04-04-91	19.44	758	7.3	13.5	--	.2	100	31
		06-18-91	--	756	7.1	17.5	--	1.2	100	32
		11-13-91	25.01	759	7.3	12.5	.167	.1	93	29
R6-MP2	390207083024012	04-04-91	19.44	683	7.4	13.5	--	5.7	95	29
		06-18-91	26.90	643	7.4	16.0	--	4.6	97	30
		11-13-91	25.02	580	7.1	12.0	.180	3.2	91	28
R6-MP3	390207083024013	11-14-91	25.06	680	8.0	10.5	.203	.2	90	29
R6-MP4	390207083024014	06-19-91	23.49	689	7.3	16.0	--	5.3	99	31
R6-MP5	390207083024015	04-04-91	19.35	630	7.0	14.0	--	7.0	94	30
R7-BR	390156083022830	06-19-91	18.50	2,090	7.9	15.5	--	.2	140	38
		11-13-91	25.97	2,210	7.7	13.0	--	--	150	42
R7-MP1	390156083022811	04-09-91	--	760	6.9	14.0	--	1.5	110	30
		06-18-91	18.28	845	7.3	16.0	--	.5	120	36
		11-14-91	18.47	806	7.3	13.0	.102	.0	110	30
R7-MP2	390156083022812	04-09-91	--	660	7.6	14.5	--	2.6	89	26
		06-18-91	18.39	699	7.3	14.5	--	0.2	98	30
		11-14-91	18.47	761	7.3	13.5	.139	2.2	110	32

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Water level (feet bls) or discharge (ft^3/s)	Specific conductance (ms/cm)	pH, field (standard units)	Temperature (deg C)	Eh (v)	Oxygen, dissolved (mg/L)	Calcium, dissolved (mg/L as Ca)	Magnesium, dissolved (mg/L as Mg)
R7-MP4	390156083022814	06-18-91	18.27	671	7.5	17.0	--	3.6	98	25
		11-14-91	18.47	722	7.4	14.0	.184	4.6	100	27
R7-MP6	390156083022816	04-09-91	11.08	585	7.6	10.0	--	6.7	82	24
R8-MP1	390156083014311	04-05-91	--	687	7.4	12.5	--	.2	81	29
		06-11-91	14.92	717	7.6	13.5	--	.3	92	32
		11-06-91	18.02	627	7.4	13.5	.400	.5	77	26
R8-MP3	390156083014313	06-11-91	14.85	828	7.1	17.0	--	3.7	100	37
		11-06-91	18.02	606	7.4	14.0	.225	.8	70	26
R8-MP4	390156083014314	04-05-91	--	697	7.3	11.0	--	7.1	100	30
S5-MP1	390241083020511	04-03-91	7.46	1,850	7.6	14.5	--	.3	220	66
		06-12-91	11.42	1,890	7.5	16.0	--	.2	220	64
		11-05-91	16.54	1,920	7.4	13.0	.128	.3	220	61
S5-MP2	390241083020512	04-03-91	--	864	7.7	14.5	--	.3	98	26
		06-12-91	11.41	944	7.6	16.0	--	.0	110	25
		11-05-91	16.54	892	7.6	12.5	.126	.6	94	24
S5-MP4	390241083020514	11-05-91	16.54	787	7.4	14.5	.209	3.4	88	28
S5-MP5	390241083020515	06-12-91	11.41	679	7.4	13.5	--	5.6	76	32
S5-MP6	390241083020516	04-03-91	--	674	--	11.5	--	4.2	62	27
S10-MP1	390233083015511	04-10-91	11.05	1,150	7.6	12.0	--	1.9	110	30
		06-10-91	--	1,170	8.0	15.0	--	.0	88	25
		11-06-91	19.73	1,140	7.5	12.5	.231	.3	89	24
S10-MP2	390233083015512	04-02-91	10.97	711	7.6	13.0	--	.4	95	27
		06-10-91	14.67	972	7.5	18.0	--	.1	120	35
		11-06-91	19.73	799	7.5	13.0	.258	0.3	92	26
S10-MP5	390233083015515	11-06-91	19.73	892	7.2	14.5	.278	1.4	110	31

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Water level (feet bls) or discharge (ft^3/s)	Specific conductance (ms/cm)	pH, field (standard units)	Temperature (deg C)	Eh (v)	Oxygen, dissolved (mg/L)	Calcium, dissolved (mg/L as Ca)	Magnesium, dissolved (mg/L as Mg)
S10-MP6	390233083015516	04-02-91	10.96	796	7.7	12.0	--	3.5	110	46
		06-10-91	14.70	889	7.4	14.5	--	2.8	110	42
S12-MP1	390223083020611	04-03-91	--	761	7.4	14.0	--	1.1	94	28
		06-13-91	15.04	718	7.2	14.0	--	5.3	96	30
S12-MP3	390223083020613	06-13-91	14.80	699	7.3	14.0	--	7.3	94	30
S12-MP4	390223083020614	04-03-91	--	647	7.4	12.0	--	7.3	91	27
S14-MP1	390224083020111	04-02-91	11.45	534	7.7	13.0	--	.2	77	22
		06-13-91	15.81	587	7.6	15.0	--	1.0	77	21
		11-05-91	20.38	577	7.4	12.5	.105	.0	77	22
S14-MP2	390224083020112	04-02-91	11.44	583	7.5	13.0	--	.1	82	24
		06-13-91	15.79	625	7.3	14.0	--	.5	81	24
		11-05-91	20.38	619	7.3	11.5	.132	.5	83	24
S14-MP4	390224083020114	11-05-91	20.38	684	7.3	14.0	.219	4.8	98	29
S14-MP5	390224083020115	06-13-91	15.79	696	7.4	16.0	--	7.1	93	31
S14-MP6	390224083020116	04-02-91	11.56	643	7.7	10.5	--	7.7	82	31
VAN METER WATER-SUPPLY WELL	390238083014100	12-04-90	32.84	775	7.4	12.5	--	1.7	73	20
BIG BEAVER CREEK	390241083011801	04-04-91	45	231	8.5	14.5	--	11.0	16	13
		06-11-91	.67	385	7.9	23.5	--	8.1	30	19
SCIOTO RIVER	390250083030001	04-03-91	5,750	538	8.0	13.5	--	11.3	64	22
		06-11-91	1,780	671	8.7	27.5	--	10.0	75	26
		11-19-91	675.5	907	8.2	15.5	--	10.1	79	28

