

GEOHYDROLOGY OF, AND NITROGEN AND CHLORIDE IN, THE GLACIAL AQUIFER, MILFORD-MATAMORAS AREA, PIKE COUNTY, PENNSYLVANIA

by Lisa A. Senior

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

<u>Multiply</u>	<u>By</u>	<u>To obtain</u>
Length		
inch (in.)	25.40	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
foot per mile (ft/mi)	0.1177	meter per kilometer
Volume		
gallon (gal)	3.785	liter
Specific Capacity		
gallon per minute per foot [(gal/min)/ft]	0.207	liter per second per meter
Area		
square mile (mi ²)	2.590	square kilometer
Mass		
pound (lb)	2.2	kilogram
pound per square mile (lb/mi ²)	.1755	kilogram per square kilometer
ton per mile (ton/mi)	.3503	megagram per kilometer
Transmissivity		
square foot per day (ft ² /d)	0.09290	square meter per day
Temperature		
degree Fahrenheit	°C=5/9 (°F-32)	degree Celsius

Sea level: In this report “sea level” refers to the National Geodetic Vertical Datum of 1929—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:

milligram per liters (mg/L)
micrograms per liter (µg/L)
microsiemen per centimeter at 25 degrees Celsius (µS/cm)

Geohydrology Of, and Nitrogen and Chloride In, the Glacial Aquifer, Milford-Matamoras Area, Pike County, Pennsylvania

by Lisa A. Senior

ABSTRACT

The glacial aquifer that underlies the Routes 209 and 6 corridor between Milford and Matamoras, Pa., is one of the most productive in Pike County. The aquifer is comprised of unconsolidated glacial outwash and kame-terrace deposits that lie within a glacially carved valley now occupied by the Delaware River. Most businesses and residences along this narrow, 7-mile-long corridor rely on individual wells for water supply and septic systems for waste-water disposal. A study of nutrients and chloride in ground water in the glacial aquifer was conducted to determine the effect of these constituents contributed from septic systems and road runoff on ground-water quality.

Sources of nutrients and chloride in the recharge zone upgradient of the aquifer include road and parking-lot runoff, septic systems, and precipitation. Nitrate and chloride from these sources can infiltrate and move in the direction of ground-water flow in the saturated zone of the aquifer. A water-table map based on 29 water levels measured in August 1991 indicates that the direction of ground-water flow is from the edges of the valley toward the Delaware River but is nearly parallel to the Delaware River in the central area of the valley.

The average concentrations of nitrogen and chloride in recharge and total annual loads of nitrogen and chloride to ground water were estimated for six areas with different population densities. These estimates assumed a recharge rate to the glacial aquifer of 20 inches per year and a 15 percent loss of chloride and nitrogen in the atmospheric precipitation to surface runoff. The estimated average concentration of nitrogen in recharge ranged from 2.5 to 10 mg/L (milligrams per liter), which corresponds to a total annual load of nitrogen that ranged from 7.3 to 28.6×10^3 lb/mi² (pounds per square mile). The calculation assumed that all nitrogen as ammonium released from septic tanks and present in precipitation was oxidized to nitrate as the dominant nitrogen species in ground water. Contributions of nitrogen from septic tanks were greater than contributions from runoff. Observed concentrations of nitrate, which was the most abundant nitrogen species in ground water in the glacial aquifer, ranged from less than 0.05 to 5.1 mg/L as nitrogen, with a median of 1.1 mg/L as nitrogen. Concentrations of nitrogen measured in ground water were lower than estimated concentrations for recharge suggesting that dissolved nitrogen species may not be conservative in ground water. Nitrate is unstable in anoxic ground water and can be removed by denitrification. Ammonium can be sorbed onto the aquifer materials. Evidence for reducing conditions included a positive correlation between low concentrations of dissolved oxygen and low concentrations of nitrate.

The estimated concentration of chloride in recharge ranged from 6.7 to 21 mg/L, and total annual load of chloride to ground water ranged from 19.4 to 50.6×10^3 lb/mi². Chloride is considered to be a chemically conservative ion in ground water. Contributions of chloride to ground water from road salting were greater than contributions from septic tanks. Observed concentrations of chloride in 18 ground-water samples from the glacial aquifer ranged from 2.1 to 32 mg/L, with a median of 17.5 mg/L. Local contamination is indicated by the elevated concentrations of chloride (up to 680 mg/L) detected in four wells located downgradient of an abandoned industry that may have released salts in processing waste. Chloride concentrations in ground water appeared to be greater near major roads and in areas of relatively greater septic-system density than in areas upgradient of roads, farther downgradient from roads, or with less densely spaced septic systems.

INTRODUCTION

The glacial aquifer that underlies the Routes 209 and 6 corridor between Milford and Matamoras, Pa., is one of the most productive in Pike County (Davis, 1989). Most businesses and residences along this narrow, 7-mi-long corridor rely on individual wells for water supply and septic systems for waste-water disposal. Much of the roadway and parking-lot runoff is routed to swales, basins, and dry wells or areas at the edges of lots where the runoff infiltrates to the ground-water system. Ground-water quality in the glacial aquifer is threatened because of potential contamination by nutrients, bacteria, and viruses from septic systems and salt from roadway runoff. Elevated concentrations of dissolved nitrate in ground water in the glacial aquifer along the Routes 209 and 6 corridor between Milford and Matamoras were measured by the Pennsylvania Department of Environmental Resources (PaDER) (Pennsylvania Department of Environmental Resources, unpublished data).

These glacial deposits are comprised of highly permeable sands and gravels that allow salts and septic-system effluent to readily infiltrate to the ground-water system. Contamination of ground water in glacial aquifers by septic systems and road salts has been documented in similar hydrogeologic settings in Wisconsin, Massachusetts, and Ontario, Canada (Tinker, 1991; Coogan, 1971; Robertson and others, 1991). Because of bacteria and viruses present in domestic sewage, contamination of ground water by septic systems can result in outbreaks of disease (Yates, 1985). Contamination of ground water by septic tanks is more likely to occur where depth to water is shallow, septic tanks are closely spaced, and aquifer materials and soils have high permeability than where these factors are absent; sand and gravel aquifers are among the aquifers most susceptible to contamination (Yates, 1985; Canter and Knox, 1985).

Chloride and nitrate from septic-system effluent and road runoff are highly mobile, dissolved constituents in ground water. Additionally, contaminants in ground water associated with residential, agricultural, commercial, and industrial land uses can include pesticides, fertilizer components, metals, and organic compounds, such as solvents, lubricants, and fuels. Many of these other contaminants are less mobile in ground water than chloride or nitrate. For example, although lead and zinc commonly are present in road runoff (German, 1989; Fisher and Katz, 1988; Ku and Simmons, 1986), they are less commonly a problem in ground water than chloride because they are less mobile than chloride. In a study of the effect of urban storm runoff on ground-water quality beneath recharge basins in glacial deposits on Long Island, N.Y., chloride and nitrate infiltrated into the ground-water system, but bacteria, metals, and organic compounds did not (Ku and Simmons, 1986). Elevated concentrations of chloride and nitrate in ground water could, however, indicate other potentially more serious contamination by metals, bacteria, viruses, and organic compounds.

Elevated concentrations of nitrate in drinking water can have adverse effects on human health. The U.S. Environmental Protection Agency (USEPA) maximum contaminant level (MCL) for nitrate in regulated drinking water supplies is 10 mg/L as nitrogen (U.S. Environmental Protection Agency, 1989). Chloride has a USEPA secondary maximum contaminant level (SMCL) of 250 mg/L.

The U.S. Geological Survey (USGS), in cooperation with the Pike County Planning Commission, conducted a 1-year study of water quality in the glacial aquifer in the Routes 209 and 6 corridor. The purpose of the study was to assess and quantify the effects of nutrient and chloride contributions from septic systems and road runoff to ground water. Data were collected from August through December 1991.

Purpose and Scope

This report describes geohydrology of the glacial aquifer, including the physical boundaries and controls on the ground-water system, and nitrogen and chloride in the aquifer, including sources, spatial distribution, relation to other chemical constituents, and loadings. Runoff-water-quality and ground-water-quality data collected for the study and calculated nutrient and chloride loads to ground water are presented, and some effects of nitrate and chloride loading on ground-water quality are discussed. Water quality in the glacial aquifer is characterized from ground-water samples collected in October 1991.

The contributions of nitrogen as nitrate and chloride from septic systems and road runoff to ground water in the glacial aquifer in the Routes 209 and 6 corridor, Milford to Matamoras, Pa., are estimated by use of a conservative, mass-balance approach. The loading calculations were limited to nitrate and chloride because these constituents commonly are indicators of other contaminants and because nitrate and chloride are conservatively transported in ground water in this area.

Description of the Study Area

Location and Physiography

The study area is defined by the extent of glacial deposits in the Delaware River valley between Milford and Matamoras, Pike County, in northeastern Pennsylvania. The 3.5-mi² area underlain by these deposits lies within Milford and Westfall Townships, the boroughs of Milford and Matamoras, and the National Park Service Delaware River Scenic and Recreation Area. The study area is less than 1 mi wide and trends northeastward, paralleling the Delaware River for about 7 mi (fig. 1). The study area is bounded by the boroughs of Matamoras and Milford at the northeastern and southwestern ends, respectively, and the Delaware River and bedrock cliffs on the southeastern and northwestern sides, respectively. The Delaware River forms the boundary between Pennsylvania, New Jersey, and New York in the vicinity.

The study area is in the glaciated Appalachian Mountain section of the Valley and Ridge Physiographic Province adjacent to the glaciated Low Plateau section of the Appalachian Plateau Physiographic Province (Berg and others, 1989). The bedrock cliffs that form the northwestern boundary of the study area, rising as much as 600 ft above the valley floor, are in the Plateau Physiographic Province. The glacial topography in the valley is generally flat, with terraces in the southern part of the study area. Land-surface altitudes for the glacial deposits range from 370 ft above sea level near the Delaware River to 515 ft at the edge of the valley near the bedrock cliffs.

The study area, which is included entirely within Pike County, has a humid continental climate (Davis, 1989). The normal annual precipitation at the National Oceanographic and Atmospheric Administration station at Matamoras for 1951-1980 was 44.2 in.; on the average, precipitation is distributed about evenly throughout the year.

Geologic Setting

During the most recent glaciation in the late Wisconsinan stage of the Pleistocene Epoch, about 22,000 years ago, glaciers moved into northeastern Pennsylvania from the northeast (Sevon and others, 1989). The glaciers occupied the Delaware River valley, carving downward into the shale, siltstone, and sandstone bedrock. Upon retreat of the glaciers, sand and gravel deposits were left behind, filling the bedrock valley. The depth to bedrock is as much as 250 ft below the channel of the present Delaware River, which is about 25 to 30 ft deep between Matamoras and Bushkill, Pa., located 25 mi to the south (Crowl, 1971, p. 8).

The bedrock valley that underlies the glacial deposits between Matamoras and Milford is Silurian and Devonian in age (200-300 million years before present). In the study area, two Devonian units have been mapped, the Marcellus Shale and the Mahantango Formation (Crowl, 1971; Davis, 1989; Sevon and others, 1989). The Devonian units are nearly flat-lying, with the younger Mahantango Formation lying over the Marcellus Shale. The Mahantango Formation forms the cliffs along the northwestern boundary of the valley. Southeast of the Mahantango Shale, the Marcellus Shale underlies the Delaware River and is buried beneath the glacial deposits (Sevon and others, 1989). The Marcellus Shale is a dark gray, silty shale, and the Mahantango Formation is a gray siltstone and silty shale. These units were extensively eroded and the resultant basic landforms in this area of the Delaware River valley were developed before the Pleistocene Epoch (Crowl, 1971, p. 7).

The glacial deposits in the valley between Matamoras and Milford have been mapped (plate 1) as outwash, kame, and kame-terrace deposits (Crowl, 1971; Sevon and others, 1989; Davis 1989). The outwash deposits are covered by thin (less than 20 ft) alluvial and alluvial fan deposits (younger than

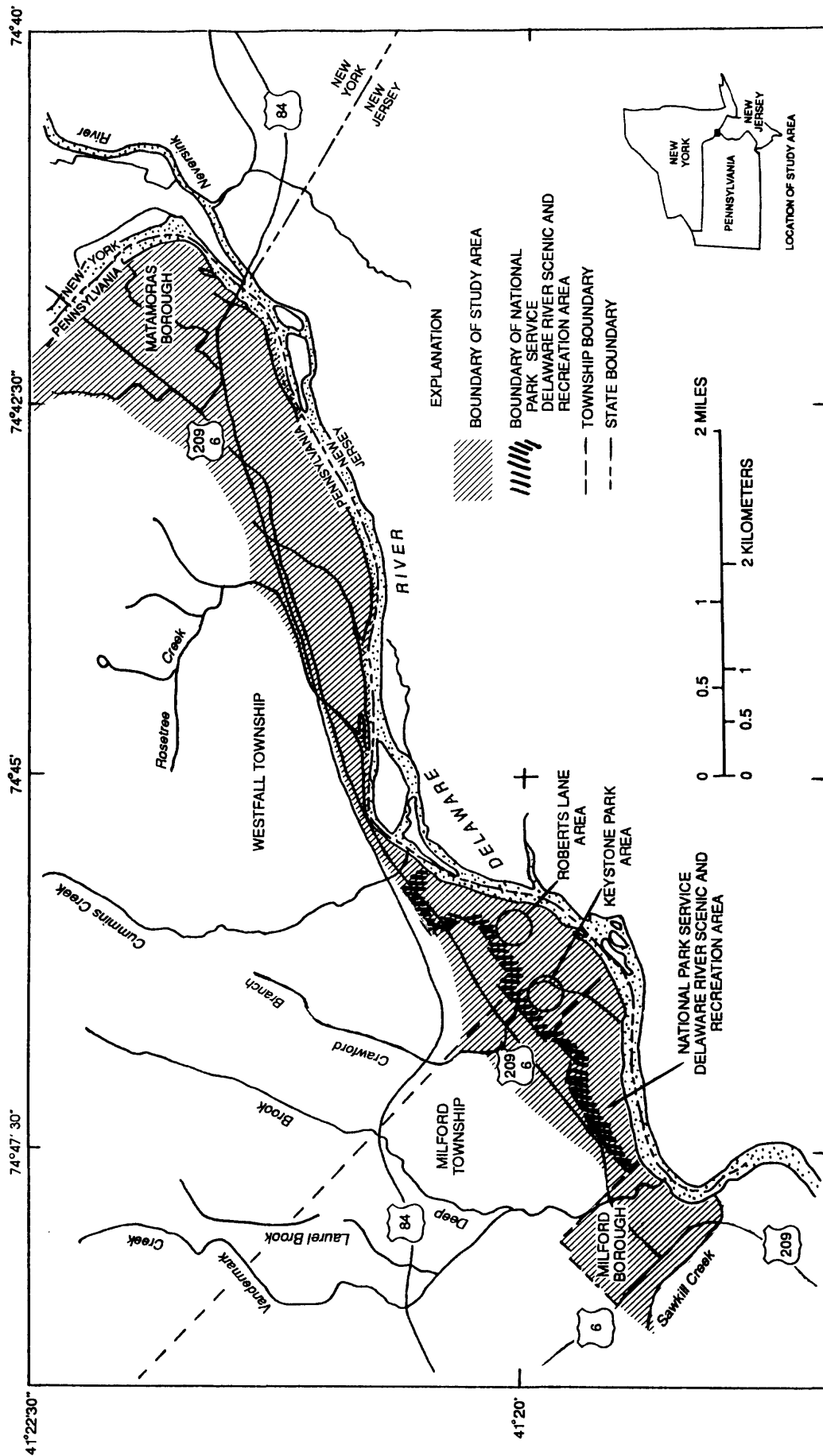


Figure 1. Location of study area in the Routes 209 and 6 corridor, Milford to Matamoras, Pike County, Pennsylvania.

Pleistocene), and alluvial fans crop out at the mouth of stream valleys (Davis, 1989; Sevon and others, 1989). Pebbles in the glacial deposits are mostly from nearby geologic formations including the Mahantango Shale (Crowl, 1971). Some patches of the unconsolidated glacial deposits are cemented by calcium carbonate or iron oxide (Crowl, 1971, p. 17-18).

The outwash deposits are unconsolidated, stratified sand and gravel with some boulders that were deposited by streams flowing from the melting glacier (Davis, 1989; Sevon and others, 1989). Thickness of the outwash deposits is reported to be as much as 500 ft near Matamoras (Davis, 1989). Driller's logs indicate that thickness of the outwash deposits is about 100 to 150 ft near the Delaware River both north and south of Matamoras. Depth to bedrock (thickness of unconsolidated deposits) reported from drillers' logs is variable, which suggests that the underlying bedrock surface was unevenly scoured by the glaciers. The outwash deposits are more extensive near Matamoras than near Milford, where most of these deposits are within the National Park boundary.

The kame-terrace features are unconsolidated, stratified sand and gravel deposits, commonly with large boulders; rock fragments of underlying bedrock are incorporated into these deposits, which have abrupt textural changes between beds and lenses (Davis, 1989; Sevon and others, 1989). The linear kame terraces were deposited in channels by glacial-melt streams in depressions between the ice edge and valley wall (Crowl, 1971). Drillers' logs indicate that the kame-terrace deposits are thinnest (10 to 14 ft) near the edge of the valley and thicken to at least 230 ft toward the Delaware River. The kame-terrace features are more extensive near Milford, where they form a terrace 60 ft above the outwash deposits, than near Matamoras. The kame terrace that extends from Cummins Creek to 4 mi south of Milford is the longest in the Delaware River valley in Pennsylvania (Crowl, 1971).

