EVALUATION OF AGRICULTURAL BEST-MANAGEMENT PRACTICES IN THE CONESTOGA RIVER HEADWATERS, PENNSYLVANIA:

Hydrology of a Small Carbonate Site near Ephrata, Pennsylvania, Prior to Implementation of Nutrient Management

WATER-QUALITY STUDY OF THE CONESTOGA RIVER HEADWATERS, PENNSYLVANIA

by Edward H. Koerkle, David W. Hall, Dennis W. Risser, Patricia L. Lietman, and Douglas C. Chichester

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CONVERSION FACTORS AND ABBREVIATED WATER-QUALITY UNITS

Multiply	by	To obtain
	Length	
inch (in.)	25.4	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
	Area	
acre	4.047	square kilometer
acre	0.4047	hectare
square mile (mi^2)	2.590	square kilometer
	Volume	
cubic foot (ft^3)	0.02832	cubic meter
	Mass	
pound (lb)	0.4545	kilogram
pound per acre (lb/acre)	1.123	kilogram per hectare
	Specific capacity	
gallons per minute per foot [(gal/min)/ft]	0.2070	liter per second per meter
	Transmissivity	
cubic foot per day per square foot times foot of aquifer thickness [(ft^3/d)ft^2] ft [1]		cubic meter per day per square meter times meter of aquifer thickness
	Temperature	
degree Fahrenheit (°F)	°C=5/9 (°F-32)	degree Celsius
	Other Abbreviations	

Abbreviated water-quality units used in report:

(mg/L) milligrams per liter

(µS/cm) microsiemens per centimeter at 25 degrees Celsius

[1] This unit of measurement reduces to foot squared per day (ft^2/d).

EVALUATION OF AGRICULTURAL BEST-MANAGEMENT PRACTICES IN THE CONESTOGA RIVER HEADWATERS, PENNSYLVANIA:

Hydrology of a Small Carbonate-Rock Site Near Ephrata, Pennsylvania, Prior to Implementation of Nutrient Management

By Edward H. Koerkle, David W. Hall, Dennis W. Risser, Patricia L. Lietman, and Douglas C. Chichester

ABSTRACT

The U.S. Geological Survey, in cooperation with the U.S. Department of Agriculture and the Pennsylvania Department of Environmental Protection[1], investigated the effects of agricultural best-management practices on water quality in the Conestoga River headwaters watershed. This report describes environmental factors and the surface-water and ground-water quality of one 47.5-acre field site, Field-Site 2, from October 1984 through September 1986, prior to implementation of nutrient management.

The site is partially terraced agricultural cropland underlain by carbonate rock. Twenty-seven acres are terraced, pipe-drained, and are under no-till cultivation. The remaining acreage is under minimum-till cultivation. Corn is the primary crop. The average annual rate of fertilization at the site was 480 pounds per acre of nitrogen and 110 pounds per acre of phosphorus.

An unconfined limestone and dolomitic aquifer underlies the site. Depth to bedrock ranges from 5 to 30 feet below land surface. Estimated specific yields range from 0.05 to 0.10, specific capacities of wells range from less than 1 to about 20 gallons per minute per foot of drawdown, and estimates of transmissivities range from 10 to 10,000 square feet per day. Average ground-water recharge was estimated to be about 23 inches per year.

The specific capacity and transmissivity data indicate that two aquifer regimes are present at the site. Wells drilled into dolomites in the eastern part of the site have larger specific capacities (averaging 20 gallons per minute per foot of drawdown) relative to specific capacities (averaging less than 1 gallon per minute per foot of drawdown) of wells drilled into limestones in the western part of the site.

Median concentrations of soil-soluble nitrate and soluble phosphorus in the top 4 feet of silt- or silty-clay-loam soil ranged from 177 to 329 and 8.5 to 35 pounds per acre, respectively.

Measured runoff from the pipe-drained terraces ranged from 10 to 48,000 cubic feet and was 1.7 and 0.8 percent, respectively, of the 1985 and 1986 annual precipitation. An estimated 90,700 cubic feet of surface runoff carried 87 pounds of total nitrogen and 37 pounds of total phosphorus, or less than 0.65 percent of the amount of either nutrient applied during the study period. Rainfall on the snow-covered, frozen ground produced more than half of the runoff and nitrogen and phosphorus loads measured in pipe-drained runoff.

Graphical and regression analyses of surface runoff suggest that (1) mean-storm concentrations of total nitrogen species and total phosphorus decreased with increasing time between a runoff event and the last previous nutrient application, and (2) mean total-phosphorus concentrations approached a baseline value (estimated at 2 to 5 milligrams per liter for total-phosphorus concentrations) after several months without nutrient applications.

Dissolved nitrate concentrations in ground water in wells unaffected by an on-site ammonia spill ranged from 7.4 to 100 milligrams per liter.

Average annual additions and removals of nitrogen were estimated. Nitrogen was added to the site by applications of manure and commercial fertilizer nitrogen, as well as by precipitation and ground water entering across the western site boundary. These sources of nitrogen accounted for 95, 3, 1, and 1 percent, respectively, of estimated additions. Nitrogen was removed from the site in harvested crops, by ground-water discharge, by volatilization, and in surface runoff, which accounted for 42, 28, 29, and less than 1 percent, respectively, of estimated removals.

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[1] Prior to 1995, the Pennsylvania Department of Environmental Protection (PaDEP) was known as the Pennsylvania Department of Environmental Resources (PaDER).

INTRODUCTION

The U.S. Geological Survey (USGS), in cooperation with the Pennsylvania Department of Environmental Protection (PaDEP)[2], conducted a study of the waters in the Conestoga River Basin to determine the effects of agricultural best-management practices (BMP's) on surface-water and ground-water quality. This study, part of the U.S. Department of Agriculture (USDA) Conestoga Headwaters Rural Clean Water Program (RCWP) and named the Conestoga Headwaters RCWP Project, began in 1981.

The RCWP, approved by the U.S. Congress in 1980, has the primary objective of improving surface-water and ground-water quality through the voluntary implementation of agricultural BMP's. With the help of RCWP cost-sharing funds, participating agricultural operations implement one or more BMP's such as manure storage, nutrient and pesticide management, field terracing, and conservation tillage.

By 1979, the Conestoga River had been identified in Pennsylvania's Section 208 Nonpoint Source Water-Quality Plan as a top-priority watershed in need of further water-quality study. The area's waters, serving more than 120,000 people, are used for domestic water supply both public and private, agriculture, wildlife, and recreation. The primary sources of the watershed's water-quality impairment have been identified as nutrients from manure and fertilizer, pesticides from agricultural applications, and sediment from cropland erosion (U.S. Department of Agriculture, 1982).

The Conestoga Headwaters RCWP is 1 of 20 RCWP projects nationwide and is 1 of 5 RCWP projects selected for Comprehensive Monitoring and Evaluation (CM&E). CM&E for the Conestoga Headwaters project includes monitoring of surface-water and ground-water quality before and after BMP implementation in order to detect water-quality changes. The project is established on three different scales: (1) Regional, one 188-mi^2 site; (2) Small Watershed, one 5.82-mi^2 site; and (3) Field, one 22.1-acre site (Field-Site 1) and one 47.5-acre site (Field-Site 2). A more detailed description of the study is available in the USGS report "Evaluation of Agricultural Best-Management Practices in the Conestoga River Headwaters, Pennsylvania: Methods of Data Collection and Analysis, and Description of Study Areas" (Chichester, 1988).

Purpose and Scope

This report presents the results of a study of water quality and the associated environmental conditions at Field-Site 2 from October 1, 1984, to September 30, 1986, prior to the implementation of nutrient management (pre-BMP). Surface-runoff and ground-water-quantity and -quality data, as well as climatological, agricultural-activity, soil, and hydrogeologic data, are summarized to characterize the site. Runoff represents discharge from 27 acres of pipe-drained terraces. Runoff quantity and mean concentrations and loads of nutrients in storm runoff are discussed. Ground-water levels and nutrient concentrations also are discussed. The results of all nutrient analyses are expressed as nitrogen or phosphorus in their elemental form unless otherwise stated. Temporal variation and correlations between water quantity and quality and selected environmental factors were examined by use of statistical techniques. In addition, these efforts will permit a future evaluation of what effect nutrient management may have had on surface-water and ground-water quality at Field-Site 2 after 2 years of BMP implementation.

Acknowledgments

The authors thank the following organizations and individuals whose efforts are an integral part of the Conestoga Headwaters RCWP Project. The USDA Soil Conservation Service (SCS) and the USDA Agricultural Stabilization and Conservation Service (ASCS) provided planning and technical assistance for the installation and implementation of agricultural BMP's. The Pennsylvania State University collected and analyzed soil-nutrient samples. John Hauenstein and Gary Lesher of the Susquehanna River Basin Commission surveyed and mapped the site. Aaron Z. Stauffer and Clark Stauffer supplied agricultural-activity data and allowed the use of their farm for the study site.

[2] Prior to 1995, the Pennsylvania Department of Environmental Protection (PaDEP) was known as the Pennsylvania Department of Environmental Resources (PaDER).

SITE LOCATION AND DESCRIPTION

Field-Site 2 is a 47.5-acre parcel of cropland on a 55-acre farm located adjacent to Indian Run Creek near Ephrata, Lancaster County, Pa. (fig. 1). Situated in carbonate rock terrane, the site is agricultural cropland with soils that are typically 5 to 12 ft deep. Slopes range from 2 to 9 percent; the median slope is 5 percent. The site is terraced for control of soil erosion. Runoff from 27 acres of the terraces is intercepted by 10 standpipes and discharged from a single pipe outlet that drains to Indian Run (fig. 2). The site is cultivated primarily in corn and a few acres of tobacco by use of both minimum-till and no-till methods. Manure produced by the farm's annual animal population of 100 beef cattle, 1,500 hogs (3 groups of 500), and 110,000 chickens (5 groups of 22,000) is applied to the site as fertilizer.

The sampling network consisted of a recording rain gage, a continuous-record surface-water runoff gage, 14 wells, and 1 spring. Seven monitoring wells and the spring were sampled monthly and during selected storms for water quality (fig. 2). The remaining seven wells were sampled less frequently and provided data used in characterization of the site.

DATA COLLECTION AND ANALYSIS, AND QUALITY ASSURANCE

A summary of data collection and analysis methods is presented in this section. A more detailed description of data-collection procedures and methods of analysis is given by Chichester (1988).

Data Collection

Data were collected from October 1984 through September 1986. In this report, all references to 1985 and 1986 refer to their respective water years[3] unless stated otherwise. Water-quality, soil-nutrient, manure-nutrient, agricultural-activity, and precipitation data were collected at the site (table 1).

All water-quality samples to be analyzed for nutrients were stored at approximately 4°C from time of collection to time of analysis, preserved with mercuric chloride, and analyzed by the PaDEP Bureau of Laboratories, located in Harrisburg, Pa., and all suspended-sediment and particle-size samples were analyzed by the USGS Sediment Laboratory, located in Lemoyne, Pa. Soil-nutrient samples were analyzed by the Pennsylvania State University Soil and Environmental Chemistry Laboratory. Manure-nutrient samples were analyzed by A&L Eastern Agricultural Laboratories.[4] The detection limits for the constituents analyzed are listed in table 2.

Water-quality data collected during this study are published in USGS Water-Resources Data Reports PA-85-2 and PA-86-2 (Loper and others, 1987, 1988). The data are catalogued by the USGS local identification numbers shown in this report.

Precipitation-quantity, duration, and intensity data were collected by use of a rain gage equipped with an Analog Digital Recorder (ADR). The ADR recorded cumulative rainfall every 5 minutes. Precipitation data were compared with long-term records from the National Oceanic and Atmospheric Administration (NOAA) station at Ephrata, Pa. When snow, ice, and equipment malfunction affected data accuracy, missing precipitation record was estimated by use of data from nearby USGS rain gages or NOAA stations. Missing precipitation quantity data was estimated to be the daily mean of data from surrounding USGS and NOAA precipitation gages. In cases where multiple storms occurred in a single day, the precipitation quantity for an individual storm was calculated by prorating daily precipitation recorded at the nearest continuous-record precipitation gage. Missing duration data were estimated by averaging distance weighted duration values recorded at surrounding USGS and NOAA gages. Missing 30-minute intensity data were estimated by least squares regression. Maximum 30-minute intensities recorded at the site and the maximum hourly intensities recorded at the Landisville, Pa., NOAA gage were regressed ($R = 0.90$, $p < 0.001$). Storms with durations of approximately 1 hour or less noticeably weakened

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[3] A water year is the period from October 1 to September 30, and is designated by the calendar year in which it ends.

[4] Use of firm names in this report is for identification purposes only and does not constitute endorsement by the U.S. Geological Survey.

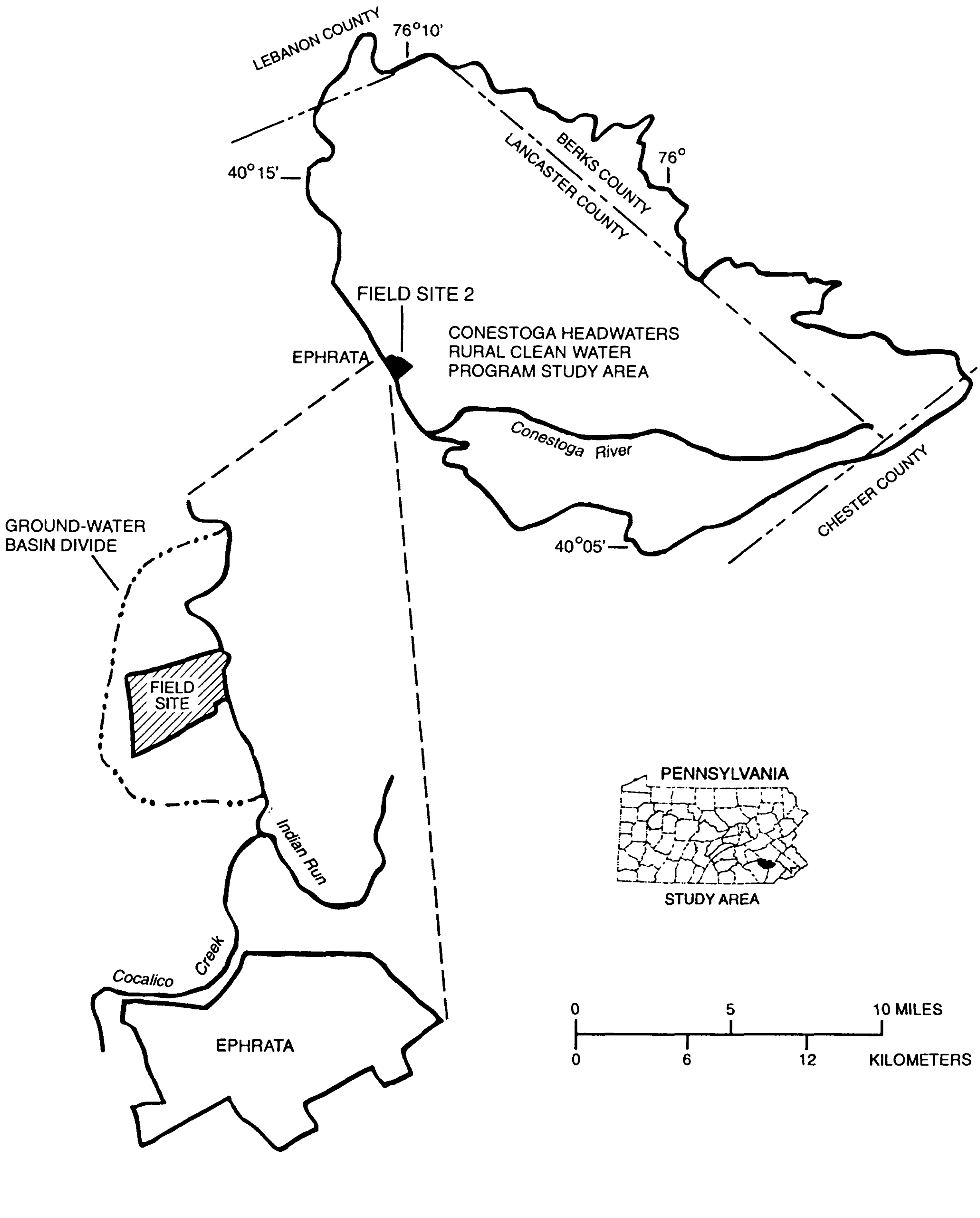


Figure 1. Location of the Conestoga Headwaters Rural Clean Water Program area and Field-Site 2.

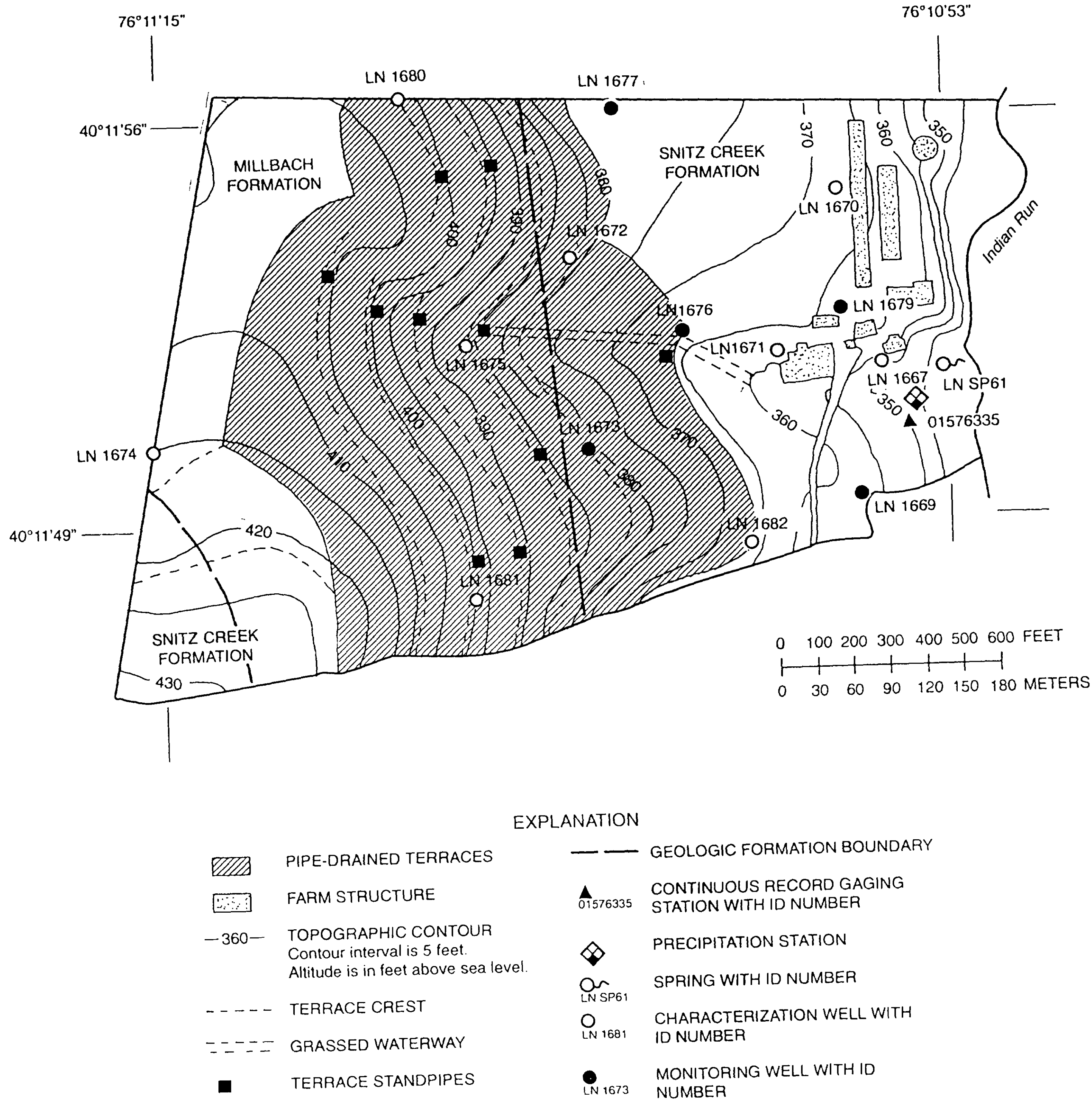


Figure 2. Topography, geologic units, terrace arrangement, and data-collection locations at Field-Site 2.

Table 1. Data-collection protocol for Field-Site 2, Conestoga Headwaters study

[NA, not applicable]

Data types	Number of collection locations	Collection frequency
Agricultural activities		
Nutrient applications	NA	Biweekly
Tillage dates	NA	Biweekly
Planting and harvesting dates	NA	Biweekly
Manure nutrients	3	Poultry - one time
		Hog - three times
		Steer - one time
Soil nutrients	3-17	Spring, before planting, and fall, after harvest
Surface-water runoff		
Discharge	1	Continuous
Nutrients and suspended sediment	1	Selected runoff events
Ground water		
Water levels	13	One well continuous
		Two wells intermittent
Nutrients	15	Six wells and 1 spring monthly
		One well quarterly
		Seven wells intermittent
Precipitation		
Quantity	1	5-minute intervals
Nutrients	1	Three times

Table 2. Physical and chemical analyses performed on water-quality samples collected at Field-Site 2 from October 1984 through September 1986

[μS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligrams per liter; °C, degrees Celsius]

Characteristic or constituent	Detection limit
Temperature (field)	Measured to nearest 0.5°C
Specific conductance (field)	1 to 10 μS/cm
Suspended sediment	1 mg/L
Total and dissolved nutrients	
Ammonium[1] + organic nitrogen	.2 mg/L
Ammonium[1]	.02 mg/L
Nitrate[2] + nitrite	.04 mg/L
Nitrite	.01 mg/L
Phosphorus	.02 mg/L

[1] Ammonium in this report represents ammonia + ammonium.

[2] Maximum Contaminant Level is 10 mg/L as nitrogen, National Primary and Secondary Drinking Water Regulation (U.S. Environmental Protection Agency, 1989).

the regression. The U.S. Weather Bureau (1957) reported that storms of shorter duration are generally smaller in their coverage area. A smaller coverage area reduces the possibility that distant gages will have similar records. On this basis, storms with durations of approximately 1 hour or less were not included in the regression.

Precipitation samples for water-quality analysis were collected in a 13-in. funnel channeled to quart jars. After collection, the samples were preserved with mercuric chloride and kept chilled at approximately 4°C until analyzed for dissolved nutrients.

Agricultural activities in this report refer to cultivation and farm operational practices associated with crop production. Agricultural-activities data were provided by the farmer every 2 to 3 weeks. Manure or commercial fertilizer application data included type of fertilizer, date, area, amount, and method of application. Tillage type, dates, and area, as well as planting and harvesting dates, were recorded.

Manure samples were collected at various times during the pre-BMP period. Sample collection was timed such that the samples were collected near the time of field application. Four manure sources were sampled—manure/bedding mix from a steer pen, hog manure from a storage tank, hog manure from a pit below a hog-finishing facility, and poultry manure from the floor of a poultry house. The samples were analyzed for nitrogen, phosphorus, and percent moisture.

Soil-sample cores from 0-2 ft and 0-4 ft were collected and analyzed for soluble nitrate and soluble phosphorus. All samples, except those collected during active crop growth, were collected in the upper 4 ft or "root zone" of the soil. Hand-augered samples 0-2 ft deep were obtained when the presence of crops precluded the use of machinery necessary to collect 0-4 ft deep samples. Each soil sample is actually a composite of three cores extracted within an area of approximately a 25-ft radius.

After collection, the three cores were divided into depth increments and composited into one sample by depth. The 2-ft cores were divided into 0-8 in. depth and 8-24 in. depth increments. The 4-ft cores were divided into four increments: 0-8 in. depth, 8-24 in. depth, 24-36 in. depth, and 36-48 in. depth. Each depth increment was analyzed for soluble nitrate and phosphorus, by extraction with strontium chloride, and the results summed for the total core depth.

Surface runoff originating from the pipe-drained terraces was discharged through a 6-in. Parshall flume. The stage in the flume was recorded on a continuous graphic recorder and an ADR. Discharge was computed by use of a standard Parshall flume rating that was field checked and corrected for poor entrance conditions. The outlet section of the flume contained a perforated intake for the collection of runoff samples.

Runoff samples were collected by use of a modified PS-69 automatic pumping sampler triggered by a stage-operated float switch. When the stage was at or above the float-switch setting, samples were collected every 15 minutes. The runoff samples were kept chilled to approximately 4°C by a refrigeration unit until retrieval, usually within 24 hours. Selected samples were preserved with mercuric chloride and analyzed for concentrations of total nutrients and a more limited number of samples were also analyzed for concentrations of dissolved nutrients and suspended sediment.

Mean water-weighted nutrient and suspended-sediment concentrations and loads were computed for individual runoff events. Estimated loads were calculated by use of equations generated by regressing the log of the constituent load against the log of total event runoff for measured runoff events (table 3). Estimated loads accounted for 0.60 percent of the total nitrogen and 0.70 percent of the total phosphorus in runoff for the pre-BMP period.

Thirteen air-rotary drilled wells, an existing hand-dug well, and a spring were used to provide ground-water data for characterization of the site (table 4). The drilled wells were cased to bedrock, then continued as uncased holes in the unconfined carbonate aquifer. The 6-in.-diameter steel casings were pressure grouted, and bentonite was used as a surface seal at each well. Continuous water-level recorders were installed on 11 wells, and intermittent steel-tape water-level measurements were made at 2 wells.

Table 3. Regression statistics for the log of nutrient loads, in pounds, as a function of the log of total event runoff, in cubic feet, for Field-Site 2

[n, number; <, less than]

Dependent variable	n	Regression coefficient: Log of total event runoff	Regression coefficient: t-test	Regression coefficient: p-value	Intercept	Coefficient of determination (Adj. R^2)[1]	Standard error: Log units	Standard error: Percent[2] Plus	Standard error: Percent[2] Minus
Total nitrogen	22	0.963	12.248	<0.001	-2.959	0.89	0.260	82	45
Total ammonia + organic nitrogen	22	.957	10.723	<.001	-3.246	.84	.310	104	51
Total nitrate + nitrite nitrogen	22	.832	12.636	<.001	-2.881	.88	.228	69	41
Total phosphorus	22	.951	12.651	<.001	-3.325	.88	.261	82	45

[1] Coefficient of determination (R^2) adjusted for degrees of freedom.
[2] G.D. Tasker, U.S. Geological Survey, written commun., 1978.

Table 4. Ground-water data-collection locations at Field-Site 2 and descriptions

[All depths shown in feet below land surface; (gal/min)/ft, gallon per minute per foot; <, less than; --, no data; E, estimated value; N/A, not applicable. See text for geologic description of the Formation.]