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Sodium, dissolved (mg/L as Na)	Potassium, dissolved (mg/L as K)	Alkalinity, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L as SO_4)	Chloride, dissolved (mg/L as Cl)	Silica, dissolved (mg/L as SiO_2)	Solids, residue at 180 deg. C dissolved (mg/L)	Nitrogen, nitrite, dissolved (mg/L as N)
R1-BR	390227083013330	06-19-91	180	9.1	260	0.7	350	11	876	<0.01
		11-12-91	170	10	340	.4	390	9.7	882	<.01
R1-MP1	390227083013311	04-08-91	18	3.5	290	81	19	8.6	354	<.01
		06-19-91	20	3.5	270	68	37	9.1	414	<.01
		11-14-91	20	3.9	290	71	39	8.4	429	<.01
R1-MP2	390227083013312	04-08-91	9.9	3.4	280	82	8.8	8.9	475	.01
		06-19-91	8.4	3.1	310	64	12	9.4	400	<.01
		11-14-91	8.9	3.3	300	70	12	8.3	410	0.02
R1-MP5	390227083013315	11-15-91	3.8	2.3	390	37	8.3	8.3	437	<.01
R1-MP6	390227083013316	04-08-91	4.0	3.1	370	41	5.5	8.6	431	<.01
		06-19-91	3.3	2.3	330	21	5.3	8.3	352	<.01
R2-BR	390239083015130	04-01-91	660	35	160	7.5	2,000	9.7	3,410	<.01
		06-18-91	670	33	170	7.1	2,100	9.9	4,080	<.01
		11-13-91	770	41	140	<.1	2,000	11	3,930	<.01
R2-MP1	390239083015111	04-01-91	150	5.5	190	72	260	9.2	712	<.01
		06-17-91	160	5.6	190	68	270	9.8	728	<.01
		11-04-91	150	6.0	190	58	290	9.7	840	<.01
R2-MP2	390239083015112	04-01-91	72	6.0	220	77	77	8.9	441	<.01
		06-17-91	65	5.8	230	71	64	9.5	397	<.01
		11-04-91	66	6.7	200	65	110	9.6	520	<.01
R2-MP5	390239083015115	06-17-91	78	3.5	300	76	100	10	590	.03
		11-04-91	68	2.0	300	74	80	8.4	564	.37
R2-MP6	390239083015116	04-01-91	6.0	1.4	240	72	4.5	8.7	348	<.01
R4-MP1	390229083022911	04-09-91	44	5.0	240	63	140	11	550	<.01
		06-12-91	40	4.5	240	65	120	12	500	.02
		11-07-91	53	5.3	260	72	140	10	588	<.01

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Sodium, dissolved (mg/L as Na)	Potassium, dissolved (mg/L as K)	Alkalinity, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L as SO_4)	Chloride, dissolved (mg/L as Cl)	Silica, dissolved (mg/L as SiO_2)	Solids, residue at 180 deg. C dissolved (mg/L)	Nitrogen, nitrite, dissolved (mg/L as N)
R4-MP2	390229083022912	04-09-91	52	4.8	320	99	77	10	588	<.01
		06-12-91	49	4.4	360	86	59	11	471	.01
		11-07-91	49	4.3	310	95	80	9.3	460	<.01
R4-MP4	390229083022914	11-07-91	28	2.7	370	60	53	11	525	<.01
R4-MP5	390229083022915	06-12-91	23	2.2	380	49	34	11	474	.01
R4-MP6	390229083022916	04-09-91	12	1.1	390	33	18	10	477	<.01
R6-BR	390207083024030	04-04-91	260	13	230	96	620	11	1,240	<.01
		06-19-91	270	12	250	1.0	600	12	1,300	<.01
		11-13-91	260	13	230	<0.1	560	11	1,280	<.01
R6-MP1	390207083024011	04-04-91	16	3.0	310	97	30	10	450	<.01
		06-18-91	15	2.9	280	93	31	11	443	<.01
		11-13-91	18	3.1	280	94	38	9.2	456	<.01
R6-MP2	390207083024012	04-04-91	7.3	1.9	300	39	13	9.8	384	<.01
		06-18-91	7.1	1.5	290	44	16	9.9	387	<.01
		11-13-91	7.4	1.8	290	35	12	8.9	406	.01
R6-MP3	390207083024013	11-14-91	8.7	1.9	300	49	17	9.0	412	<.01
R6-MP4	390207083024014	06-19-91	8.1	1.5	320	46	16	10	402	<.01
R6-MP5	390207083024015	04-04-91	6.1	1.5	300	34	12	9.5	385	<.01
R7-BR	390156083022830	06-19-91	190	20	190	14	550	10	1,310	<.01
		11-13-91	210	17	190	2.6	550	10	1,300	<.01
R7-MP1	390156083022811	04-09-91	11	2.4	320	71	26	12	466	<.01
		06-18-91	9.8	2.4	380	69	23	12	487	<.01
		11-14-91	13	2.5	370	79	67	11	492	<.01
R7-MP2	390156083022812	04-09-91	17	2.7	240	110	25	9.8	389	<.01
		06-18-91	16	2.5	290	81	27	10	436	<.01
		11-14-91	11	2.7	320	75	25	9.3	390	<.01