Well-Numbering System and Well Construction

The well-numbering system used in this report is the county name abbreviation followed by a sequentially assigned local well number. All local well numbers in Pike County have the prefix Pi. Wells included in this study are listed in numerical sequence in table 9 (at the end of report), and their locations are shown in plate 2.

All wells drilled in the unconsolidated materials of the glacial aquifer require casing or well screen along their entire depth to prevent borehole collapse. Most of the wells completed in the glacial aquifer are 6 in. in diameter, are cased with steel, and are open-ended. Some larger supply wells may have 10 ft or more of well screen installed in a productive zone of the aquifer. Drive-point wells typically are about 2 in. in diameter, about 20 ft deep, and have an open screened point up to 3 ft long. Wells completed in bedrock are cased through the overlying unconsolidated materials, but are left as open holes along the length drilled into bedrock; water-bearing fractures supply the well.

Previous Investigations

The hydrogeology and ground-water development of the glacial deposits were briefly described by Lohman (1937). Davis (1989) provided a more detailed description of the geology and hydrology of the glacial aquifer and included data on water quality, well yield, and well-construction characteristics. The geology of the glacial deposits was described by Crowl (1971) and Sevon and others (1989). The bedrock geology of Pike County was described by Sevon and others (1989) and Davis (1989).

Acknowledgments

The cooperation of private well owners in Milford and Westfall Townships and the Matamoras Borough Water Authority in providing data and access to their wells for sampling and measuring water levels is gratefully acknowledged. Data on road and parking-lot areal coverage and the number of housing units was provided by Peter Wulforth of the Pike County Planning Commission. Laboratory space was made available by the National Park Service at the Delaware River Scenic and Recreation Area. Data on well construction were provided by the PaDER, Bureau of Topographic and Geologic Survey, and on water quality by the PaDER, Bureau of Water Quality.

GEOHYDROLOGY

The unconsolidated glacial deposits that overlie bedrock in the Delaware River valley form an unconfined aquifer. These glacial deposits mainly are recharged directly by precipitation, additionally by flow from the adjacent and underlying bedrock, and by infiltration of streamflow. The amount of recharge to the ground-water system in an unconfined aquifer depends upon climatic and hydrologic conditions and permeability of the aquifer materials. Most recharge to the glacial aquifer is discharged to the Delaware River or withdrawn from wells; near the Delaware River, however, some ground water may flow parallel to the river and discharge into the river downstream. The Delaware River probably is a discharge zone for ground water from both the bedrock and glacial aquifers along most of the 8-mi stream reach from Matamoras to Milford.

Seven streams join the Delaware River in this reach, six from the Pennsylvania side and two from the New Jersey side (fig. 1). In Pennsylvania the streams are, in downstream order, an unnamed stream in Westfall Township, Rosetree Creek, Cummins Creek, Crawford Branch, Vandermark Creek, and Sawkill Creek (fig. 1). Only Cummins, Vandermark, and Sawkill Creeks are perennial in reaches over the glacial deposits. Crawford Branch, Rosetree Creek, and the unnamed stream are intermittent; these streams go dry in the summer and fall months as the streams flow over the glacial deposits and lose water to the ground-water system.

Recharge to the glacial deposits by precipitation largely depends on the amount and timing of the precipitation. In northeastern Pennsylvania, recharge is greatest in the late fall, winter, or early spring months when evapotranspiration is least and the ground is not frozen. Davis (1989, p. 10) estimated that annual recharge for the Bushkill Basin, located about 20 mi to the southwest of the study area, ranged from 10.4 in. in a dry year (31 in. precipitation) to 27.9 in. in a wet year (52 in. precipitation). The mean recharge for the Bushkill Basin in an average year (43 in. precipitation) was 20.6 in. More recharge takes place through the glacial deposits, especially the outwash deposits along the Delaware River, than through bedrock in the Bushkill Basin (Davis, 1989, p. 10). On the basis of the normal annual precipitation of 44.2 in. at Matamoras, average recharge to the glacial aquifer is about 20 in. per year. This recharge rate is similar to the net recharge rate of about 21 in. that can be estimated from an assumed porosity of 0.35 (Freeze and Cherry, 1979, p. 37) and a maximum annual water-level difference of about 5 ft in a well (Pi-223) completed in the glacial aquifer.

In Milford and Westfall Townships, most ground water that is pumped is returned to the aquifer through discharge by septic systems. A general estimate of domestic consumptive use of ground water is about 10 percent of the volume withdrawn from the aquifer (Frimpter and others, 1990, p. 223). In Matamoras Borough, some of the water supply is derived from wells drilled in the bedrock aquifer. In Milford Borough, all of the water supply is derived from springs in the bedrock aquifer. In both boroughs, all waste water is discharged to the glacial aquifer by septic systems.

Aquifer Characteristics

Aquifer productivity is directly related to the permeability of aquifer materials. Hydraulic conductivity is a measure of permeability and is defined as the rate at which a unit volume of a fluid is transmitted through a unit cross-sectional area of aquifer in a direction perpendicular to flow under a unit hydraulic gradient per unit thickness of saturated aquifer material (Lohman, 1972). Units of hydraulic conductivity are length per time, such as feet per day. Transmissivity (T) is hydraulic conductivity multiplied by the saturated thickness of an aquifer (Lohman, 1972). Aquifer transmissivity can be estimated using specific-capacity data from single-well tests by a method described by Theis (1963). The specific capacity of a well is defined as the pumping rate divided by the drawdown; it measures the combined efficiency of the well and the aquifer transmissivity. In the study area, the specific-capacity method (Theis, 1963) was used to estimate transmissivity. Units of transmissivity are length squared per time, such as square foot per day.

In Pike County, wells in outwash deposits have a greater maximum and median specific capacity than wells in kame-terrace deposits or in bedrock aquifers (Davis, 1989). Nondomestic wells commonly are constructed for higher yields than domestic wells. The transmissivity of the glacial aquifers estimated from median specific capacity is about 270 ft²/d for the kame-terrace deposits and 670 ft²/d for glacial

outwash deposits. These transmissivities are from 6 to 100 times greater than transmissivities estimated from specific capacities for the Mahantango Formation and Marcellus Shale bedrock aquifers (Davis, 1989).

The rate at which water and conservative water-soluble salts or contaminants move through an aquifer depends largely on the permeability of aquifer materials and hydraulic gradient. Ground-water discharge (volume of ground water flow per unit of time) given by Darcy's Law is proportional to hydraulic conductivity multiplied by the cross-sectional area of the aquifer and hydraulic gradient. The velocity of ground-water flow can be calculated by dividing ground-water discharge by the cross-sectional area and by aquifer porosity. Conservative constituents travel at the same rate as ground water, whereas nonconservative constituents travel more slowly than ground water because of adsorption onto aquifer materials. Although the transmissivity of the glacial aquifers is much greater than that of the bedrock aquifers, ground-water velocities in the two aquifers probably do not differ as much because of steeper hydraulic gradients and lower porosities of the bedrock.

Water Levels and Ground-Water Flow

Water levels in wells penetrating an unconfined aquifer indicate the altitude of the water table. Water levels commonly are measured from the top of the well opening and are reported as depth to water below the land surface; this measurement can be converted to an altitude of the water table above sea level by use of the elevation of the land surface. The distribution of static water levels in an aquifer can indicate the direction of ground-water flow. Ground water flows from higher to lower altitudes (potentials). The ratio of the change in altitude of the water table over a given distance is the hydraulic gradient in an unconfined aquifer.

In August 1991, the USGS measured water levels in 18 wells completed in the glacial aquifer and 11 wells completed in the underlying bedrock aquifer. Differences in water level for adjacent wells completed at different depths in the glacial aquifer or in the bedrock aquifers were not detected, indicating a good hydraulic connection between the glacial and bedrock aquifers. The altitude and configuration of the water table was estimated from water levels measured in 31 wells during August 1991, including levels reported by the Matamoras Borough Authority in 2 wells (plate 3). One of the Matamoras Borough Authority wells (Pi-271) is not shown on plate 3 because it is just north of the study area; data for the well were used for water-level control in the area near the Delaware River. Most of the water levels were measured in nonpumping wells or in wells that had not been recently pumped and, thus, represent static conditions.

The water-table gradient determined from the water-level data indicates that ground water flows from the edges of the valley through the glacial and underlying bedrock aquifers toward the Delaware River (plate 3). Near the Delaware River, ground-water flow becomes almost parallel to the river in the downstream direction. The gradient is steeper at the edges of the valley than near the river. In the Matamoras area, ground-water levels along the northern river boundary are at the same or lower altitude than the water surface of the Delaware River, indicating that Delaware River water may infiltrate into the ground-water system along this boundary. The high water table in Matamoras (water levels are about 15 ft below land surface) may be sustained in part by river inflow. Downstream from Matamoras, ground-water altitudes are the same or greater than the water surface of the river, indicating that ground water discharges to the river.

The lack of difference detected in water levels between the bedrock and glacial aquifers suggests that little or no vertical flow occurs and that the direction of ground-water flow is similar in both aquifers. The Delaware River is a discharge zone for both aquifers. Because the glacial aquifer is thin at the edge of the valley and thickens toward the river in the southern part of the study area, ground water from the bedrock aquifer at the edges of the valley probably flows through the glacial aquifer toward the Delaware River.

Water levels can decrease in response to withdrawals (discharge) from the aquifer, such as pumping or evapotranspiration. Water levels can increase in response to recharge by infiltration of precipitation, streamflow, irrigation, or other artificial diversions. The change in water levels indicates a net gain or loss of ground water in the aquifer (change in storage). Water levels in well Pi-223, which penetrates the glacial aquifer, show that changes in this well are seasonal with a range of about 5 ft (Davis, 1989, p. 17-18). A rise

in the water table (net recharge) typically occurs during the winter and early spring period during which evapotranspiration is at a minimum and infiltration is greater than discharge. Water levels measured during periods when the aquifer is not being stressed are static.

Water levels measured in three nonpumping wells (Pi-456, Pi-463, and Pi-477) in August 1991 and again in October 1991 indicate that water levels declined from 0 to about 5 ft in the 2-month period (table 9). However, no temporal differences in static water levels greater than 10 ft were indicated in these three wells by 1991 and older data for other wells (table 9). Seasonal changes in water levels between August and October generally are 5 ft or less. The water table may be a few feet higher in the winter and spring, but it probably has a similar configuration.

Depth to water is greater in the southern part of the study area where many wells are drilled into kame-terrace deposits than in the northern part of the study area where most wells are drilled into outwash deposits. In the southern part, depth to water in 1991 ranged from 98 ft below land surface in the kame terrace deposits to 30 ft in the outwash deposits. In the northern part, depth to water in 1991 ranged from 26 ft below land surface in the kame terrace deposits to 13 ft in the outwash deposits. The water levels in wells completed in bedrock and glacial aquifers near the Delaware River are close to the altitude of the river's water surface. The altitude of the Delaware River water surface drops about 40 ft from the streamflow-gaging station 01434000 at Port Jervis, N.Y. (across from Matamoras), to the streamflow-gaging station 01438500 at Montague, N.J. (just below Milford) (plate 2). Water levels in the glacial aquifer are below the elevation of the streams tributary to the Delaware River that traverse the glacial aquifer. A cross section showing water levels and thickness of the glacial deposits estimated from driller logs is shown in figure 2.

Water Quality

The chemical composition of ground water under natural conditions is derived from atmospheric precipitation, weathering of aquifer minerals, biological processes in soils and aquifer materials, and other reactions during infiltration and flow in the aquifer. Major inorganic ions and constituents in ground water include calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^+), potassium (K^+), chloride (Cl^-), sulfate (SO_4^{2-}), bicarbonate (HCO_3^-), nitrate (NO_3^-), and silica (SiO_2). The minor ions include iron ($\text{Fe}^{2+, 3+}$), manganese ($\text{Mn}^{2+, 4+}$), and fluoride (F^-) in addition to other trace constituents. In addition, water commonly is characterized chemically by the pH (a measure of hydrogen ion concentration), alkalinity, specific conductance, concentration of dissolved oxygen, and oxidation-reduction potential. Specific conductance measures the electrical conductivity of a solution and can be used to estimate the quantity of dissolved ions in solution.

Twenty-three wells completed in the glacial aquifer and one well (Pi-483) completed in the underlying bedrock aquifer were sampled in October 1991 to characterize general water quality of the glacial aquifer in areas selected for chloride and nutrient loading evaluation. All ground-water samples were collected before reaching filters or treatment systems. Wells were pumped to flush standing water from plumbing and the well bore, and water samples were collected as close to the well head as possible to obtain a sample representative of ground water in the aquifer. The pH, alkalinity, specific conductance, temperature, and concentration of dissolved oxygen of the ground-water samples were measured in the field by use of standard methods (Wood, 1976). In the pH range of ground water in the study area, bicarbonate is assumed to be the dominant component of alkalinity. The ground-water samples were analyzed for concentrations of dissolved calcium, magnesium, sodium, potassium, silica, chloride, fluoride, sulfate, iron, manganese, lead (Pb), and nutrients, consisting of nitrate, nitrite (NO_2^-), ammonium (NH_4^+), ammonium plus organic nitrogen (org-N), and phosphate (PO_4). All samples were analyzed by the USGS Central Laboratory in Arvada, Colo. (table 10, at the end of report).

The ranges and medians of measured physical properties and chemical constituents for 18 wells in the glacial aquifer not known to be highly contaminated are presented in table 1. Five of the 23 sampled wells are in a contaminated area of the glacial aquifer, and results for these 5 wells are discussed separately. The minimum concentrations of calcium, magnesium, sodium, and chloride were measured in a water sample from well Pi-485. Concentrations of all ions in ground water from this well probably represent natural or nearly natural background concentrations in the shallow glacial aquifer (table 10).

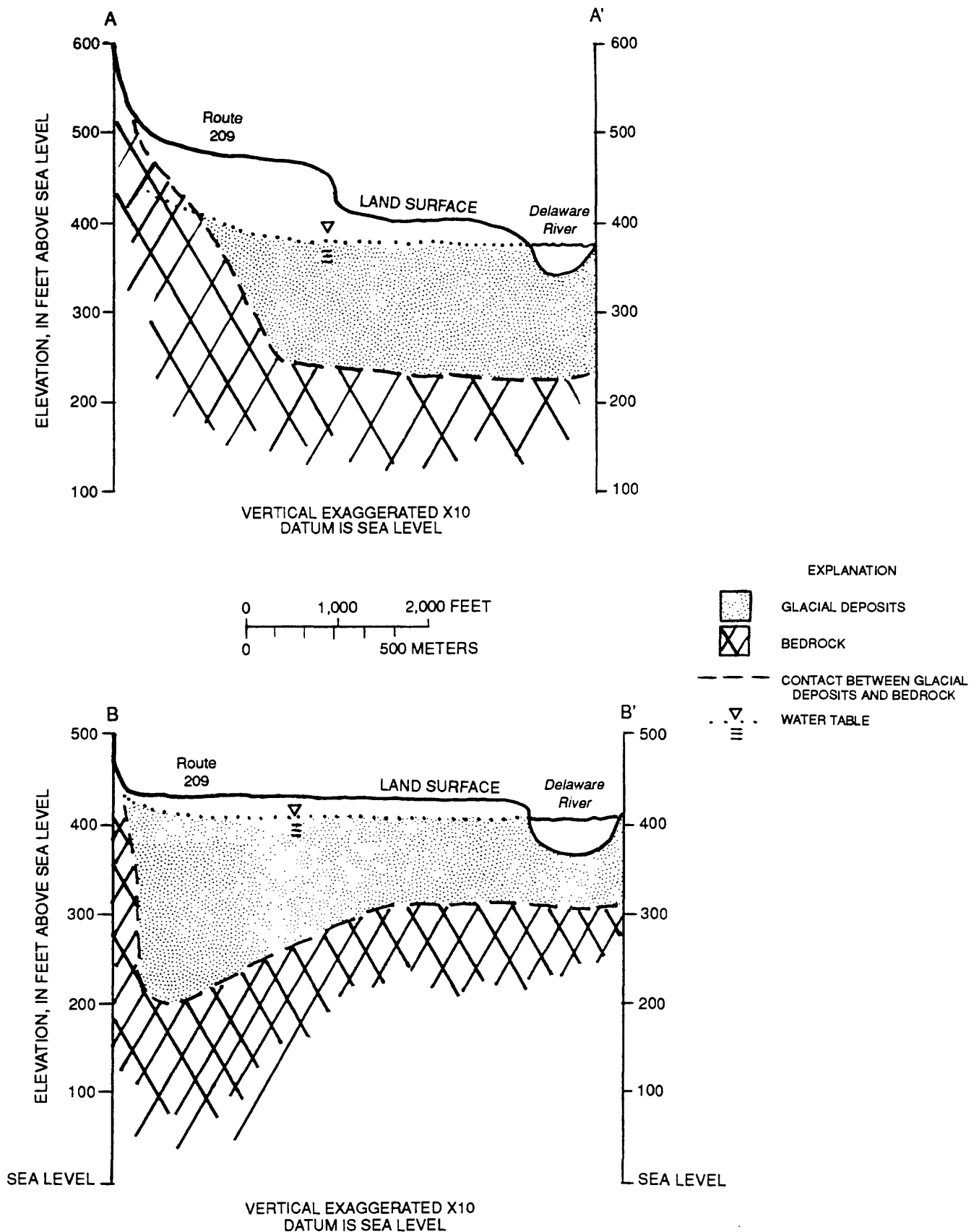


Figure 2. Generalized hydrogeologic sections showing water levels and thickness of the glacial aquifer.