Well number	Station location: Latitude	Station location: Longitude	Total depth of well	Depth to casing opening (overburden thickness)	Depth to bedrock	Bedrock elevation (feet)	Sampling depth (feet)	Estimated specific capacity [(gal/min)/ft]	Data collected: Nutrients	Data collected: Water level
LN 1667	40°11'52"	76°10'55"	15	--	--	--	At outside faucet, pump at 12	--	Occasionally	--
LN 1669	40°11'49"	76°10'55"	100	11	6.5	351.5	85	1	Monthly	Continuous
LN 1670	40°11'56"	76°10'57"	75	9.77	5.5	361.4	65	1	Quarterly	Continuous
LN 1671	40°11'52"	76°10'58"	28	18.81	13	342.0	--	<1	Occasionally	Intermittent
LN 1672	40°11'52"	76°11'05"	10	10.88	10	370.4	--	<1	Occasionally	Intermittent
LN 1673	40°11'48"	76°11'03"	46	13.80	12	367.47	35	1	Monthly	Continuous
LN 1674	40°11'45"	76°11'15"	125	25.16	19	396.0	--	<1	Occasionally	Continuous
LN 1675	40°11'50"	76°11'07"	55	17.17	14	374.5	--	<1	Occasionally	Continuous
LN 1676	40°11'52"	76°11'01"	40	8.78	11	356.3	35	1	Monthly	Continuous
LN 1677	40°11'56"	76°11'05"	50	29.99	28	349.0	35	20	Monthly	Continuous
LN 1679	40°11'52"	76°10'57"	60	13.40	10	354E	35	20	Monthly	Continuous
LN 1680	40°11'56"	76°11'09"	60	7.8	7	375E	--	<1	Occasionally	Continuous
LN 1681	40°11'47"	76°11'08"	60	8.8	8	40E	35	<1	Occasionally	Continuous
LN 1682	40°11'48"	76°10'59"	350	18.6	18	350E	35	<1	Occasionally	Continuous
LN SP61	40°11'52"	76°10'53"	--	--	--	--	--	--	Monthly	N/A

Initially, ground-water samples were collected at the 14 wells and the spring. On the basis of successful well development and specific capacity, seven of the most productive wells were selected to be monitoring points for continued sampling. Samples of water from six of the monitoring wells and the spring were analyzed for specific conductance and nutrient concentrations every 3 to 4 weeks during nonrecharge periods, and more frequently over several recharge periods. Water from the remaining well was sampled quarterly for specific conductance and nutrient concentrations. Additionally, in May 1986, water samples from eight wells were measured for temperature and analyzed for dissolved-oxygen and major-ion concentrations. Prior to sampling, all wells except LN 1667 and LN 1677 were pumped until water levels were approximately 5 ft below sampling depth. After water levels recovered to the sampling depth, ground-water samples were collected by use of a distilled-water rinsed Kemmerer type sampler. Ground-water samples were chilled to 4°C and preserved with mercuric chloride. Well LN 1677 was not pumped prior to sampling because access to the well was restricted during the growing season and because sample analyses indicated little change in water quality before and after pumping. Well LN 1667 (the house well) was sampled from an outside spigot because the pumping equipment was inappropriate for use in the well.

Data Analysis

In this report, statistical procedures were used for summarizing data, making statistically supported inferénces about the data, and defining explanatory relations between various data sets. Data summarization was accomplished with descriptive statistics such as means, medians, ranges, standard deviations, and percentiles. Statistical inferences about data normality, equality of data-set means or medians, and linear-regression analyses were based on the results of hypothesis testing. All hypotheses tests were performed at the 95-percent confidence level. Because most of the data sets have small sizes and nonnormal distributions, nonparametric procedures were used for all hypothesis tests with the exception of the linear regression F- and t-tests. Equality of the means or medians between data sets was tested with the Wilcoxon Signed-Rank, Mann-Whitney Rank Sum, or Kruskal-Wallis tests. The Wilcoxon Signed-Rank test, for matched-pair data, was used with duplicate sample data. The Mann-Whitney or Kruskal-Wallis tests, for independent sample data, were used for two or more independent data sets such as water-quality data. Explanatory relations between hydrologic, climatic, and agricultural-activities data were explored with multiple linear regression. The regression analyses were evaluated by use of the F-test for overall regression significance and the t-test for significance of regression coefficients.

Multiple linear regressions were completed as follows. Potential explanatory variables were regressed in all possible combinations against a single dependent variable. From these regressions the best-fit models were selected by use of Mallow's Cp statistic. These models were evaluated on the basis of significance of the regressions coefficients and the distribution of the residuals. The models were further examined for potential problems such as multicollinearity, high leverage, and high influence.

A more complete discussion of the basic descriptive, parametric, and nonparametric procedures used can be found in Iman and Conover (1983), and Conover (1980). All statistical procedures were run on software from the Statistical Analysis System (SAS) Institute, Inc. (1982a, 1982b), P-STAT, Inc. (1986), and SYSTAT, Inc. (Wilkinson, 1988). Because adequate descriptive terms and techniques have not been developed to describe anisotropic, heterogenous, and irregularly fractured, carbonate ground-water systems, descriptive terms and techniques typically used to analyze isotropic, homogeneous aquifers have been applied in a general manner to yield useful information about aquifer properties at the study area. Descriptive terms and techniques that have been borrowed from the isotropic, homogeneous domain include Rorabaugh's recession slope analysis (Rorabaugh, 1960) and the use of Darcy's Law (Freeze and Cherry, 1979).

Quality Assurance

Quality assurance (QA) of laboratory nutrient analyses was maintained by use of three types of QA samples. Blank samples of distilled water were submitted for the evaluation of laboratory baseline capabilities at or near detection-limit concentrations. USGS standard-reference water samples (Janzer, 1985a, 1985b, 1986a, 1986b) were used for the determination of analytical accuracy. Field-split duplicate water samples were used for the assessment of analytical repeatability for the individual constituents. The constituents analyzed and summary statistics for the QA data are shown in table 5.

For each blank sample, the difference between the reported value and the detection limit for each constituent was calculated. The results of a Wilcoxon Signed-Rank test indicate that the analytical concentrations for the blanks varied significantly from their respective detection limits for total and dissolved ammonium, total and dissolved ammonium + organic nitrogen, and total phosphorus. The results suggest either a possible positive analytical bias for these constituents at concentration levels near detection limits or contaminated distilled water blanks.

A comparison of median concentrations of dissolved ammonium in ground-water and blank-water samples also showed a positive bias for the blank samples. Because more than half of the ground-water samples had measured concentrations of dissolved ammonium at or below the detection limit, the detection limit bias in blank samples was suspect. Further investigation determined that ammonia contamination of the blank water probably occurred when blank-water samples were transported in close proximity to surface-water samples that contained measurable concentrations of ammonium.

Table 5. Summary statistics for quality-assurance analyses at Field-Site 2

[All values shown in milligrams per liter; Min, minimum; Max, maximum; n, number of samples; <, less than; --, no data]

Constituent	Detection limit	Concentrations, range of all samples		Blanks				Reference samples					Duplicates			
					Concentrations, range of blanks		Median difference from detection limit		Concentrations, range of reference samples		Absolute value of difference between known and measured values			Concentration range of duplicate samples		Absolute value of difference between pairs
		Min	Max	n	Min	Max		n	Min	Max	Max	Median	n	Min	Max	Max
Total nitrate + nitrite	0.04	2.4	140	11	<0.04	<0.04	0.00	--	--	--	--	--	31	1.3	57	2.1
Dissolved nitrate +nitrite	.04	2.6	130	9	<.04	<.04	.00	--	--	--	--	--	87	.96	115	9.6
Total nitrate	.04	--	--	--	--	--	--	27	0.19	17	1.4	0.10	--	--	--	--
Dissolved nitrate	.04	--	--	--	--	--	--	13	.80	17	1.0	.10	--	--	--	--
Total nitrite	.01	<.01	5.3	8	<.01	.01	.00	--	--	--	--	--	31	<.01	2.7	.40
Dissolved nitrite	.01	<.01	3.6	8	<.01	.01	.00	--	--	--	--	--	44	<.01	.44	.07
Total ammonia + organic nitrogen	.20	.36	46	11	<.20	1.0	.06	25	.45	21	5.7	1.1	30	.22	22.	2.0
Dissolved ammonia + organic nitrogen	.20	<.20	21	9	<.20	.80	.06	15	4.9	16	3.9	1.3	87	<.20	2.6	.52
Total ammonium	.02	<.02	16	8	.04	.11	.04	22	.15	23	3.2	-.06	31	<.02	10	.57
Dissolved ammonium	.02	<.02	13	10	<.02	.11	.03	13	.76	23	2.5	-.10	44	<.02	.44	.10
Total phosphorus	.02	.02	18	11	<.02	.06	.00	30	.02	4.7	3.5	.10	31	.04	4.3	.19
Dissolved phosphorus	.02	<.02	14	8	<.02	.05	.00	15	.71	3.5	.60	.20	45	<.02	25	5.6

For reference samples, a relative percent difference (RPD) was calculated for each measured concentration and expected concentration pair. The RPD was calculated as follows:

$$RPD = \frac{|\text{Measured concentration} - \text{expected concentration}|}{\frac{(\text{Measured concentration} + \text{expected concentration})}{2}} \times 100 \qquad (1)$$

The RPD's indicated that overall analytical accuracy was marginal for all constituents except dissolved ammonium, total nitrate, and dissolved nitrate. Total and dissolved ammonium + organic nitrogen and total and dissolved phosphorus had the least accuracy. RPD's for these four constituents were greater than 15 percent for more than 50 percent of the samples. A Wilcoxon Signed-rank test indicated significant bias for all constituents except dissolved ammonium + organic nitrogen and dissolved ammonium. All significant biases were positive except for the total ammonium. For runoff data, the constituent biases represented less than 5 percent of their respective median measured concentrations with the exception of total and dissolved ammonium + organic nitrogen. The estimated bias for total ammonium + organic nitrogen represented about 20 percent of the median concentration measured in runoff. For ground-water data, estimated biases in the reported total and dissolved ammonium + organic nitrogen and the total- and dissolved-phosphorus concentration represent a large source of error and caution should be used in their interpretation. However, in this study, the constituent of primary concern in ground water was nitrate nitrogen. The estimated bias for total and dissolved nitrate nitrogen concentrations represented 2 percent or less of the nitrate-nitrogen concentrations measured in ground-water samples.

For duplicate data, determination of acceptable analytical repeatability was made by comparing the RPD for each duplicate pair with RPD goals. Duplicate RPD's were calculated as follows:

$$RPD = \frac{|\text{Sample concentration} - \text{duplicate concentration}|}{\frac{(\text{Sample concentration} + \text{duplicate concentration})}{2}} \times 100 \qquad (2)$$

RPD goals are listed below.

Concentration (in detection limits)	RPD goal[1] (in detection limits or percent)
0-5	1
5-20	2 or 20 percent
>20	10 percent

[1] M.A. Hardy, U.S. Geological Survey, written commun., 1990.

The RPD's for all constituents with the exception of total nitrate plus nitrite and dissolved nitrate plus nitrite were within RPD limits for 90 percent or more of the duplicate samples analyzed. RPD's for total nitrate plus nitrite and dissolved nitrate plus nitrite were within the RPD limits for 87 and 86 percent of the samples, respectively.

Results from the QA program indicate that bias and accuracy limitations existed for most of the constituents. Therefore, in terms of absolute value, the water-quality data for nutrients should be interpreted with caution. Caution should particularly be used when the data are approaching the detection limit. However, in the framework of this study, the bias and accuracy limitations are of minor importance when compared to the magnitude and natural variation in concentrations of those constituents that are likely to be affected by BMP's.

FACTORS THAT CAN AFFECT WATER QUANTITY AND QUALITY

Many physical and climatological factors can affect surface-runoff and ground-water quantity of a basin. Detailed characterizations of the factors thought to have a major hydrologic influence at Field-Site 2 are presented in this section.

Topography

Land surface elevation in the study area ranges from 342 ft in the southeastern corner to 430 ft at the southwestern corner, a total relief of 88 ft. The slope of the land surface ranges from 2 percent in the north central part to about 9 percent in the central part of the study area. Median slope of the entire site is about 5 percent. Terraces were constructed in 1965 and restructured in 1981 to include a pipe-outlet system that drained 27 of the terraced acres.

Physiography and Geology

The study area is located within the Piedmont Physiographic Province in the Conestoga Valley Section (Custer, 1985; Meisler and Becher, 1971). This section is characterized by the presence of carbonate and shale rocks that have been repeatedly deformed by folding and faulting.

Approximately two-thirds of the site is underlain by limestone of the Millbach Formation, and the other one-third of the site is underlain by dolomite of the Snitz Creek Formation. Both formations are of Cambrian age. The Millbach Formation is a light-pinkish gray to medium-dark gray, finely to very finely crystalline limestone with light-gray laminae of dolomite. The Snitz Creek Formation is a light to dark gray, finely to very finely crystalline dolomite.

Precipitation

Quantity

Measured annual precipitation at the site was 35.9 in. in the 1985 water year and 38.8 in. in the 1986 water year. Average annual precipitation is 43.5 in. on the basis of 30 years of record from the NOAA observation site at Ephrata, Pa. (National Oceanic and Atmospheric Administration, 1984). Compared to the annual average, precipitation deficits for 1985 and 1986 were 17 and 11 percent, respectively.

Monthly precipitation for 1985 ranged from 0.60 in. in April to 5.20 in. in July (fig. 3). Growing-season precipitation, from May through October, accounted for 60 percent (21.7 in.) of the annual total. Precipitation greater than or equal to 0.05 in. was recorded on 46 days of the period. A hurricane in September accounted for 87 percent (4.1 in.) of the monthly total and 10 percent of the annual total.

In 1986, monthly precipitation ranged from 0.85 in. in December to 7.95 in. in July. Growing-season precipitation accounted for 52 percent (20.0 in.) of the total annual precipitation. Precipitation greater than or equal to 0.05 in. was recorded on 39 days of the period. July and August had the largest monthly totals and together accounted for 35 percent of the annual total. In July, 4.3 in. of rain occurred over a 2-day period and accounted for 54 percent of the monthly total and 11 percent of the annual total of precipitation.

In the 6-month period prior to the start of the study, the NOAA site at Ephrata, Pa. (National Oceanic and Atmospheric Administration, 1984), recorded 30.4 in. of precipitation. This was 70 percent of the 30-year average total annual precipitation and 25 percent (6.0 in.) greater than the normal for the 6-month period. However, the last 2 months of the period, August and September, had below-normal rainfall.

For this study, a storm was defined as any precipitation event bounded by precipitation-free periods of 1 hour or more and having 0.10 in. or greater total precipitation. There were 58 storms in 1985 and 59 in 1986. These storms accounted for 83 percent and 89 percent, respectively, of the total annual precipitation. The number and sizes of the storms are summarized, by month, in figure 4.

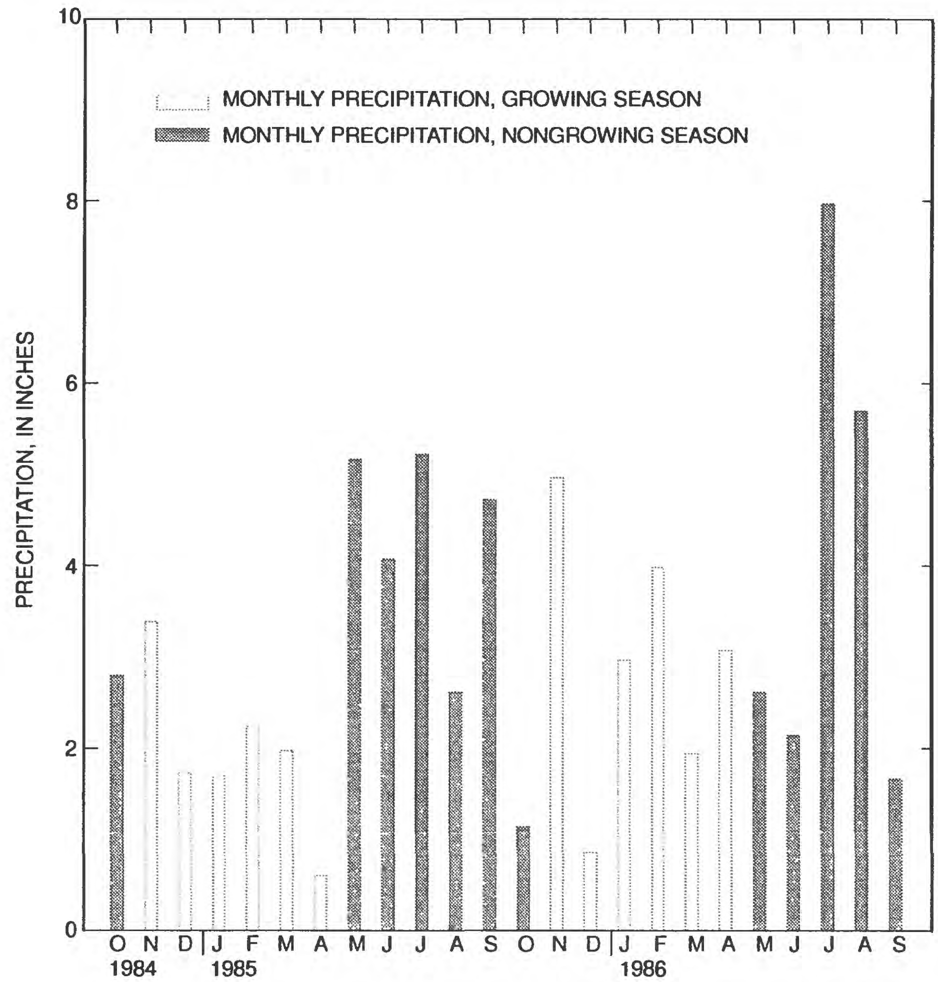


Figure 3. Monthly precipitation at Field-Site 2.

Maximum 30-minute precipitation intensity and precipitation duration was determined for runoff events that were sampled for runoff water quality. Maximum 30-minute precipitation intensity was estimated for 12 runoff events and precipitation duration was estimated for 10 runoff events. Maximum 30-minute precipitation intensity had median and maximum values of 0.4 and 1.3 in. per 30 minutes, respectively, in 1985 and 0.4 and 1.8 in. per 30 minutes, respectively, in 1986. Precipitation durations ranged from 1.0 to 24 hours. Median precipitation duration was 4.0 hours.

Quality

Nitrogen enters the site in precipitation in the form of ammonium and nitrate ions. Average annual concentrations of ammonium and nitrate in precipitation in southeastern Pennsylvania were published by Lynch and others (1986, 1987)(table 6). Quantities of precipitation recorded at the Field-Site 2 gage were multiplied by the concentrations of ammonium and nitrate in precipitation in samples collected by Lynch and others (1986, 1987) to calculate loads of nitrogen in precipitation input to the 55-acre farm during 1985-86 (table 6).

The calculated annual loads of nitrogen in precipitation in table 6 contain an unknown amount of error. Errors in the measurement of precipitation at the site gage are small. A much greater error in the calculated loads comes from the use of ammonium and nitrate values reported by Lynch and others (1986, 1987) for southeastern Pennsylvania. Farms with large animal populations (such as Field-Site 2) are sites of

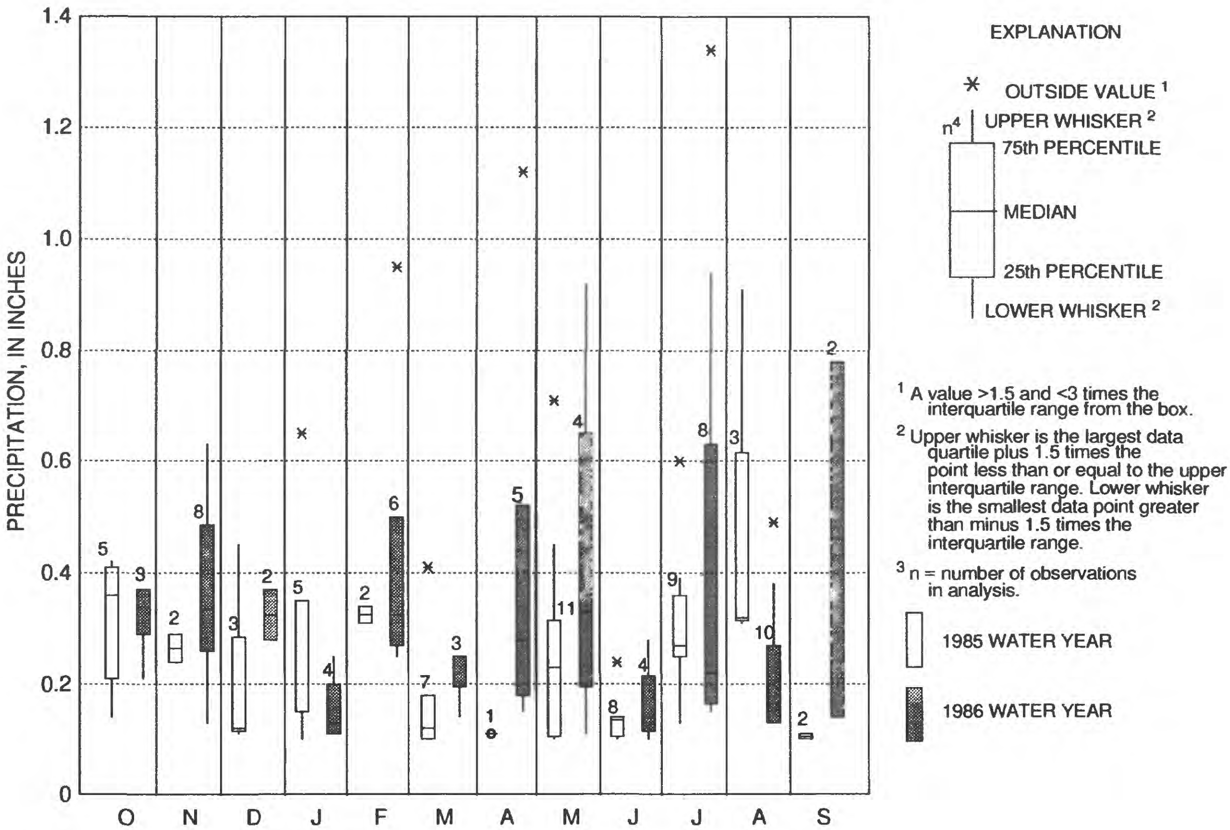


Figure 4. Monthly distribution of storm precipitation at Field-Site 2.

Table 6. Calculation of annual nitrogen loads in precipitation, by use of annual nitrate and ammonium concentrations reported by Lynch and others (1986, 1987) and precipitation-quantity data from Field-Site 2

[mg/L, milligrams per liter; pounds of nitrogen in precipitation were calculated by multiplying total ammonium plus nitrate concentrations (as nitrogen) by annual precipitation (in liters) and a milligram-to-pound conversion factor (2.205×10^{-6} pounds per milligram)]

Water year	Annual dissolved ammonium concentrations as N, in mg/L	Annual dissolved nitrate concentrations as N, in mg/L	Total ammonium plus nitrate as N, in mg/L	×	Annual precipitation in liters	×	$\left(\frac{1g}{1,000mg}\right)\left(\frac{2.205lb}{1,000g}\right)$	=	Annual pounds nitrogen in precipitation
1985	0.19	0.45	0.64		202,624,000				290
1986	.19	.44	.63		219,528,000				300

active volatilization of manure nitrogen. For example, measurements of the nitrogen content of precipitation made at a chicken and hog farm in Adams County, Pa. (Langland, 1992), indicated that precipitation had elevated concentrations of ammonium from manure volatilization. In the Langland (1992) study, concentrations of ammonium in precipitation became gradationally lower as distance from a manure-storage lagoon increased, and estimated annual concentrations of nitrogen in precipitation were significantly higher than those reported by Lynch and others for that region of Pennsylvania. Because Field-Site 2 has a manure-storage facility and is a site of concentrated animal populations, volatilization of manure nitrogen at the site probably caused elevated concentrations of nitrogen in precipitation relative to the nitrogen-concentration data published in Lynch and others (1986, 1987). Therefore, actual loads of nitrogen in precipitation at Field-Site 2 during 1985-86 were probably larger than those reported in table 6. A precipitation sample collected at Field-Site 2 during a storm that occurred on April 16-17, 1986, contained concentrations of 0.48 mg/L ammonium, 0.44 mg/L nitrate, and <0.01 mg/L nitrite. All three constituents were primarily in the dissolved phase.

Agricultural Activities

Cropping

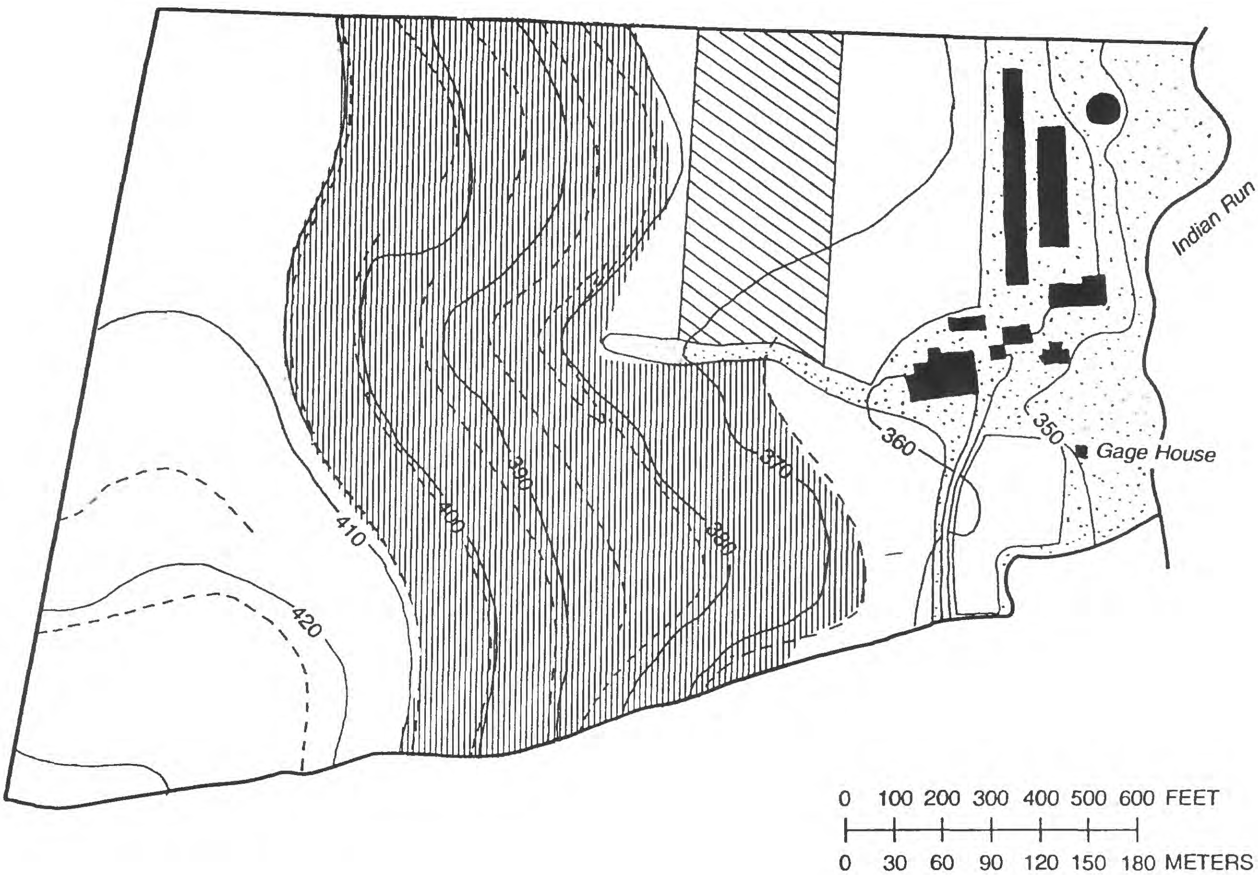
Planting and harvesting were scheduled by growth requirements of the crops and prevailing field conditions. Corn, the primary crop, was planted during the last 2 weeks in April and harvested from late September to mid-October. Tobacco, which requires a shorter, warmer season, was transplanted from starting beds to the field in mid-June and harvested in mid-August. A winter cover crop of rye was broadcast seeded after corn harvesting and covered primarily the pipe-drained terraces. The rye was not harvested but was killed with herbicide prior to the planting of corn.