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Sodium, dissolved (mg/L as Na)	Potassium, dissolved (mg/L as K)	Alkalinity, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L as SO_4)	Chloride, dissolved (mg/L as Cl)	Silica, dissolved (mg/L as SiO_2)	Solids, residue at 180 deg. C dissolved (mg/L)	Nitrogen, nitrite, dissolved (mg/L as N)
R7-MP4	390156083022814	06-18-91	11	1.6	270	49	19	9.1	377	<.01
		11-14-91	9.6	1.8	300	66	18	9.1	412	<.01
R7-MP6	390156083022816	04-09-91	6.3	1.2	230	51	11	7.4	321	<.01
R8-MP1	390156083014311	04-05-91	27	3.2	280	53	30	9.5	402	<.01
		06-11-91	25	2.9	280	56	45	10	415	.03
		11-06-91	20	2.9	270	73	30	9.0	378	<.01
R8-MP3	390156083014313	06-11-91	25	2.0	380	29	32	12	459	.02
		11-06-91	19	2.9	210	94	38	8.7	362	<.01
R8-MP4	390156083014314	04-05-91	7.3	1.3	320	44	11	11	406	<.01
S5-MP1	390241083020511	04-03-91	17	4.3	88	19	560	10	1,120	<.01
		06-12-91	17	3.7	110	28	550	11	1,100	.02
		11-05-91	16	4.0	180	32	510	9.7	1,150	<.01
S5-MP2	390241083020512	04-03-91	41	4.0	220	62	130	10	514	<.01
		06-12-91	43	4.0	240	67	130	11	525	0.020
		11-05-91	50	4.4	220	68	130	9.9	506	<.01
S5-MP4	390241083020514	11-05-91	33	3.0	290	53	46	11	438	<.01
S5-MP5	390241083020515	06-12-91	18	7.8	270	36	21	9.7	391	.06
S5-MP6	390241083020516	04-03-91	27	--	210	75	21	7.6	427	0.56
S10-MP1	390233083015511	04-10-91	85	5.4	210	78	230	9.8	670	.01
		06-10-91	100	5.2	180	76	220	9.8	653	<.01
		11-06-91	100	5.4	190	80	230	9.4	640	<.01
S10-MP2	390233083015512	04-02-91	33	7.2	240	85	65	10	467	.03
		06-10-91	37	7.1	280	72	69	11	575	.06
		11-06-91	30	6.8	240	83	65	9.6	484	.07
S10-MP5	390233083015515	11-06-91	32	12	310	83	70	11	571	<.01

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Sodium, dissolved (mg/L as Na)	Potassium, dissolved (mg/L as K)	Alkalinity, field (mg/L as $CaCO_3$)	Sulfate, dissolved (mg/L as SO_4)	Chloride, dissolved (mg/L as Cl)	Silica, dissolved (mg/L as SIO_2)	Solids, residue at 180 deg. C dissolved (mg/L)	Nitrogen, nitrite, dissolved (mg/L as N)
S10-MP6	390233083015516	04-02-91	8.0	1.0	370	47	22	13	498	.03
		06-10-91	14	4.1	330	45	33	13	484	<.01
S12-MP1	390223083020611	04-03-91	15	3.3	260	57	37	10	415	<.01
		06-13-91	14	2.9	290	49	34	11	416	<.01
S12-MP3	390223083020613	06-13-91	12	2.5	280	44	23	10	408	<.01
S12-MP4	390223083020614	04-03-91	9.1	1.7	280	40	18	9.6	381	<.01
S14-MP1	390224083020111	04-02-91	13	.6	210	78	23	10	341	<.01
		06-13-91	12	3.2	270	85	24	11	329	<.01
		11-05-91	15	3.1	220	76	21	11	344	<.01
S14-MP2	390224083020112	04-02-91	18	3.7	250	74	26	9.9	386	.01
		06-13-91	16	3.7	250	72	26	10	333	<.01
		11-05-91	17	3.7	250	79	25	9.9	367	<.01
S14-MP4	390224083020114	11-05-91	20	2.9	300	64	35	11	432	<.01
S14-MP5	390224083020115	06-13-91	14	2.3	280	62	26	11	419	<.01
S14-MP6	390224083020116	04-02-91	5.3	.5	270	29	12	8.6	362	<.01
VAN METER WATER-SUPPLY-WELL	390238083014100	12-04-90	60	4.2	140	80	135	--	498	<.02
BIG BEAVER CREEK	390241083011801	04-04-91	8.5	2.0	23	47	5.0	9.3	158	<.01
		06-11-91	14	3.8	96	74	11	6.6	211	<.01
SCIOTO RIVER	390250083030001	04-03-91	16	2.6	190	58	24	7.0	339	.02
		06-11-91	34	4.3	220	93	25	4.5	417	.01
		11-19-91	72	6.8	220	140	68	2.8	523	.02