The ground water in the glacial aquifer is acidic to neutral. For the ground-water samples collected in October 1991, the median pH was 6.5, and thus half the samples were more acidic than the USEPA SMCL range for pH (table 1). Total dissolved solids in most of the ground-water samples were less than 200 mg/L. Because recharge water from precipitation is acidic, the slight acidity and low concentration of dissolved solids of the ground water reflects limited amounts of chemical reaction and silicate-mineral dissolution. Dissolution of minerals in aquifer materials releases ions and other constituents into ground water and can change the pH and composition of the water. For example, the dissolution of feldspar and other silicate minerals results in increases in concentrations of silica and major cations (calcium, magnesium, sodium, or potassium), pH, and bicarbonate alkalinity. Concentrations of dissolved iron and manganese in some ground-water samples exceeded the USEPA SMCL's for these constituents (tables 1 and 10).

Concentrations of dissolved oxygen ranged from near 0.2 to 10.6 mg/L. The presence of dissolved oxygen in ground water can indicate rapid or recent recharge or short residence time, although oxygen can be consumed by reduction-oxidation reactions in the soil zone or aquifer rapidly or at any time. Some oxidation-reduction reactions are biologically-mediated reactions that involve the oxidation of organic carbon and the sequential consumption of dissolved oxygen and reduction of nitrate, manganese, iron, and sulfate in ground water (Stumm and Morgan, 1980).

Ground water in contaminated areas of the glacial aquifer has greater concentrations of most ions than ground water in areas not known to be contaminated. The range and median of measured physical properties for four wells (Pi-232, Pi-481, Pi-482, and Pi-486) downgradient from a tanning industry and one well (Pi-463) in the borough of Matamoras are presented in table 2. The ground water downgradient from the tanning operation contains elevated concentrations of calcium, sodium, and chloride and appears

Table 1. Minimum, maximum, and median values of physical properties and concentrations of selected ions in ground-water samples from 18 wells completed in the glacial aquifer, October 1991

[°C, degree Celsius; $\mu\text{S}/\text{cm}$, microsiemen per centimeter at 25 degrees Celsius; mg/L, milligram per liter; $\mu\text{g}/\text{L}$, microgram per liter; CaCO_3 , calcium carbonate; N, nitrogen; org-N, organic nitrogen; <, less than; --, no data]

Constituent or property	Minimum	Maximum	Median	Primary or secondary drinking-water regulations ¹
Field measurements and physical properties				
Temperature (°C)	9.5	14	11	—
Specific conductance ($\mu\text{S}/\text{cm}$)	85	300	196	—
Dissolved oxygen (mg/L)	.2	10.6	6.9	—
pH	5.6	7.4	6.4	² 6.5 - 8.5
Alkalinity (mg/L as CaCO_3)	9	92	44	—
Dissolved constituents determined by laboratory analyses				
Calcium (mg/L)	7.9	47	20.5	—
Magnesium (mg/L)	2.6	6.4	4.1	—
Sodium (mg/L)	2.8	16	6.3	—
Potassium (mg/L)	.4	2.4	.5	—
Chloride (mg/L)	2.1	32	17.5	² 250
Sulfate (mg/L)	11	42	25	² 250
Fluoride (mg/L)	<.1	.2	<.1	³ 2
Silica (mg/L)	6.9	11	8.5	—
Iron ($\mu\text{g}/\text{L}$)	6	310	58	² 300
Manganese ($\mu\text{g}/\text{L}$)	<1	290	7	² 50
Lead ($\mu\text{g}/\text{L}$)	<1	2	<1	⁴ 15
Ammonium (mg/L as N)	<.01	.02	<.01	—
Nitrite (mg/L as N)	<.01	.47	<.01	³ 10
Ammonium + org-N (mg/L as N)	<.2	<.2	<.2	—
Nitrate + nitrite (mg/L as N)	<.05	5.10	1.07	³ 10
Phosphate (mg/L)	<.01	.04	<.01	—

¹ Primary and secondary regulations established by the U.S. Environmental Protection Agency (1989).

² Secondary regulation recommended for public supplies.

³ Primary regulation required for public supplies.

⁴ Action level required for public supplies (U.S. Environmental Protection Agency, 1991).

to be contaminated by salts released in process waste water through an on-lot disposal system (Stan Leaman, Pennsylvania Department of Environmental Resources, oral commun., 1993). The well in the borough of Matamoras also contains elevated concentrations of all major ions and probably is affected by septic-system effluent and road salts in the borough. Contamination of the glacial aquifer is discussed in the following section on nutrients and chloride.

Ground water in aquifers with different mineralogies commonly have different characteristic chemical compositions. The relative proportion of the major cations and anions can be used to identify ground water of different types. In most ground-water samples from the glacial aquifer, calcium is the dominant cation, and bicarbonate and sulfate are relatively important naturally occurring anions that may be derived from calcite or gypsum in the aquifer. The difference in water type between ground water in the bedrock aquifer and the glacial aquifer is shown in a piper diagram (fig. 3), which shows the relative concentration of cations and anions. The ground-water sample from well Pi-483 in the Mahantango Formation, a bedrock aquifer, is relatively high in sodium and bicarbonate compared to the samples from the glacial aquifer. Clay minerals or sodium-bearing feldspars may be the source of sodium and bicarbonate in the shale.

Most ground-water samples from the outwash deposits in the northern part of the study area (wells Pi-213, Pi-456, Pi-463, Pi-485, and Pi-487) have slightly greater relative concentrations of sulfate than elsewhere in the glacial aquifer, suggesting spatial variability of ground-water quality. The greater concentrations of sodium and chloride plus nitrate in ground-water samples probably have anthropogenic sources.

Table 2. Minimum, maximum, and median values of physical properties and concentrations of selected ions in ground-water samples from five contaminated wells completed in the glacial aquifer, October 1991

[°C, degree Celsius; $\mu\text{S}/\text{cm}$, microsiemen per centimeter at 25 degrees Celsius; mg/L, milligram per liter; $\mu\text{g}/\text{L}$, microgram per liter; CaCO_3 , calcium carbonate; N, nitrogen; org-N, organic nitrogen; <, less than]

Constituent or property	Minimum	Maximum	Median
Field measurements and physical properties			
Temperature (°C)	10.5	13.5	11
Specific conductance ($\mu\text{S}/\text{cm}$)	525	2,480	1,570
Dissolved oxygen (mg/L)	.3	5.6	.5
pH	6.6	7.5	7.0
Alkalinity (mg/L as CaCO_3)	31	142	84
Dissolved constituents determined by laboratory analyses			
Calcium (mg/L)	80	300	100
Magnesium (mg/L)	11	47	21
Sodium (mg/L)	5	180	97
Potassium (mg/L)	.6	1.6	1.2
Chloride (mg/L)	77	680	370
Sulfate (mg/L)	33	190	53
Fluoride (mg/L)	<.1	.2	.2
Silica (mg/L)	6.3	10	8.2
Iron ($\mu\text{g}/\text{L}$)	14	450	110
Manganese ($\mu\text{g}/\text{L}$)	1	2,100	220
Lead ($\mu\text{g}/\text{L}$)	<1	<1	<1
Ammonium (mg/L as N)	<.01	.04	.03
Nitrite (mg/L as N)	<.01	<.01	<.01
Ammonium + org-N (mg/L as N)	<.2	<.2	<.2
Nitrate + nitrite (mg/L as N)	<.05	4.0	<.05
Phosphate (mg/L)	<.01	<.01	<.01



Figure 3. Relative concentration of major anions and cations in ground-water samples from 23 wells completed in the glacial aquifer and 1 well completed in the bedrock aquifer. Percent calculated for concentration in milliequivalents per liter.

NITROGEN AND CHLORIDE IN THE GLACIAL AQUIFER

Chloride and the nutrients nitrogen and phosphorus are present in septic-system effluent, and chloride is present in road runoff. A nutrient is an element that is essential to plant and animal life; only some forms or compounds containing that element are favored by or available to biological processes. The dissolved nutrient constituents addressed in this study are nitrate, ammonium, nitrite, organic nitrogen, and phosphate. The presence of chloride and these nutrients in ground water can indicate contamination by road runoff or septic systems. Some of these constituents also are present in leachate and in other contaminant sources, such as fertilizer. Therefore, these constituents are not necessarily unique identifiers of the sources of contamination.

Nitrate and chloride are relatively chemically unreactive ions in ground water compared to other contaminants and may persist once introduced into the ground-water system (Freeze and Cherry, 1979). Nitrate and chloride in ground water can be removed through the process of flushing (if sources are eliminated) or they can be diluted through the processes of dispersion and mixing. Recharge water with low concentrations of chloride and nitrate also will dilute elevated ground-water concentrations.

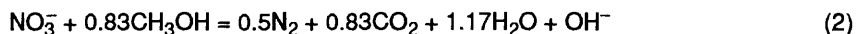
The chloride ion is effectively conservative and will travel at the same rate and in the same direction as ground-water flow. Chloride is very soluble in water and does not sorb onto aquifer materials readily, whereas ions that precipitate as solids or are sorbed onto aquifer materials may be removed from ground water or retarded in transport.

Nitrate also is conservative with respect to physical transport in ground water, but can be chemically reactive. In ground water under oxidizing conditions, nitrate is more stable than ammonium, nitrite, or organic nitrogen. These nitrogen compounds are part of the nitrogen cycle (fig. 4). Nitrogen is taken up and released by plant and animals in some of its reduced forms, such as ammonia, but is broken down to its oxidized form by bacteria in the environment. Bacteria are important in both the processes of nitrification and denitrification. Denitrification in ground water under reducing conditions can decrease nitrate concentrations.

The bacterially mediated reactions of nitrification and denitrification can change the chemical composition of ground water; simplified reactions are given in equations 1 and 2 (Andreoli and others, 1979, p. 844). Ammonium is transformed into nitrate during nitrification (eq. 1), a process that consumes oxygen, releases hydrogen ions, and results in greater acidity (lower pH) and lower bicarbonate alkalinity. The nitrification reaction is



Nitrate is transformed into nitrogen gas (N_2) during denitrification (eq. 2), a process that oxidizes carbon, releases hydroxyl ions (OH^-), and results in less acidity (higher pH). Nitrogen in gaseous form may escape to the atmosphere. The denitrification reaction that uses methanol (CH_3OH) as a carbon source is



Ammonium (NH_4^+) is a cation (positively charged ion), whereas nitrate (NO_3^-) is an anion (negatively charged ion). Ammonium is much more readily sorbed onto negatively charged aquifer materials than nitrate and is less chemically stable than nitrate in an oxidizing environment. Thus, ammonium commonly is not as mobile in ground water as nitrate, especially in the presence of oxygen.

Phosphorus is another nutrient that is taken up and released by plant and animal life. Dissolved phosphorus is most stable as the phosphate ion. Elevated concentrations of phosphate usually are not observed in ground water because of biological uptake, low solubility, or adsorption onto aquifer materials (Hem, 1985).

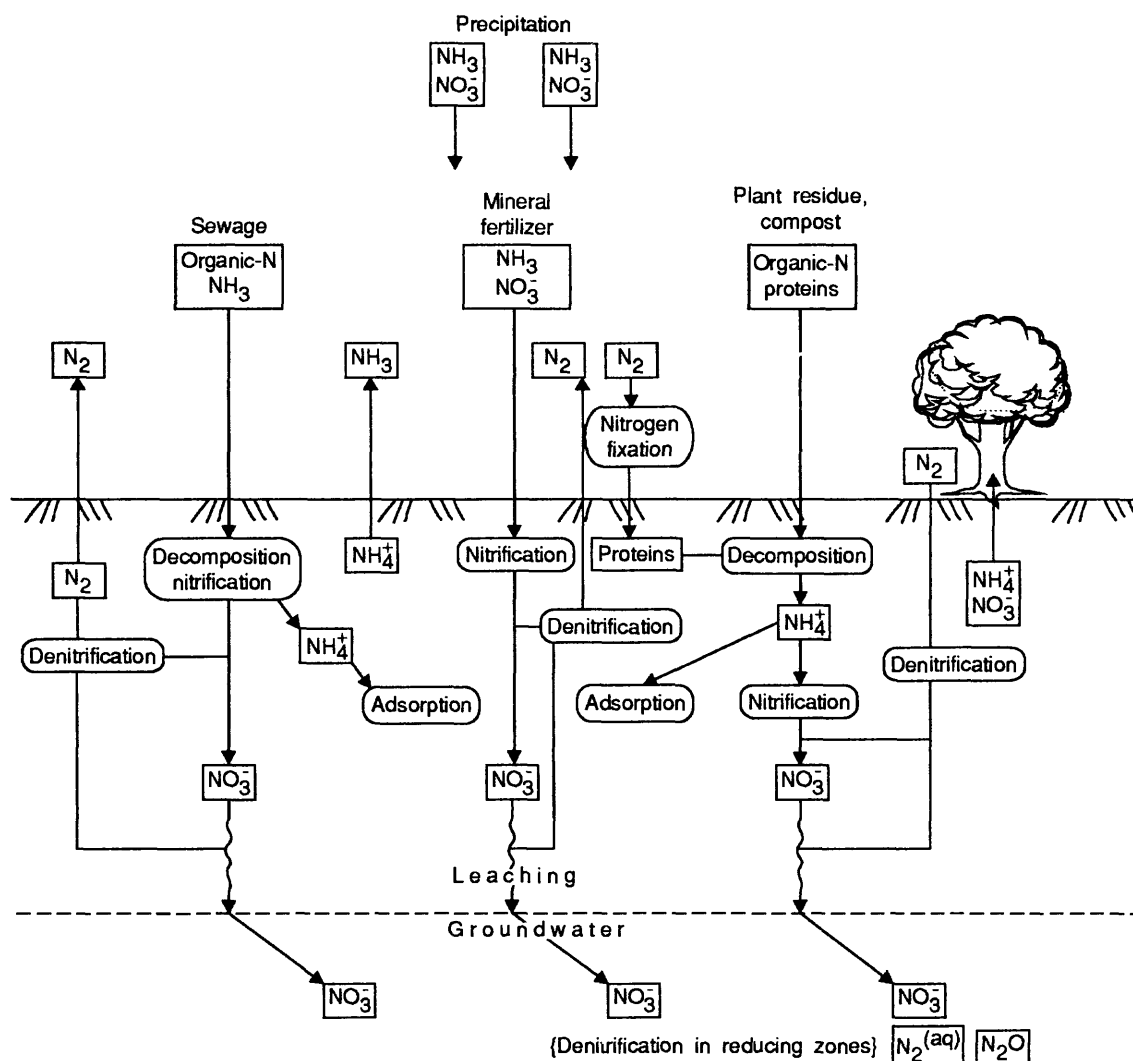


Figure 4. Sources and pathways of nitrogen in the unsaturated zone and ground water.
(modified from Freeze and Cherry, 1979, p. 414)

Sources

Nutrients and chloride are introduced by recharge to the ground-water system from both natural and human sources. Natural sources of nutrients and chloride include atmospheric precipitation and animal, plant, and bacterial processes that involve the release, as well as the uptake, of nutrients. Anthropogenic sources of both nutrients and salts, in addition to other constituents such as organic compounds and metals, include agricultural fertilizer, road runoff, atmospheric emissions, and septic-system waste.

Atmospheric precipitation, road runoff, and septic-system effluent are the sources considered and quantified in the calculation of nitrogen and chloride loads reaching ground water in the glacial aquifer. Natural sources other than precipitation are assumed to contribute negligible amounts of nitrogen and chloride to ground water. Agricultural activities may be sources of nitrogen or chloride, but these sources are not considered in this study because present agricultural activity is small and the effects on ground water in the corridor of past agricultural activities are unknown. At present, agriculture is limited to corn cultivation in the low lying lands near the Delaware River that are downgradient from residential and commercial areas, and only one active dairy farm operates along Routes 209 and 6. However, past agriculture was more extensive and included poultry production, a possible source of nitrogen and chloride.

Atmospheric Precipitation

Atmospheric precipitation is the major natural source of chloride and is an important source of nitrogen to ground water in the glacial aquifer. The chemical composition of precipitation varies, depending upon preceding atmospheric conditions, season, and origin and type of precipitation event. In eastern Pennsylvania, winter storms commonly are from the Atlantic Ocean, and precipitation has relatively high concentrations of sodium and chloride. Summer storms commonly are continental, move easterly, carry relatively higher concentrations of sulfate and nitrogen compounds, and have a lower pH in precipitation than winter storms (Joseph Scudlark, University of Delaware, oral commun., 1992; Lynch and others, 1992). Elevated concentrations of nitrate and sulfate in precipitation and the acidic quality of precipitation in Pennsylvania is related to emissions from automobiles and fossil-fuel burning power plants in Pennsylvania and the upper Ohio River Valley (Slade, 1990).