Crop acreage for 1985-86 are listed in table 7, and the cropping patterns for 1985 and 1986 are shown in figures 5 and 6, respectively. The cropping pattern was changed from 1985 to 1986 with the relocation of the tobacco acreage. This change moved approximately 1 acre of tobacco out of the pipe-drained terraces.

Table 7. Crop acreage at Field-Site 2 for 1985 and 1986 water years

Crop	1985 Pipe-drained terraces	1985 Other	1985 Total	1986 Pipe-drained terraces	1986 Other	1986 Total
Corn	27	16.5	43.5	26	16.5	43.5
Tobacco	1	3.0	4.0	1	4.0	4.0
Rye (winter)	22.5	0	22.5	22.5	2.5	25.0

Crops potentially influence surface-water and ground-water quantity and quality through processes such as interception of precipitation, transpiration, and uptake of water and nutrients. Interception of precipitation by crop foliage reduces the kinetic energy of precipitation, and subsequently reduces the ability of precipitation to seal the surface against infiltration and to dislodge sediment and nutrient-bearing materials on or in the soil. These conditions can continue after harvest if sufficient crop residue



EXPLANATION

CORN WITH NO WINTER COVER

NO-TILL CORN WITH RYE WINTER COVER

TOBACCO WITH NO WINTER COVER

GRASS

IMPERVIOUS SURFACES

—360— TOPOGRAPHIC CONTOUR
Contour interval is 10 feet.
Altitude is in feet above sea level.

– – – – – TERRACE CREST

Figure 5. Land cover for 1985 water year at Field-Site 2.

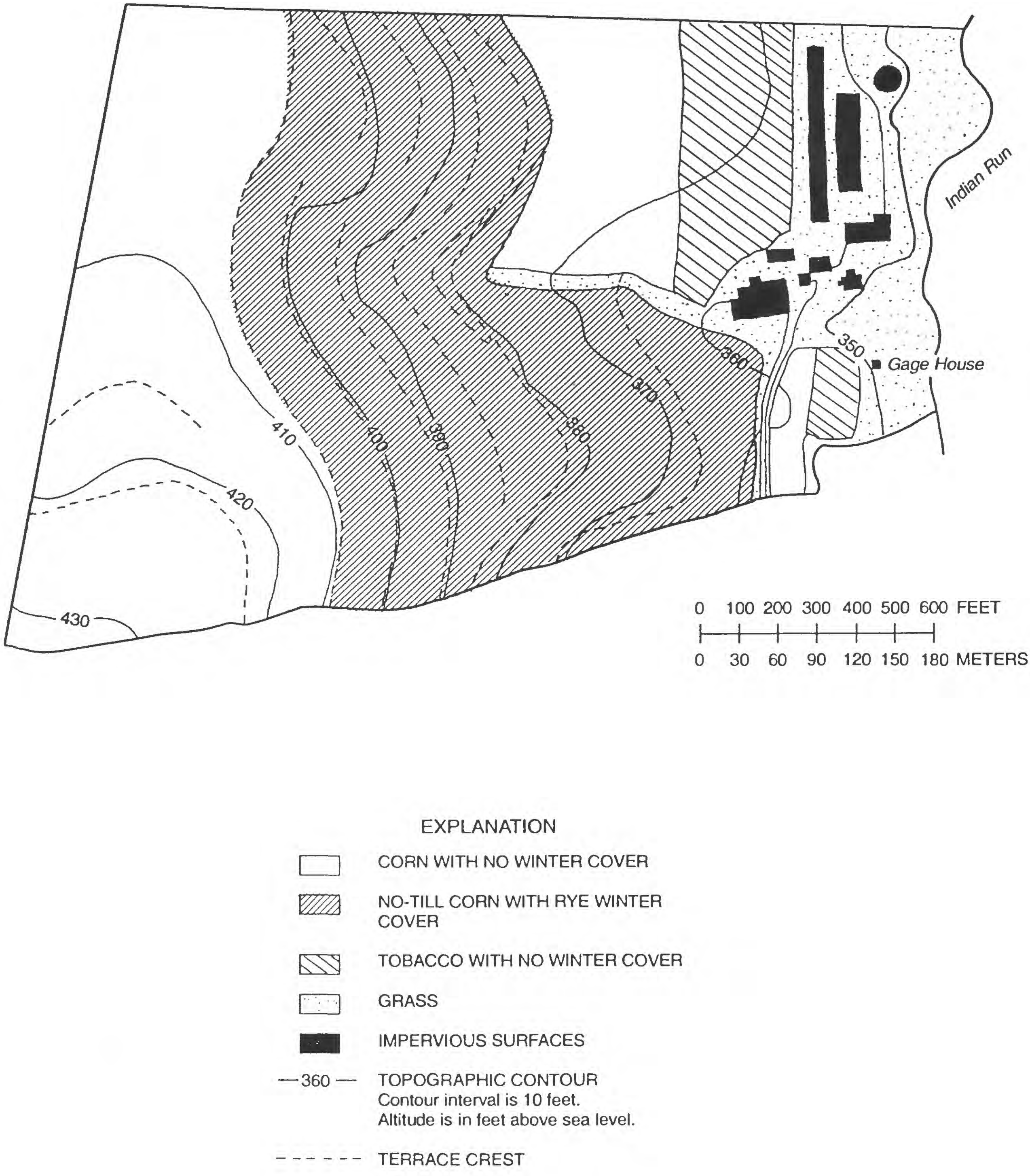


Figure 6. Land cover for 1986 water year at Fleld-Site 2.

remains. As a result, the sediment and nutrient concentrations in surface runoff may vary as the amount of crop cover varies. At the same time, infiltration amounts and nutrient concentrations characteristically increase with reduced impact consolidation (Musgrave and Holtan, 1964). Additionally, during periods of active growth, crops take up a substantial amount of soil water and nutrients, in turn reducing the amount available for ground-water recharge.

In this report, the combination of crop cover and crop residue was used as an estimator of the influence these processes might have on water quantity and quality. Effective crop cover (fig. 7) is expressed in percent and was estimated by use of planting and harvesting dates along with field observations.

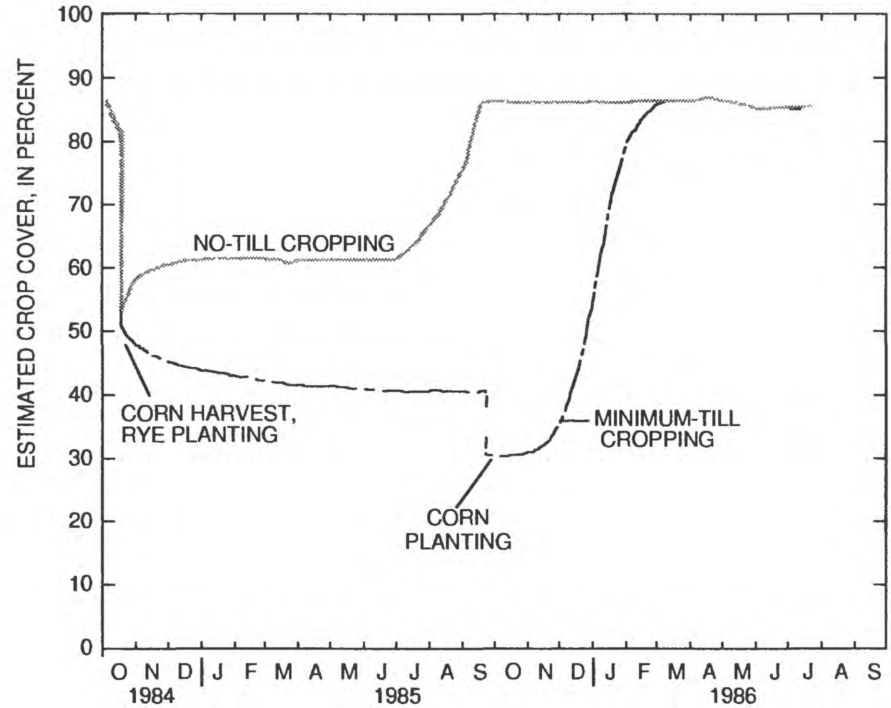


Figure 7. Estimated annual crop-cover cycle for no till and minimum till at Field-Site 2 during the 1985-86 water years.

Tillage

The conservation tillage methods of minimum till and no till were employed during the pre-BMP period. Minimum till consisted of chisel plowing in the fall and cultivating before spring planting. No-till methods involve no direct tillage of the soil. Rather, the soil is kept under continuous crop cover permitting natural processes to maximize infiltration and minimize erosion. A winter cover crop of rye was planted after harvest and remained on the field until spring planting. The rye was then killed with herbicide and corn was planted through the rye residue. Minimum-tillage practices were used on approximately 18 acres having low or no slopes. On higher-slope areas, no-till practices were used. The timing of tillage practices were recorded in data supplied by the farmer. During the pre-BMP period, the tillage practices did not change.

Tillage practices and water quantity and quality are related primarily by the effect the former has on infiltration and erosion. The processes by which tillage affects infiltration and erosion, although thoroughly investigated, are not easily quantified (Baker, 1987). During the pre-BMP period, the pipe-drained terraces were cultivated almost exclusively in no till; therefore, tillage induced variations in surface-water quantity and quality were expected to be minimal. However, the ground-water wells were located in both no-till and minimum-till areas.

Fertilization

Fertilization at the site was accomplished by use of manure and commercial fertilizer. The manure was of three types—hog manure from gestation and finishing operations, steer manure and bedding mixture from a feedlot, and poultry manure from a poultry house. The steer-manure mix and poultry manure were applied by surface spreading. Hog manure was injected into the soil 8 to 10 in. below the surface unless the soil was frozen, in which case, all manures were surface spread. Commercial fertilizers applied were ammonium sulfate, broadcast prior to planting, and liquid nitrogen co-applied with pre-emergence pesticides.

Other on-site agricultural activities contributed unknown amounts of nitrogen and phosphorus to the site.

The nutrient content of the fertilizers was established by laboratory analysis for the manures and by the labeled analysis for commercial fertilizer. Representative samples of the manures were collected about the time of major spring and fall field applications. Fertilizer analyses are listed in table 8.

Table 8. Average nitrogen, phosphorus, and moisture in samples of manure collected at Field-Site 2 and in commercial fertilizers

[--, not applicable]

Fertilizer type	Nitrogen (pounds per ton)	Phosphorus (pounds per ton)	Moisture (percent)
Hog manure	9.2	2.1	92
Steer manure	19	4.4	69
Poultry manure	66	26	36
Ammonium nitrate	670	0	--
Liquid nitrogen	600	0	--

The nitrogen and phosphorus load from each fertilizer application was calculated by multiplying the average nutrient content of the manure(s) or commercial fertilizer by the application amount. Figures 8 and 9 show the dates, amounts, and method of nitrogen and phosphorus applications to the 27 acres monitored for surface runoff and to the entire 47.5-acre site, respectively. Each bar in figure 8 represents a 3-day total. Individual nitrogen and phosphorus applications were summed to yield the total annual nitrogen and phosphorus applications (table 9).

Annual nitrogen and phosphorus applications to the entire site decreased from 1985 to 1986 by 28 and 44 percent, respectively. Less poultry manure was produced in 1986 than in 1985, and other manure was stockpiled or exported from the site during 1986.

If fertilizer had been uniformly distributed over the site, 57 percent of the annual nitrogen and phosphorus load would be placed on the pipe-drained terraces. In 1985, the pipe-drained terrace received 64 percent of the nitrogen and phosphorus applied. In 1986, the pipe-drained terraces received 75 percent of the nitrogen and 76 percent of the phosphorus load (table 9).

Nitrogen and phosphorus application rates varied substantially across the site. Sources of the variation were (1) single applications typically covered no more than 5 to 10 acres and subsequent applications often overlapped previous applications; (2) field conditions and the presence of crops at times restricted the placement of manure applications; (3) field areas judged by the farmer to be 'nutrient poor' received larger applications; and (4) different manure types were applied at differing rates to different areas. Estimates of the average annual nitrogen and phosphorus application rates for different areas of the field site are shown on figures 10 and 11. In 1985 (fig. 10), application rates varied over a 30-to-1 range; the minimum was less than 50 lb of nitrogen and 10 lb of phosphorus per acre and the maximum rate was about 1,400 lb of nitrogen and 350 lb of phosphorus per acre. The maximum application rate occurred when large quantities of manure accumulated during the growing season were applied on a harvested tobacco plot. In 1986, application rates (fig. 11) were generally smaller and had less spacial variation. These smaller application rates reflect the reduced manure production in 1986. On average, the greatest application rates were on the pipe-drained terraces. In both 1985 and 1986, field areas nearest Indian Run received low nitrogen (less than 200 lb) and phosphorus applications.

The recommended fertilization rate for a corn grain yield of 175 bushels per acre is 230 lb of nitrogen and 55 lb of phosphorus per acre (the Pennsylvania State University, 1980). In 1985, actual application rates, over 60 percent of the site, exceeded the recommended rate by a factor of two or more. In 1986, application rates were between one and two times the recommended rate over 50 percent of the site and at or below the recommended rate for the remaining area.

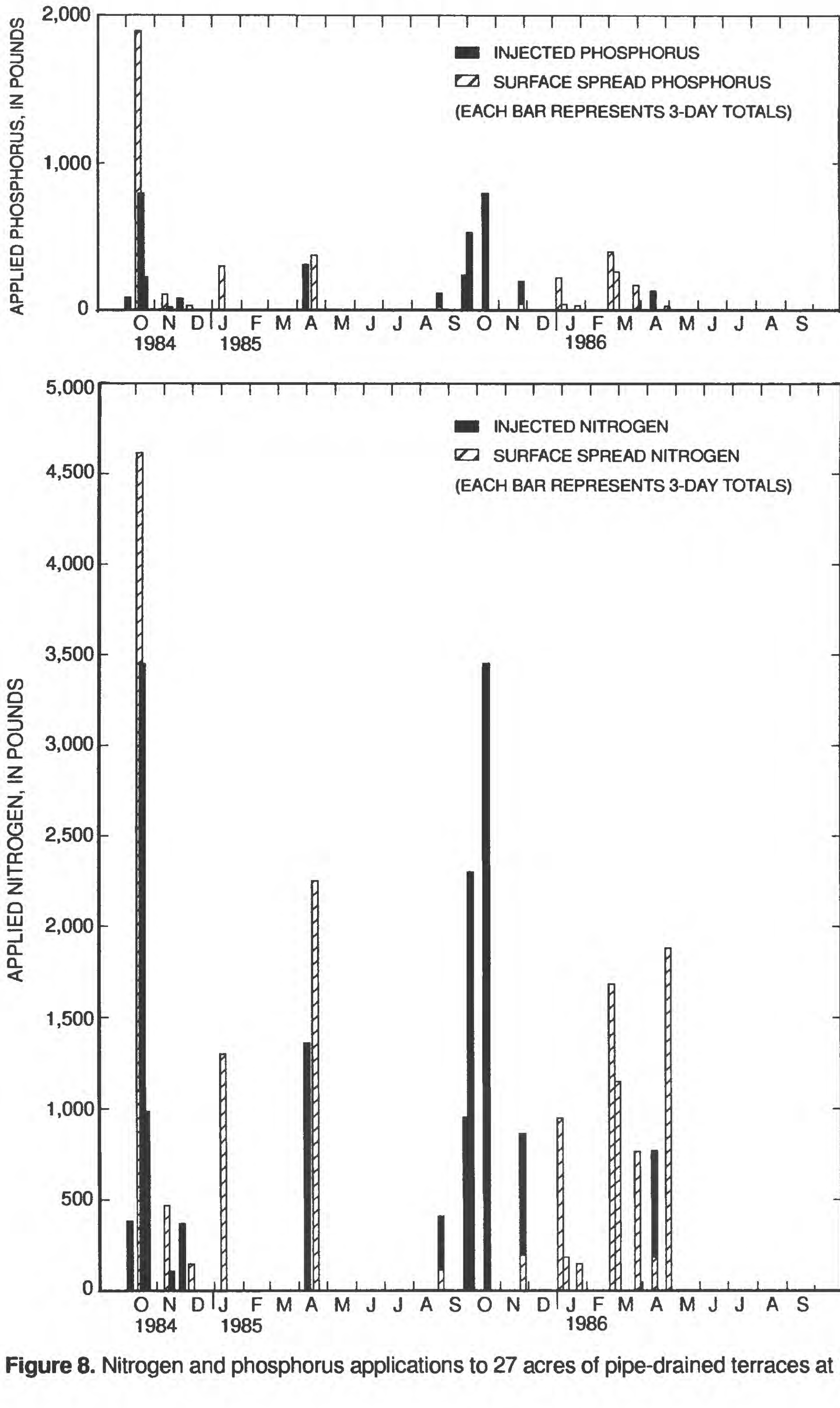


Figure 8. Nitrogen and phosphorus applications to 27 acres of pipe-drained terraces at Field-Site 2.

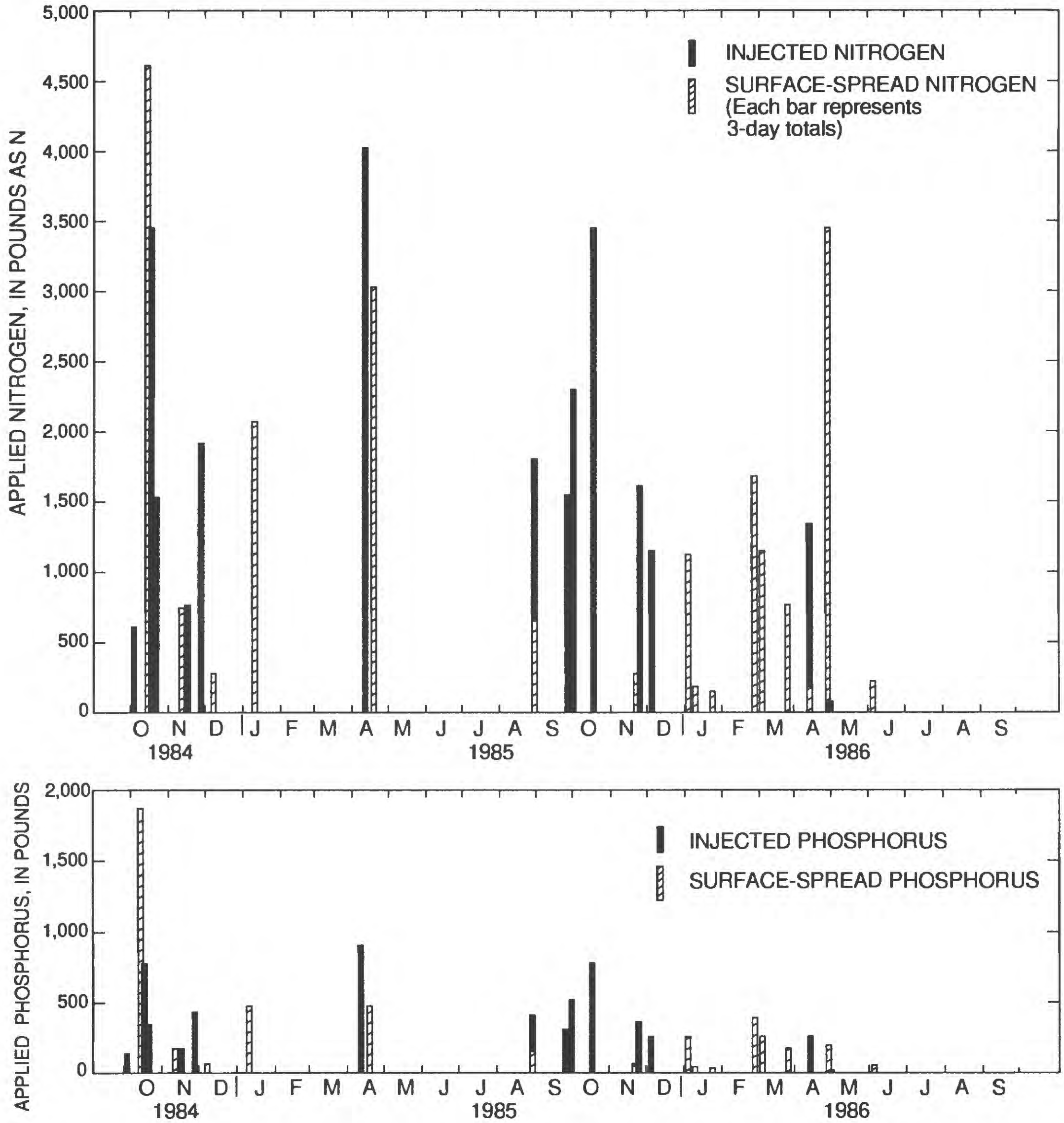
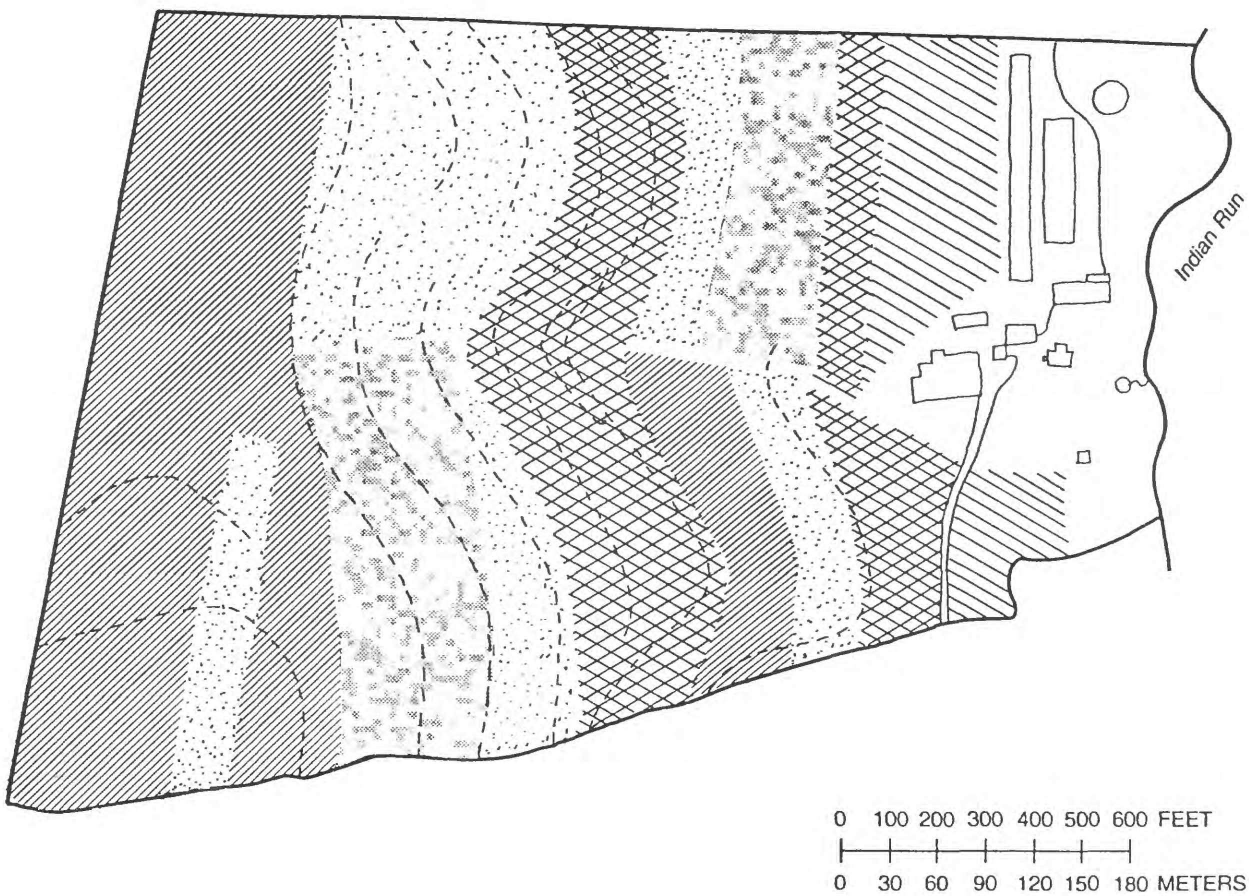


Figure 9. Nitrogen and phosphorus applications to the entire 47.5-acre Field-Site 2.

Table 9. Annual application of nitrogen and phosphorus to the entire 47.5-acre site and to the 27-acre pipe-drained terraces at Field-Site 2

	Annual applications, in pounds			
	1985		1986	
Fertilizer type	Nitrogen	Phosphorus	Nitrogen	Phosphorus
Steer	4,490	1,060	4,490	1,060
Hog	16,200	3,690	11,700	2,660
Poultry	4,610	1,880	0	0
Commercial	960	0	2,800	0
Total, entire site	26,300	6,630	18,900	3,720
Total, pipe-drained terraces	16,800	4,250	14,200	2,820



EXPLANATION

APPLICATION RATES, IN POUNDS PER ACRE

	Nitrogen	Phosphorus
	0-200	0-50
	200-400	50-100
	400-600	100-150
	600-800	150-200
	800-1400	200-350
	Noncropped	Noncropped
- - - - -	TERRACE CREST	

Figure 10. Nitrogen and phosphorus application rates at Field-Site 2 in 1985.