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S.Geological Survey surface-water station or well number	Date	Nitrogen, NO2+NO3, dissolved (mg/L as N)	Nitrogen, ammonia, dissolved (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Phosphorus, ortho, dissolved (mg/L as P)	Iron, total (µg/L as Fe)	Iron, dissolved (µg/L as Fe)	Manganese, total (µg/L as Mn)	Manganese, dissolved (µg/L as Mn)
R1-BR	390227083013330	06-19-91	0.06	1.5	.01	<.01	280	290	160	160
		11-12-91	<.05	1.7	.02	<.01	320	310	180	170
R1-MP1	390227083013311	04-08-91	.32	<.01	.03	<.01	50	6	<10	4
		06-19-91	.40	.01	<.01	<.01	<10	<3	<10	<1
		11-14-91	.44	<.01	<.01	<.01	30	6	<10	2
R1-MP2	390227083013312	04-08-91	1.1	<.01	<.01	<.01	<10	3	50	55
		06-19-91	.37	.01	<.01	<.01	<10	<3	<10	15
		11-14-91	.69	.02	<.01	<.01	80	6	<10	8
R1-MP5	390227083013315	11-15-91	1.2	.02	<.01	<.01	20	5	<10	2
R1-MP6	390227083013316	04-08-91	0.92	<.01	<.01	<.01	10	<3	<10	2
		06-19-91	2.0	.01	<.01	<.01	<10	3	<10	<1
R2-BR	390239083015130	04-01-91	<.11	3.9	<.01	<.01	320	14	260	250
		06-18-91	.06	4.9	.02	<.01	740	70	240	280
		11-13-91	<.05	6.0	<.01	<.01	2,700	790	1,200	770
R2-MP1	390239083015111	04-01-91	<.05	.25	<.01	<.01	2,000	1,800	260	250
		06-17-91	<.05	.24	<.01	.01	1,900	1,800	250	270
		11-04-91	.13	.28	<.01	.02	2,400	2,500	330	340
R2-MP2	390239083015112	04-01-91	<.05	.06	<.01	<.01	760	740	780	790
		06-17-91	<.05	.08	<.01	.01	700	720	370	390
		11-04-91	1.4	.09	<.01	.01	1,100	1,200	510	550
R2-MP5	390239083015115	06-17-91	8.4	.03	.02	.01	240	190	940	1,000
		11-04-91	4.1	.05	<.01	<.01	--	100	830	810
R2-MP6	390239083015116	04-01-91	.23	.02	<.01	<.01	280	200	440	430
R4-MP1	390229083022911	04-09-91	<.05	.10	<.01	<.01	2,600	2,600	440	460
		06-12-91	<.05	.08	<.01	<.01	2,500	2,800	440	480
		11-07-91	.22	.07	<.01	.03	2,800	2,600	440	450

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S.Geological Survey surface-water station or well number	Date	Nitrogen, NO2+NO3, dissolved (mg/L as N)	Nitrogen, ammonia, dissolved (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Phosphorus, ortho, dissolved (mg/L as P)	Iron, total (µg/L as Fe)	Iron, dissolved (µg/L as Fe)	Manganese, total (µg/L as Mn)	Manganese, dissolved (µg/L as Mn)
R4-MP2	390229083022912	04-09-91	.19	.04	<.01	<.01	930	980	470	480
		06-12-91	<.05	.01	<.01	<.01	840	930	380	410
		11-07-91	<.05	.05	<.01	<.01	680	--	380	380
R4-MP4	390229083022914	11-07-91	3.1	<.01	<.01	.02	<10	9	<10	<1
R4-MP5	390229083022915	06-12-91	3.4	<.01	<.01	<.01	20	<3	<10	<1
R4-MP6	390229083022916	04-09-91	1.2	.03	<.01	<.01	<10	8	10	10
R6-BR	390207083024030	04-04-91	<.05	.40	<.01	<.01	410	510	--	85
		06-19-91	<.05	2.1	<.01	<.01	410	410	80	83
		11-13-91	<.05	2.2	<.01	<.01	410	430	140	120
R6-MP1	390207083024011	04-04-91	.06	.01	<.01	<.01	1,900	1,900	510	530
		06-18-91	.13	.02	<.01	<.01	1,800	1,900	480	530
		11-13-91	<.05	.05	<.01	<.01	2,000	1,700	460	480
R6-MP2	390207083024012	04-04-91	1.2	<.01	<.01	<.01	<10	<3	<10	6
		06-18-91	5.2	.01	<.01	<.01	<10	<3	<10	<1
		11-13-91	<.05	<.01	.01	<.01	30	4	<10	3
R6-MP3	390207083024013	11-14-91	4.4	.02	<.01	<.01	--	14	--	3
R6-MP4	390207083024014	06-19-91	5.4	<.01	<.01	<.01	<10	7	<10	<1
R6-MP5	390207083024015	04-04-91	5.1	<.01	<.01	<.01	<10	5	<10	3
R7-BR	390156083022830	06-19-91	.06	1.7	<.01	<.01	150	7	330	360
		11-13-91	<.05	2.0	<.01	<.01	330	270	710	660
R7-MP1	390156083022811	04-09-91	<.05	.05	<.01	<.01	5,800	5,300	460	470
		06-18-91	<.05	.06	<.01	<.01	5,800	6,100	400	440
		11-14-91	<.05	.10	<.01	<.01	5,300	5,500	380	390
R7-MP2	390156083022812	04-09-91	<.05	.03	<.01	<.01	1,800	1,700	490	520
		06-18-91	<.05	.04	<.01	<.01	1,300	1,300	500	540
		11-14-91	.10	.06	<.01	<.01	1,400	1,300	600	560