The average concentration and total annual load of chloride, nitrate, ammonium, and other major constituents in precipitation at Milford from 1982 to 1991 are given in table 3. The average concentrations of chloride and other components of precipitation were determined from precipitation samples collected weekly by the U.S. Forest Service in Milford at a site included in the National Atmospheric Deposition Program (Lynch, 1990; Lynch and others, 1992). Sulfate, nitrate, and ammonium, compared to chloride, sodium, calcium, magnesium, and potassium, are relatively abundant ions in the precipitation.

The maximum possible concentrations of chloride and nitrogen compounds contributed to ground water from precipitation are calculated by assuming total conservation of mass. Some of the nitrogen and chloride deposited by precipitation on the land surface may not infiltrate into the ground-water system because of losses to surface runoff, uptake by plants, or retention in the unsaturated zone. The maximum concentration of a constituent in recharge is calculated by use of the equation:

$$C_r = M_p P_i / V_r, \quad (3)$$

where C_r is concentration in recharge,

M_p is mass load in precipitation,

P_i is percent of mass that infiltrates, and

V_r is volume of recharge.

The maximum concentrations of chloride and nitrogen in recharge water were calculated using equation 3. The calculations were based on the following: (1) loss to surface-water runoff is estimated to be 15 percent of the water not evaporated or transpired on the basis of water budgets by Davis (1989); (2) average annual recharge is 20 in.; and (3) average annual precipitation loads as given in table 3. This calculation indicates that the estimated maximum concentrations in ground-water recharge contributed by precipitation are 0.5 mg/L for chloride and 1.1 mg/L as N for nitrate plus ammonium. This maximum recharge concentration estimated from precipitation probably is much more accurate for chloride, a relatively conservative ion, than for nitrate or ammonium. Some of the nitrogen compounds are taken up by plants or are transformed by bacteria to nitrogen gases at or near the land surface. Chloride, however, is not retained significantly by biological processes and is lost mostly to surface-water runoff.

Table 3. Average concentrations and annual loads of chemical constituents in wet precipitation at Milford, Pa., 1982-91

[Concentration is weighted by volume. Source of data is Lynch and others (1992); mg/L; milligram per liter; lb/mi², pound per square mile; N, nitrogen]

Chemical constituent	Concentration (mg/L)	Annual load (lb/mi ²)
Hydrogen	¹ —	383
Sodium	0.10	1,160
Calcium	.098	757
Magnesium	.035	229
Potassium	.023	147
Chloride	.258	2,070
Sulfate	2.50	16,400
Nitrate as N	.415	2,600
Ammonium as N	.166	255

¹ Median pH = 4.2

Septic Systems

Chloride and the nutrients nitrogen and phosphorus are present in septic-system effluent. Chloride is present in sewage because of salt in the typical diet. Nitrogen is a metabolic waste product and is present mostly as ammonium in septic-tank effluent. Phosphorus is introduced into domestic sewage as another waste product of animal metabolism and from sodium phosphate detergents (Hem, 1985, p. 126-127); about 85 percent of total phosphate is present primarily as orthophosphate in septic-tank effluent (Canter and Knox, 1985). The septic-tank effluent changes chemically once discharged into the unsaturated zone where ammonium and other reduced nitrogen compounds commonly are oxidized to form nitrate under aerobic conditions (Andreoli and others, 1979), and phosphate tends to be sorbed, precipitated, or biologically removed (Hem, 1985).

Typical or average concentrations of chloride, nitrogen, and phosphate as phosphorus in domestic septic-tank effluent are reported by Walker and others (1973a), Andreoli and others (1979), Canter and Knox (1985), and Robertson and others (1991) (table 4). Estimates for phosphate concentrations in septic-system effluent in table 4 may be greater than those in septic effluent in the study area because the sale of phosphate-bearing detergents has been banned in Pennsylvania for several years. Other constituents in domestic sewage listed in table 4 include calcium, magnesium, sodium, potassium, and sulfate. Elevated concentrations of these constituents have been detected elsewhere in ground water in unconfined sand aquifers contaminated by septic-system effluent (Robertson and others, 1991; Denver, 1989).

Typical volumes of residential septic-tank effluent releases are estimated to range from about 45 (gal/person)/d (Canter and Knox, 1985; Reneau and others, 1989) to 65 (gal/person)/d (Frimpter and others, 1990). Leaching rates for nitrate as nitrogen from residential septic tanks into ground water in glacial outwash aquifers in Wisconsin were estimated to equal 8.2 kg N per person per year (Walker and others, 1973a), which is equivalent to an effluent volume of 75 (gal/person)/d and a concentration of 80 mg/L as N for the system studied by Walker and others (1973b).

Table 4. Reported minimum, maximum, and average concentrations of selected ions in septic-tank effluent

[Sources: 1 = Canter and Knox (1985); 2 = Walker and others (1973a); 3 = Andreoli and others (1979); 4 = Robertson and others (1991); mg/L, milligram per liter; N, nitrogen; P, phosphorus; -, not reported]

Constituent or property	Minimum	Maximum	Average	Sources
Nitrate (mg/L as N)	0.1	1	0.1	3, 4
Ammonium (mg/L as N)	40	100	70	2
Organic N (mg/L as N)	5	25	15	2
Total N (mg/L as N)	-	-	40	1
Phosphate (mg/L as P)	-	-	15	1
Chloride (mg/L)	37	200	53	1, 2, 4
Iron (mg/L)	0	20	2.6	1
pH	6.5	7.5	6.9	1, 3
Alkalinity (mg/L)	-	-	186	3
Calcium (mg/L) ¹	0	6	-	4
Magnesium (mg/L) ¹	0	2	-	4
Sodium (mg/L) ¹	88	94	-	4
Potassium (mg/L) ¹	10	20	-	4
Sulfate (mg/L) ¹	0	36	-	4

¹ Constituents adjusted for background concentrations in ground water.

Road and Parking-Lot Runoff

Chloride and nutrient concentrations present in precipitation can increase in runoff as storm water flows over paved or other surfaces that have accumulated solids that include dry atmospheric deposition, road salt, leaf and other plant litter, animal wastes, and refuse.

Road, roof, and parking-lot runoff and precipitation were sampled at three sites (plate 2) in November 1991 over a period of 3 days of rain to assess the load of nutrients and chloride from these sources to ground water. Twenty-one runoff samples were collected at the three sites at various time intervals: two samples of roof runoff at the Head Injury Center (HIC); three samples from Route 209 runoff at the HIC; eight samples from combined parking-lot and roof runoff at the Milford Valley Convalescent Home (MVCH); and eight samples of parking-lot and roof runoff from the Delaware Valley High School (DVHS) (plate 2). The lot coverage at DVHS site (at least 4 acres) was greater than the lot coverage at the MVCH site (about 1.5 acres). Cumulative rainfall of 3.22 in. was measured by the USGS for the period of sampling at the MVCH site. The concentrations of chloride, sulfate, ammonium, nitrate, and nitrite in the precipitation and runoff from the November 1991 storm are given in table 5; chemical analyses for the runoff samples and the precipitation sample are given in table 11 (at the end of report).

A single storm sample does not account for seasonality, such as road salting in winter, changes in storm patterns and chemistry, preceding conditions of high evapotranspiration or recharge, or the growth and decay cycles of plants. Therefore, the concentrations of nutrients and chloride in runoff and precipitation from the one storm sampled in November 1991 may not be representative of annual loads of nutrients and chloride in runoff and precipitation.

The concentrations of most ions were greater in the initial samples than in the later samples during the November 1991 storm. Figure 5 shows the cumulative rainfall and nitrite plus nitrate concentrations for runoff samples collected at the DVHS site. The concentration of nitrite plus nitrate is inversely related to cumulative rainfall and rainfall intensity. Similarly, specific conductance and concentrations of ammonium and sulfate in runoff at the DVHS site decreased as the storm progressed (table 11). Concentrations of sulfate, nitrite plus nitrate, and nitrite, and specific conductance were less in runoff from the MVCH site than in runoff from the DVHS site; concentrations of ammonium and nitrite plus nitrate in runoff at the MVCH site were similar to the concentrations of those constituents in the composite precipitation sample (fig. 6). The storm followed 6 weeks of little or no precipitation, and the initially higher concentrations were the result of the first flush of storm water carrying salts and other deposits accumulated during the dry period. Additionally, the chemical composition of precipitation can vary during a storm, and some of the change in runoff concentrations may reflect the change in precipitation concentrations. Compared to the 10-year average for Milford (table 3), the November 1991 storm precipitation contained similar concentrations of chloride, but lower concentrations of nitrate, ammonium, and sulfate.

The greater concentrations of chloride in road runoff than in precipitation or parking-lot and roof runoff probably were the result of road salt application. The average annual road salt application in Pike County by the Pennsylvania Department of Transportation is about 2 tons of sodium chloride (NaCl) per lane mile on highways such as Routes 6 and 209 and Interstate 84 (C. Johnson, Pennsylvania Department of Transportation, oral commun., 1992). Concentrations of chloride in road and parking-lot runoff vary seasonally and can be much greater than those detected in the November 1991 sampling. During a 2-year water-quality study of runoff from parking lots in Maryland (Wilde, 1989), chloride concentrations in runoff ranged from about 4 to 2,240 mg/L, with the maximum concentration occurring during winter runoff events.

The concentrations of nitrogen species in runoff were similar or greater than those in precipitation. In both precipitation and runoff, nitrate is the dominant nitrogen ion; it is 2 to 3 times more abundant than ammonium and up to 25 times more abundant than nitrite. Runoff contained up to twice as much nitrogen as precipitation but still less than the average dormant season (November through April) precipitation concentration of 0.38 mg/L as N (Lynch and others, 1992).

In runoff studies of highways and commercial areas in Florida, total nitrogen concentrations were up to six times greater in runoff than in precipitation, although many runoff samples contained similar concentrations of total nitrogen (Hardee and others, 1978; Miller and others, 1979). These findings for runoff in other studies may be similar for runoff from highways and commercial areas in Pike County, where small to insignificant increases in nitrogen concentrations from rain to runoff were observed (fig. 6 and table 5). Without significant sources of nitrogen, such as fertilizer, in the study area, nitrogen loads in road, parking-lot, and roof runoff may not be much greater than the input from atmospheric wet and dry deposition that includes nitrogen from automobile emissions.

Table 5. Average concentrations of chloride, sulfate, and nutrients in precipitation and parking-lot, road, and roof runoff, November 21-23, 1991

[mg/L, milligram per liter; N, nitrogen; NH₄, ammonium; NO₂, nitrite; NO₃, nitrate; MVCH, Milford Valley Convalescent Home; DVHS, Delaware Valley High School; HIC, Head Injury Center; <, less than]

Constituent	Precipitation ¹ (mg/L)	Runoff (mg/L) ²			
		Lot at MVCH	Lot at DVHS	Road at HIC	Roof at HIC
Chloride	0.5	0.5	0.3	2.2	<0.1
Sulfate	1.2	1.4	5.2	1.9	1.2
NH ₄ as N	.05	.05	.06	.14	.09
NO ₂ as N	<.01	.01	.03	.02	.01
NO ₂ + NO ₃ as N	.16	.17	.36	.25	.24
Total N	.21	.22	.42	.39	.31

¹ Concentration in composite precipitation sample.

² Volume-weighted average concentration in runoff samples.

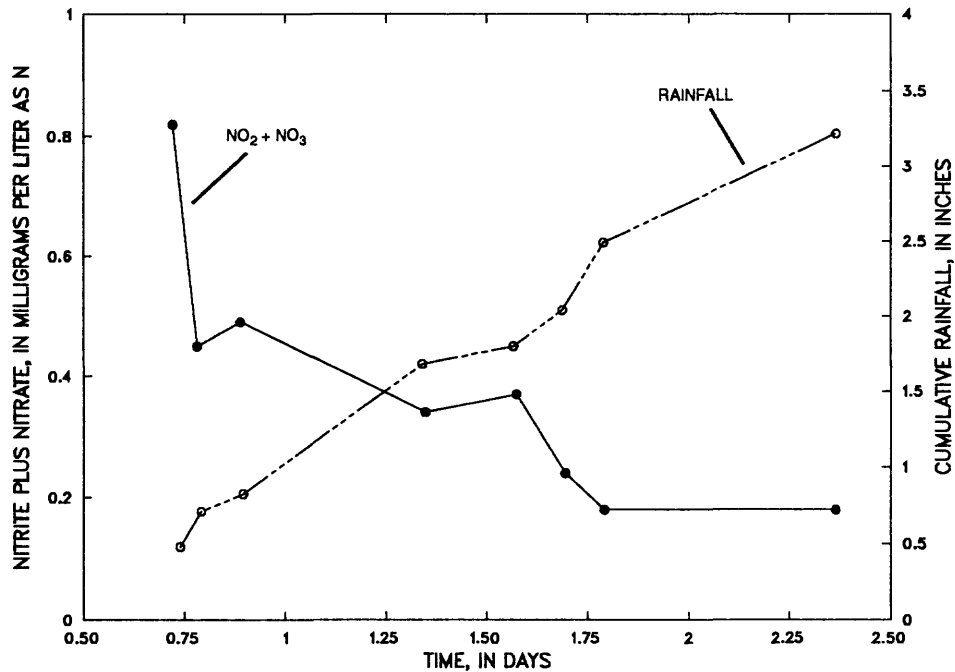


Figure 5. Cumulative rainfall and nitrite plus nitrate concentrations in runoff from the Delaware Valley High School sampling site, November 21-23, 1991.

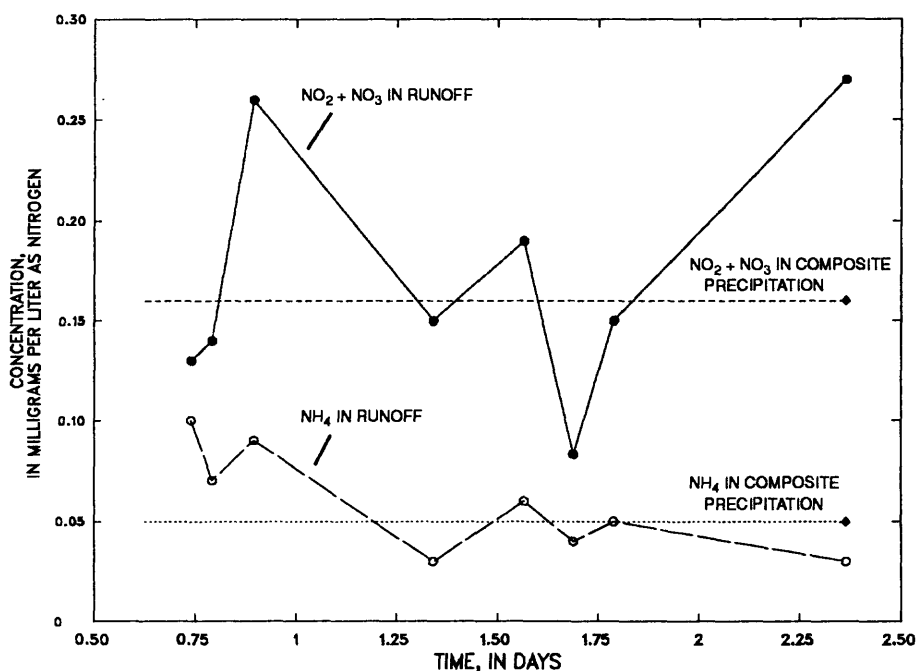


Figure 6. Nitrite plus nitrate and ammonium concentrations in eight runoff samples and one composite precipitation sample from the Milford Valley Convalescent Home sampling site, November 21-23, 1991.

Spatial Distribution

The effects of septic-tank discharge and highway runoff on ground-water quality in the glacial aquifer were evaluated by sampling water from wells both upgradient and downgradient from these sources. Most of the wells in the study area potentially could be affected by contamination from both road runoff and septic-tank effluent. Sampled wells upgradient from Interstate 84 and Routes 209 and 6 were Pi-485 and Pi-456; sampled wells very near and downgradient from the highways were Pi-113, Pi-213, Pi-473, and Pi-477 (plate 2). Sampled wells in three areas of septic-system density greater than average in the study area were, in the following general upgradient to downgradient order: Pi-479, Pi-229, Pi-230, Pi-478, Pi-227, Pi-480, and Pi-455 in Keystone Park; Pi-484, Pi-233, Pi-486, Pi-232, Pi-481, and Pi-482 on Roberts Lane; and Pi-463 in Matamoras (plate 2). Sampled wells not close to but downgradient from highways in areas of relatively less densely spaced septic systems were Pi-464, Pi-487, and Pi-472 (plate 2). Water from one well (Pi-484) completed in the bedrock aquifer upgradient of both highway and septic tank sources also was sampled (plate 2).

The concentrations of chloride in water from the 24 sampled wells ranged from 2.1 to 680 mg/L (fig. 7 and table 10). Upgradient of most roads and septic systems (wells Pi-456, Pi-483, and Pi-485), concentrations of chloride in ground water are less than 10 mg/L and represent nearly natural background concentrations. Concentrations of chloride greater than the USEPA SMCL of 250 mg/L were measured in three of four ground-water samples in the Roberts Lane area downgradient from a tanning operation that may have released salts in wastewater. In other areas in Westfall and Milford Townships (plate 2), concentrations of chloride were less than or equal to 32 mg/L in ground water. In these areas, chloride concentrations generally were greater in water samples from wells nearer major roads than in samples from wells farther from major roads; chloride concentrations were greater in water samples from wells downgradient from major roads than in samples from wells upgradient from major roads (fig. 7).