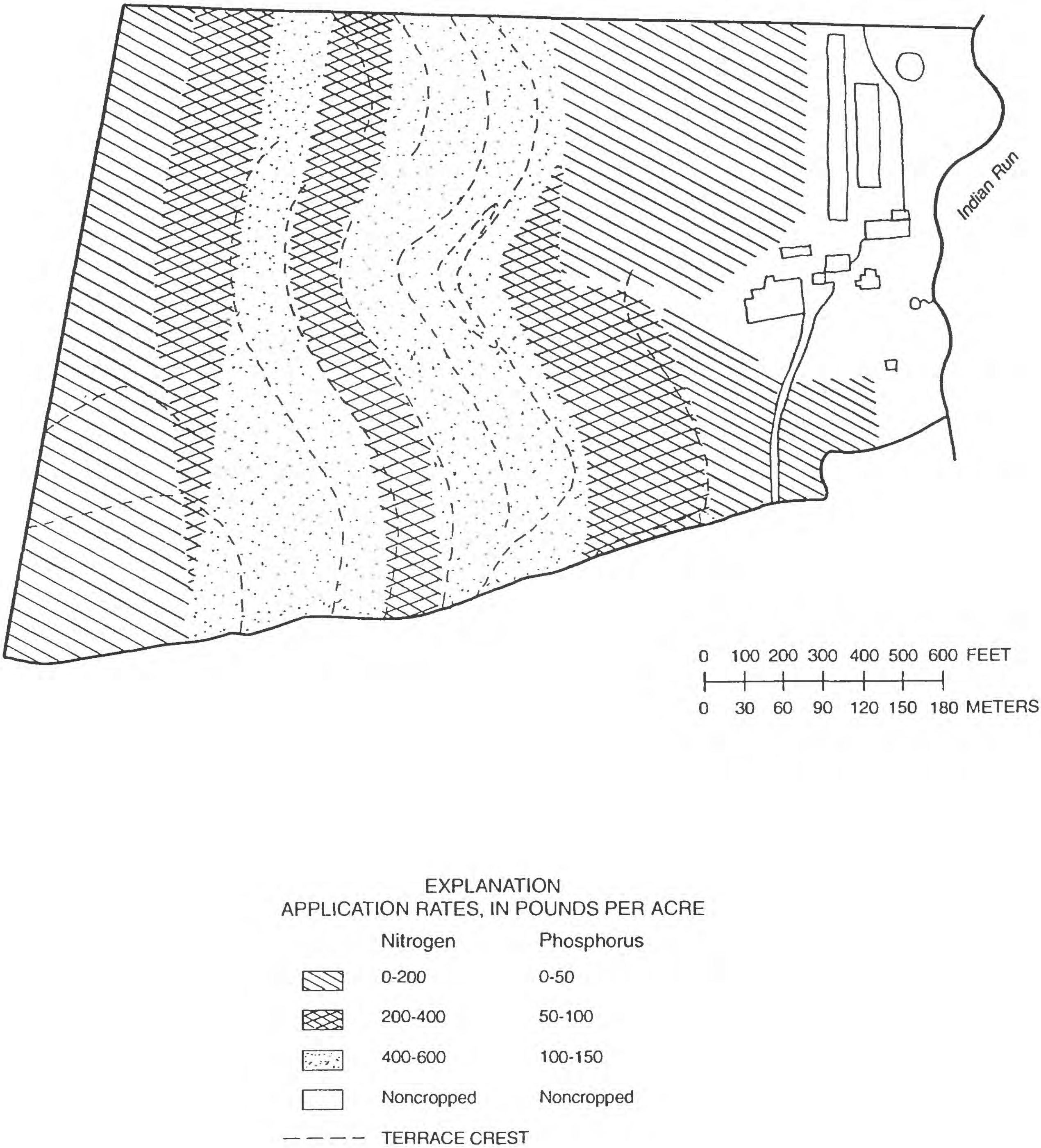


Figure 11. Nitrogen and phosphorus application rates at Field-Site 2 in 1986.

Soil

Because surface runoff and ground-water recharge are in contact with the soil, soils at the site were characterized by their physical structure, tilth, and chemical content. Additionally, the soil serves as a medium for nutrient transformation, storage, and release.

Soils at the site are classified as Hagerstown silt loams and silty-clay loams. These soils, formed in residuum from weathered limestone, are typically deep and well-drained (Custer, 1985). Soil depth to bedrock ranges from 5 to 30 ft. Particle-size analysis of the top 2 in. of soil shows a silt content of 56 percent and a clay content of 27 percent. Clay content of the upper 6 in. typically ranges from 13 to 25 percent and increases to 50 percent at 20 in. From there, clay content remains about 50 percent to 60-in. depth. Clay content decreases below 60-in. depth (Custer, 1985).

The relative amount of soil compaction was dependent primarily on farming practices. The soil under no-till cultivation (27 acres) was disturbed minimally twice a year. In the fall and early winter, slits approximately 1.5 in. wide, 8-10 in. deep, and about 4 ft apart were made to inject liquid manure; in the spring, bands approximately 2 in. wide, 1-2 in. deep, and about 30 in. apart were made when corn was seeded. The soil under minimum till (18 acres) was disturbed more than no-till soil. In the late fall, the soil was loosened to a depth of 6-7 in. in rows 4 ft apart by chisel plowing. Slits for manure injection were made as described above. Prior to planting in the spring, the soil was loosened to a depth of about 2 in. by cultivation.

Soil pH and cation-exchange capacity vary with depth from the surface. The pH ranges from 4.5 to 6.5 with the maximum pH measured in the upper 20 in. Below 20 in., pH decreases and approaches 4.5 at 80 in. depth. Cation-exchange capacity is between 15 and 20 milliequivalents per 100 grams down to 60 in. Below 60 in., cation-exchange capacity decreases (Custer, 1985).

Soil cores to depths of 2 and 4 ft were collected prior to the growing season, after harvest, and during the period of active crop growth. Median concentrations of nitrate in the soil ranged from 116 to 161 lb/acre for the 0- to 2-ft sample cores and from 177 to 329 lb/acre for the 0- to 4-ft sample cores (table 10). Median concentrations of soluble phosphorus in the soil ranged from 8.1 to 40.8 lb/acre for the 0- to 2-ft depth and from 8.5 to 35 lb/acre for the 0- to 4-ft depth. (Reported phosphorus concentrations may be converted to phosphate (P_2O_5) by multiplying the phosphorus concentration by 2.3.) Median concentrations of soluble nitrate and phosphorus in the soil for both the 0- to 2-ft and 0- to 4-ft cores were tested for significant differences between sampling dates. Median concentrations of nitrate were not significantly different during the pre-BMP period (Wilcoxon-Mann-Whitney test). Median phosphorus concentrations increased significantly through all samplings except the October 1986 sampling, which showed a significant decrease.

Table 10. Summary statistics of soluble nitrate and phosphorus content of soil samples from Field-Site 2

[values are in pounds per acre as nitrogen or phosphorus; ft, foot; Min, minimum; Max, maximum; --, no data]

		Soluble nitrate						Phosphorus					
	Number of	0- to 2-ft depth			0- to 4-ft depth			0- to 2-ft depth			0- to 4-ft depth		
Date	samples	Median	Min	Max	Median	Min	Max	Median	Min	Max	Median	Min	Max
November 24, 1984 [1]	3	122	107	133	177	166	272	8.1	4.2	10.7	8.5	4.6	11.1
October 8, 1985 [1]	8	161	67.0	250	329	128	428	11.1	4.7	52.7	11.6	5.0	53.0
April 1, 1986 [2]	17	118	31.7	336	224	76	633	31.9	4.0	71.8	34.8	4.3	72.3
June 24, 1986 [3]	15	128	53.6	249	--	--	--	40.8	4.7	71.6	--	--	--
October 21, 1986 [1]	9	116	61.5	222	215	104	364	26.3	2.2	44.9	26.5	2.4	45.0

[1] After harvest.

[2] Prior to planting.

[3] During growing season.

A profile of the median concentrations of soluble nitrate and phosphorus in the soil by depth is shown in figure 12. Although there was considerable variation in the nitrate concentrations of individual samples in any specific depth segment, median concentrations of nitrate showed, on the average, a gradual decrease with increasing depth. The only substantial departure from this pattern occurred in the October 1985 sampling when the 8- to 24-in.-depth nitrate concentrations exceeded the 0- to 8-in.-depth concentrations. Seven days prior to this sampling, 2,800 lb of manure nitrogen were injected to a depth of 8- to 10-in. Median concentrations of soluble phosphorus in the soil were near or below detection limits for all but the 0- to 8-in.-depth segment. Binding of the phosphorus with soil particles near the surface prevented phosphorus movement beyond the 0- to 8-in. depth.

Correlations between measured nutrient concentrations in the soil and nutrient applications in a given area were generally poor. Some clustering of large nitrate concentrations was noted, however, in the northeast corner of the site. As previously mentioned, this area was the site of an anhydrous ammonia spill in September 1983. The spill was estimated to have contained up to 2,700 lb of nitrogen. Concentrations of soluble phosphorus did not show the same spacial distribution as the nitrate concentrations. But again, the northeast corner did appear to have marginally greater phosphorus concentrations than the remainder of the site. The poor correlation between nitrate concentrations in the soil and nutrient applications is probably caused by a variety of factors. Some of these factors are (1) nitrification processes that affect nitrate nitrogen speciation in the soil without greatly affecting total nitrogen content; (2) small-scale irregularities in manure application rates (particularly with injected manure and bedding/manure mixes); (3) variability in the nitrate to total nitrogen ratio of manure; and (4) local variation in soil composition and moisture (Schmidt, 1982). Because of these factors, the small number of samples analyzed for nitrate concentrations, except in April and June 1986, probably did not adequately represent average concentrations of nitrate in the soil at the site.

Median concentrations of nitrate in the soil measured during the pre-BMP period were substantially greater than those recommended to allow satisfactory crop yields while minimizing movement of nitrate to the ground water. Rehm and others (1983) and Hallberg and others (1984) recommend a residual of 100 lb/acre of nitrate nitrogen in the top 4 ft. Recommendations for phosphorus are not available.

SURFACE RUNOFF

Water-quantity and water-quality data were used to characterize surface runoff from pipe-drained terraces at Field-Site 2. Water-quality, climate, and agricultural-activities data were examined for significant relations linking water quality to nutrient application. Because conservation practices to limit runoff from the site were in place at the start of the study, runoff from the pipe-drained terraces accounted for only a small percentage of water and nutrient losses during the pre-BMP period.

Quantity

During the pre-BMP period, a total of 36 storms (rainfall, precipitation, snowmelt, or both) produced measurable runoff. In addition, another 10 storms produced a quantity of runoff that was too small to measure.

Annual and Monthly Runoff

Runoff ranged from 10 to 48,000 ft^3 during 15 events in the 1985 water year and from 15 to 12,500 ft^3 during 21 events in the 1986 water year. The total, mean, and peak discharges for runoff events are shown in figure 13.

In 1985, the annual runoff was 60,900 ft^3 or 1.7 percent of the annual precipitation. Ninety-five percent of the annual runoff occurred during three storms that delivered 20 percent of the 1985 annual precipitation. A storm on February 12-13, 1985, under conditions of frozen soil and snowcover, produced 79 percent of the annual runoff. An additional 9 percent ran off during a tropical storm on September 27, 1985. The remaining 7 percent ran off during a July 31, 1985, storm that had rainfall intensities exceeding 1.2 in. in 30 minutes. Of the remaining 12 storms with runoff, none had amounts greater than 1 percent of the 1985 annual runoff.

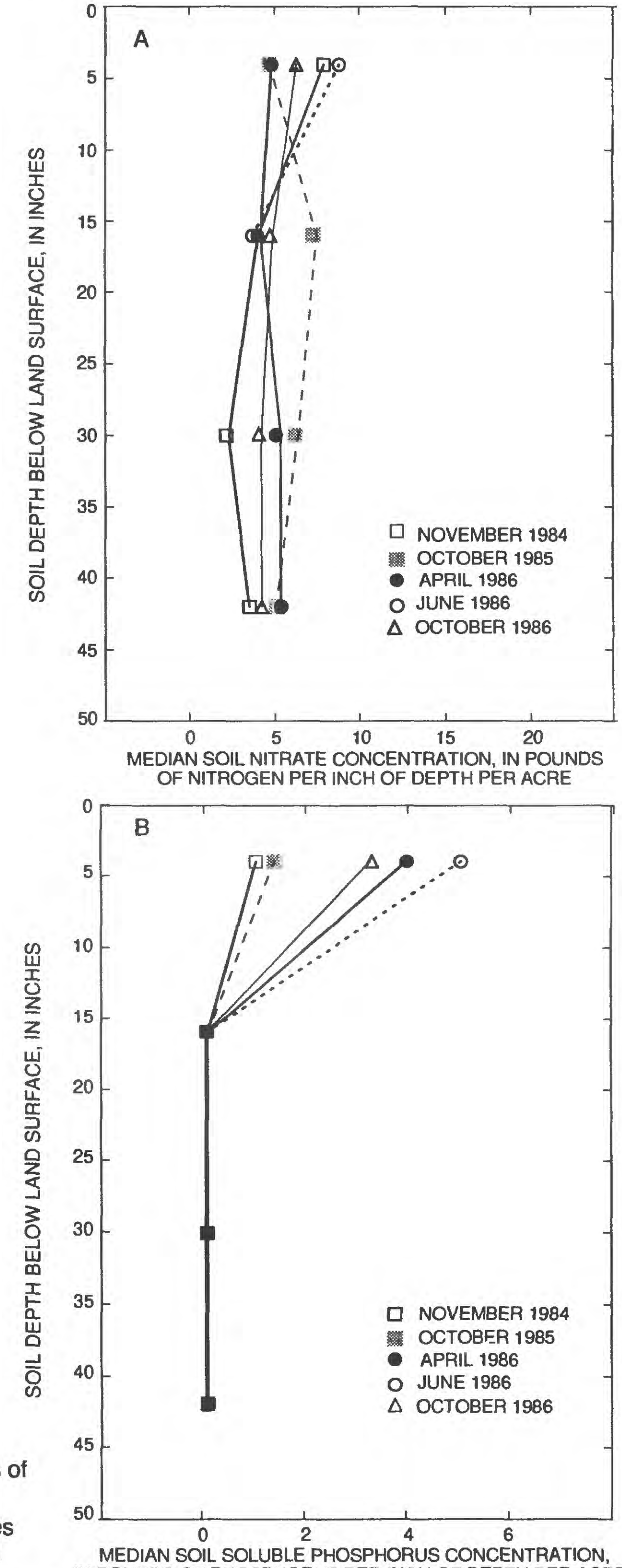


Figure 12. Median concentrations of soluble nitrate (above) and phosphorus (below) in soil samples from Field-Site 2, November 1984 through October 1986.

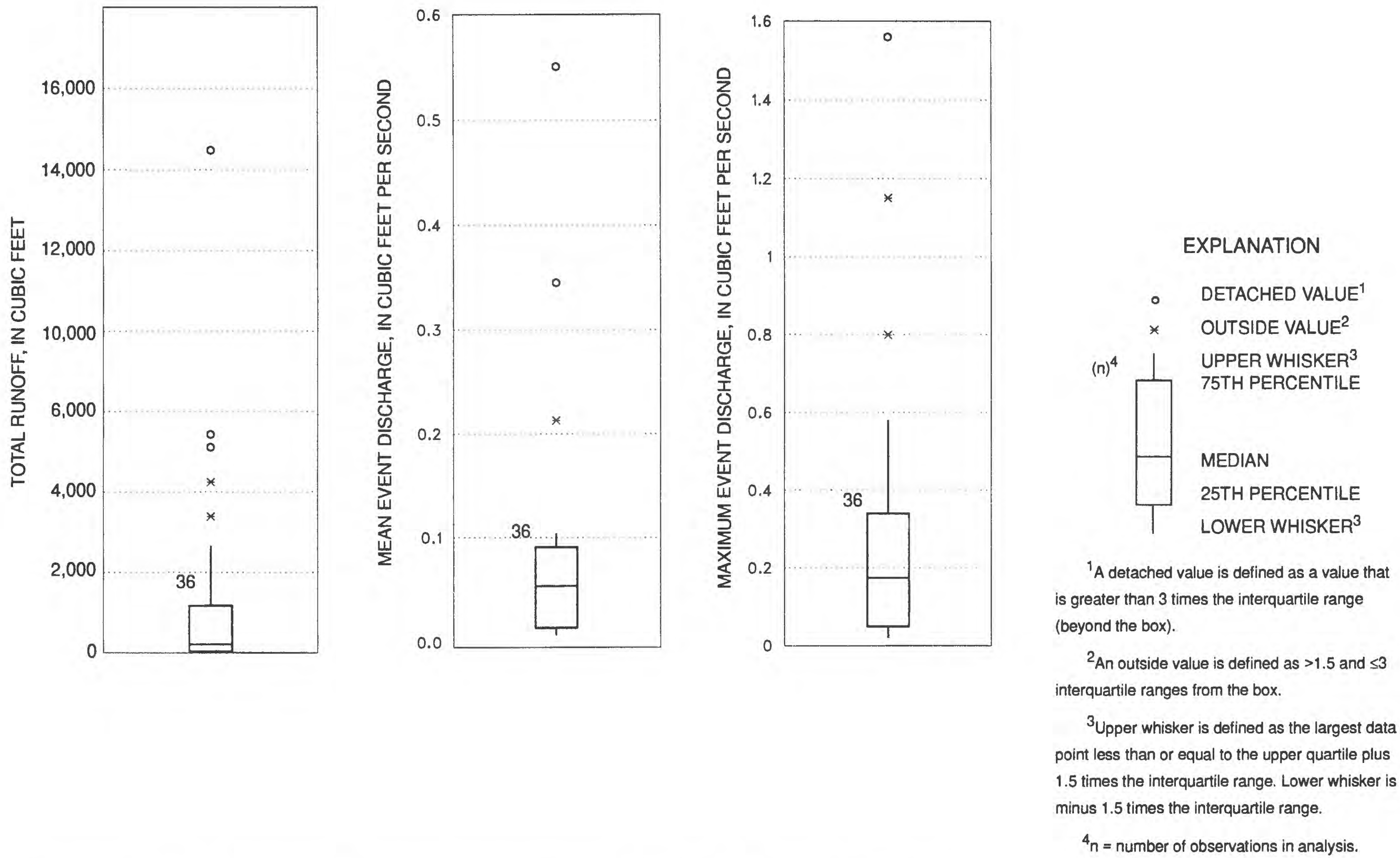


Figure 13. Distribution of total runoff and mean and maximum discharges for runoff events from the pipe-drained terraces at Field-Site 2, October 1984 through September 1986.

In 1986, annual runoff was 30,000 ft^3 or 0.8 percent of the 1986 annual precipitation. Eighty-eight percent of the annual runoff occurred during six storms that delivered 15 percent of the 1986 annual precipitation. Five storms in January and February 1986 on frozen snow-covered soils accounted for 46 percent and a July 26, 1986, thunderstorm that had 30-minute rainfall intensities of up to 1.1 in. accounted for the other 42 percent of the annual runoff. Of the remaining 15 storms that produced runoff, none accounted for more than 3 percent of the 1986 annual runoff.

Monthly runoff for 1985 and 1986 are shown in figure 14. The maximum monthly runoff occurred in February 1985. There were 7 months when no measurable runoff occurred.

At Field-Site 2 during the pre-BMP period, runoff was typically less than 0.5 percent of storm precipitation. Median runoff for runoff events during the pre-BMP period was 202 ft^3. Only under conditions of frozen soil or intense precipitation, with frozen soil having the greatest effect, did runoff exceed 1 percent of storm precipitation.

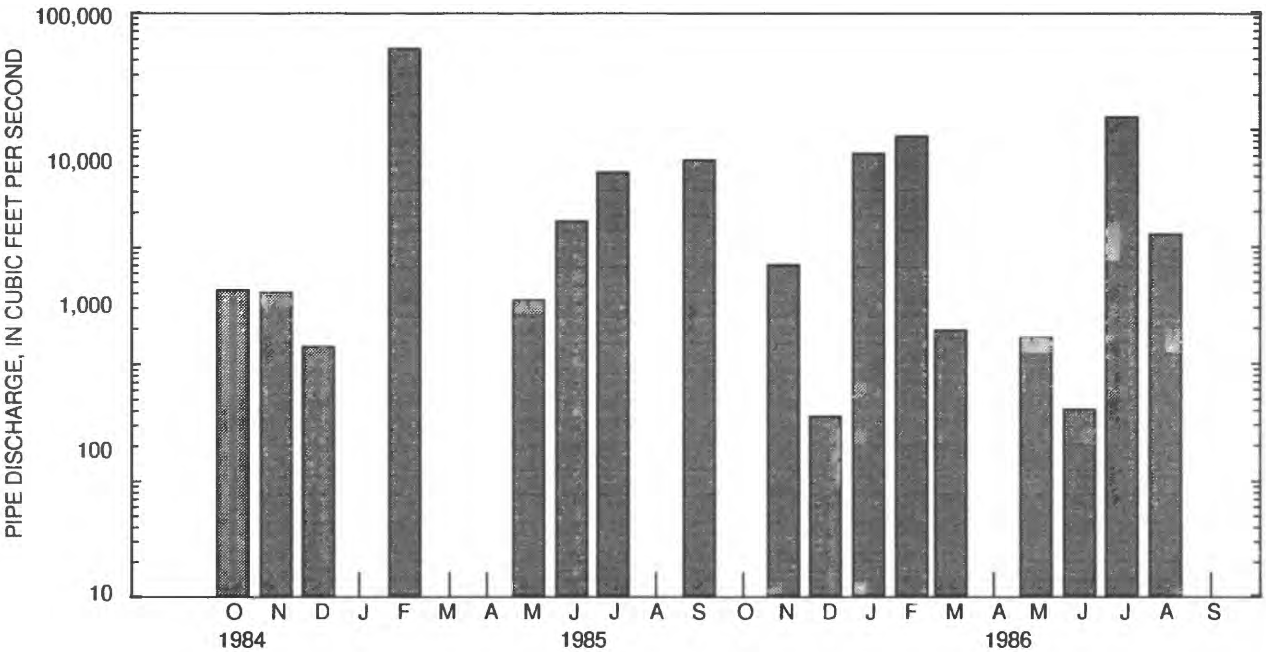


Figure 14. Monthly surface runoff from the pipe-drained terraces at Field Site 2.

Relation to Climate and Agricultural Activities

The relation between runoff and precipitation is dependent, in part, on the following factors: precipitation type, intensity, and duration; and prevailing land-surface conditions. For example, for a given amount of precipitation, an intense cloudburst on fallow, compacted soil would be expected to result in a greater percentage of runoff than would result from precipitation of moderate intensity falling on recently tilled or fully cropped soil. Moreover, in an agricultural setting, cultivation practices can radically alter land-surface conditions in a short period of time.

Variables that 'explain' the variation in runoff were identified by use of graphical, correlation, and regression methods. Total event runoff and the logarithm of runoff were plotted against precipitation quantity, maximum 30-minute precipitation intensity, precipitation duration, and percent crop cover. Of these variables, the logarithm of event runoff was positively correlated with precipitation quantity (fig. 15); the other variables were not significantly correlated with runoff. Further, when runoff data were grouped by soil condition (thawed or frozen) the correlation between the logarithm of event runoff and precipitation quantity was stronger for both groups individually than combined; Pearson correlation coefficients are shown in table 11. Additional grouping of runoff data by estimated soil-compaction conditions revealed no discernible correlations.

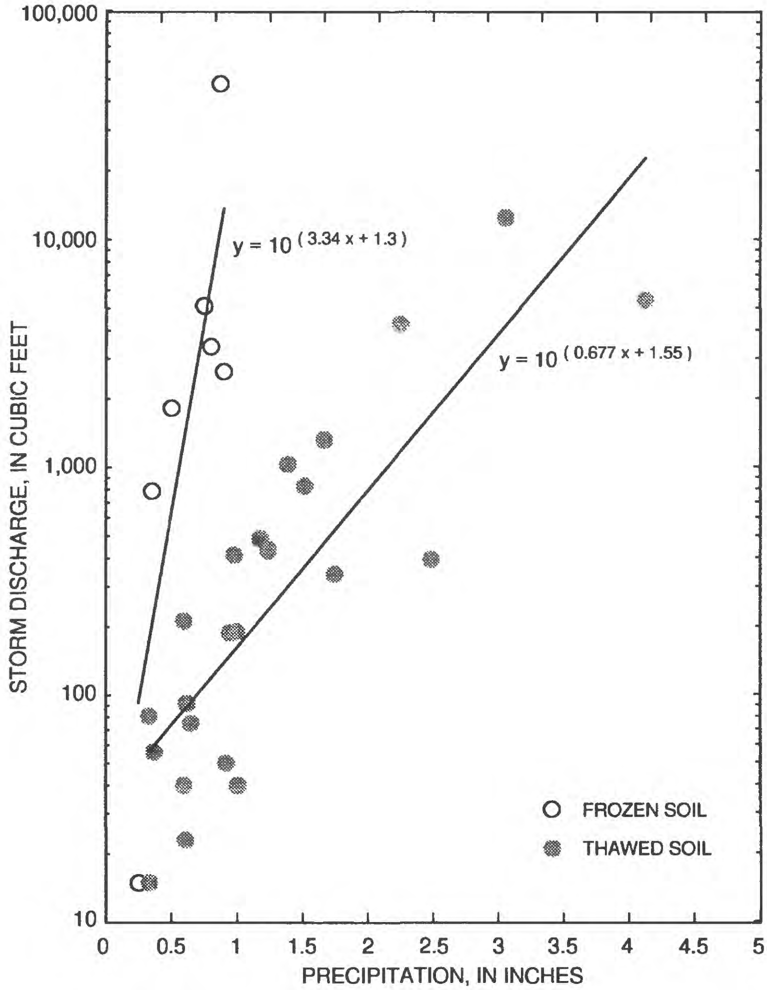


Figure 15. Relation between runoff from the pipe-drained terraces at Field-Site 2 and precipitation for runoff events occurring when soil was thawed or frozen, excluding runoff from snowmelt only, October 1984 through September 1986.

Table 11. Correlation coefficients for the relation between the logarithm of event runoff and precipitation quantity for Field-Site 2, October 1984 through September 1986

Data set	n	Person correlation coefficient (r)
Frozen and thawed soil runoff events	29	0.57
Thawed soil runoff	22	.87
Frozen soil runoff	7	.87

Because graphical method of analysis allows only one explanatory variable at a time to be examined, linear multiple regression was used to evaluate complex multiple variable relations. On the basis of the strong distinction between the amount of runoff produced from storms occurring when the soil was thawed or frozen, separate regression models were run for runoff events occurring during each soil condition; precipitation quantity, maximum 30-minute intensity, precipitation duration, and percentage crop cover were the independent variables. As with the plotting technique, precipitation quantity was found to be the only significant explanatory variable of event runoff (table 12). The predicted runoff from precipitation on thawed and frozen soil is indicated by the regression equations shown in figure 15. Precipitation quantity explained about 69 and 71 percent of the variation in the logarithm of thawed- and frozen-soil storm discharges, respectively.

Table 12. Regression statistics for the log of runoff in cubic feet as a function of precipitation quantity in inches for runoff events at Field-Site 2, October 1984 through September 1986

[<, less than]

Dependent variable	Data set	n	Regression coefficient: Precip-itation	Regression coefficient: t-test	Regression coefficient: p-value	Intercept	Coefficient of determination (adj. R^2)[1]	Standard error: Log units	Standard error: Percent[2] Plus	Standard error: Percent[2] Minus
Log of runoff	Thawed soil	23	0.677	7.128	<0.001	1.554	0.69	0.42	163	62
Log of runoff	Frozen soil	7	3.337	3.238	<.023	1.132	.61	.66	357	78

[1] Coefficient of determination (R^2) adjusted for degrees of freedom.
[2] G.D. Tasker, U.S. Geological Survey, written commun., 1978.