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S.Geological Survey surface-water station or well number	Date	Nitrogen, NO2+NO3, dissolved (mg/L as N)	Nitrogen, ammonia, dissolved (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Phosphorus, ortho, dissolved (mg/L as P)	Iron, total (µg/L as Fe)	Iron, dissolved (µg/L as Fe)	Manganese, total (µg/L as Mn)	Manganese, dissolved (µg/L as Mn)
R7-MP4	390156083022814	06-18-91	4.8	.02	<.01	<.01	<10	<3	<10	<1
		11-14-91	2.9	.00	<.01	<.01	20	<3	<10	<1
R7-MP6	390156083022816	04-09-91	2.3	<.01	<.01	<.01	570	4	20	11
R8-MP1	390156083014311	04-05-91	1.4	.02	<.01	<.01	70	26	510	520
		06-11-91	1.1	.01	<.01	<.01	20	5	370	370
		11-06-91	1.3	.03	<.01	<.01	<10	--	150	150
R8-MP3	390156083014313	06-11-91	1.2	<.01	<.01	<.01	<10	<3	<10	2
		11-06-91	.27	.03	<.01	<.01	10	6	<10	<1
R8-MP4	390156083014314	04-05-91	.70	.01	<.01	<.01	70	22	90	88
S5-MP1	390241083020511	04-03-91	<.05	.30	<.01	<.01	4,400	4,100	1,500	1,500
		06-12-91	<.05	.29	<.01	<.01	4,400	4,900	1,200	1,200
		11-05-91	.77	.35	<.01	<.01	5,500	5,300	1,100	1,100
S5-MP2	390241083020512	04-03-91	<.05	.07	<.01	<.01	2,200	2,100	--	510
		06-12-91	<.05	.05	<.01	<.01	2,400	2,400	380	400
		11-05-91	.40	.10	<.01	<.01	2,000	2,000	350	350
S5-MP4	390241083020514	11-05-91	4.5	.03	<.01	<.01	<10	--	<10	1
S5-MP5	390241083020515	06-12-91	11.0	<.01	<.01	<.01	<10	<3	<10	6
S5-MP6	390241083020516	04-03-91	9.7	.11	<.01	<.01	<10	--	<10	13
S10-MP1	390233083015511	04-10-91	<.05	.40	<.01	<.01	1,300	1,100	850	900
		06-10-91	.05	.30	<.01	<.01	1,600	1,400	580	580
		11-06-91	.21	.39	<.01	<.01	1,900	1,800	450	460
S10-MP2	390233083015512	04-02-91	2.7	.02	<.01	<.01	140	160	180	180
		06-10-91	10.0	.02	.01	<.01	<10	<3	120	130
		11-06-91	5.5	.02	<.01	<.01	10	<3	120	110
S10-MP5	390233083015515	11-06-91	3.7	.21	<.01	<.01	<10	<3	40	<1

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S.Geological Survey surface-water station or well number	Date	Nitrogen, NO2+NO3, dissolved (mg/L as N)	Nitrogen, ammonia, dissolved (mg/L as N)	Phosphorus, dissolved (mg/L as P)	Phosphorus, ortho, dissolved (mg/L as P)	Iron, total (µg/L as Fe)	Iron, dissolved (µg/L as Fe)	Manganese, total (µg/L as Mn)	Manganese, dissolved (µg/L as Mn)
S10-MP6	390233083015516	04-02-91	2.8	.03	< .01	< .01	<10	8	290	290
		06-10-91	8.8	.03	< .01	< .01	<10	<3	90	79
S12-MP1	390223083020611	04-03-91	4.3	< .01	< .01	< .01	<10	7	70	54
		06-13-91	6.2	.02	< .01	< .01	<10	4	10	7
S12-MP3	390223083020613	06-13-91	6.8	.02	< .01	< .01	40	<3	10	<1
S12-MP4	390223083020614	04-03-91	1.4	< .01	< .01	< .01	<10	5	80	89
S14-MP1	390224083020111	04-02-91	< .05	.07	< .01	< .01	1,600	1,600	430	410
		06-13-91	< .05	.08	< .01	< .01	1,900	2,000	300	320
		11-05-91	< .05	.11	< .01	< .01	2,100	2,100	330	330
S14-MP2	390224083020112	04-02-91	< .05	.04	< .01	< .01	1,000	960	370	380
		06-13-91	< .05	.04	< .01	< .01	1,000	1,000	270	280
		11-05-91	.05	.04	< .01	< .01	1,200	1,100	290	270
S14-MP4	390224083020114	11-05-91	2.9	.03	< .01	< .01	<10	7	<10	1
S14-MP5	390224083020115	06-13-91	5.7	.02	< .01	< .01	20	<3	10	<1
S14-MP6	390224083020116	04-02-91	2.2	< .01	< .01	< .01	160	4	<10	7
VAN METER-WATER-SUPPLY WELL	390238083014100	12-04-90	<.01	.06	<.05	--	1,160	850	250	255
BIG BEAVER CREEK	390241083011801	04-04-91	.72	.01	< .01	< .01	920	160	110	100
		06-11-91	.10	.04	< .01	< .01	230	18	120	91
SCIOTO RIVER	390250083030001	04-03-91	2.8	.09	.07	.06	1,600	76	60	25
		06-11-91	2.3	.05	.05	.05	1,200	9	70	5
		11-19-91	3.6	.04	.32	.28	360	60	100	89