The distribution of chloride in ground water in the glacial aquifer does not appear to be related to the distribution of septic systems except possibly in the borough of Matamoras, where a ground-water sample from well Pi-463 contained a chloride concentration of 77 mg/L. This concentration is much greater than the chloride concentration of 9.2 mg/L measured in a water sample from well Pi-456, which is upgradient of the borough. Both septic systems and road salt are likely sources of ground-water contamination in Matamoras.

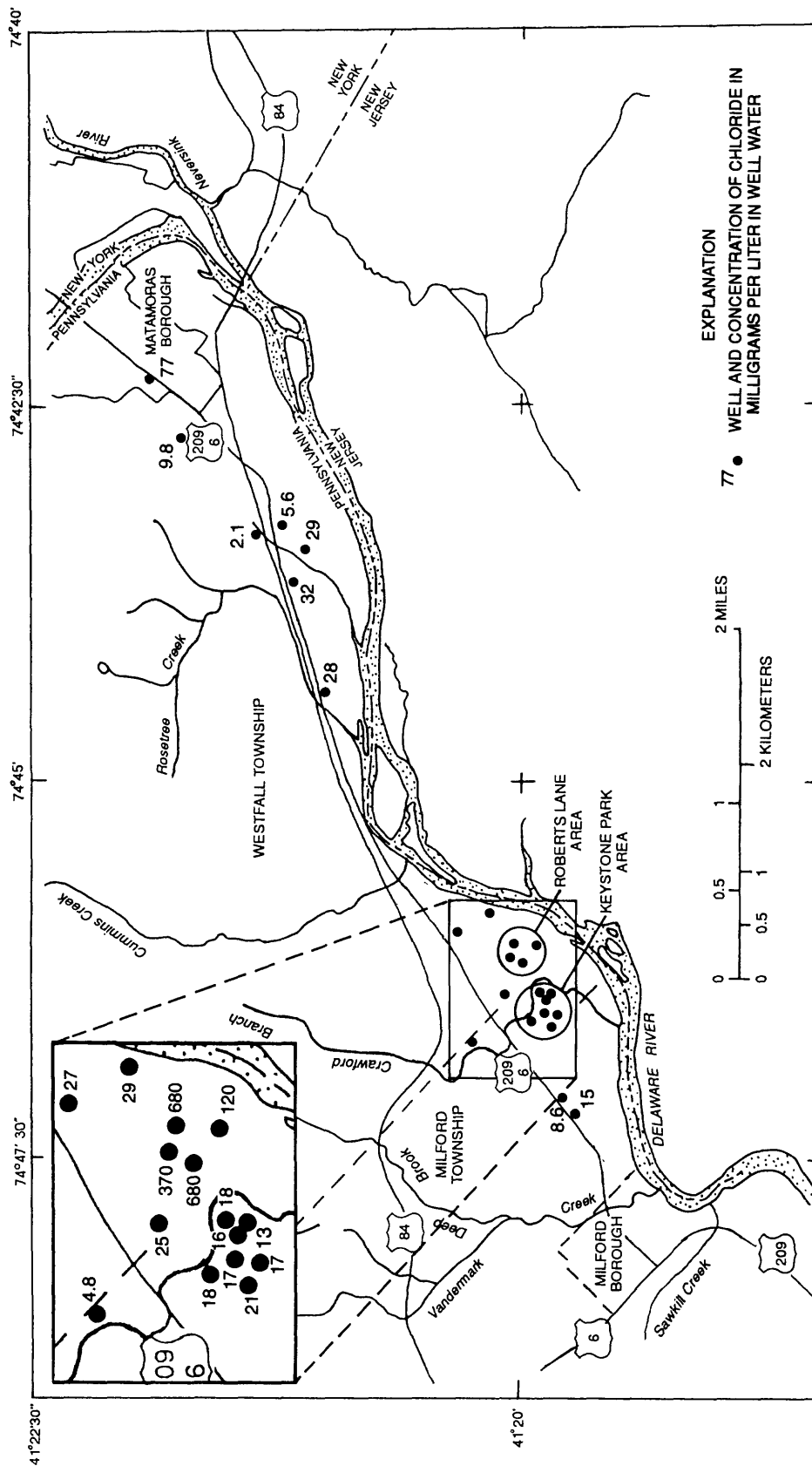


Figure 7. Concentrations of chloride in ground-water samples from 24 wells, October 1991.

The concentrations of nitrate in ground water ranged from less than 0.05 to 5.1 mg/L as N (fig. 8 and table 10). Nitrate concentrations in the 24 ground-water samples did not exceed the USEPA MCL of 10 mg/L as N for nitrate. The maximum nitrate concentration measured, 5.1 mg/L as N, was in a water sample collected from well Pi-213 at the DVHS and is equal to the nitrate concentration measured in a sample collected 10 years before (table 10). Nitrate concentrations of about 1 mg/L as N in oxygenated ground water probably represent background concentrations from infiltration of precipitation.

In the three areas sampled to assess effects of septic systems, the concentrations of nitrate were greater in water samples from wells upgradient from most septic systems than in samples from wells downgradient from most septic systems (fig. 8).

The spatial distribution of nitrate in ground water is different in these three areas than that observed elsewhere in areas with similar hydrogeologic settings (unconfined sand and gravel glacial aquifers) where septic-system contamination of ground water was detected. In other studies, water samples from wells downgradient from septic systems contain greater concentrations of nitrate and chloride, in addition to other constituents, than water samples from wells upgradient of septic systems (Rea and Upchurch, 1980; Robertson and others, 1991; Tinker, 1991). The spatial distribution of nitrate in ground water in the glacial aquifer in the study area suggests that nitrate removal or dilution takes place downgradient from septic systems. Other studies have shown that aquifers consisting of sand and gravel have high permeabilities, which should enhance nitrification and infiltration of nitrate into ground water (Reneau and others, 1979; Walker and others, 1973a) and that nitrate from septic tanks may be unlikely to undergo denitrification in ground water in unconfined sand and gravel aquifers (Walker and others, 1973a, b). Nevertheless, nitrogen could be removed by denitrification or remain unoxidized (reduced) as ammonium, a less soluble form, that tends to sorb onto aquifer materials. Ammonium concentrations were below the level of detection in water samples both downgradient and upgradient of septic systems.

Discharge from septic systems can sink and disperse in ground water, forming irregular plumes (Rea and Upchurch, 1980; Robertson and others, 1991). The ground-water samples collected may not have intercepted the septic-effluent plumes. Additionally, ground water supplying the wells downgradient of septic systems may not contain greater chloride and nitrate concentrations than wells upgradient of septic systems because of dispersion, dilution in the glacial aquifer that increases in thickness (volume) downgradient toward the Delaware River, dilution by inflow from the underlying bedrock aquifer, and (or) incomplete characterization of the aquifer by ground-water samples collected from zones not affected by septic contamination.

Concentrations of chloride and nitrate in ground water apparently are not related to depth to water below land surface or well depth in the glacial aquifer. Low nitrate concentrations (less than 0.05 mg/L) were measured in water samples from wells with a depth to water ranging from 15 to 80 ft below land surface and from wells ranging from 20 to 226 ft in depth.

The concentrations of other dissolved nutrients, phosphate, and the reduced forms of nitrogen as ammonium, nitrite, and organic nitrogen, generally were lower than nitrate concentrations and detected in fewer ground-water samples (table 11). Phosphate concentrations up to 0.04 mg/L as P were measured in water samples only from well Pi-213 (sample also contained concentrations of nitrate of 5.1 mg/L as N and ammonium of 0.02 mg/L as N) and well Pi-464 (sample contained the only nitrite detected in ground water of 0.47 mg/L as N). Because background phosphate concentrations were below the reporting level of 0.01 mg/L as P, the detection of phosphate may indicate contamination of ground water by fertilizers or septic systems.

Ammonium was detected in ground water from only 6 of 24 wells sampled; the concentrations of ammonium in the 6 samples ranged from 0.01 to 0.19 mg/L as N. The maximum ammonium concentration, 0.19 mg/L as N, was measured in a water sample from well Pi-483, which is completed in the Mahantango shale bedrock aquifer. In the glacial aquifer, ammonium up to 0.04 mg/L as N was detected in water samples from three wells (Pi-481, Pi-482, and Pi-463) that are completed in the outwash deposits, and in two wells (Pi-213 and Pi-473) that are completed in the kame-terrace deposits. Except for wells Pi-473 and Pi-483, the depth to water in wells with detectable ammonium concentrations in water samples was less than 30 ft. Contamination by septic systems or fertilizer is possible where ammonium was detected in ground-water samples. Ground-water samples with detectable ammonium concentrations generally were from wells near Route 209 or in areas of relatively high septic-system density, such as Matamoras Borough or the Roberts Lane area.

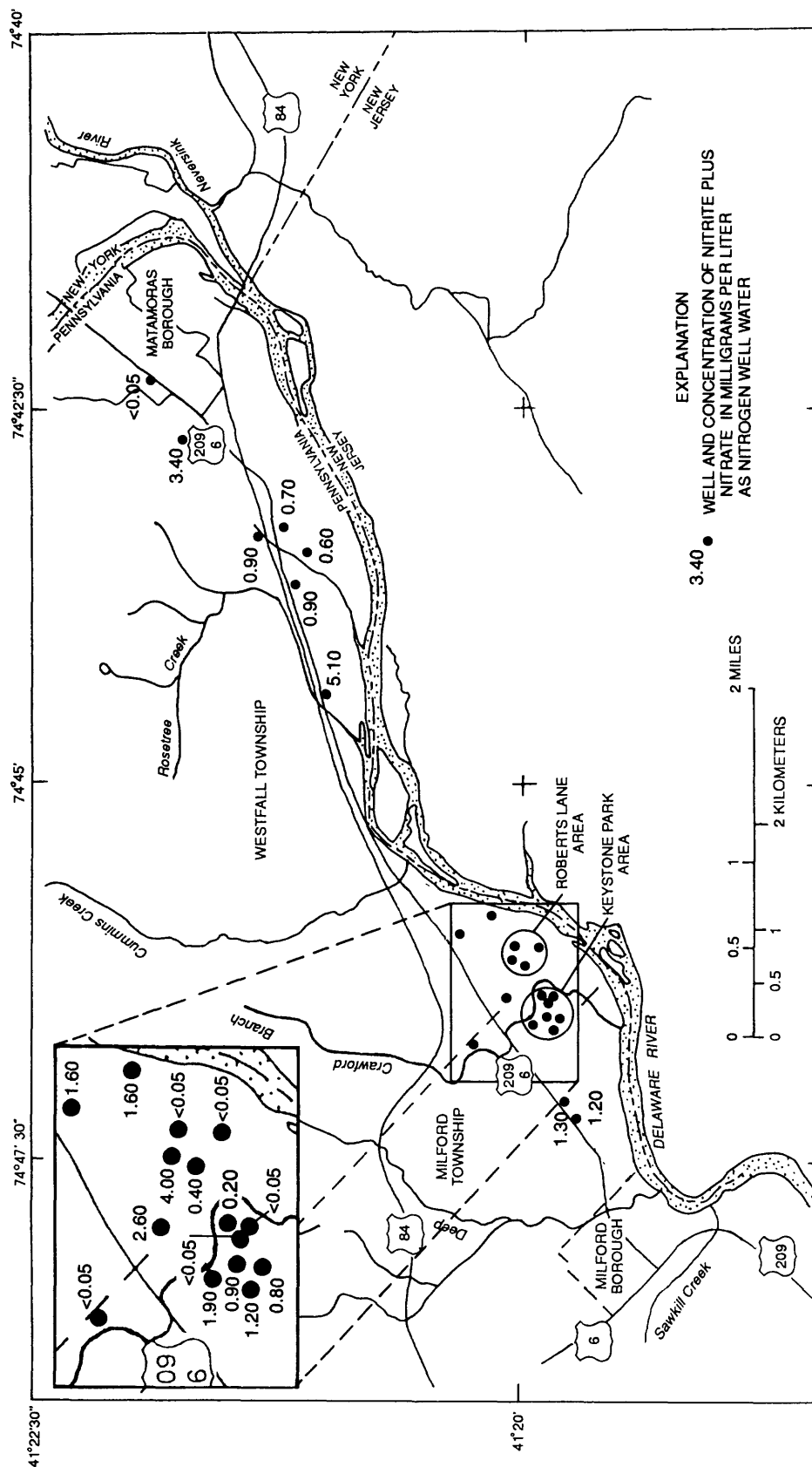


Figure 8. Concentrations of nitrite plus nitrate in ground-water samples from 24 wells, October 1991.

Relation to Other Chemical Constituents

The relations between chloride, nitrate, and the other nutrients to some other chemical constituents in ground water can provide information about sources contributing to and chemical processes controlling ground-water quality. In the study area, ground water that probably is contaminated by both road salt and septic-system effluent (such as the sample from well Pi-463) contains greater concentrations of calcium, sodium, magnesium, sulfate, chloride, alkalinity, and ammonium and lower concentrations of nitrate and dissolved oxygen than ground water from less contaminated or uncontaminated upgradient areas (such as samples from wells Pi-485 and Pi-456). Concentrations greater than the background amounts for calcium, sodium, potassium, alkalinity, chloride, nitrate, and dissolved organic carbon and concentrations lower than the background amounts for dissolved oxygen have been observed in ground water affected by septic-system effluent in unconfined sand aquifers in Canada (Robertson and others, 1991).

The strong association between chloride concentrations and sodium concentrations in ground-water samples (fig. 9) from the glacial aquifer near highways indicates that road salt (NaCl) is a major source of chloride in the ground water. In the Roberts Lane area, elevated concentrations of chloride in well-water samples also are associated with elevated concentrations of calcium, indicating that calcium chloride (CaCl₂) was a component in the industrial waste water discharged to the aquifer.

Concentrations of nitrate are positively correlated with concentrations of dissolved oxygen in ground water in the glacial aquifer (fig. 10) because nitrate is stable in an oxidizing environment but is unstable in a low-oxygen or reducing environment. Water from wells downgradient from areas of relatively greater septic-system density (Keystone Park, Roberts Lane, and Matamoras borough, plate 2) contained low concentrations of dissolved oxygen and nitrate, indicating reducing conditions in the aquifer. Bacteria can reduce nitrate through denitrification after oxygen concentrations are depleted if there is a carbon source in ground water. Further studies are needed to determine if denitrification is occurring.

Elevated alkalinities and pH, and elevated concentrations of calcium, sulfate, and manganese also were measured in water samples from most of the wells downgradient of septic systems. The elevated concentrations of these ions could be caused by mineral weathering along a ground-water flow path, a change in aquifer reduction-oxidation conditions, and (or) inflow from the underlying bedrock aquifer. Calcite (CaCO₃), gypsum (CaSO₄), and calcium-bearing feldspar could contribute calcium, bicarbonate alkalinity, or sulfate to the ground water. Dissolved calcium and sulfate also could be from septic-system effluent. Calcite dissolution, feldspar weathering, and denitrification can increase the pH of ground water. Reduction of manganese oxides also results in increases in pH and concentrations of dissolved manganese.

Dissolved manganese in ground water may reflect reducing conditions. Elevated manganese concentrations are associated with low concentrations of dissolved oxygen (fig. 11). Manganese in minerals can be reduced and dissolved in ground water through bacterially mediated reactions after available oxygen and nitrate have been consumed (Stumm and Morgan, 1980). Elevated dissolved manganese concentrations are associated with low concentrations of nitrate in ground water (fig. 12). Thus, elevated dissolved manganese concentrations in ground water also may indicate reducing conditions favoring denitrification.

Ground-water inflow from the underlying shale also may contribute to the differences in water chemistry between the upgradient and downgradient wells. For example, the piper diagram (fig. 3) shows that the ground water in the bedrock aquifer contains relatively more sodium and bicarbonate than the ground water in the glacial aquifer. However, the water sample from upgradient bedrock well Pi-483 contains low concentrations of nitrate and dissolved oxygen like the water samples from wells downgradient of septic systems. Therefore, processes more complex than mixing of inflow are required to explain entirely the water quality observed in the glacial aquifer.