While noting that frozen-soil runoff events were few in number and were in six of seven cases accompanied by existing snow cover, it can be seen that, as precipitation quantities increased, event runoff on frozen soil increased at an exponentially faster rate than runoff on thawed soil. By use of regression estimates, 1 in. of precipitation on frozen soil with snow cover would have resulted in 10 to 100 times more discharge compared to 1 in. of precipitation on thawed soil. Thus, storms occurring when soils are frozen have the potential for transporting extremely large loads per inch of precipitation.

Quality

Water-quality samples of surface runoff were collected during 22 of the 36 runoff events in the pre-BMP period. Runoff samples were collected at 30-minute intervals when the stage in the flume measured 0.05 ft or greater. Concentrations of total nutrients were determined for 125 instantaneous samples collected during 22 runoff events; of these samples, concentrations of dissolved nutrients were determined for 23 samples collected during 10 runoff events. Suspended-sediment analysis was performed on 67 instantaneous samples collected during 13 runoff events.

Nitrite concentrations are not discussed separately from nitrite concentrations because nitrite concentrations reported for this study typically are one to two orders of magnitude lower than nitrate concentrations.

Reported concentrations of total ammonium and total nitrate plus nitrite represent primarily dissolved ammonium and dissolved nitrate plus nitrite concentrations. Samples for these constituents were allowed to settle prior to analysis and, after settling, the liquid part of the sample only was analyzed. For total ammonium, the analysis represents an unknown part of the total ammonium present in the sample because of an affinity between ammonium and soil particles. However, total nitrate plus nitrite is highly soluble, and therefore, the analysis closely represents a total concentration.

Instantaneous Sample Water Quality

Time-series plots showing the range and median concentrations of total nutrients for instantaneous samples collected from runoff events are shown in figure 16. The concentration of total ammonium and total ammonium plus organic nitrogen show a seasonal variation; maximum concentrations tend to occur during the fall and winter months and minimum concentrations tend to occur during the spring and summer months. To a lesser degree, concentrations of total nitrate plus nitrite and total phosphorus also show a seasonal variation. Concentrations generally are greater during the nongrowing season than during the growing season.

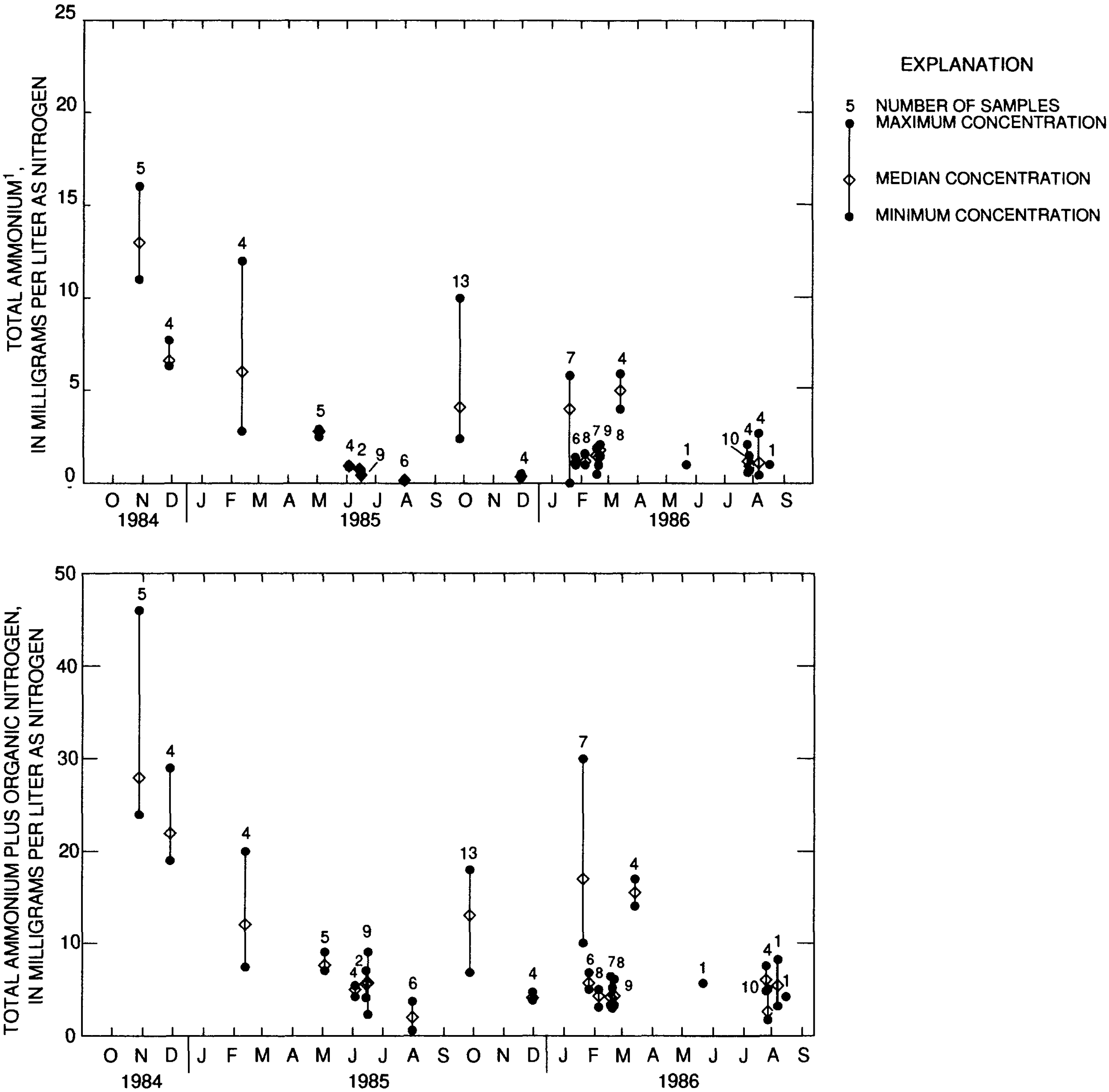
In the 23 samples analyzed for both total and dissolved nutrients, dissolved nutrients typically accounted for a large percentage of the total concentrations. Dissolved-ammonia nitrogen and dissolved nitrate plus nitrite nitrogen accounted for greater than 90 percent of their respective total concentrations in more than 75 percent of the samples because of the laboratory analysis method and because of the high solubility of nitrate in water. Dissolved ammonia plus organic nitrogen, which is partially derived from organic matter of varying solubilities, accounted for about 60 to 95 percent of the total concentrations. Dissolved phosphorus typically comprised about 75 to 90 percent of the total phosphorus.

BMP's intended to control soil/sediment losses were already in place at the start of the study and little change was expected in average suspended-sediment yields after nutrient-management implementation. Thus, the measurement of suspended sediment was not a priority. However, because there often exists an association between suspended sediment and other water-quality constituents, suspended-sediment samples were collected during seven runoff events in 1985 and six in 1986. These events represented 92 and 45 percent of the annual pipe discharge for 1985 and 1986, respectively. The time-series plot in figure 17 indicates seasonal variation in suspended-sediment concentration. Although the pipe discharge originated primarily from the no-till area, the maximum concentrations of suspended sediment were associated with storms occurring shortly after spring planting and fall harvest. The relatively minor soil disturbances occurring at these times had a noticeable effect on suspended-sediment concentrations. Late-winter and early-spring storms typically had the lowest suspended-sediment concentrations. Storm loads of suspended sediment were generally small; the maximum load was 970 lb, and the median load was 20 lb. Loads for all but one storm were less than 160 lb.

The silt and clay fraction (shown below) was greater than 90 percent of the total suspended sediment for seven of the eight samples analyzed. The samples were mostly collected at or near discharge maximums when silt/clay fractions are typically near minimum.

		Percent silt and clay (0.62 millimeter or less)	
Number of samples	Median	Minimum	Maximum
8	98	75	100

Correlation analysis of 27 instantaneous runoff samples from Field-Site 2 analyzed for both suspended sediment and total nutrients resulted in no significant relations between concentrations of these constituents. The large percentage of nutrients found in the dissolved phase, which is not associated with suspended matter, supports the lack of correlations. At a 23-acre field site in Lancaster County, Lietman and others (1996) found correlations between total organic nitrogen, total phosphorus, and suspended-sediment concentrations in runoff. It was suggested that the presence of nutrient-bearing organic matter in the sediment and the binding of nutrients, phosphorus in particular, to suspended silt and clay particles were likely causes for the correlations at that site. Possible reasons for the differences between the two sites are (1) lower slopes and the use of conservation tillage at Field-Site 2 resulted in suspended-sediment concentrations that are typically much less than those reported by Lietman and others (1996); and (2) the liquid hog manure and poultry manure applied at Field-Site 2 did not include the large amounts of bedding material, which does not readily dissolve, found in dairy manure applied at the other field site.



[1]Total ammonium is ammonium in the liquid portion of an unfiltered, settled sample.

Figure 16. Distribution of instantaneous nutrient concentrations in runoff from the pipe-drained terraces at Field-Site 2, October 1984 through September 1986.

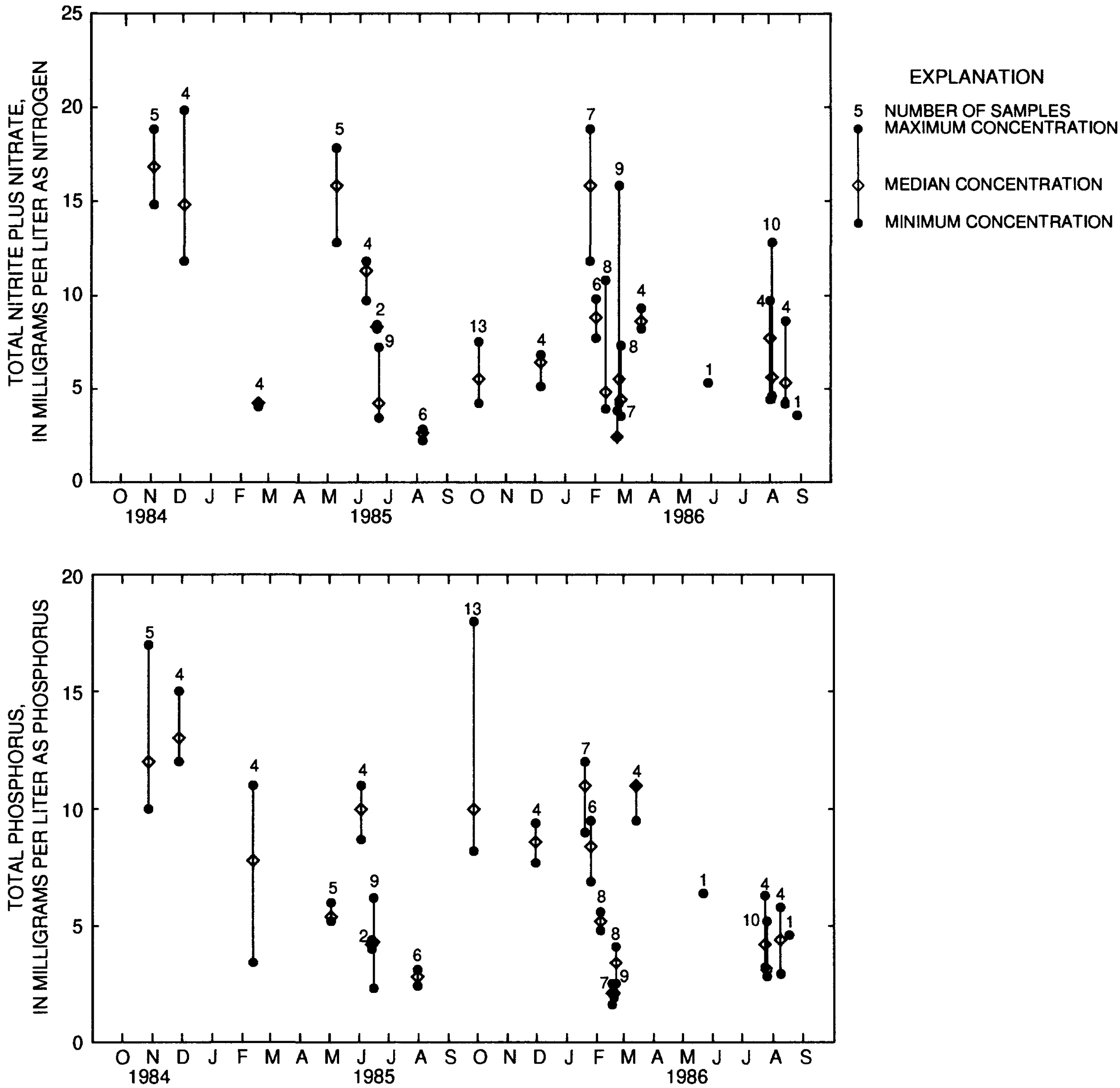


Figure 16. Distribution of instantaneous nutrient concentrations in runoff from the pipe-drained terraces at Field-Site 2, October 1984 through September 1986--Continued.

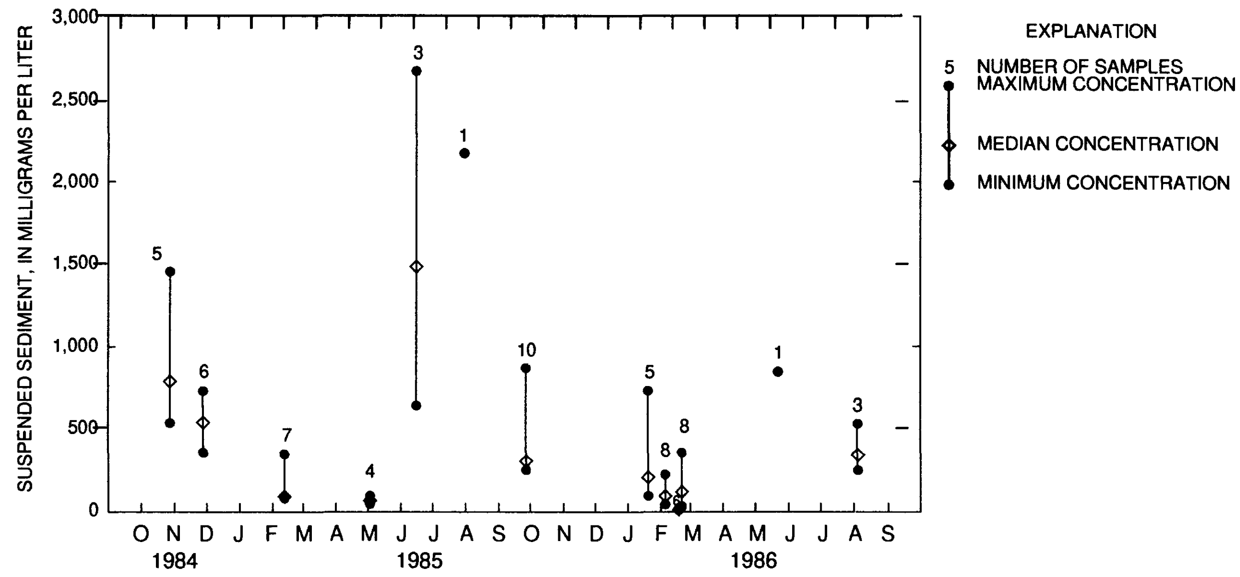


Figure 17. Distribution of instantaneous suspended-sediment concentrations in runoff from the pipe-drained terraces at Field-Site 2, October 1984 through September 1986.

Variation During Runoff Events

The quality of storm runoff results from the complex interaction of many variables, including runoff duration, precipitation quantity and intensity, soil-moisture conditions, temperature, and nutrient availability. These variables differ not only for each runoff event but also change during the course of an event. For many of the pre-BMP runoff events at the site, these changes were observed to follow a characteristic pattern. For most events, nutrient concentrations peaked early in the rising stage of the event, dropped to a minimum shortly after maximum discharge, and became constant in the recession stage. For runoff events of long duration or on saturated soil, concentrations of nitrate and total phosphorus increased late in the recession. Occasionally the recession stage was punctuated by peaks in one or more nutrient concentrations. Examples of the temporal variations in runoff water quality over the course of three storms—October 28, 1984, February 19-20, 1986, and July 26, 1986—are shown in figures 18-20.

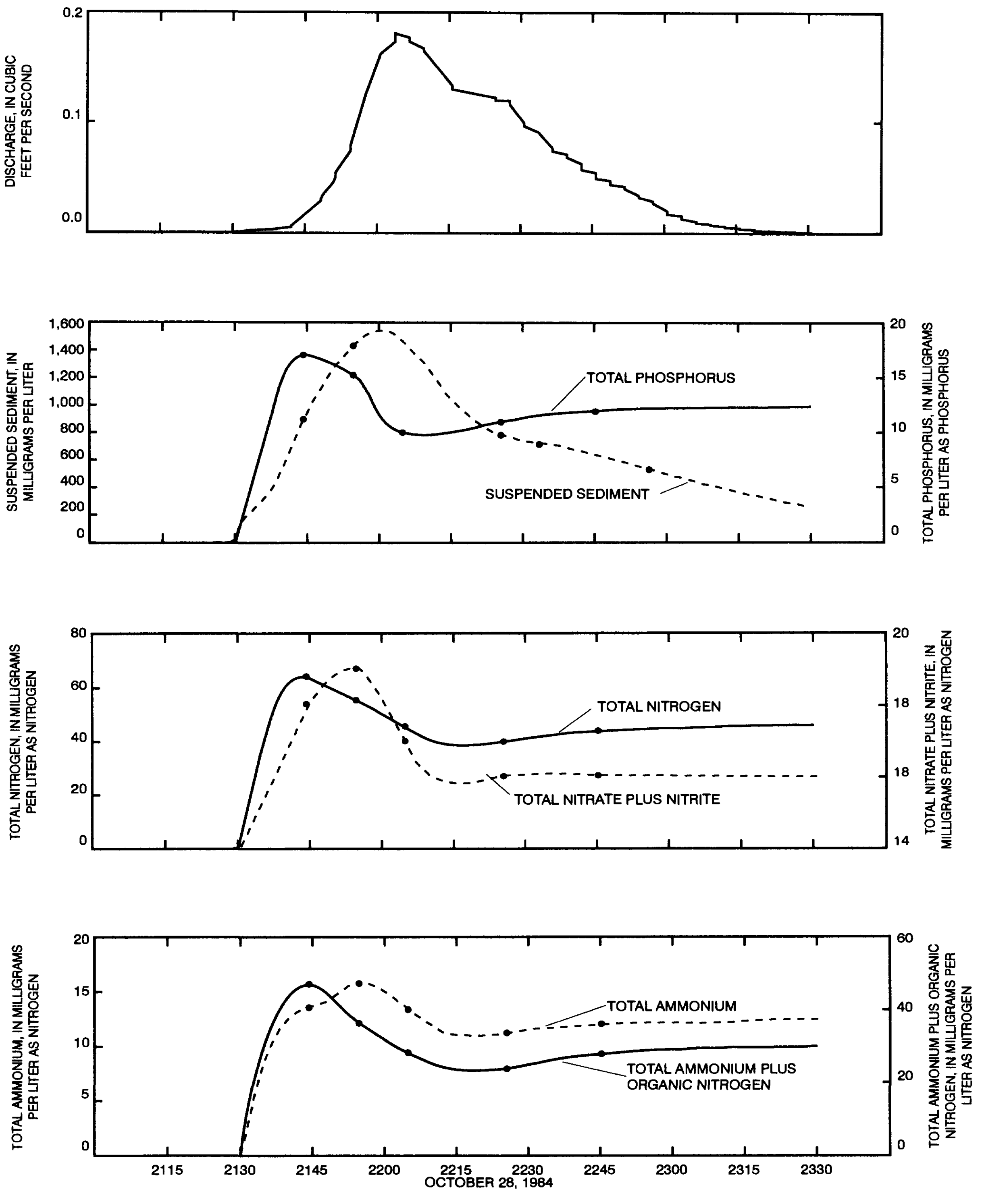


Figure 18. Discharge and nutrient and suspended-sediment concentrations in runoff from the pipe-drained terraces at Field-Site 2 during a storm on October 28, 1984.

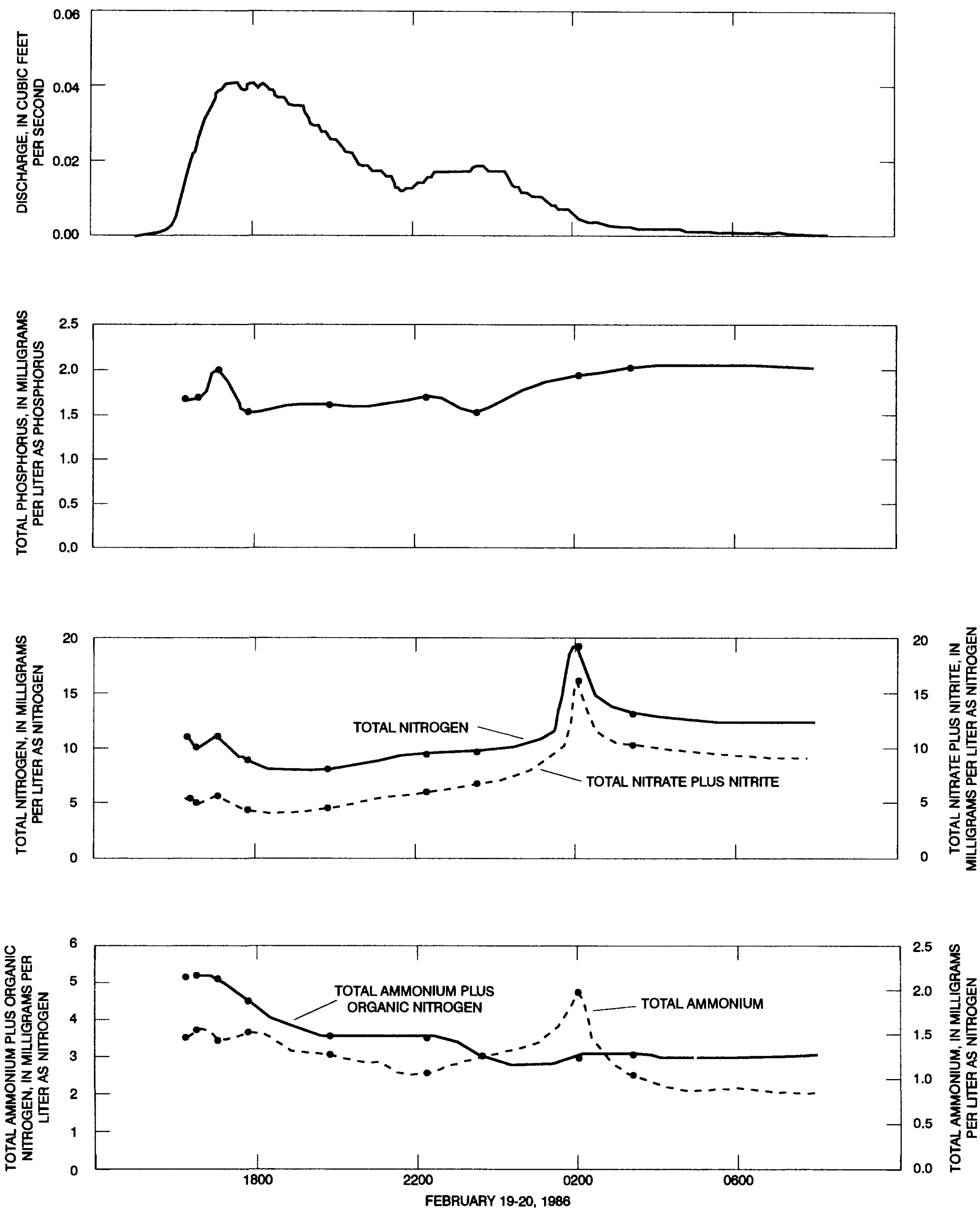


Figure 19. Discharge and nutrient concentrations in runoff from the pipe-drained terraces at Field-Site 2 during a storm on February 19-20, 1986.

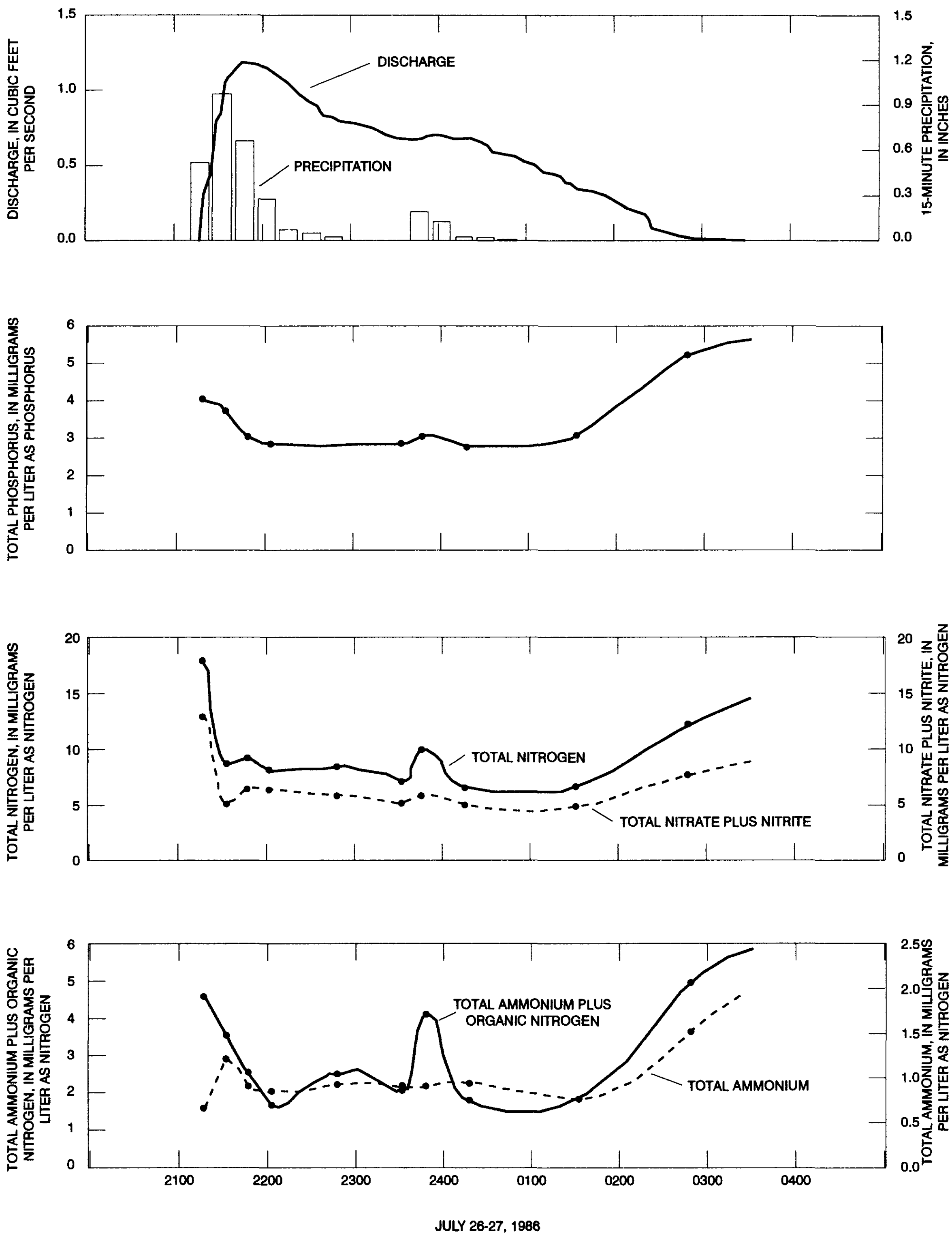


Figure 20. Discharge, precipitation, and nutrient concentrations in runoff from the pipe-drained terraces at Field-Site 2 during a storm on July 26-27, 1986.