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Carbon, organic, total (mg/L as C)	Carbon, organic, dissolved (mg/L as C)	Alachlor, dissolved (µg/L)	Ametryn, dissolved (µg/L)	Atrazine, dissolved (µg/L)	Cyanazine, dissolved (µg/L)	Desethyl-atrazine, dissolved (µg/L)
R1-BR	390227083013330	06-19-91	.6	--	< 0.05	< 0.05	< 0.05	< 0.20	< 0.05
		11-12-91	1.8	1.3	< .05	< .05	< .05	< .20	< .05
R1-MP1	390227083013311	04-08-91	.7	--	< .05	< .05	< .05	< .20	< .05
		06-19-91	.9	--	< .05	< .05	< .05	< .20	< .05
		11-14-91	1.0		< .05	< .05	< .05	< .20	< .05
R1-MP2	390227083013312	04-08-91	1.0	--	< .05	< .05	< .05	< .20	< .05
		06-19-91	.8	--	< .05	< .05	< .05	< .20	< .05
		11-14-91	1.2		< .05	< .05	< .05	< .20	< .05
R1-MP5	390227083013315	11-15-91	1.0		< .05	< .05	< .05	< .20	< .05
R1-MP6	390227083013316	04-08-91	1.0	--	< .05	< .05	< .05	< .20	< .05
		06-19-91	.6	--	< .05	< .05	< .05	< .20	< .05
R2-BR	390239083015130	04-01-91	.9	--	< .05	< .05	< .05	< .20	< .05
		06-18-91	.6	--	< .05	< .05	< .05	< .20	< .05
		11-13-91	2.7		< .05	< .05	< .05	< .20	< .05
R2-MP1	390239083015111	04-01-91	0.9	--	< .05	< .05	< .05	< .20	< .05
		06-17-91	0.7	--	< .05	< .05	< .05	< .20	< .05
		11-04-91	1.4	1.2	< .05	< .05	< .05	< .20	< .05
R2-MP2	390239083015112	04-01-91	0.7	--	< .05	< .05	< .05	< .20	< .05
		06-17-91	0.9	--	< .05	< .05	< .05	< .20	< .05
		11-04-91	1.5	1.0	< .05	< .05	< .05	< .20	< .05
R2-MP5	390239083015115	06-17-91	0.7	--	< .05	< .05	< .05	< .20	< .05
		11-04-91	1.8	1.0	< .05	< .05	< .05	< .20	< .05
R2-MP6	390239083015116	04-01-91	0.8	--	< .05	< .05	< .05	< .20	< .05
R4-MP1	390229083022911	04-09-91	1.4	--	< .05	< .05	< .05	< .05	< .05
		06-12-91	0.4	--	< .05	< .05	< .05	< .20	< .05
		11-07-91	0.8	.9	< .05	< .05	< .05	< .05	< .05

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Carbon, organic, total (mg/L as C)	Carbon, organic, dissolved (mg/L as C)	Alachlor, dissolved (µg/L)	Ametryn, dissolved (µg/L)	Atrazine, dissolved (µg/L)	Cyanazine, dissolved (µg/L)	Desethyl-atrazine, dissolved (µg/L)
R4-MP2	390229083022912	04-09-91	0.8	--	<.05	<.05	<.05	<.05	<.05
		06-12-91	0.3	--	<.05	<.05	<.05	<.20	<.05
		11-07-91	0.7	.7	<.05	<.05	<.05	<.20	<.05
R4-MP4	390229083022914	11-07-91	0.7	.6	<.05	<.05	<.05	<.20	<.05
R4-MP5	390229083022915	06-12-91	0.4	--	<.05	<.05	<.05	<.20	<.05
R4-MP6	390229083022916	04-09-91	0.9	--	<.05	<.05	<.05	<.20	<.05
R6-BR	390207083024030	04-04-91	0.3	--	<.05	<.05	<.05	<.20	<.05
		06-19-91	0.2	--	<.05	<.05	<.05	<.20	<.05
		11-13-91	0.8		<.05	<.05	<.05	<.20	<.05
R6-MP1	390207083024011	04-04-91	0.4	--	<.05	<.05	<.05	<.20	<.05
		06-18-91	0.3	--	<.05	<.05	<.05	<.20	<.05
		11-13-91	.6		<.05	<.05	<.05	<.20	<.05
R6-MP2	390207083024012	04-04-91	0.7	--	<.05	<.05	<.05	<.20	<.05
		06-18-91	0.5	--	<.05	<.05	<.05	<.20	<.05
		11-13-91	0.7		<.05	<.05	<.05	<.20	<.05
R6-MP3	390207083024013	11-14-91	--		<.05	<.05	<.05	<.20	<.05
R6-MP4	390207083024014	06-19-91	0.3	--	<.05	<.05	<.05	<.20	<.05
R6-MP5	390207083024015	04-04-91	0.4	--	<.05	<.05	<.05	<.20	<.05
R7-BR	390156083022830	06-19-91	1.0	--	<.05	<.05	<.05	<.20	<.05
		11-13-91	1.6		<.05	<.05	<.05	<.20	<.05
R7-MP1	390156083022811	04-09-91	1.0	--	<.05	<.05	<.05	<.20	<.05
		06-18-91	0.6	--	<.05	<.05	<.05	<.20	<.05
		11-14-91	0.9	1.0	<.05	<.05	<.05	<.20	<.05
R7-MP2	390156083022812	04-09-91	1.0	--	<.05	<.05	<.05	<.20	<.05
		06-18-91	0.8	--	<.05	<.05	<.05	<.20	<.05
		11-14-91	1.4	1.0	<.05	<.05	<.05	<.20	<.05