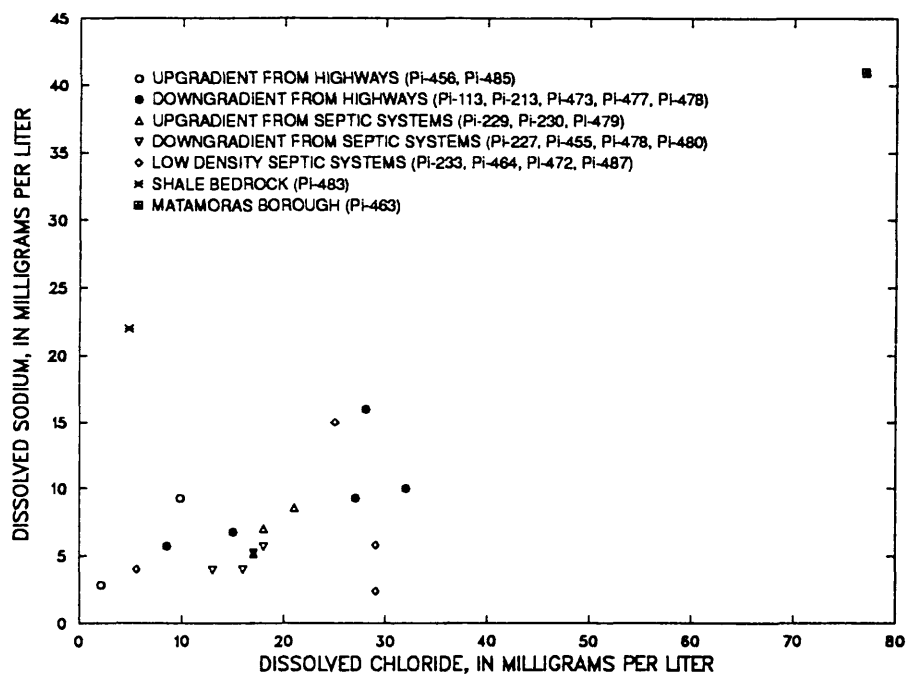


Figure 9. Relation between concentrations of dissolved sodium and dissolved chloride in water from 20 wells sampled in October 1991.

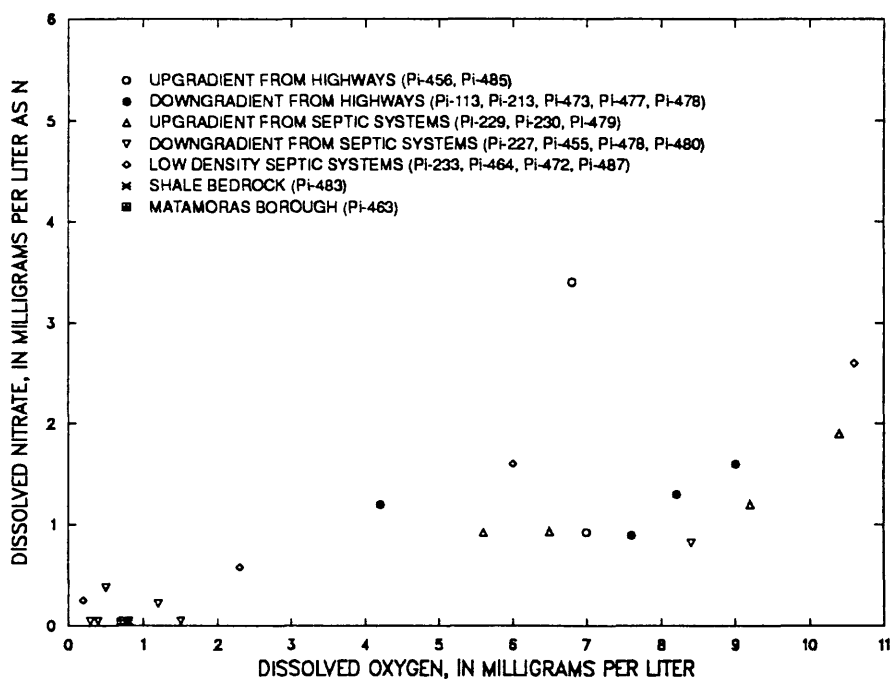


Figure 10. Relation between concentrations of dissolved nitrate and dissolved oxygen in water from 24 wells sampled in October 1991.

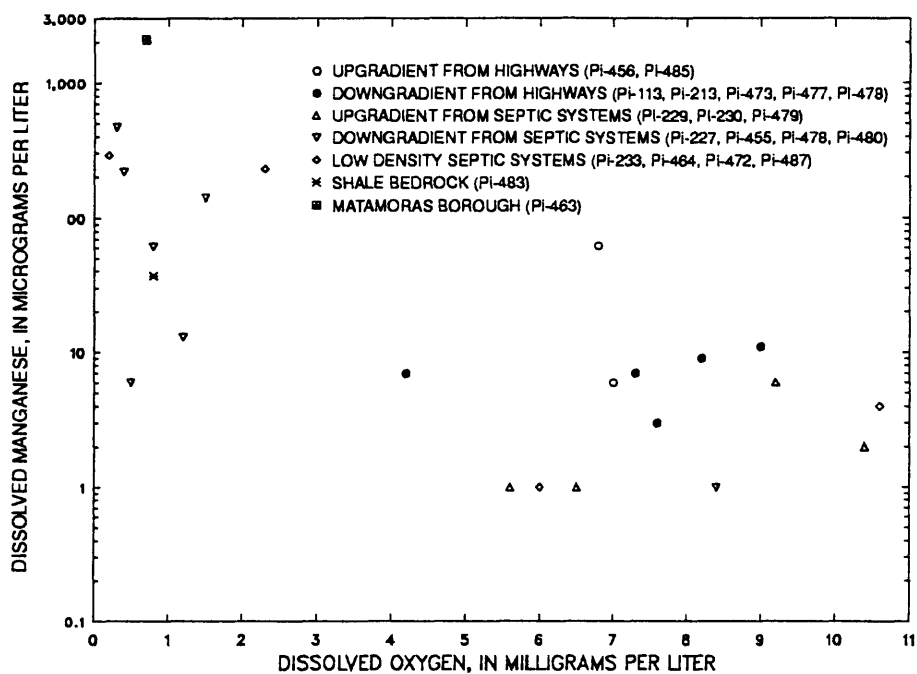


Figure 11. Relation between concentrations of dissolved manganese and dissolved oxygen in water from 24 wells sampled in October 1991.

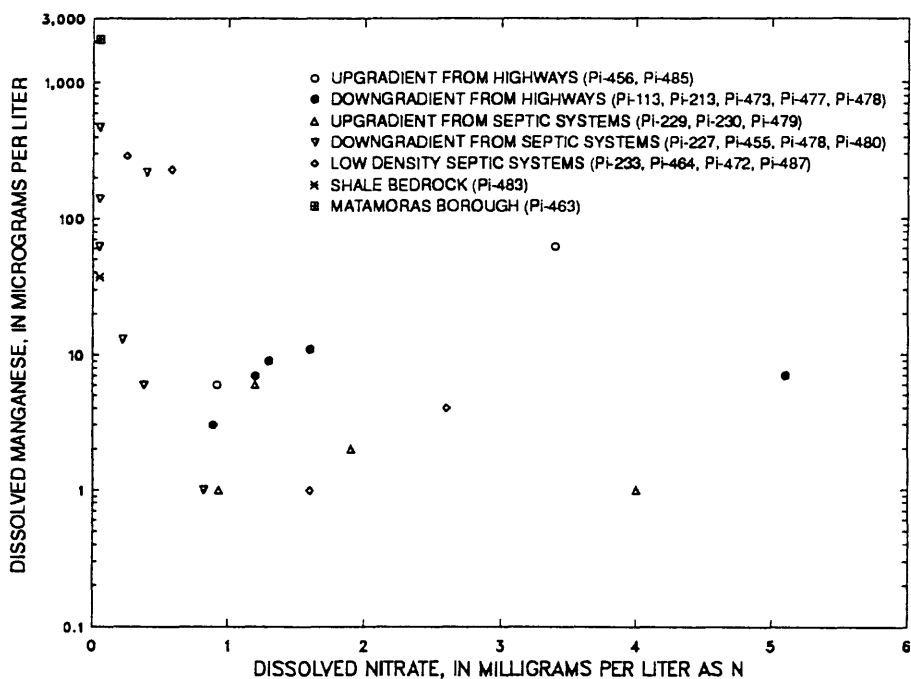


Figure 12. Relation between concentrations of dissolved manganese and dissolved nitrate in water from 24 wells sampled in October 1991.

Nitrogen and Chloride Loading to Ground Water

The average annual contributions of nitrogen and chloride from atmospheric precipitation, road runoff, and septic systems to ground water in the study area were calculated by use of a mass-balance, conservative-ion approach. The average concentration of nitrate and chloride in recharge was estimated on an annual basis. This approach assumes average annual loads, average annual recharge rates, steady-state conditions, conservative ions (no loss of mass), and a well-mixed, uniform ground-water system. It does not account for chemical reactions, transient behavior, variable recharge and input concentrations, preferential flow paths or incomplete mixing in the aquifer, or ground-water withdrawals and effects of pumping on solute transport in the aquifer.

The mass-balance loading approach is useful for estimating the possible consequences of inputs to the aquifer. For example, the mass-balance approach for nitrate loading to glacial aquifers in Massachusetts was used by Frimpter and others (1990) and Nelson and others (1988) to predict future potential nitrogen concentrations in supply wells in areas with septic systems. Knowledge of input loads is important in restricted glacial valley aquifer systems because all loads to the recharge area of the aquifer can infiltrate and affect the quality of ground water withdrawn by a pumping well (fig. 13).

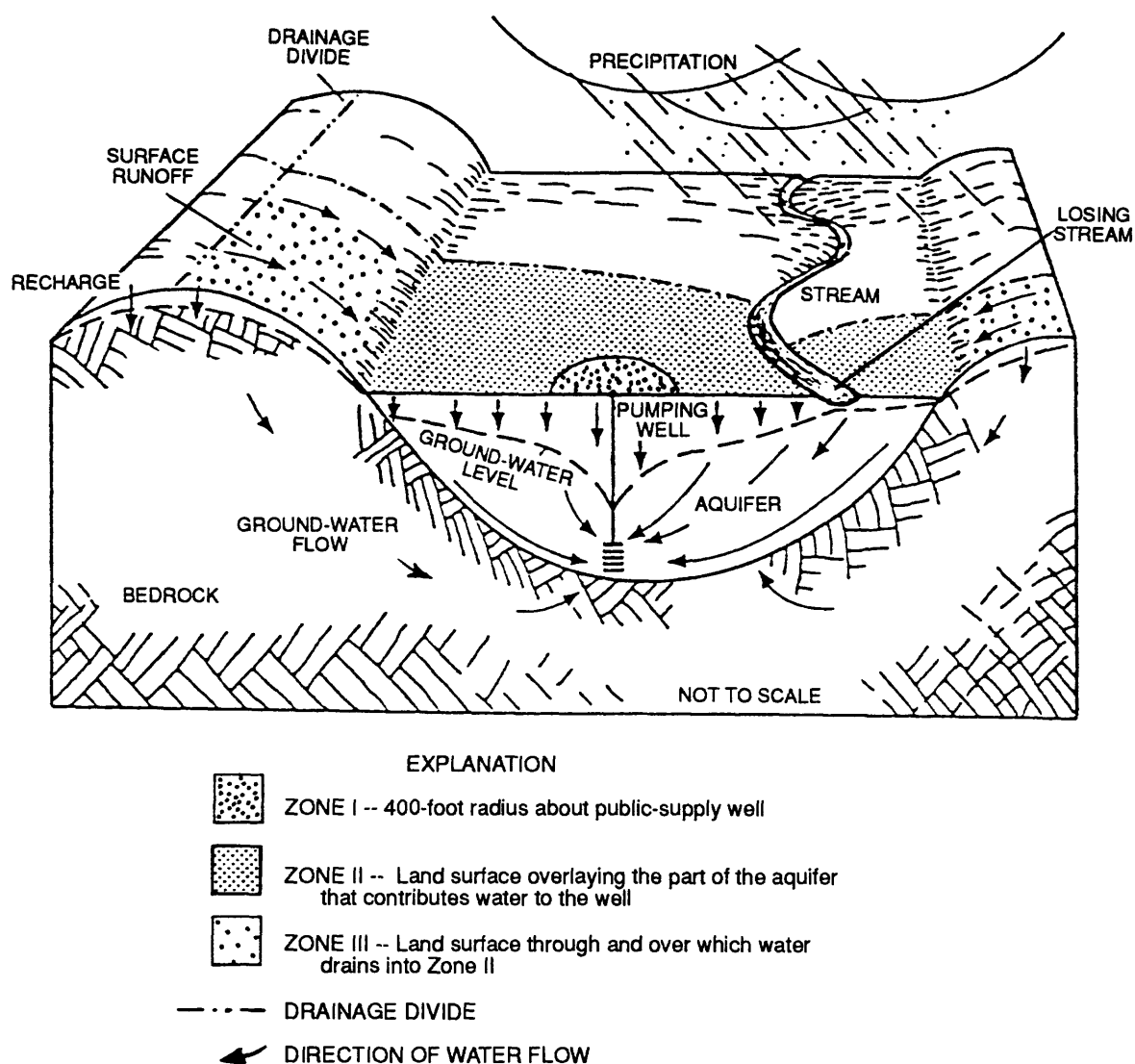


Figure 13. Contributing sources and recharge area of a valley fill aquifer that supplies a well.

The contributions of nitrogen and chloride to the loading of the aquifer were computed for three sources: septic-tank effluent, precipitation, and road runoff. Nitrogen from all sources was assumed to be converted to nitrate. For the septic-tank effluent, mass flux was calculated from the products of average flow of 65 (gal/person)/d, average concentrations for nitrate of 40 mg/L as N and for chloride of 50 mg/L, and population. Populations were provided as the number of people in the boroughs of Milford and Matamoras and the number of housing units in Milford and Westfall Townships multiplied by 2.6 persons per housing unit (Peter Wulforth, Pike Co. Planning Commission, written and oral commun., 1991 and 1992). For precipitation, contributions of chloride and nitrate as nitrogen from precipitation were taken from the average concentrations for 1982-1991 at Milford (Lynch and others, 1992), assuming a 15 percent loss of nitrogen and chloride to runoff. For road runoff, road salt (NaCl) as a source of chloride to the aquifer was calculated by use of an average annual application of 2 tons per lane mile multiplied by the number of lane miles of state-maintained highways in each area. No additional nitrogen was added from road runoff.

The average annual concentration of chloride and nitrate in water recharging the glacial aquifer was calculated by use of the equation:

$$C_r = M_s/V_r, \quad (4)$$

where C_r is concentration of chloride or nitrate in recharge,

M_s is mass of net sum of loads from chloride and nitrogen sources per year, and

V_r is volume of recharge per year.

Loads were calculated for six sub-areas: (1) Milford Township; (2) Westfall Township; (3) Milford Borough; (4) Matamoras Borough; (5) a hypothetical, but representative, residential half-acre lot; and (6) a hypothetical, but representative, residential 1-acre lot. The land area within the present and future National Park Service boundary was not included in the load calculations.

Land area, annual recharge volume, and population density for each of the six areas are given in table 6. The calculated concentrations and total loads of chloride and nitrate in recharge for the six areas are given in tables 7 and 8. Greater concentrations of nitrate and chloride are predicted for ground water in the boroughs of Milford and Matamoras than in the townships of Milford and Westfall because of differences in population and road density. At the estimated loading levels, septic-system effluent is a greater source of nitrogen and chloride than precipitation and road salt in the boroughs of Matamoras and Milford. However, in the less densely populated townships of Westfall and Milford, the loading calculations indicate that precipitation and road salt are greater sources of chloride than septic systems, although most of the calculated nitrogen load in Milford and Westfall Townships is from septic systems.

The concentrations of chloride measured in the ground-water samples from 24 wells are similar in magnitude to the concentrations calculated by use of the average inputs. Chloride concentrations calculated for the hypothetical half-acre and 1-acre-lot areas (table 7) fall within the range observed at Keystone Park (fig. 7), a residential area of half-acre to 1-acre lots that was built in the late 1970's and early 1980's. However, the concentration of chloride in water from well Pi-463 in Matamoras is 77 mg/L (table 10), about twice that calculated for recharge water in the borough by use of estimated average inputs (table 7). The median chloride concentration for 18 samples from the glacial aquifer in Westfall and Milford Townships was 17.5 mg/L (table 1), which is about two times the concentration calculated for these townships. The difference between measured ground-water concentrations and calculated concentrations in recharge may be caused by transient behavior or by greater amounts of road salting, consumption of ground water, and (or) less recharge than used in the calculations. The calculated concentrations are spatially averaged and do not account for differences in the spatial distribution of sources and recharge rates.

The concentrations of nitrate measured in ground water are much less than calculated concentrations in recharge by use of a conservative-ion, mass balance approach. Much of the nitrogen load from precipitation may be lost in runoff or taken up by plants before infiltration into the ground water. Also, nitrate probably is not chemically conservative in ground water and may be lost by denitrification or retained on aquifer materials in a reduced form (ammonium) in areas where the ground-water system is anoxic. Nevertheless, the mass balance model is useful for predicting the maximum possible nitrate concentration from estimated loads. Only in the hypothetical half-acre lot do the calculated loading rates

for nitrate result in recharge water concentrations that exceed the USEPA MCL of 10 mg/L as N in drinking water. The calculations also indicate that elevated nitrate concentrations in ground water are possible in the boroughs of Milford and Matamoras. If septic-tank density increases and recharge is reduced, elevated nitrate concentrations in ground water could occur in Westfall and Milford Townships. Studies of the effect of septic-tank density on ground water have determined minimum lot sizes for residential lots to protect ground-water quality (Yates, 1985; Perkins, 1984). Areas with a septic-tank density greater than 40 systems per square mile (one per 16 acres) have been designated as regions of potential ground-water contamination by the USEPA (Yates, 1985).