A 1.0-in. storm occurring during 2 hours on October 28, 1984, produced 420 ft^3 of runoff (fig. 18). Precipitation intensity, as estimated from hydrograph slopes, was moderate compared to other storms. Soils were wet but not saturated; 0.66 in. of rain had fallen in the 7 days prior to the storm, but the 3 days immediately preceding were dry. Daily mean temperature was 68°F. In the 2-week period prior to the storm, 4,600 lb of nitrogen and 1,900 lb of phosphorus were surface applied and 5,000 lb of nitrogen and 1,400 lb of phosphorus were injected. This was the largest nutrient application prior to a discharge event during the pre-BMP period. Measured concentrations of total nitrogen in runoff ranged from 40 to 64 mg/L. Total ammonium plus organic nitrogen accounted for between 60 to 72 percent of the total nitrogen; the greatest percentage was during the early stages of discharge. As discharge increased, total ammonium plus organic nitrogen decreased, reaching a minimum concentration during mid-recession. In the late-recession phase, concentrations of total ammonia plus organic nitrogen increased slightly. The transport of easily dislodged organic matter at the onset of the storm followed by dilution at larger discharges could explain the behavior of total ammonia plus organic nitrogen concentrations during the storm. Concentrations of total ammonium and total nitrate plus nitrite remained relatively constant throughout the storm, decreasing slightly during the recession phase. Measured concentrations of total phosphorus ranged from 11 to 17 mg/L. The temporal response of total phosphorus concentrations were similar to that of total ammonium and organic nitrogen, suggesting a similar source and transport mechanism for both.

On February 19-20, 1986, a 0.4-in. rainfall of approximately 17 hours duration and snowmelt from 2 in. of existing snow produced 800 ft^3 of runoff (fig. 19). Estimated maximum precipitation intensity was 0.1 in. per hour. The soil was saturated and, at the onset of the storm, frozen. Daily mean temperature was 35°F. Three earlier runoff events followed the last surface application of 150 lb of nitrogen and 45 lb of phosphorus on January 26, 1986. Measured concentrations of total nitrogen in runoff ranged from 8 to 19 mg/L and were dominated by the total nitrate plus nitrite fraction. Early in the event, concentrations of total nitrogen were 11 mg/L, with total ammonium plus organic nitrogen and total nitrate plus nitrite each contributing about half of the nitrogen. As in the October storm, concentrations of total nitrogen decreased as discharge increased. About 12 hours after discharge began the maximum concentrations of total ammonium (2.0 mg/L) and total nitrate plus nitrite (16 mg/L) were measured. Concentrations of total nitrate plus nitrite at the end of the storm were greater than at the beginning; at the end of the storm, the nitrate plus nitrite fraction accounted for about 75 percent of the total nitrogen.

The end-of-storm rise in nitrate plus nitrite is thought to result from an increasing percentage of nitrate-enriched subsurface water in the runoff. Alberts and Spomer (1985) found that on terraced no-till watersheds, nitrate concentrations in soil water flowing just below the soil surface can be substantially greater than nitrate concentrations in surface runoff. At Field-Site 2, such "subsurface flow" could be discharged with surface runoff by entering the terrace standpipes through subsurface perforations. For storms of short duration, such as the October 28, 1984, event, or for those occurring on drier soil, subsurface flow contributes minimally to discharge. However, under saturated-soil conditions nitrate-enriched subsurface soil water can readily move into the terrace standpipes, increasing nitrate concentrations in storm discharge. This effect is more pronounced late in the event when the surface-runoff component of discharge subsides.

Measured concentrations of total phosphorus ranged from 1.9 to 2.5 mg/L. The lowest concentrations were measured during the highest discharges.

The third storm occurred on July 26-27, 1986; 3.1 in. of rain with a duration of 6.5 hours produced 12,500 ft^3 of runoff (fig. 20). Maximum 30-minute precipitation intensity was 1.8 in. The soil was saturated; about 9 hours previous to this storm, 1 in. of rain fell causing 190 ft^3 of discharge. The daily mean temperature was 79°F. The last nutrient application prior to the event was 440 lb of nitrogen and 34 lb of phosphorus that was surface applied on April 28-29. A total of 8.5 in. of rain fell between this storm and the time of last application. Measured concentrations of total nitrogen in runoff ranged from 6.5 to 18 mg/L and were composed primarily of total nitrate plus nitrite. The greatest concentrations of total nitrogen occurred at the onset of discharge. The nitrate plus nitrite fraction was greater than 60 percent of the total nitrogen in all runoff samples. That percentage is considerably greater than that observed for other moderate duration thunderstorms recorded during the pre-BMP period and most likely results from the flushing of nitrate-enriched soil water remaining from the previous storm. After the initial peak, concentrations of nitrate plus nitrite decreased by about 50 percent and remained relatively steady until late in the recession when they began to increase. Concentrations of total ammonium plus organic nitrogen had the typical early peak, a secondary peak that coincided with an increase in rainfall, and a late recession increase to the maximum concentration of 5 mg/L near the end of the event. Concentrations of total ammonium nitrogen were relatively constant throughout the storm with the exception of a slight increase occurring near the end of the event. Measured concentrations of total phosphorus ranged from 2.8 to 5.3 mg/L. After an initial peak, total phosphorus concentrations remained relatively constant until late in the event when the maximum concentration was measured. Because this was a large storm that occurred on wet soil, the late-recession rise in nutrient concentrations probably resulted from increasing amounts of subsurface soil water in the discharge.

Mean-Event Concentrations and Loads

Discharge-weighted mean concentrations of total nitrogen in runoff ranged from 4.7 to 44 mg/L; the median concentration was 12 mg/L (fig. 21). Variation in mean concentration of total nitrogen was greatest in 1985; both the maximum and minimum mean concentrations for the study period occurred in that year. Seasonality was observed in mean concentrations of total nitrogen. Most of this seasonal variation probably results from a corresponding seasonal variation in nitrogen applications; greater mean nitrogen concentrations were generally observed during the nongrowing season when nutrient applications were greater than during the growing season (fig. 22). Concentrations of total nitrogen were composed of an average of 48 percent ammonia plus organic nitrogen and 52 percent nitrate plus nitrite.

Loads of total nitrogen loads in runoff ranged from less than 0.1 to 50 lb; the median load was 1.0 lb (fig. 23). The maximum load of 50 lb was transported by a February 12-13, 1985, runoff event and was 460 percent greater than the next largest total-nitrogen load.

Mean concentrations of total phosphorus in runoff ranged from 2.0 to 14 mg/L; the median concentration was 5.1 mg/L (fig. 21). The maximum mean concentration was measured in November 1984 and the minimum in February 1986 (fig. 22). The temporal behavior of the mean concentrations of total phosphorus and total nitrogen were much the same. Total-phosphorus loads ranged from less than 0.1 to 22 lb; the median load was 0.3 lb (fig. 23). The maximum load of 22 lb, transported by the February 12-13, 1985, runoff event, was 620 percent greater than the next largest total-phosphorus load.

Mean concentrations of suspended sediment ranged from 18 to 1,700 mg/L; the median concentration was 420 mg/L (fig. 21). Suspended-sediment loads ranged from less than 1.0 to 970 lb; the median load was 20 lb (fig. 23).

Annual Loads

A total of 90,700 ft^3 of runoff was discharged from the pipe-drained tract during the study period (table 13) and carried with it 87 lb of total nitrogen and 37 lb of total phosphorus (total nutrient loads in pipe discharge were calculated by summing both measured and estimated discharge loads). The total-nitrogen load was 65 percent ammonium plus organic nitrogen and 35 percent nitrate plus nitrite. For the study period, 67 percent of the total discharge, 71 percent of the total-nitrogen load, and 75 percent of the total-phosphorus load occurred during the 1985 water year. Annual nutrient loads in runoff represent only 0.37 and 0.18 percent of the total nitrogen and 0.65 and 0.32 percent of the total phosphorus applied to the pipe-drained tract during 1985 and 1986, respectively.

Because discharge is factored into the computation of loads, variations in the annual discharge, which result primarily from climatic variability, will be reflected in the total annual nutrient loads. In order to express the loads in a manner that allows a more equitable comparison between time periods, discharge variation was removed by dividing out total annual discharge. Discharge-weighted loads were 1.0 and 0.83 lb per 1,000 ft^3 of total nitrogen and 0.46 and 0.31 lb per 1,000 ft^3 of total phosphorus for 1985 and 1986, respectively. From 1985 to 1986, discharge-weighted loads were reduced 17 percent for nitrogen and 33 percent for phosphorus. These reductions compare closely with reductions in nutrient applications of 15 percent in nitrogen and 34 percent in phosphorus from 1985 to 1986.

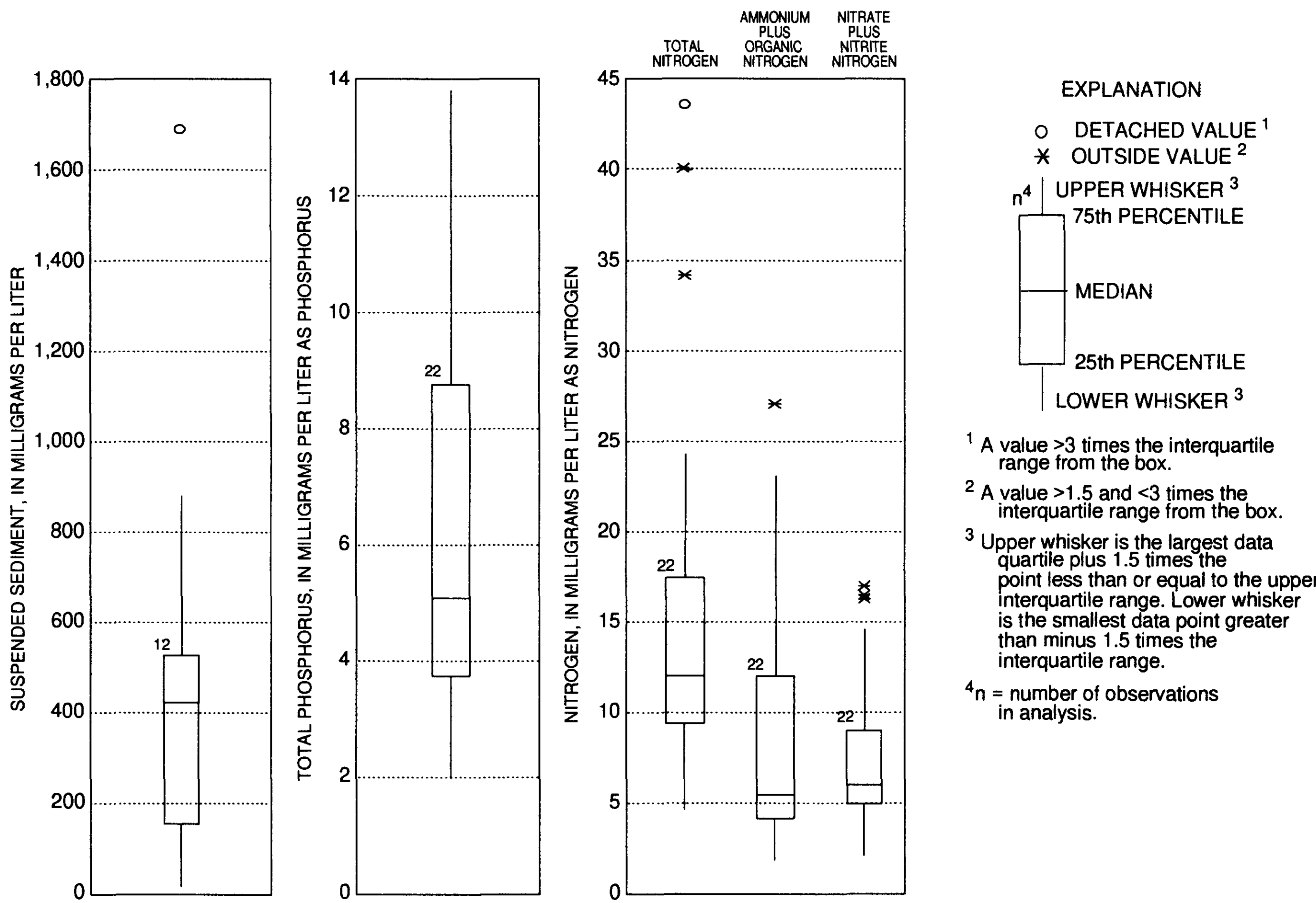


Figure 21. Distribution of discharge-weighted, mean-event-constituent concentrations in runoff from the pipe-drained terraces at Field-Site 2, October 1984 through September 1986.

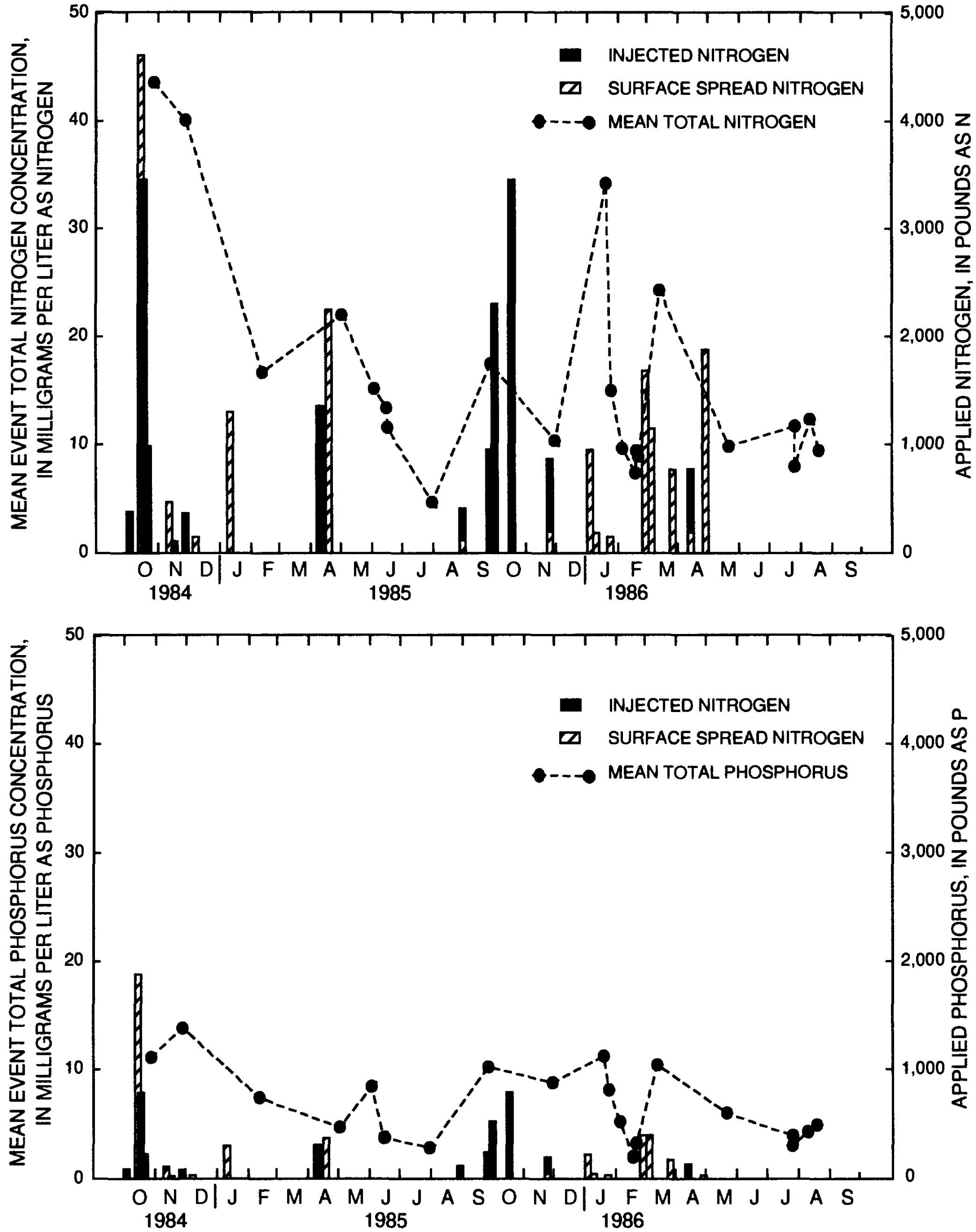


Figure 22. Discharge-weighted, mean-event concentrations of total ammonium plus organic nitrogen, total nitrate plus nitrite, total nitrogen and phosphorus in runoff from the pipe-drained terraces and nitrogen and phosphorus applied to the pipe-drained terraces at Field-Site 2, October 1984 through September 1986.

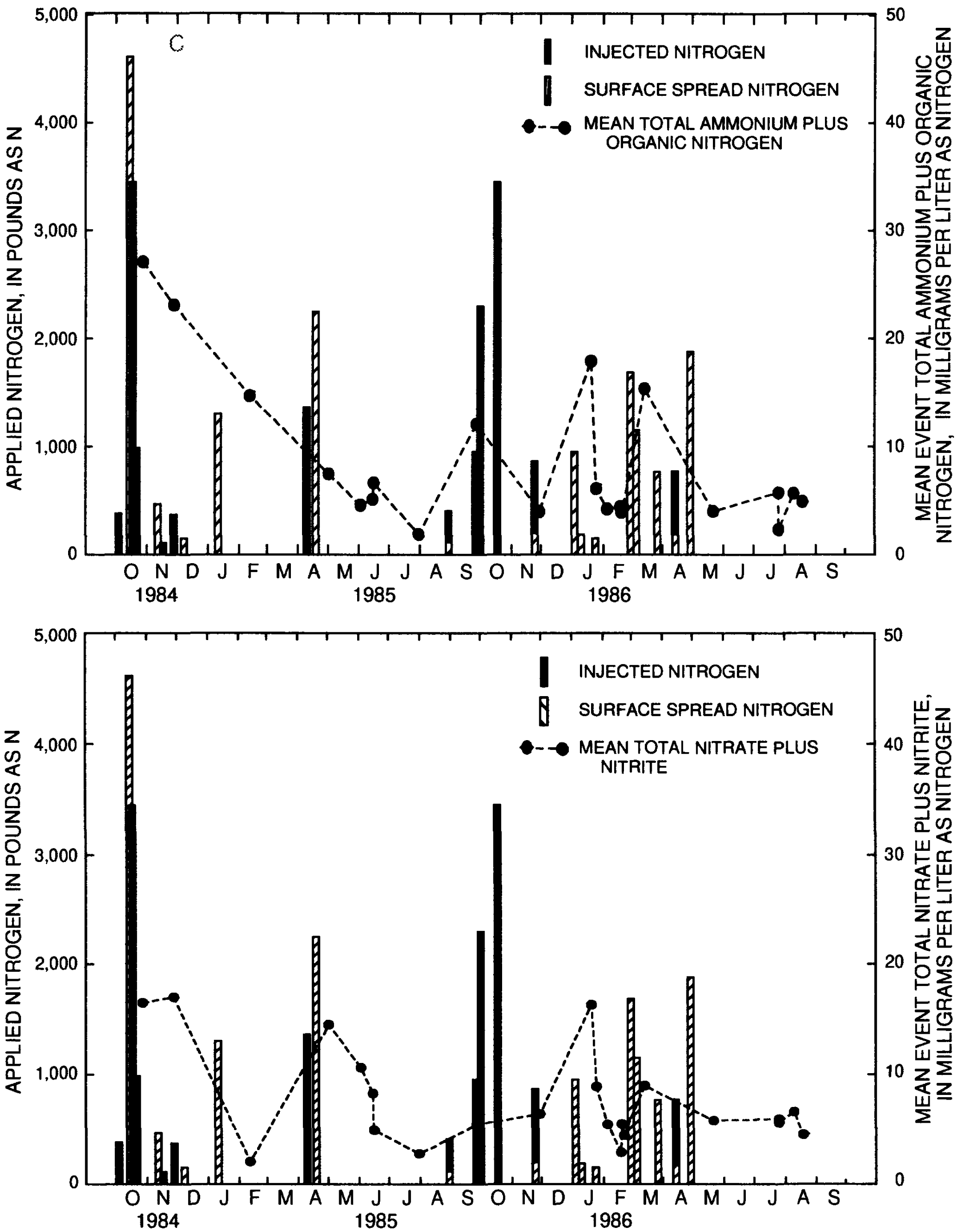


Figure 22. Discharge-weighted, mean-event concentrations of total ammonium plus organic nitrogen, total nitrate plus nitrite, total nitrogen and phosphorus in runoff from the pipe-drained terraces and nitrogen and phosphorus applied to the pipe-drained terraces at Field-Site 2, October 1984 through September 1986—Continued.

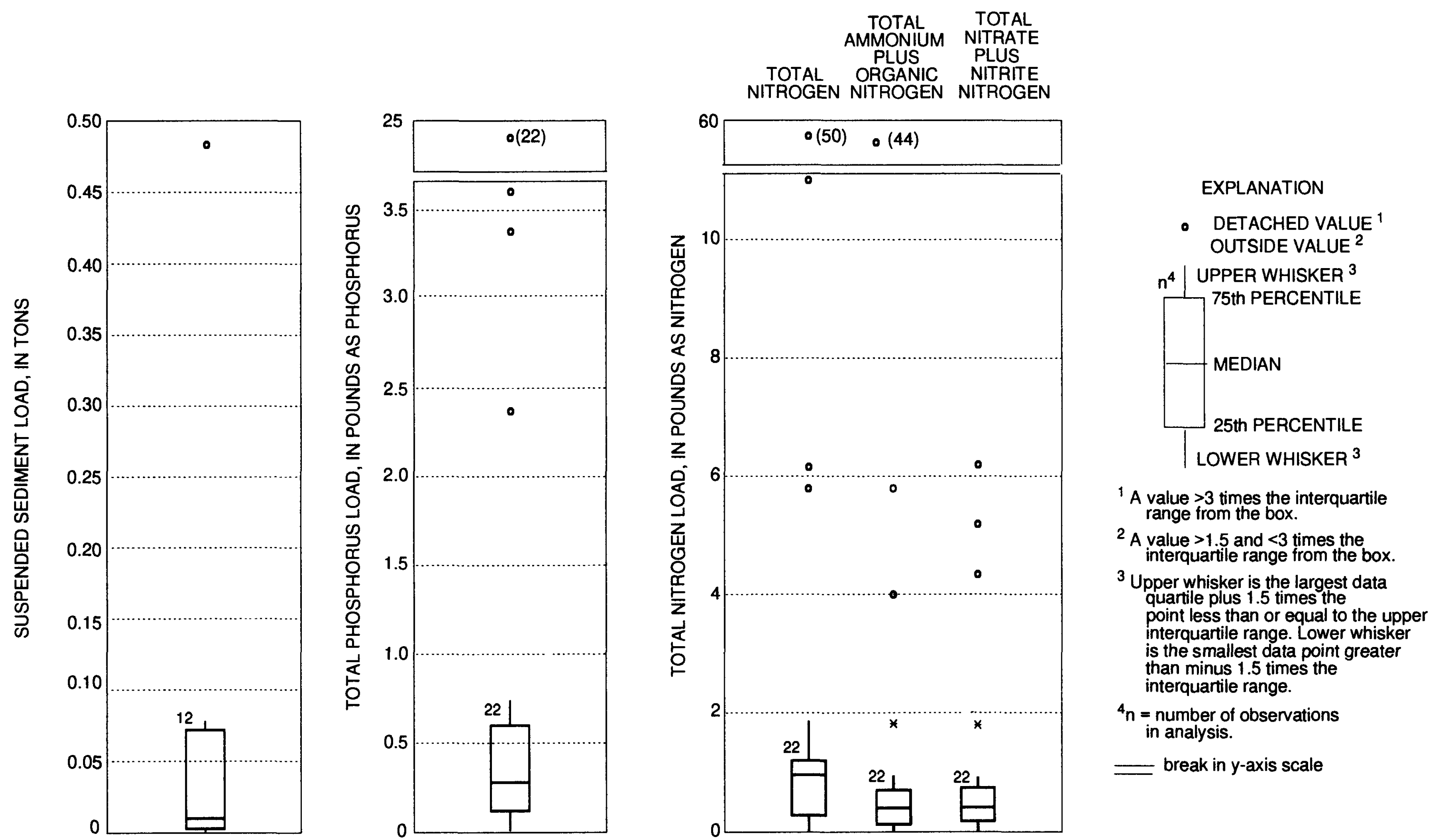


Figure 23. Distribution of storm loads in runoff from the pipe-drained terraces at Field-Site 2, October 1984 through September 1986.

Table 13. Annual runoff and nutrient loads from the pipe-drained terraces at Field-Site 2 for the 1985 and 1986 water years[1]

[Total annual runoff in cubic feet and nutrient loads in pounds]

	1985	1986	Total
Total runoff	60,900	29,800	90,700
Total nitrogen	62	25	87
Total ammonia + organic nitrogen	51	11	62
Total nitrate + nitrite nitrogen	11	13	24
Total phosphorus	28	9.1	37

[1] The 1985 and 1986 water years include the period October 1984 through September 1986.

Runoff on frozen ground accounted for 76 percent of the total-nitrogen load and 75 percent of the total-phosphorus load. These loads were carried by 68 percent of the pre-BMP pipe discharge. In February 1985, two storms on frozen ground with snowcover, which occurred 1 month after surface application of manure, made up 82 percent of the load of total nitrogen and 80 percent of the load of total phosphorus for the 1985 water year, and 58 percent of the load of total nitrogen and 61 percent of the load of total phosphorus for the study period. Total ammonium plus organic nitrogen accounted for 88 percent of the total-nitrogen load discharge from the storm. By comparison, total ammonium plus organic nitrogen accounted for 31 to 69 percent of the total-nitrogen load in other storms. A January 1986 frozen-ground event, which occurred within 15 days of surface application of manure, discharged 44 percent of the nitrogen and 40 percent of the phosphorus in the 1986 total-nutrient load.

Relation to Climate and Agricultural Activities

Evaluating the effect of nutrient management on runoff water quality can be simplified if a quantitative relation between water quality and nutrient applications can be demonstrated. Graphical and regression methods were used to identify variables that explain variation in mean-storm nutrient concentrations. Variables considered include storm discharge and duration, storm precipitation, 5- and 7-day antecedent precipitation, nutrient application quantity and days since last application, and crop cover. These variables were entered into regression procedures to estimate a quantitative relation among mean-storm nutrient concentrations, climate, and agricultural activities.

The regressions were run on the complete data set and on the thawed- and the frozen-soil data subsets (table 14). Significant explanatory variables were not found for either the complete data set or the frozen-soil only data set. For the thawed-soil data set, the number of days since the previous nutrient application was found to be a significant explanatory variable (table 14).