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Carbon, organic, total (mg/L as C)	Carbon, organic, dissolved (mg/L as C)	Alachlor, dissolved (μg/L)	Ametryn, dissolved (μg/L)	Atrazine, dissolved (μg/L)	Cyanazine, dissolved (μg/L)	Desethyl-atrazine, dissolved (μg/L)
R7-MP4	390156083022814	06-18-91	0.5	--	<.05	<.05	<.05	<.20	<.05
		11-14-91	0.6	.6	<.05	<.05	<.05	<.20	<.05
R7-MP6	390156083022816	04-09-91	0.8	--	<.05	<.05	<.05	<.20	<.05
R8-MP1	390156083014311	04-05-91	0.6	--	<.05	<.05	<.05	<.20	<.05
		06-11-91	0.7	--	<.05	<.05	<.05	<.20	<.05
		11-06-91	1.3	.8	<.05	<.05	<.05	<.20	<.05
R8-MP3	390156083014313	06-11-91	0.6	--	<.05	<.05	<.05	<.20	<.05
		11-06-91	1.1	.7	<.05	<.05	<.05	<.20	<.05
R8-MP4	390156083014314	04-05-91	0.8	--	<.05	<.05	<.05	<.20	<.05
S5-MP1	390241083020511	04-03-91	0.5	--	<.05	<.05	<.05	<.20	<.05
		06-12-91	0.4	--	<.05	<.05	<.05	<.20	<.05
		11-05-91	0.7		<.05	<.05	<.05	<.20	<.05
S5-MP2	390241083020512	04-03-91	0.6	--	<.05	<.05	<.05	<.20	<.05
		06-12-91	0.6	--	<.05	<.05	<.05	<.20	<.05
		11-05-91	1.1		<.05	<.05	<.05	<.20	<.05
S5-MP4	390241083020514	11-05-91	1.0		<.05	<.05	<.05	<.20	<.05
S5-MP5	390241083020515	06-12-91	0.5	--	<.05	<.05	<.05	<.20	<.05
S5-MP6	390241083020516	04-03-91	1.3	--	<.05	<.05	0.05	<.20	<.05
S10-MP1	390233083015511	04-10-91	1.3	--	<.05	<.05	<.05	<.20	<.05
		06-10-91	0.9	--	<.05	<.05	<.05	<.20	<.05
		11-06-91	1.2	1.0	<.05	<.05	<.05	<.20	<.05
S10-MP2	390233083015512	04-02-91	0.5	--	<.05	<.05	<.05	<.20	<.05
		06-10-91	0.8	--	<.05	<.05	<.05	<.05	<.05
		11-06-91	1.2	.9	<.05	<.05	<.05	<.05	<.05
S10-MP5	390233083015515	11-06-91	1.0	.9	<.05	<.05	<.05	<.20	<.05

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Carbon, organic, total (mg/L as C)	Carbon, organic, dissolved (mg/L as C)	Alachlor, dissolved (µg/L)	Ametryn, dissolved (µg/L)	Atrazine, dissolved (µg/L)	Cyanazine, dissolved (µg/L)	Desethyl-atrazine, dissolved (µg/L)
S10-MP6	390233083015516	04-02-91	0.5	--	<.05	<.05	<.05	<.05	<.05
		06-10-91	0.5	--	<.05	<.05	<.05	<.20	<.05
S12-MP1	390223083020611	04-03-91	1.4	--	<.05	<.05	<.05	<.20	<.05
		06-13-91	0.4	--	<.05	<.05	<.05	<.20	<.05
S12-MP3	390223083020613	06-13-91	0.4	--	<.05	<.05	<.05	<.20	<.05
S12-MP4	390223083020614	04-03-91	0.5	--	<.05	<.05	<.05	<.20	<.05
S14-MP1	390224083020111	04-02-91	0.8	--	<.05	<.05	<.05	<.20	<.05
		06-13-91	0.9	--	<.05	<.05	<.05	<.20	<.05
		11-05-91	1.2	1.1	<.05	<.05	<.05	<.20	<.05
S14-MP2	390224083020112	04-02-91	0.8	--	<.05	<.05	<.05	<.05	<.05
		06-13-91	0.9	--	<.05	<.05	<.05	<.20	<.05
		11-05-91	1.2	1.0	<.05	<.05	<.05	<.20	<.05
S14-MP4	390224083020114	11-05-91	1.0	.7	<.05	<.05	<.05	<.20	<.05
S14-MP5	390224083020115	06-13-91	0.6	--	<.05	<.05	<.05	<.05	<.05
S14-MP6	390224083020116	04-02-91	0.5	--	<.05	<.05	<.05	<.20	<.05
VAN METER WATER-SUPPLY WELL	390238083014100	12-04-90	<5	--	--	--	--	--	--
BIG BEAVER CREEK	390241083011801	04-04-91	2.0	--	<.05	<.05	<.05	<.20	.08
		06-11-91	4.0	--	<.05	<.05	<.05	<.20	<.05
SCIOTO RIVER	390250083030001	04-03-91	5.8	--	<.05	<.05	0.17	<.20	<.05
		06-11-91	9.0	--	.3	<.05	2.7	<.20	.13
		11-19-91	11	9.1	<.05	<.05	1.9	<.20	.47