The assumption that ground-water concentrations of chloride and nitrate will eventually equal the total net recharge concentrations calculated in tables 7 and 8 assumes little or no inflow from other aquifers or sources. Average ground-water concentrations initially will differ from the recharge concentrations because of the delay in time for infiltration, ground-water flow, and mixing. However, the entire aquifer eventually could be affected by recharge containing nitrate and chloride from septic systems and road salting. For a steady input concentration in a well-mixed ground-water system, the length of time until average ground-water concentrations equal the recharge concentrations can be calculated as the chemical response time (Gelhar and Wilson, 1974). The chemical response time, t_c , is calculated by use of the equation:

$$t_c = nh/e, \quad (5)$$

where n is effective porosity;

h is saturated thickness, in feet; and

e is recharge rate, in feet per year.

For the glacial aquifer, which has a saturated thickness of about 80 to 100 ft, an estimated porosity of 0.30 to 0.35, and an average recharge rate of 20 in. per year, the chemical response time ranges from about 14 to 21 years. In the study area, calculated concentrations of chloride in recharge are close to measured concentrations of chloride in ground water from wells in Keystone Park, a residential development about 10 to 14 years of age.

Three wells (Pi-113, Pi-213, and Pi-230) sampled by the USGS during 1981-82 were resampled in October 1991 to assess possible changes in water quality (table 10). The concentrations of chloride increased by as much as 100 percent in water samples from all three wells during this 9 to 10-year period. Concentrations of sodium, calcium, magnesium, and potassium also increased over time, but nitrate concentrations are similar in samples from both times. The increase in chloride, a relatively conservative ion in ground water, suggests that ground-water quality has changed in the 9 or 10 years between samples. The change may reflect the delay between inputs and observed ground-water concentrations (chemical response time), increases in inputs, increased ground-water pumpage, or decreased ground-water recharge caused by urbanization or periods of low precipitation.

Differences in measured concentrations of chloride and other constituents in only three samples do not indicate a statistically significant trend in water quality, and data to assess annual and seasonal variability are not available. A water-quality sampling network to collect ground-water samples in different seasons over a number of years would be necessary to assess seasonal and annual variability and long-term changes in water quality. Monthly sampling probably is sufficient to assess seasonal variability in ground-water quality, and sampling for several years is required to assess annual variability and long-term change in ground-water quality. Ground-water quality networks could be used to describe typical ground-water quality in areas of the glacial aquifer affected by different rates of chloride and nutrient loading. The number of wells, sampling frequency, methods of sample collection and analysis, and sample density need to be considered in design of networks, especially if statistically significant results are desired (Spruill, 1990).

Table 6. Estimated annual recharge and population density for Milford and Matamoras Boroughs, Milford and Westfall Townships, and two hypothetical lot areas

[mi², square mile; in/yr, inch per year; L/yr, liter per year]

	Milford Borough	Matamoras Borough	Milford Township	Westfall Township	One- acre lot	Half- acre lot
Land area (mi ²)	0.43	0.65	0.32	1.44	0.001553	0.000776
Recharge						
Rate (in/yr)	20	20	20	20	20	20
Volume (× 10 ⁸ L/yr)	5.7	8.6	4.2	19	.02	.01
Population density						
Housing units	564	921	65	515	1	1
Number of persons	1,064	1,934	169	1,339	2.6	2.6
Persons per mi ²	2,474	2,975	528	930	1,674	3,349
Persons per acre	3.8	4.6	.8	1.4	2.6	5.2

Table 7. Estimated contribution of chloride from septic systems, road salt, and precipitation in average annual recharge to the glacial aquifer

[L/yr, liter per year; mg/L, milligram per liter; Cl, chloride; NaCl, sodium chloride; kg/yr, kilogram per year; (ton/mi)/yr, ton per mile per year; mi, mile; lb/mi², pound per square mile]

	Milford Borough	Matamoras Borough	Milford Township	Westfall Township	One- acre lot	Half- acre lot
Contribution from septic systems						
Housing units	564	921	65	515	1	1
Number of people	1,064	1,934	169	1,339	2.6	2.6
Septic outflow (× 10 ⁷ L/yr)	9.6	17	1.5	12	.023	.023
Cl concentration (mg/L)	50	50	50	50	50	50
Cl load (× 10 ³ kg/yr)	4.8	8.7	.76	6.0	.012	.012
Contribution from road salt (NaCl)						
Application rate [(ton/mi)/yr]	2	2	2	2	2	2
Road miles (mi)	5	6	2	6.5	.02	.01
Cl load (× 10 ³ kg/yr)	5.5	6.6	2.2	7.1	.022	.011
Net ¹ load (× 10 ³ kg/yr)	4.7	5.6	1.9	6.1	.019	.009
Contribution from precipitation						
Cl concentration (mg/L)	.258	.258	.258	.258	.258	.258
Cl load (× 10 ³ kg/yr)	.32	.49	.24	1.1	.0012	.0006
Net ¹ load (× 10 ³ kg/yr)	.27	.41	.20	.91	.0010	.0005
Total net contribution						
Cl load (× 10 ³ kg/yr)	9.8	15	2.9	13	.031	.021
Load/area (× 10 ³ lb/mi ²)	51	51	19	20	44	62
Percent septic ²	49	59	27	46	37	54
Recharge concentration (mg/L)	17	17	6.7	6.8	15	21

¹ Net load is calculated by assuming that 15 percent of chloride mass is lost to surface runoff and does not enter recharge.

² Percent septic is the percent of total load of chloride in recharge that is contributed by septic systems.

Table 8. *Estimated contribution of nitrate from septic systems, precipitation, and road runoff in average annual recharge to the glacial aquifer*

[in/yr, inch per year; L/yr, liter per year; NO₃, nitrate; mg/L as N, milligram per liter as nitrogen; kg/yr as N, kilogram per year as nitrogen; lb/mi², pound per square mile]

	Milford Borough	Matamoras Borough	Milford Township	Westfall Township	One acre lot	Half- acre lot
Contribution from septic systems						
Housing units	564	921	65	515	1	1
Number of people	1,064	1,934	169	1,339	2.6	2.6
Septic outflow (× 10 ⁷ L/yr)	9.6	17	1.5	12	.023	.023
NO ₃ concentration (mg/L as N)	40	40	40	40	40	40
NO ₃ load (× 10 ³ kg/yr as N)	3.8	6.9	.61	4.8	.0093	.0093
Contribution from precipitation and runoff						
Volume (in/yr)	44	44	44	44	44	44
NO ₃ concentration (mg/L as N)	.58	.58	.58	.58	.58	.58
NO ₃ load (× 10 ³ kg/yr as N)	.73	1.1	.55	2.4	.0026	.0013
Net load ¹ (× 10 ³ kg/yr as N)	.61	.93	.46	2.1	.0022	.0011
Total contribution						
NO ₃ load (× 10 ³ kg/yr as N)	4.4	7.9	1.1	6.9	.012	.010
Load/area (× 10 ³ lb/mi ²)	22	26	7.3	11	16	29
Percent septic ²	86	88	57	70	81	89
Recharge concentration (mg/L as N)	7.8	9.2	2.5	3.6	5.6	10.2

¹ Net load is calculated by assuming that 15 percent of nitrogen mass is lost to surface runoff and does not enter recharge.

² Percent septic is the percent of the total load of nitrogen that is contributed by septic systems.

SUMMARY

The glacial aquifer that underlies the Routes 209 and 6 corridor between Milford and Matamoras, Pa., is one of the most productive in Pike County. The aquifer is comprised of unconsolidated glacial outwash and kame-terrace deposits that lie within a glacially-carved valley now occupied by the Delaware River. Ground-water quality in the glacial aquifer is threatened because of contamination by salts and nutrients from septic systems and salts in road-way runoff. Most businesses and residences along this narrow, 7-mi-long corridor rely on individual wells for water supply and septic systems for waste-water disposal.

Water levels were measured in 29 wells in August 1991 to determine the direction and gradient of ground-water flow. Generally, ground water flows from the edges of the valley toward the Delaware River; near the center of the valley, ground water flows nearly parallel to the Delaware River in the downstream direction. Depth to water below land surface is as great as 90 ft in the area underlain by kame terraces near Milford and as little as 13 ft in the area underlain by glacial outwash near Matamoras. Ground-water levels in the Matamoras area may be sustained in part by river inflow. The gradient of ground-water flow in the glacial aquifer ranges from 10 to 20 ft/mi, reflecting differences in aquifer transmissivities across the area.

Ground-water samples were collected in October 1991 from 23 wells completed in the glacial aquifer and from 1 well completed in the bedrock aquifer to determine concentrations of dissolved nutrients and chloride in ground water in residential areas and near highways and to provide baseline data for future water-quality assessments. Concentrations of nitrate in some ground-water samples and of chloride in most ground-water samples were elevated above estimated natural background concentrations.

The median concentration of nitrate was about 1.1 mg/L as N, which is close to the estimated background concentration of 1 mg/L as N. The maximum concentration of nitrate measured was 5.1 mg/L as N. Concentrations of the other nitrogen species and phosphate generally were low or below level of reporting, suggesting limited contamination of ground water by nutrients from septic systems or fertilizers applied in areas sampled. No water samples contained nitrate in concentrations that exceeded the USEPA MCL of 10 mg/L as N. Low concentrations of nitrate (near or less than 0.5 mg/L as N) were measured in ground-water samples with low concentrations of dissolved oxygen (less than 1 mg/L), suggesting that nitrate may be removed from ground water through denitrification or nitrate reduction. Elevated concentrations of dissolved manganese were associated with low concentrations of dissolved oxygen and nitrate in ground-water samples. Six of the 23 ground-water samples contained concentrations of dissolved manganese that exceeded the USEPA SMCL of 50 µg/L.

Concentrations of chloride in ground water near highways and in residential areas were greater than the estimated background concentration of about 3 mg/L, suggesting contamination of ground water by salts from road salting and/or septic systems. The maximum concentration of chloride measured in a ground-water sample was 680 mg/L; the sample was collected from a well in an area where ground water may be locally contaminated by industrial waste. Water from three of four wells sampled in that area contained concentrations of chloride that exceeded the USEPA SMCL of 250 mg/L. For 18 wells not known to be affected by industrial contamination, the median concentration of chloride was 17.5 mg/L.

Storm water, parking-lot runoff, and precipitation were sampled at three sites in November 1991 over a period of 3 days of rain (cumulative rainfall was 3.22 in.) to help assess the load of nitrogen and chloride from these sources of recharge. The concentration of chloride was greater in road runoff than in precipitation. The concentrations of nitrogen in runoff samples were similar to those in precipitation samples (about 1 mg/L as N for total nitrogen species). Because only a single storm in late autumn was sampled for chemical analysis, the results do not represent annual averages. The single sampling also does not account for seasonality, such as road salting in winter.

The total loads of nitrogen and chloride from septic systems and road runoff (including parking lots) were calculated by use of data obtained from the literature for septic-system contributions and from the Pike County Planning Commission for population and roadway coverage for six areas with different population densities. Estimated total annual loads to ground water in the glacial aquifer ranged from 20,000 to 51,000 lb/mi² for chloride, and from 7,300 to 29,000 lb/mi² for nitrogen; estimated average concentrations in recharge (20 in. per year) to the glacial aquifer ranged from 6.7 to 21 mg/L for chloride and 3.6 to 10 mg/L for nitrogen. Road salt is a greater source of chloride than septic systems in Westfall and Milford Townships. The average concentrations of chloride and nitrate in ground water that would result from estimated average loading, assuming conservative behavior, do not exceed the USEPA SMCL for chloride but approach the USEPA MCL for nitrate. Chloride is a conservative ion, but nitrate does not appear to be conservative in ground water in the study area because of anoxic conditions that render the nitrate unstable. Present ground-water concentrations reflect past input loads from upgradient sources; the chemical response lag time for ground water concentrations to equal recharge concentrations is estimated to be 14 to 21 years.

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Table 9. Record of wells

USGS well number: local well number preceded by county identification prefix Pi for Pike County.

Township or borough: Name refers to township (Twp) or borough (Boro).

Driller license number: -231, William Holden; -229, E. J. Shields; 0145, William E. Martin; 0425, Alan J. Greening; 0999, Jack Ziegler; 1020, H. C. Horton; 1073, Douglas H. Steele; 1123, Peter A. Kestler; 1134, Bruce Steele; 1590, Dave Weber, Jr.; 1736, Dunn & Dunn, Inc.

Use of site: U, unused; W, withdrawal.

Use of water: C, commercial; H, domestic; I, irrigation; P, public supply; T, institutional; U, unused; Z, other.

Hydrogeologic unit: 344MNNG, Mahantango Formation; 344MRCL, Marcellus Shale; 112KMTC, kame-terrace deposits; 112OTSH, outwash.

gal/min: gallon per minute.

(gal/min)/ft: gallon per minute per foot.

Elevation of land surface is estimated from topographic maps. Elevation is in feet above sea level.

Water level is in feet below land surface; * denotes recent pumping in well.

Table 9. Record of wells—Continued

USGS well number	Location		Township or borough	Owner	Driller license number	Year drilled	Primary		Elevatio of land surface (feet)	Hydro geologic unit
	Latitude (degrees)	Longitude (degrees)					Use of site	Use of water		
Pi-94	412229	0744234	Matamoras Boro	Matamoras Municiple Auth.	-231	—	W	P	480	344MNNG
112	412101	0744412	Westfall Township	Delaware Valley High Sch.	—	1956	U	U	430	344MRCL
113	412110	0744340	Westfall Township	Mclaughlin	-229	—	W	H	420	112OTSH
213	412100	0744423	Westfall Township	Delaware Valley High Sch.	0245	1969	W	T	430	112KMTC
223	412149	0744228	Westfall Township	Matamoras Municiple Auth.	0245	1981	W	P	430	112OTSH
225	412100	0744428	Westfall Township	Mil. Val. Conval.	0245	1979	W	T	420	112KMTC
227	411955	0744629	Westfall Township	Willis, Charles M.	1123	1976	W	H	470	112KMTC
229	411955	0744640	Milford Township	Shishak, Ayro	0425	1972	W	H	470	112KMTC
230	411956	0744636	Westfall Township	Jenkins, Norman	0245	1979	W	H	470	112KMTC
232	412003	0744605	Westfall Township	Albanese, N.	1134	1971	W	H	410	112OTSH
233	412009	0744554	Westfall Township	Gruehr, R.A.	1134	1972	W	H	440	112KMTC
238	412120	0744204	Westfall Township	Bonnell, G.A.	0245	1976	W	H	420	344MRCL
271	412234	0744224	Westfall Township	Matamoras Municiple Auth.	0245	1977	W	P	420	112OTSH
348	412052	0744456	Westfall Township	Kettner, Emil	0245	1981	W	H	430	344MNNG
452	411954	0744639	Westfall Township	Orben	—	1979	W	H	474	112KMTC
453	411954	0744642	Milford Township	Burke	0145	1984	U	H	478	344MNNG
454	411954	0744642	Milford Township	Burke	1123	1987	W	H	478	112KMTC
455	411954	0744627	Westfall Township	Butler, Joyce	1020	1985	W	H	475	112KMTC
456	412146	0744244	Westfall Township	Ewbank, Gale	1590	1986	W	C	441	112OTSH
457	412122	0744340	Westfall Township	Peters, Keith	—	—	W	H	465	—
458	412130	0744305	Westfall Township	Brown, Albert	—	1990	U	H	419	112OTSH
459	412130	0744305	Westfall Township	Brown, A.	1590	1988	W	H	419	112OTSH
460	412148	0744232	Westfall Township	Miller	—	—	W	—	424	112OTSH
461	412121	0744202	Westfall Township	Shatt	—	—	W	—	426	—
462	412128	0744216	Westfall Township	Morales, Celestino	0999	1989	W	—	426	344MRCL
463	412154	0744220	Matamoras Boro	Whipples Bldg. Materials	0245	1985	W	Z	425	—
464	412114	0744318	Westfall Township	Farm Plus	1123	1984	W	I	430	112OTSH
465	412106	0744254	Westfall Township	Hunt, Michael	1123	1987	W	—	423	—
466	412109	0744304	Westfall Township	Hunt, Michael	—	—	U	U	421	—
468	411948	0744719	Milford Township	Swanson, Kathy	1073	1977	W	H	510	344MNNG