Table 14. Regression statistics for the log of mean-event concentrations of total nitrogen species and total phosphorus, in milligrams per liter, in runoff, as a function of the number of days since the last nutrient application at Field-Site 2, October 1984 through September 1986; thawed-soil storms only

Dependent variable (log transformed)	n	Regression coefficient: Days since previous nutrient application	t-test	p-value	Intercept	Coefficient of determination (Adj. R^2)[1]	Standard error: Log units	Standard error: Percent[2] Plus	Standard error: Percent[2] Minus
Mean total nitrogen	16	-0.004	3.362	0.005	1.33	0.41	0.19	55	35
Mean total ammonium plus organic nitrogen	16	-.005	2.830	.013	1.02	.32	.26	82	45
Mean total nitrite plus nitrate	16	-.003	2.967	.010	1.01	.34	.17	48	32
Mean total phosphorus	16	-.004	4.847	<.001	.97	.60	.14	38	28

[1] Coefficient of determination (R^2) adjusted for degrees of freedom.
[2] G. D. Tasker, U.S. Geological Survey, written commun., 1978.

The relation between the number of days since the previous nutrient application and the logarithm of mean concentrations of total nitrogen and total phosphorus in runoff for the thawed-soil subset is shown in figure 24. In general, mean concentrations of total nitrogen and total phosphorus in runoff decreased as the number of days since the previous nutrient application increased. However, mean concentrations of total phosphorus appear to behave asymptotically; there is no apparent decrease in mean total-phosphorus concentrations in runoff as the number of days after the previous nutrient application increases beyond about 60 days. This asymptotic behavior suggests the existence of a baseline concentration of total phosphorus in the 2 to 5 mg/L range. Mean concentrations of total nitrogen, however, appear to be decreasing as long as 120 days after the previous nutrient application.

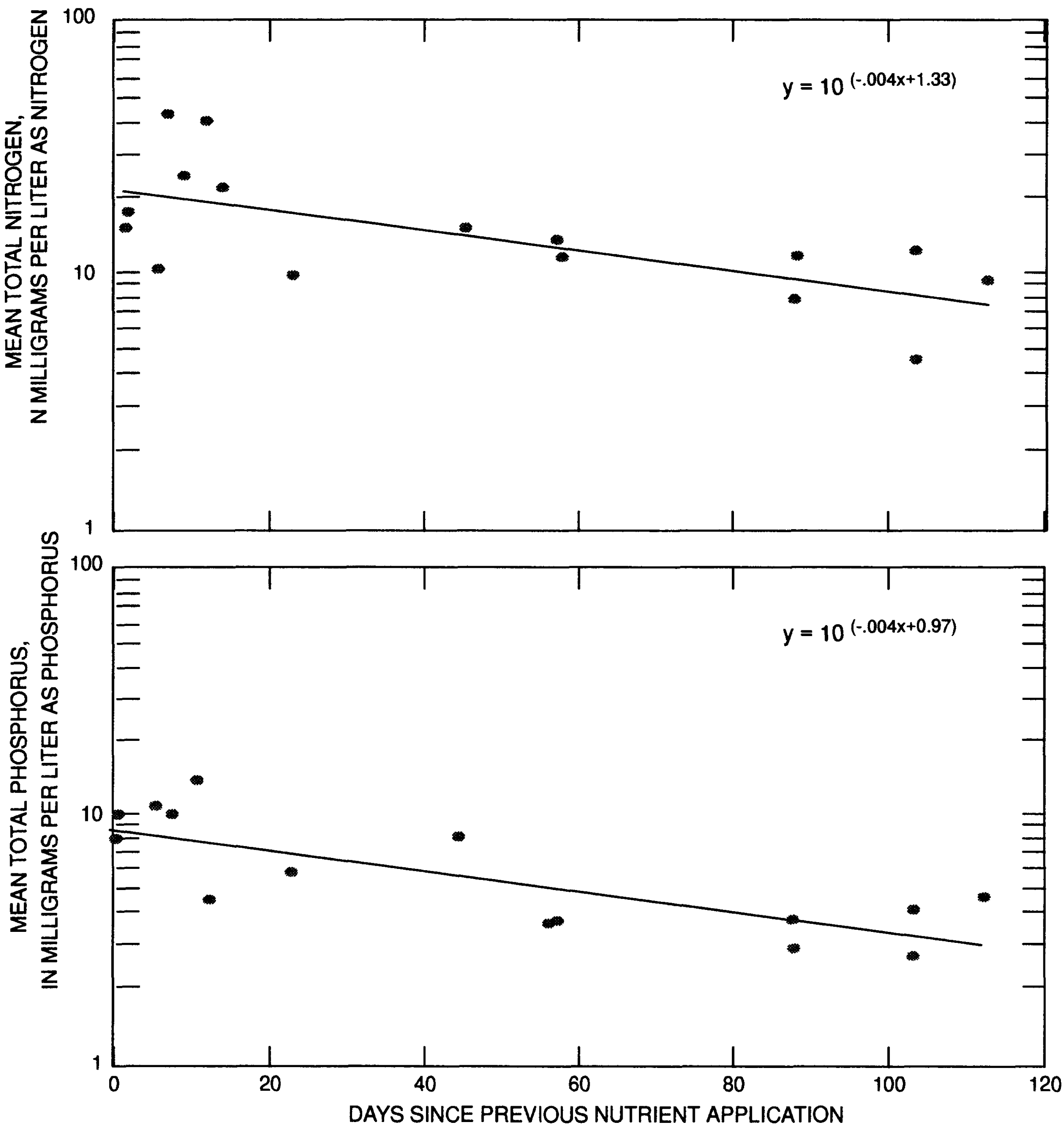


Figure 24. Relation between mean concentrations of total nitrogen (above) and total phosphorus (below) and the number of days since the previous nutrient application for runoff events on thawed soil at Field-Site 2, October 1984 through September 1986. (Regression statistics shown on table 15)

The regression procedures described above did not show the quantity of nutrients applied to be a significant explanatory variable. Possible explanations for the lack of significance in the regressions are (1) the relation between application quantities and mean-storm concentrations was nonlinear, and (2) nitrogen availability for transport in discharge was substantially different from actual nitrogen applications because of application method, volatilization, denitrification, and leaching into the soil by precipitation.

Of the factors affecting nutrient availability, all but application method have time-dependent probabilities. That is, with increasing elapsed time after nitrogen application there is an increasing chance of one or more of these factors reducing the nitrogen available for transport in discharge. For the mean concentrations of total nitrogen this time-dependent behavior could explain why the number of days since the previous nutrient application was the only significant explanatory variable.

GROUND WATER

The ground-water system in the study area was characterized on the basis of water levels, flow directions, and chemical quality. The three-dimensional configuration of the ground-water system was interpreted from outcrops of bedrock and from well logs. Water-level data from six wells with continuous water-level recorders were used to help define the water-table configuration and ground-water flow directions (fig. 25). Preliminary qualitative analyses were made to relate agricultural activities and recharge events to ground-water quality. For ground-water discussions, the site is considered to consist of the entire 55-acre farm, although only 47.5 acres are cropped. Inclusion of the additional 7.5 acres in ground-water discussions permits the use of data from several wells and a spring adjacent to the 47.5 cropped acres and from a stream adjacent to the farm.

Description of the System Framework and Properties

The ground-water basin that contains Field-Site 2 is shown in figure 1. The western boundary of Field-Site 2 is approximately 500 ft downgradient from the western ground-water basin divide (fig. 26). The location of the basin divide is based on regional hydrogeology, surface topography, and water-level data collected at and adjacent to Field-Site 2.

The water table in this region is typically a subdued replica of the land surface (Gerhart and Lazorchick, 1984). Permeable weathered regolith overlies carbonate bedrock of small primary porosity. Secondary bedrock features, including solutionally developed fractures, joints, cleavage planes, and faults transmit water through the bedrock. The shallow water table in the region commonly is found in the lower part of the weathered regolith but also may be located in the upper bedrock.

Observations made during well construction and well sampling and the results of ground-water-quality data analysis all indicate that aquifer properties such as specific yield, specific capacity, and transmissivity vary widely at the site. This variation reflects the folded and faulted nature of the aquifer, and the fact that fractures, which control the flow of ground water, are irregularly connected. There are no wells adjacent to the site that are pumped sufficiently to affect water-table altitudes or ground-water flow directions at the site.

Bedrock at the site is overlain by 5 to 30 ft of soils and weathered regolith formed in the residuum of the carbonate bedrock. The water-table surface is approximately coincident with the bedrock surface over most of the field site and may rise into regolith during periods of recharge. Water may pool and move along the bedrock-regolith contact in places where the bedrock is not sufficiently fractured to admit infiltrating water.

Ground-water recharge to the site occurs as infiltration of precipitation and ground-water flow into the site across the western field-site boundary. Ground-water discharge occurs as flow across the northern, eastern, and southern-site boundaries and is discussed in detail in the *Occurrence and Flow* section of this report. The base of the flow system and vertical flow characteristics in the bedrock have not been determined. Observation wells drilled to depths as great as 350 ft encounter fresh water, indicating that ground-

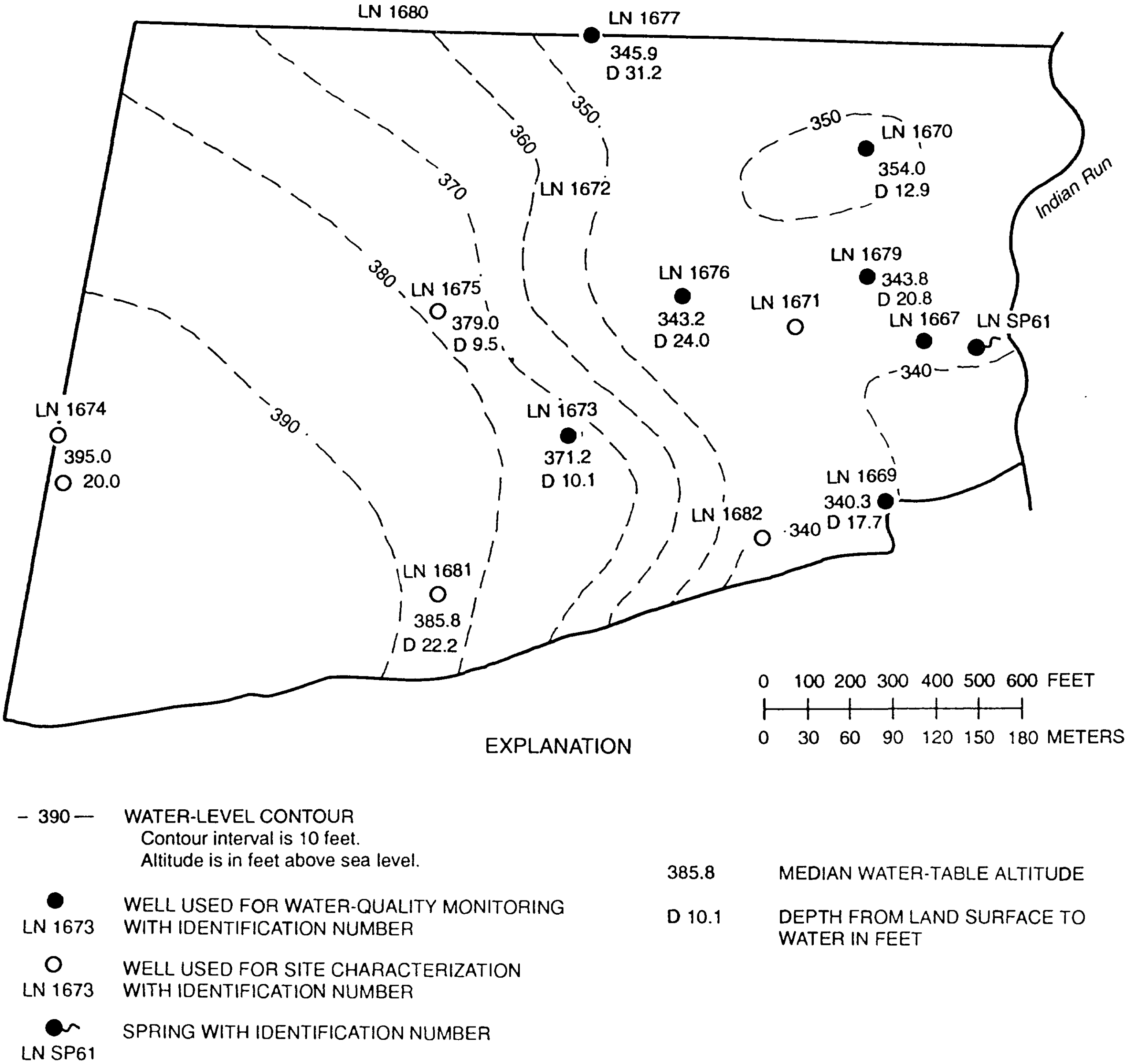


Figure 25. Median water-table altitudes, October 1, 1984, through September 30, 1986, and depth from land surface to ground water at Field-Site 2.

water flow is active at least to that depth. However, drillers logs indicated that most of the fractures, joints, and solutional openings were in the upper 50 ft of bedrock. Therefore, most of the ground-water flow probably takes place in this shallow zone (Parizek and others, 1971; Meisler and Becher, 1971).

Water-level data were analyzed, and estimates of the specific yield, specific capacities, and transmissivity were made to describe site hydrogeology. Well-completion data, water-level fluctuations, and the water-table configuration all indicate that aquifer properties vary widely across this small site, which is not unusual for a carbonate terrain.

Specific yield of a water-bearing geologic material is the volume of water it will yield by gravity drainage divided by its total volume (Lohman and others, 1972). At Field-Site 2, the specific yield of water-bearing bedrock and regolith was estimated by use of a method described by Gerhart (1986) from the water-level rise measured in wells during periods of negligible evapotranspiration and high soil-water saturation. Four storms occurring between November 1985 and May 1986 were selected for analysis. During this period, most plants were dormant and soil moisture was assumed to be at or near its field

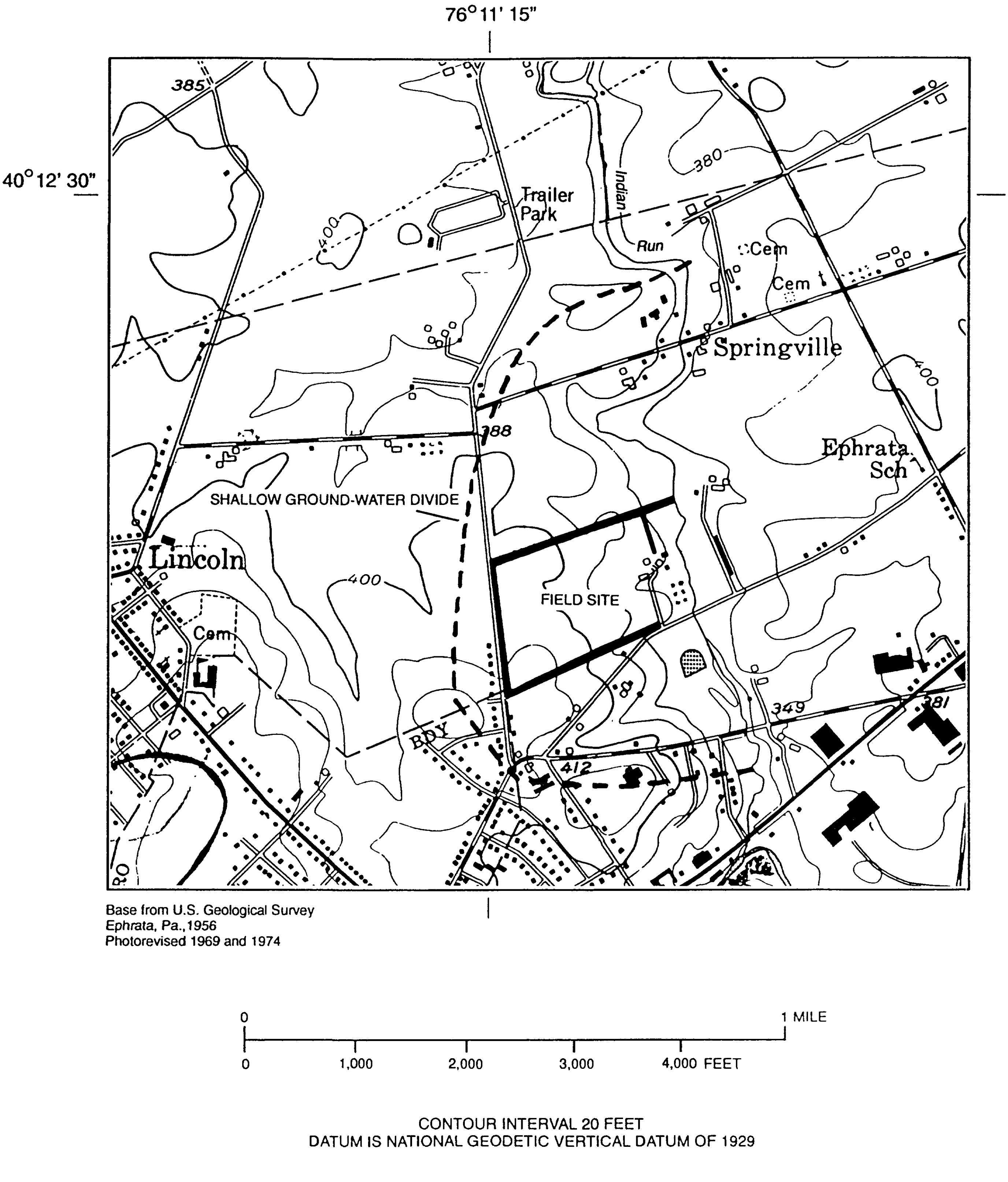


Figure 26. Location of Field-Site 2 in relation to the shallow ground-water basin and area topography.

capacity. Storms that produced rain on frozen ground, rain on a snowpack, or snowstorms were not used. Given these conditions, all precipitation during the selected storms either produced runoff or ground-water recharge.

Precipitation, runoff, and water-level rise were measured for each storm. The specific yield was computed as

$$\text{Specific Yield} = (P - R) / WLR \,, \qquad (3)$$

where P is precipitation measured at Field-Site 2, in inches;

R is runoff measured at Field-Site 2, in inches; and

WLR is water-level rise in well, in inches.

A summary of the specific yields computed at eight wells is shown in table 15. The mean specific yield for each well ranged from about 0.05 at wells LN 1669 and LN 1670 to 0.10 at well LN 1674. The variation in estimates of specific yield between storms at individual wells indicate that some of the assumed conditions do not hold for each storm. Most likely, soil moisture was not always at field capacity prior to the onset of a storm and runoff may not have been uniform throughout the site.

Table 15. Specific yields computed for observation wells at Field-Site 2, for selected storm periods, November 1985 to March 1986

[--, no data]

Storm period	Well number							
	LN 1669	LN 1670	LN 1673	LN 1674	LN 1675	LN 1676	LN 1677	LN 1679
November 13 - December 2, 1985	0.056	0.044	0.091	0.11	0.095	0.092	0.067	--
January 20-21, 1986	.038	.037	.035	.15	.055	.086	.075	--
January 25-27, 1986	.052	--	.087	.11	.082	.065	.057	--
March 13-17, 1986	.041	.077	.067	.04	.064	.059	.048	0.081
Mean specific yield	.047	.053	.070	.10	.074	.076	.062	.081

Specific capacity of a well is the rate of discharge of water from the well divided by the drawdown of the water level within the well (Lohman and others, 1972). Specific capacities of wells at the site were calculated on the basis of estimates made by the driller. Estimated specific capacities range from less than 1 (gal/min)/ft of water-level drawdown to 20 (gal/min)/ft of drawdown (table 4). Large spatial variations in specific capacity between wells are characteristic of wells in carbonate aquifers because of the presence or absence of fractures, joints, faults, and voids that may differentially store and transmit water in different parts of the aquifer (Parizek and others, 1971).

Specific-capacity data indicate that two contrasting aquifer regimes were present at the site. Wells LN 1677, LN 1679, and spring LN SP61 are apparently located in solutionally-developed, fractured zones in dolomite of the Snitz Creek Formation. Wells in the Snitz Creek Formation have large specific capacities [averaging about 20 (gal/min)/ft of drawdown] relative to specific capacities [averaging <1 (gal/min)/ft of drawdown] of wells drilled in limestone of the Millbach Formation in the western part of the site.

Transmissivity is the rate at which water is transmitted through a unit width of aquifer under a unit hydraulic gradient (Lohman and others, 1972). At Field-Site 2, transmissivity of the saturated regolith and carbonate bedrock can be estimated from the water-table configuration and Darcy's Law. For ground-water flow through a unit width of aquifer, Darcy's Law states that

$$Q = T(dh/dl) \,, \qquad (4)$$

where Q is the ground-water flow rate, in cubic feet per day;

T is the aquifer transmissivity, in feet squared per day; and

dh/dl is the hydraulic gradient (water-table slope), in feet per foot.

For example, at well LN 1675 the water-table slope (change in water-table altitude/horizontal distance) is about 0.07 (fig. 25). Ground-water flow through a 1 ft wide section of aquifer can be estimated from a steady ground-water recharge of 23.0 in. per year at a point about 1,200 ft down a flow path from the ground-water divide (fig. 26), where the slope of the water table was measured. Transmissivity can then be computed as

$$T = (6.3\ \mathrm{ft}^3/\mathrm{d})/(0.07\ \mathrm{ft}) \qquad (5)$$
$$= 90\ \mathrm{ft}^2/\mathrm{d}.$$

Transmissivity computations support the specific-capacity based observations that two contrasting aquifer regimes are present at the site. Computations at wells LN 1673, LN 1674, LN 1680, and LN 1681 indicate that saturated geologic material in the western upland part of the site probably is similar to that at well LN 1675. This area is underlain primarily by limestone of the Millbach Formation.

The eastern part of the site near Indian Run is underlain primarily by dolomite of the Snitz Creek Formation. The flatness of the water table there indicates a much larger transmissivity than in the western part of the site. At well LN 1676, an estimated hydraulic gradient of about 0.01 and flow rate of 8.8 5 ft^3/d indicates a transmissivity of about 880 ft^2/d, which is an order of magnitude greater than the upland area. Pumping of wells during sampling and well completion also showed that the aquifer was much more transmissive in the eastern part of the site at wells LN 1667, LN 1679, and LN 1677.

The large range of transmissivity across the site also is indicated by large differences in ground-water recession slopes of water levels shown on hydrographs of wells (fig. 27). Rorabaugh (1960) showed that for an idealized aquifer, ground-water recessions will approach a consistent straight-line slope given a sufficient length of time after an instantaneous recharge event. The characteristic slope depends only on the hydraulic diffusivity (transmissivity/specific yield) and aquifer geometry.

Rorabaugh's formula to calculate diffusivity is

$$\frac{T}{S} = 0.933a^2\left[\frac{\log\left(\frac{h_1}{h_2}\right)}{t_2 - t_1}\right], \qquad (6)$$

where T is transmissivity, in cubic feet per day per square feet times feet of aquifer thickness;

S is storage (dimensionless);

0.933 is used to convert base e to base 10;

a is the half-width of the aquifer, in feet;

h is hydraulic head, in feet;

t is time, in days; and

terms contained in the long brackets represent the slope of the regression line on semilog paper.

Although Field-Site 2 does not fit the idealized geometry on which the method is based because the aquifer is heterogenous and anisotropic, the large range of recession slopes from 370 to 7,400 days per log cycle of water-level change suggest that the diffusivity ranges over at least one order of magnitude at the site. Because estimates of specific yield range from only 0.05 to 0.10 (table 15), the large range of recession slopes is probably caused by the large range in transmissivity.

Occurrence and Flow

A water-table map (fig. 25), based on surface topography and measured water levels, shows the median water-table altitude at the site during the study period. Depth from the land surface to the water table ranges from 5 to 35 ft. The water-table configuration indicates the presence of multiple ground-water drainage basins, with discharge occurring across the northern, eastern, and southern site boundaries.

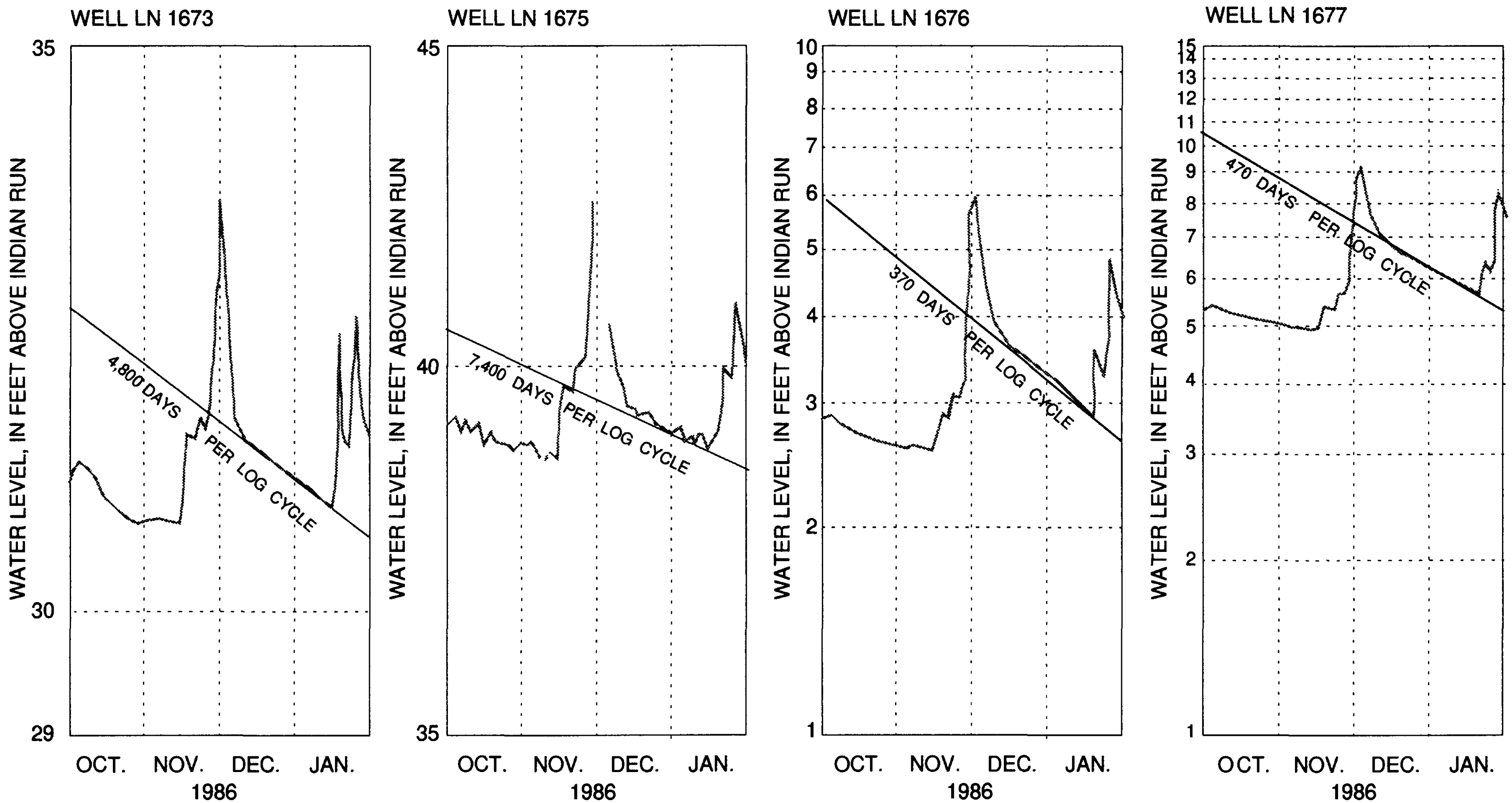


Figure 27. Characteristic ground-water recession slopes for selected wells at Field-Site 2. (Slopes are expressed as the number of days per log cycle of water-level change.)