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Des-isopropyl atrazine, dissolved (µg/L)	Metolachlor, dissolved (µg/L)	Metribuzin, dissolved (µg/L)	Prometon, dissolved (µg/L)	Prometryn, dissolved (µg/L)	Propazine, dissolved (µg/L)	Simazine, dissolved (µg/l)
R1-BR	390227083013330	06-19-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-12-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R1-MP1	390227083013311	04-08-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-19-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-14-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R1-MP2	390227083013312	04-08-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-19-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-14-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R1-MP5	390227083013315	11-15-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R1-MP6	390227083013316	04-08-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-19-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R2-BR	390239083015130	04-01-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-18-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R2-MP1	390239083015111	04-01-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-17-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-04-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R2-MP2	390239083015112	04-01-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-17-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-04-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R2-MP5	390239083015115	06-17-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-04-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R2-MP6	390239083015116	04-01-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R4-MP1	390229083022911	04-09-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-12-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-07-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Des-isopropyl atrazine, dissolved (µg/L)	Metolachlor, dissolved (µg/L)	Metribuzin, dissolved (µg/L)	Prometon, dissolved (µg/L)	Prometryn, dissolved (µg/L)	Propazine, dissolved (µg/L)	Simazine, dissolved (µg/l)
R4-MP2	390229083022912	04-09-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-12-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-07-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R4-MP4	390229083022914	11-07-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R4-MP5	390229083022915	06-12-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R4-MP6	390229083022916	04-09-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R6-BR	390207083024030	04-04-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-19-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R6-MP1	390207083024011	04-04-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-18-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R6-MP2	390207083024012	04-04-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-18-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R6-MP3	390207083024013	11-14-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R6-MP4	390207083024014	06-19-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R6-MP5	390207083024015	04-04-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R7-BR	390156083022830	06-19-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-13-91	<.05	<.05	<.05	<.05	<.05	<.05	< .05
R7-MP1	390156083022811	04-09-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-18-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-14-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R7-MP2	390156083022812	04-09-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-18-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-14-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Des-isopropyl atrazine, dissolved (μg/L)	Metolachlor, dissolved (μg/L)	Metribuzin, dissolved (μg/L)	Prometon, dissolved (μg/L)	Prometryn, dissolved (μg/L)	Propazine, dissolved (μg/L)	Simazine, dissolved (μg/l)
R7-MP4	390156083022814	06-18-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-14-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R7-MP6	390156083022816	04-09-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R8-MP1	390156083014311	04-05-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-11-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-06-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R8-MP3	390156083014313	06-11-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-06-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
R8-MP4	390156083014314	04-05-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S5-MP1	390241083020511	04-03-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-12-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-05-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S5-MP2	390241083020512	04-03-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-12-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-05-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S5-MP4	390241083020514	11-05-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S5-MP5	390241083020515	06-12-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S5-MP6	390241083020516	04-03-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S10-MP1	390233083015511	04-10-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-10-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-06-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S10-MP2	390233083015512	04-02-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-10-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-06-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S10-MP5	390233083015515	11-06-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05

Table 13. Summary of ground-water and surface-water quality at the Ohio Management Systems Evaluation Area—Continued

Local surface-water station or well number	U.S. Geological Survey surface-water station or well number	Date	Des-isopropyl atrazine, dissolved (µg/L)	Metolachlor, dissolved (µg/L)	Metribuzin, dissolved (µg/L)	Prometon, dissolved (µg/L)	Prometryn, dissolved (µg/L)	Propazine, dissolved (µg/L)	Simazine, dissolved (µg/l)
S10-MP6	390233083015516	04-02-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-10-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S12-MP1	390223083020611	04-03-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S12-MP3	390223083020613	06-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S12-MP4	390223083020614	04-03-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S14-MP1	390224083020111	04-02-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-05-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S14-MP2	390224083020112	04-02-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		11-05-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S14-MP4	390224083020114	11-05-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S14-MP5	390224083020115	06-13-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
S14-MP6	390224083020116	04-02-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
VAN METER WATER-SUPPLY WELL	390238083014100	12-04-90	--	--	--	--	--	--	--
BIG BEAVER CREEK	390241083011801	04-04-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
		06-11-91	< .05	< .05	< .05	< .05	< .05	< .05	< .05
SCIOTO RIVER	390250083030001	04-03-91	< .05	.12	< .05	< .05	< .05	< .05	.06
		06-11-91	< .05	1.8	.1	< .05	< .05	< .05	< .05
		11-19-91	.28	1.9	< .05	< .05	< .05	< .05	.09