Table 9. Record of wells—Continued

Depth of well (feet)	Casing		Depth to water-bearing zone(s) (feet)	Water level (feet)	Date water level measured	Reported yield (gal/min)	Measured yield			USGS well number
	Depth (feet)	Diameter (inches)					Specific capacity [(gal/min)/ft]	Discharge (gal/min)	Pumpin period (hours)	
304	30	8	—	30.00	08-09-30	140	—	—	—	Pi-94
				23.00	08-07-91					
918	193	6	—	30.20	08-07-91	—	—	—	—	112
68	68	6	—	—	—	30	—	—	—	113
99	84	10	84/93/97	30.00	05-15-69	—	8.5	350	24.0	213
						—	—	—	—	
84	72	8	73	24.00	06-09-81	—	33	350	24	223
				30.00	08-07-91					
114	114	6	106	26.00	10-11-76	—	3.2	110	3.0	225
				26.35	08-08-91					
214	214	6	—	95.00	09-01-76	—	.27	20	5.0	227
				96.90	10-22-91					
200	200	6	—	46.00	07-23-72	—	.39	30	3.0	229
				94.12	10-22-91					
191	191	6	—	81.00	07-15-82	—	.51	20	3.0	230
				85.95	10-21-91					
130	130	6	—	40.00	06-07-71	—	.75	15	2.0	232
145	145	6	—	75.00	09-16-72	—	1.3	20	2.0	233
1200	81	6	124	20.00	01-31-76	—	.01	3	2.0	238
				20.15	08-07-91					
78	59	6	59	31.00	07-27-77	175	—	—	24.0	271
				33.00	08-07-91	10	175	24		
225	125	6	166/204	31.40	07-14-82	—	.07	15	2.0	348
				23.92	08-08-91					
—	—	—	—	90.10	08-06-91	—	—	—	—	452
890	226	6	812/831/848	90.90	08-06-91	—	.01	8	4	453
147	147	6	147	94.10	08-06-91	20	—	—	—	454
226	226	6	226	85.00	08-06-91	25	—	—	—	455
				90.10	10-21-91					
140	140	—	140	13.00	10-30-91	—	.81	100	1	456
				23.10*	08-06-91					
—	—	—	—	61.30	08-06-91	—	—	—	—	457
250	—	—	—	9.65	08-06-91	—	—	—	—	458
275	232	6	—	10.20	08-06-91	2	—	—	—	459
23	20	1	20	13.10	08-06-91	—	—	—	—	460
—	—	—	—	20.10	08-07-91	—	—	—	—	461
423	142	6	418	18.70	08-07-91	10	—	—	—	462
—	—	—	—	16.00	08-07-91	—	—	—	—	463
				16.10	10-29-91					
142	142	6	142	19.20	10-29-91	—	.20	25	2	464
				49.48*	08-07-91					
340	40	6	—	45.60	08-07-91	—	.32	45	17	465
						30	—		—	
—	—	—	—	15.50	08-07-91	—	—	—	—	466
125	42	6	100/115	61.00	08-07-91	—	.08	6	2	468

Table 9. Record of wells—Continued

USGS well number	Location		Township or borough	Owner	Driller license number	Year drilled	Primary		Elevatio of land surface (feet)	Hydro geologic unit
	Latitude (degrees)	Longitude (degrees)					Use of site	Use of water		
Pi-469	411941	0744731	Milford Township	Santos	1134	1985	W	H	515	344MNNG
470	412029	0744555	Westfall Township	National Park Service	—	—	U	U	487	—
471	411937	0744725	Milford Township	Latwig	1123	1988	W	H	480	112KMTC
472	412006	0744628	Milford Township	Petinko, John	—	1984	W	H	477	112KMTC
473	411943	0744716	Milford Township	Milford Township	1134	1987	W	H	468	112OTSH
474	412108	0744327	Westfall Township	Schueler	—	1990	U	U	419	112OTSH
475	412002	0744606	Westfall Township	National Park Service	—	1978	W	H	422	112OTSH
476	411954	0744551	Westfall Township	National Park Service	1134	1980	U	U	416	112OTSH
477	411947	0744710	Milford Township	Barnett	0245	1984	W	H	475	112KMTC
478	411953	0744636	Westfall Township	Onuschak, John	1590	1983	W	H	465	112KMTC
479	412000	0744638	Milford Township	Sotak	—	1985	W	H	475	112KMTC
480	411957	0744626	Westfall Township	Harrison, James	0245	1981	W	H	465	112KMTC
481	412003	0744604	Westfall Township	Wachtel	—	—	W	H	415	112OTSH
482	411956	0744605	Westfall Township	Aiosa, Joseph	1736	1980	W	H	410	112OTSH
483	412016	0744647	Milford Township	Mooney, Doris	—	—	W	H	515	—
484	412019	0744602	Westfall Township	Izatt, Ian	-229	1966	W	H	475	112KMTC
485	412122	0744322	Westfall Township	Strouse, Mark	—	1988	W	H	430	112OTSH
486	412004	0744609	Westfall Township	Sosiewicz, Stanley	1736	1978	W	H	—	112OTSH
487	412107	0744328	Westfall Township	Schueler	—	—	W	H	427	112OTSH

Table 9. Record of wells—Continued

Depth of well (feet)	Casing		Depth to water-bearing zone(s) (feet)	Water level (feet)	Date water level measured	Reported yield (gal/min)	Measured yield			USGS well number
	Depth (feet)	Diameter (inches)					Specific capacity [(gal/min)/ft]	Discharge (gal/min)	Pumpin period (hours)	
138	25	6	—	58.50	08-07-91	10	—	—	—	P1-469
—	—	—	—	92.30	08-09-91	—	—	—	—	470
120	120	6	120	74.80	08-08-91	30	—	—	—	471
121	121	6	121	90.00	08-08-91	10	—	—	—	472
113	113	6	113	89.10	10-31-91	25	—	—	2	473
				98.25*	08-06-91					
320	—	—	—	15.30	08-07-91	—	—	—	—	474
138	—	—	—	31.00	08-08-91	—	—	—	—	475
90	90	6	90	28.60	08-09-91	—	—	—	—	476
146	146	6	146	60.80	10-31-91	—	0.16	15	2.0	477
				55.56	08-09-91					
158	158	6	158	89.00	10-21-91	—	20	8	3	478
—	—	—	—	—	—	—	—	—	—	479
213	213	6	213	89.80	10-25-91	—	28	35	3	480
—	—	—	—	—	—	—	—	—	—	481
142	142	6	142	—	—	—	—	—	—	482
—	—	—	—	—	—	—	—	—	—	483
118	118	6	118	90.00	10-26-91	—	—	—	—	484
20	20	—	20	—	—	—	—	—	—	485
119	118	6	118	30.00	10-29-91	20	—	—	—	486
20	20	2	20	15.00	10-00-91	—	—	—	—	487

Table 10. Results of chemical analyses for nutrients, major ions, and selected metals for ground-water samples from 24 wells

[°C, degree Celsius; µS/cm, microsiemen per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; <, less than; --, no data]

Local well number	Date	Temperature (°C)	Specific conductance (µS/cm)	Oxygen, dissolved (mg/L)	pH, field (standard units)	Alkalinity, field (mg/L as CaCO ₃)	Nitrogen, ammonia, dissolved (mg/L as N)	Nitrogen, ammonia + organic, dissolved (mg/L as N)	Nitrogen, nitrite, dissolved (mg/L as N)
Pi-113	07-15-82	11.0	135	--	6.0	--	0.03	--	<0.01
	10-24-91	11.0	163	7.6	5.9	9	<.01	<0.20	<.01
213	08-10-81	22.0	260	--	7.1	--	.01	--	<.01
	10-31-91	10.5	258	7.3	5.7	22	.02	<.20	<.01
227	10-22-91	9.5	270	1.5	6.9	81	<.01	<.20	<.01
229	10-22-91	11.0	180	9.2	6.3	25	<.01	<.20	<.01
230	07-15-82	10.0	235	--	7.8	--	.03	--	<.01
	08-10-82	10.5	230	--	7.8	--	--	--	--
	10-21-91	11.0	254	6.5	6.8	87	<.01	<.20	<.01
232	10-29-91	10.5	--	.5	6.7	54	<.01	<.20	<.01
233	10-25-91	11.0	299	6.0	7.4	81	<.01	<.20	<.01
455	10-21-91	9.5	270	.8	7.3	75	<.01	<.20	<.01
456	10-30-91	11.5	151	6.8	6.0	24	<.01	<.20	<.01
463	10-29-91	13.5	800	.7	6.6	112	.04	<.20	<.01
464	10-29-91	11.5	185	.2	7.3	64	<.01	<.20	.47
472	10-30-91	11.0	183	10.6	6.4	26	<.01	<.20	<.01
473	10-31-91	11.5	190	4.2	6.2	48	.01	<.20	<.01
477	10-31-91	12.0	170	8.2	6.8	52	<.01	<.20	<.01
478	10-21-91	11.0	195	8.4	6.7	50	<.01	<.20	<.01
479	10-24-91	11.0	208	10.4	6.5	39	<.01	<.20	<.01
480	10-25-91	10.5	300	1.2	6.9	92	<.01	<.20	<.01
481	10-25-91	11.0	1,980	.3	7.0	31	.03	<.20	<.01
482	10-25-91	10.5	525	.4	7.4	84	.03	<.20	<.01
483	10-26-91	14.0	240	.8	7.4	108	.19	<.20	<.01
484	10-26-91	11.0	197	9.0	6.0	19	<.01	<.20	<.01
485	10-28-91	14.0	85	7.0	5.6	14	<.01	<.20	<.01
486	10-29-91	11.0	--	5.6	7.5	142	<.01	<.20	<.01
487	10-30-91	13.0	199	2.3	5.6	12	<.01	<.20	<.01

Table 10. Results of chemical analyses for nutrients, major ions, and selected metals or ground-water samples from 24 wells—Continued

[°C, degree Celsius; µS/cm, microsiemen per centimeter at 25 degrees Celsius; mg/L, milligram per liter; µg/L, microgram per liter; <, less than; --, no data]

Local well number	Date	Nitrogen, NO ₂ + NO ₃ , dissolved (mg/L as N)	Phosphorus, ortho, dissolved (mg/L)	Calcium, dissolved (mg/L)	Magnesium, dissolved (mg/L as Mg)	Sodium, dissolved (mg/L)	Potassium, dissolved (mg/L)	Chloride, dissolved (mg/L)	Sulfate, dissolved (mg/L)
Pi-113	07-15-82	0.83	<0.010	9.8	3.3	8.0	<0.10	16	13
	10-24-91	.89	<.010	11	3.5	10	.50	32	11
213	08-10-81	5.10	.010	11	3.4	42	.70	20	15
	10-31-91	5.10	.020	19	6.4	16	.90	28	39
227	10-22-91	<.05	<.010	42	5.7	4.0	.40	16	39
229	10-22-91	1.20	<.010	20	3.9	8.6	.50	21	19
230	07-15-82	.71	<.010	32	3.8	4.3	<.10	8.6	22
	08-10-82	--	--	--	--	--	--	--	--
	10-21-91	.93	<.010	38	4.2	5.2	.50	17	27
232	10-29-91	.38	<.010	210	28	170	1.6	680	53
233	10-25-91	1.60	<.010	45	5.5	5.8	.50	29	29
455	10-21-91	<.05	<.010	42	5.6	4.0	.40	13	37
456	10-30-91	3.40	<.010	12	3.6	9.3	.50	9.8	24
463	10-29-91	<.05	<.010	87	21	41	1.0	77	190
464	10-29-91	.72	.040	27	4.1	4.0	.50	5.6	29
472	10-30-91	2.60	<.010	14	4.0	15	.50	25	22
473	10-31-91	1.20	<.010	20	4.9	6.8	1.5	15	25
477	10-31-91	1.30	<.010	21	3.0	5.7	.50	8.5	18
478	10-21-91	.82	<.010	28	3.4	5.2	.40	17	21
479	10-24-91	1.90	<.010	26	3.7	7.0	.40	18	21
480	10-25-91	.22	<.010	47	6.2	5.7	.50	18	42
481	10-25-91	<.05	<.010	300	47	97	1.6	680	54
482	10-25-91	<.05	<.010	80	11	5.0	.60	120	33
483	10-26-91	<.05	<.010	20	5.7	22	.40	4.8	25
484	10-26-91	1.60	<.010	18	5.2	9.3	.60	27	25
485	10-28-91	.92	<.010	7.9	2.6	2.8	.90	2.1	20
486	10-29-91	4.00	<.010	100	12	180	1.2	370	34
487	10-30-91	.58	<.010	13	3.9	13	2.4	29	29

Table 10. Results of chemical analyses for nutrients, major ions, and selected metals for ground-water samples from 24 wells—Continued

[°C, degree Celsius; $\mu\text{S}/\text{cm}$, microsiemen per centimeter at 25 degrees Celsius; mg/L, milligram per liter; $\mu\text{g}/\text{L}$, microgram per liter; <, less than; --, no data]

Local well number	Date	Fluoride, dissolved (mg/L)	Silica, dissolved (mg/L)	Iron, dissolved ($\mu\text{g}/\text{L}$)	Manganese, dissolved ($\mu\text{g}/\text{L}$)	Lead, dissolved ($\mu\text{g}/\text{L}$)	Solids, sum of constituents, dissolved (mg/L)	Hardness, total (mg/L as CaCO_3)	Sodium (percent)
Pi-113	07-15-82	0.10	8.5	34	4	--	66	38	--
	10-24-91	<.10	7.8	120	3	1	85	42	34
213	08-10-81	<.10	9.5	60	10	--	163	41	68
	10-31-91	<.10	10	7	5	<1	155	74	32
227	10-22-91	<.10	6.9	310	140	<1	163	130	6
229	10-22-91	<.10	9.1	56	6	<1	102	66	22
230	07-15-82	.10	8.1	<3	9	--	117	96	--
	08-10-82	--	--	--	--	2	--	--	--
	10-21-91	<.10	9.0	13	<1	<1	157	110	9
232	10-29-91	.20	6.5	18	6	<1	1,180	640	37
233	10-25-91	<.10	8.5	6	1	<1	179	140	9
455	10-21-91	.20	7.7	92	62	<1	155	130	6
456	10-30-91	<.10	8.1	220	15	<1	97	45	31
463	10-29-91	.20	10	450	2,100	<1	497	300	23
464	10-29-91	<.10	8.2	10	290	<1	120	84	9
472	10-30-91	<.10	7.9	60	4	2	116	51	39
473	10-31-91	<.10	8.7	100	7	<1	116	70	17
477	10-31-91	.20	11	200	9	<1	105	65	16
478	10-21-91	<.10	8.4	11	1	<1	117	84	12
479	10-24-91	.20	9.0	29	2	<1	117	80	16
480	10-25-91	<.10	8.0	16	13	<1	184	140	8
481	10-25-91	.20	6.3	230	470	<1	1,210	940	18
482	10-25-91	.10	8.2	110	220	<1	309	250	4
483	10-26-91	.10	11	12	37	<1	154	73	39
484	10-26-91	<.10	8.7	190	11	<1	112	66	23
485	10-28-91	.20	8.5	36	6	2	58	30	16
486	10-29-91	.10	10	14	<1	<1	810	300	57
487	10-30-91	.20	11	66	230	<1	112	49	35

Table 11. Results of chemical analyses for nutrients and selected anions in precipitation and runoff[μ S/cm, microsiemen per centimeter at 25 degrees Celsius; mg/L, milligram per liter; <, less than; --, no data]

Date	Begin time	Specific conductance (μ S/cm)	Ph, water, whole, lab (standard units)	Nitrogen, ammonia, total (mg/L as N)	Nitrogen, nitrite, total (mg/L as N)	Nitrogen, NO ₂ + NO ₃ , total (mg/L as N)	Phosphorus, ortho, total (mg/L as P)	Chloride, dissolved (mg/L)	Sulfate, dissolved (mg/L)	Fluoride, dissolved (mg/L)
Culvert off Rt.209 at Head Injury Center										
11-21-91	1800	27	7.0	0.18	0.02	0.160	<0.01	2.4	1.5	0.20
11-21-91	2140	27	7.0	.17	.01	.300	.01	1.8	1.9	.20
11-22-91	0745	19	6.7	.11	.02	.260	<.01	2.4	2.2	.20
Pipe from roof at Head Injury Center										
11-21-91	1805	7	6.3	.10	.01	.160	<.01	<.10	1.0	<.10
11-21-91	2145	11	6.3	.07	.01	.290	<.01	<.10	1.4	.20
Parking lot at Milford Valley Convalescent Home										
11-21-91	1745	17	6.6	.10	.01	.130	<.01	.50	1.7	.20
11-21-91	1900	17	6.6	.07	.01	.140	<.01	.70	1.3	.20
11-21-91	2130	18	6.7	.09	.01	.260	.01	.60	1.7	.20
11-22-91	0810	20	6.5	.03	.01	.150	<.01	--	1.1	.20
11-22-91	1335	10	6.0	.06	.01	.190	.02	.50	2.2	.20
11-22-91	1630	13	6.5	.04	.01	.083	.02	.60	1.7	.20
11-22-91	1855	11	6.7	.05	<.01	.150	<.01	1.1	1.5	<.10
11-22-91	¹ 1900	10	7.5	.03	<.01	.270	<.01	.50	1.2	<.10
Culvert at Delaware Valley High School										
11-21-91	1730	86	7.0	.20	.08	.820	.03	1.0	14	.20
11-21-91	1845	57	7.2	.09	.04	.450	<.01	<.10	8.0	.20
11-21-91	2120	69	7.5	.09	.04	.490	.01	.80	8.2	.20
11-22-91	0820	47	7.3	.04	.02	.340	<.01	.10	3.6	.20
11-22-91	1345	55	7.1	.05	.02	.370	.01	.10	4.8	<.10
11-22-91	1640	43	7.4	.03	.02	.240	.01	<.10	3.8	.20
11-22-91	1900	33	7.1	.03	.02	.180	<.01	<.10	2.7	.20
11-22-91	¹ 1905	41	7.3	.02	.01	.180	<.01	<.10	2.1	.20
Rain gage at Milford Valley Convalescent Home										
11-21-91	¹ 1000	10	5.7	.05	<.01	.160	.01	.50	1.2	.20

¹ Composite sample from begin time to 11-23-91 at 0845.