Water-level data were collected at 16 wells on or adjacent to the site. Water levels at the six wells that were equipped with continuous water-level recorders are shown in figure 28. Water levels in the remainder of the wells were either recorded continuously for a short time or measured periodically. The six instrumented wells show a maximum difference in water level of about 35 ft; highest water levels are beneath topographically highest parts of the site.

A ground-water flow model (Hall and Risser, 1992) was constructed of the hillslope within which the site is situated to help estimate the magnitude of ground-water flow across boundaries and to test whether the estimated transmissivities are reasonable for an average recharge rate of about 23 in/yr and the water-table configuration shown in figure 25. The hillslope was simulated as a two-dimensional, steady-state flow system in the x-y plane by use of the finite-difference model of McDonald and Harbaugh (1988). Wells at the site were drilled to depths of 350 ft without encountering saline water, indicating that ground-water flow is active to that depth. However, most of the fractures, joints, and solutioned openings were found in the upper 50 ft of saturated material, and most of the ground-water flow probably occurs in this shallow zone. Therefore, a two-dimensional model was judged to be adequate to provide general information about ground-water flow that is primarily horizontal within a relatively thin layer. The finite difference grid, aquifer properties, and boundary conditions used in the model are shown in figure 29. The modeled area consists of 977, 100 × 100-ft cells. The 55-acre farm within the modeled area is represented by 236 of the cells.

The simulated water table is shown in figure 30 with measured water levels from wells for comparison. Although the simulated surface does not fit the observed water levels on the water-table map exactly, the "character" of the water-table surface has been reproduced by the model. The model indicates that for uniform recharge, transmissivities must (1) be about 100 ft^2/d in the western upland part of the site, and (2) be at least an order of magnitude larger in the eastern part of the site near Indian Run. Simulations also indicate that the area near well LN 1670 has a very low transmissivity. The observed water level at that well was simulated by use of a transmissivity no larger than about 10 ft^2/d. Transmissivities outside of the site are speculative; however, model results indicate that one method to simulate the low water levels in wells LN 1669 and LN 1682 is by adding a very transmissive zone along Indian Run to the south of the site (fig. 29).

Water-Level Fluctuations

Water levels in each well respond quickly to recharge, and water levels peaked 1 to 3 days following precipitation. Rapid response of the water levels is a result of the very permeable soil and unsaturated regolith overlying a carbonate aquifer that contains solution-enhanced fractures.

Temporal variations in water level were similar at the six wells through the study period (fig. 27). Aquifer water levels were 1 to 2 ft higher in 1986 than in 1985, reflecting the 7 in. increase in total precipitation during 1986. Variation in water level throughout the study period ranged from about 4.1 ft at well LN 1673 to about 11.0 ft at well LN 1670 (fig. 27). Hydraulic gradients between the six wells change during wet periods as compared to dry periods because of spatial variations in the specific yield of the aquifer. For example, a storm occurring on May 1-3, 1985, caused a water-level rise of approximately 5 ft at well LN 1670, but a rise of approximately 1 ft at well LN 1677 (fig. 31); hence, the lateral hydraulic gradient between wells steepened during storm recharge.

Quantity

Quantities of ground-water recharge were estimated from the water-level rises recorded in wells in response to storms. Rapid water-level rise after storms indicates that precipitation quickly percolates to the water table. Therefore, the water-level rise multiplied by the specific yield of the geologic material should provide a reasonable estimate of the quantity of ground-water recharge from a storm (Gerhart, 1986).

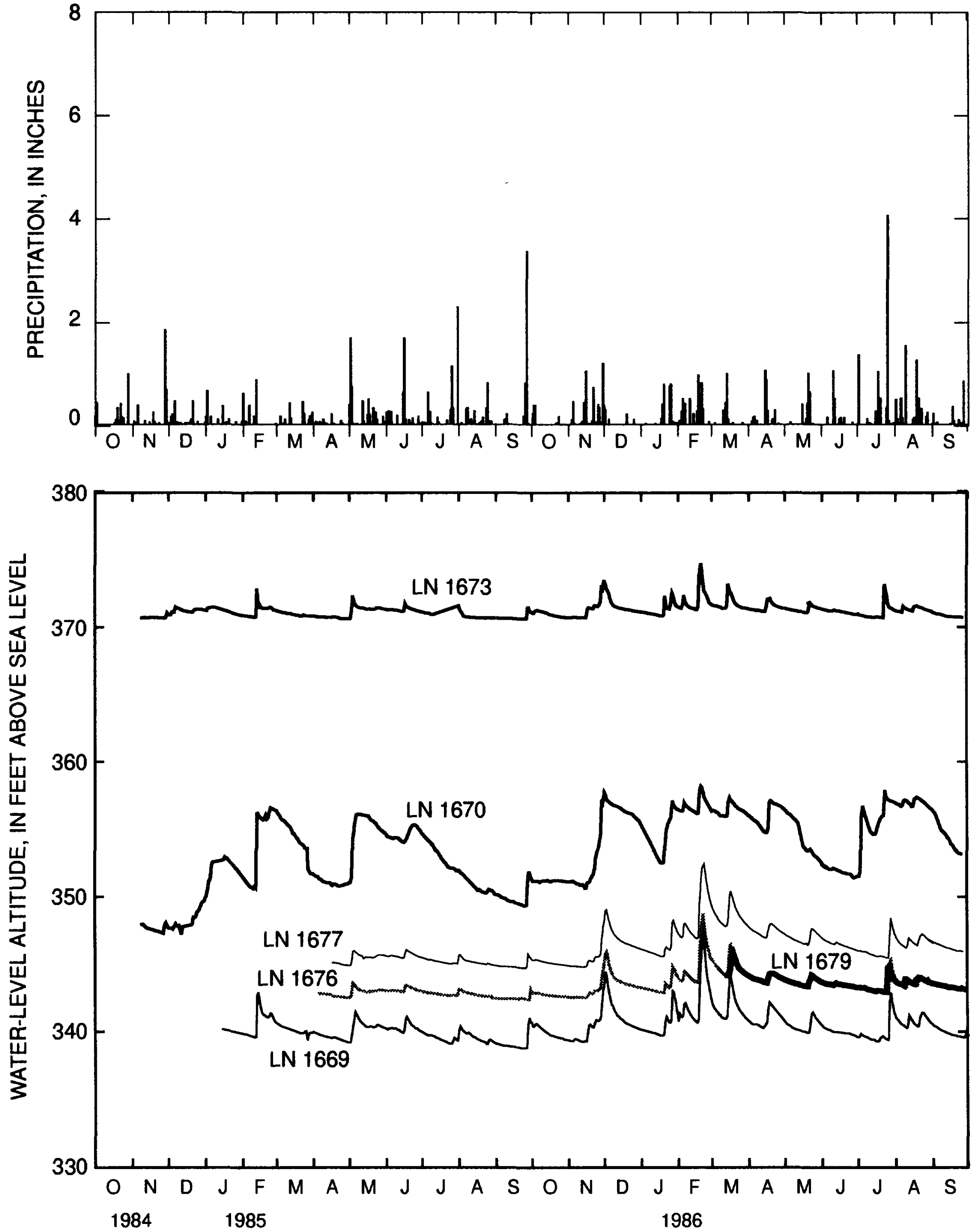


Figure 28. Precipitation and temporal variation in water-level altitude in five wells at Field-Site 2, October 1984 through September 1986.

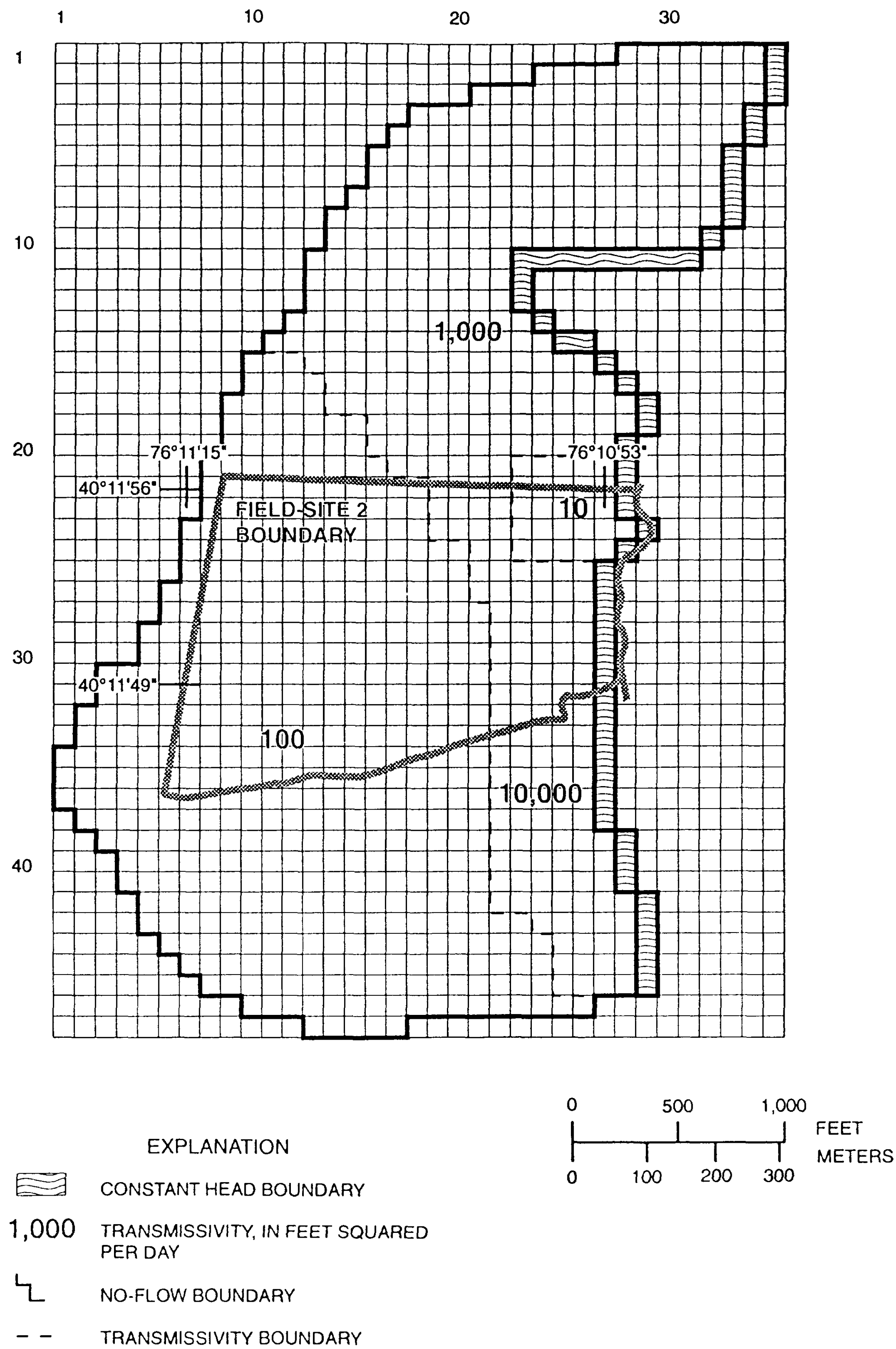


Figure 29. Finite-difference grid, hydrologic coundaries, and aquifer properties used in the ground-water flow model of Field-Site 2.

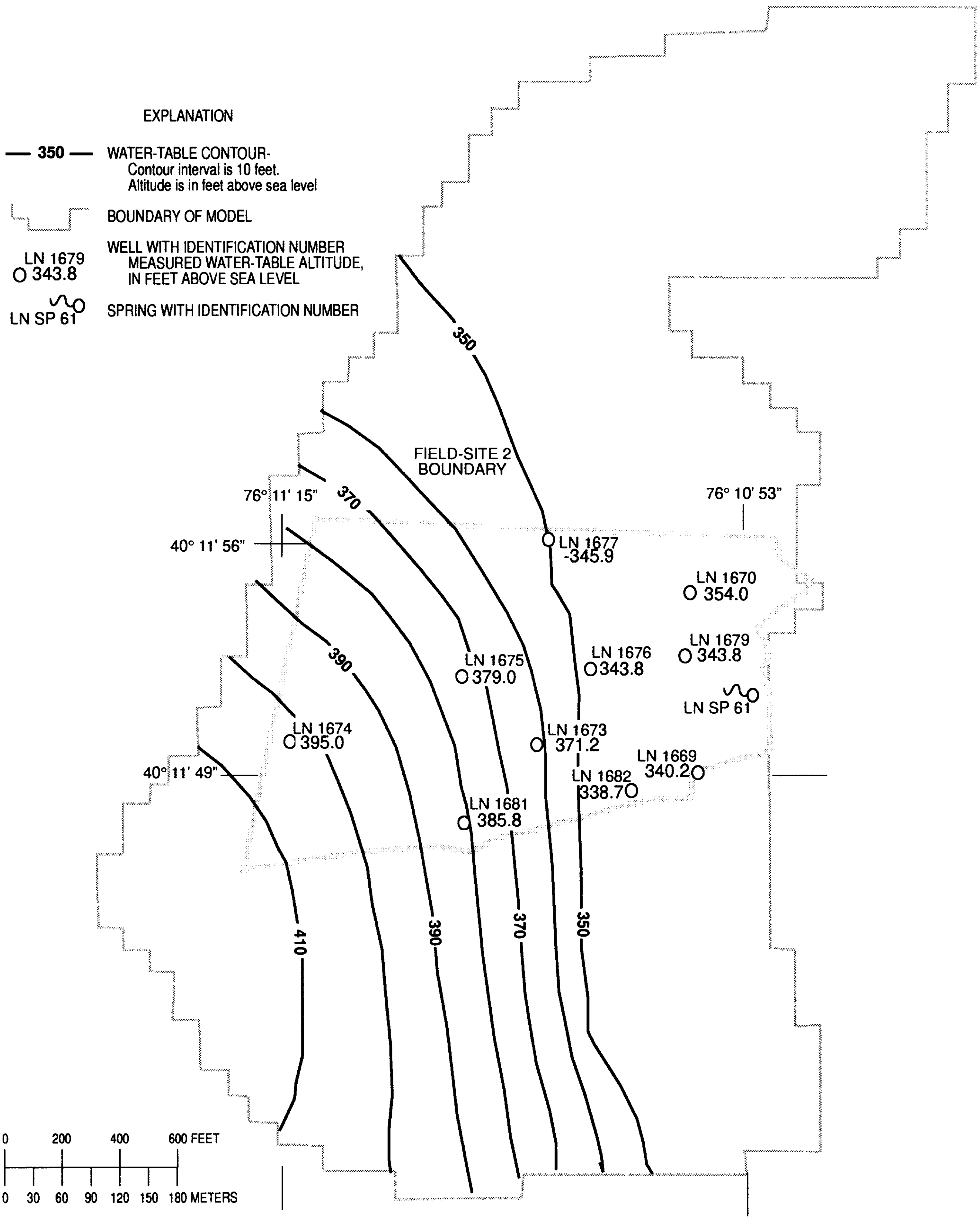


Figure 30. Simulated water-table surface at Field-Site 2 and surrounding areas.

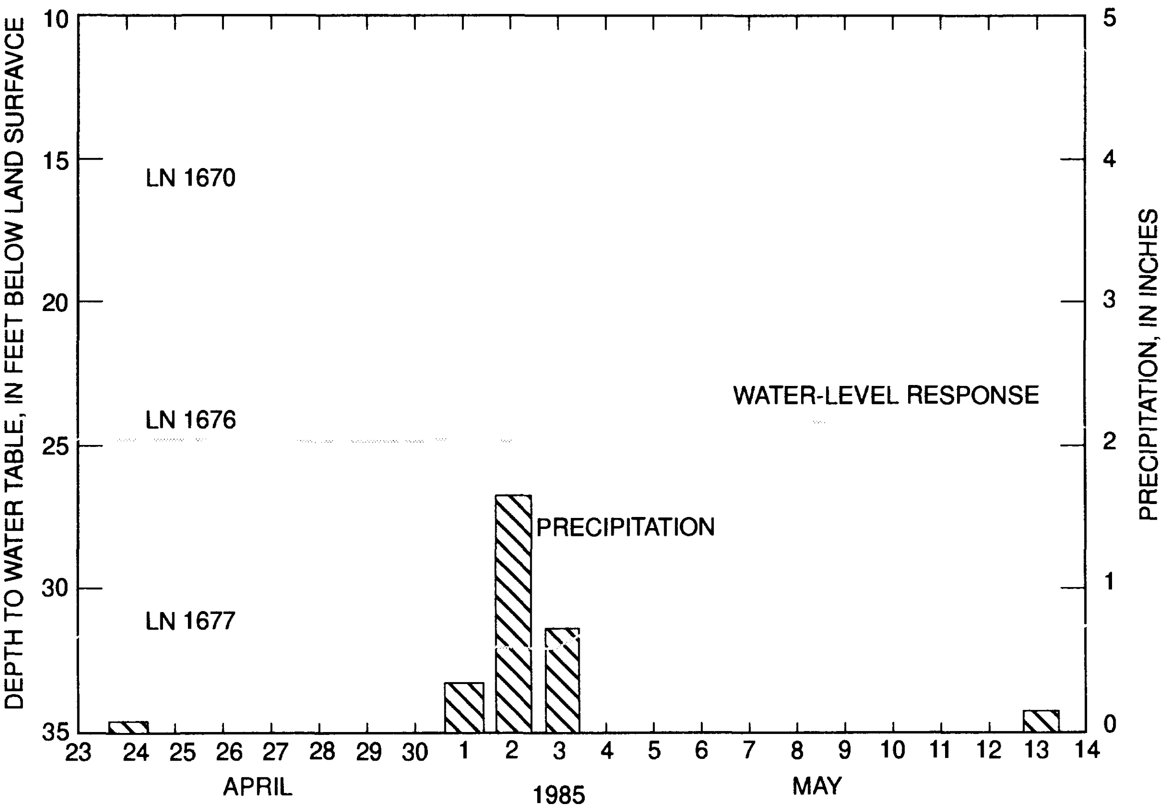


Figure 31. Response of water levels in wells LN 1670, LN 1676, and LN 1677 to precipitation at Field-Site 2, April 23-May 14, 1985.

Recharge was computed by use of water-level observations and computed specific yields (table 16) at wells LN 1673 and LN 1677. These wells were selected because together, available water-level data from these wells provided a nearly-complete record spanning the study period. Prior to May 1985, the water-level rise in well LN 1673 was multiplied by a specific yield of 0.07 to estimate recharge. Recharge from the remaining storms was computed from the water-level rise and specific yield (0.06) at observation well LN 1677, which possessed the most complete water-level record for the period after May 1985.

Recharge estimates are very sensitive to the specific yield used in this computation. For example, if a specific yield of 0.05 were used (instead of 0.06) to compute recharge for water year 1986, the estimated total recharge would be 16.9 in. and would be equal to about 43 percent of the precipitation for that period as opposed to 20.3 in. or 52 percent of precipitation.

Ground-water recharge was computed for each storm and summed by month (table 16, fig. 32) for the entire study period. The total recharge calculation for the 1985 water year contained estimated values for October 1984 and January 1985 because no water-level data were available for these months. The October 1984 rate of recharge was estimated as the mean of the rates for November 1984, September, October, and November 1985, and September 1986. The January 1985 rate of recharge was estimated from the record of well LN 1670, because data for that month were missing from the nearly-complete record at well LN 1673.

During February 1985 and February and March 1986, the estimated quantities of ground-water recharge exceeded the measured monthly precipitation. While snowmelt from snow that fell during previous months could cause monthly recharge to exceed monthly precipitation, errors in the measurement of snowfall quantity may also contribute to errors in the computations. When snowfall

Table 16. Monthly precipitation, water-level rise, computed ground-water recharge, and annual recharge as a percentage of annual precipitation at Field-Site 2

Month	Precipitation (inches)	Water-level rise (feet) at well LN 1677	Computed recharge (inches)	Estimated annual recharge, as a percentage of precipitation
Oct. 1984	2.78	[1]0.95	0.79	
Nov. 1984	3.37	[1].6	.50	
Dec. 1984	1.71	[1]1.0	.82	
Jan. 1985	1.69	[1].8	.67	
Feb. 1985	2.23	[1]3.4	2.86	
Mar. 1985	1.96	[1].4	.34	
Apr. 1985	.60	[1].0	.0	
May 1985	5.14	2.1	1.51	
June 1985	3.86	.9	.65	
July 1985	5.20	.4	.28	
Aug. 1985	2.60	.8	.58	
Sept. 1985	4.70	1.7	1.22	
Totals	35.84	13.1	10.22	29
Oct. 1985	1.13	.1	.07	
Nov. 1985	4.94	4.9	3.53	
Dec. 1985	.85	.0	.0	
Jan. 1986	2.95	3.1	2.23	
Feb. 1986	3.96	8.8	6.34	
Mar. 1986	1.93	3.2	2.30	
Apr. 1986	3.06	1.2	.86	
May 1986	2.60	1.4	1.01	
June 1986	2.13	.3	.22	
July 1986	7.95	3.3	2.38	
Aug. 1986	5.68	1.8	1.30	
Sept. 1986	1.65	.1	.07	
Totals	38.83	28.2	20.31	52

[1] Water-level rise as measured in LN 1673.

exceeded the recording capacity of the site gage (72 in.), snowfall quantities were estimated using record from the NOAA weather station located approximately 2 mi from the site. Variations in local snow conditions (particularly drifting snow) may have contributed to underestimation of snowfall quantity at the site, causing estimated recharge to exceed measured precipitation during winter months.

In the 1985 water year, total ground-water recharge was 10.22 in., or about 29 percent of the precipitation recorded during that period. In the 1986 water year, total recharge was 20.3 in., or about 52 percent of the precipitation that year. Although only a partial record was available for the 1985 water year, ground-water recharge appeared to be substantially less than in the 1986 water year.

Although absolute values of ground-water recharge may be suspect, the relative differences between estimates of monthly recharge are significant. If recharge were evenly distributed throughout the pre-BMP period, 1.27 in. would be recharged monthly. From figure 31, the importance of months with extreme recharge is evident. In the 1986 water year, for example, 49 percent of the year's recharge occurred in 2 months—November 1985 and February 1986. Over the long term, the greatest recharge could occur in any month if the timing, amount, and intensity of precipitation are favorable.

Ground-water recharge quantities were estimated for 1985 and 1986. Of these 2 years, precipitation during 1986 (38.8 in.) was nearest to the 30-year average annual precipitation of 43.5 in. Therefore, the long-term average ground-water recharge rate at the site was estimated by assuming that

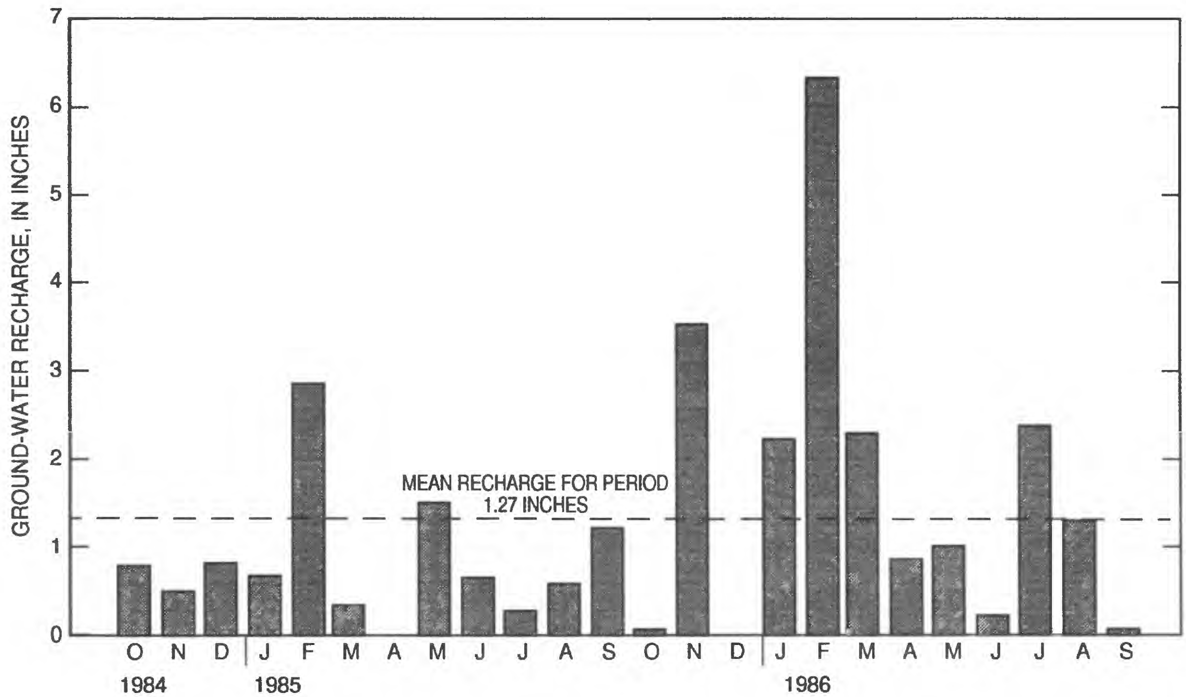


Figure 32. Estimated ground-water recharge at Field-Site 2 for water years 1985-86.

52 percent of precipitation recharges the ground-water system as computed for year 1986. By use of the 30-year average of 43.5 in. per year at Ephrata (National Oceanic and Atmospheric Administration, 1984), long-term ground-water recharge would average about 23.0 in. per year.

Ground-water recharge from infiltration of precipitation and inflow across the site boundaries is balanced by outflow across field-site boundaries and discharge to Indian Run. The boundaries of the ground-water flow system at Field-Site 2 are poorly defined. Because the site occupies only part of a larger watershed, the northern, southern, and western boundaries do not correspond to real physical boundaries. The water-table configuration (fig. 25) indicates that ground water flows across these boundaries.

An estimate of the ground-water budget derived from the flow model is shown in table 17. These values should be viewed as approximations because of the simplifying assumptions of the model and the sensitivity of flow values to the aquifer properties used. However, the simulations reveal that inflow and outflow of ground water across field-site boundaries is significant. For example, although the majority of ground-water recharge is from infiltration of direct precipitation, about 16 percent may be from ground-water inflow from areas outside of the site. Discharge of ground water is to Indian Run, but as much as 52 percent of the discharge could take place downstream of the site. Simulations also indicate that about 26 percent of ground water exits across the northern boundary of the site, eventually discharging to Indian Run upstream from the site. Evapotranspiration is probably not a significant source of ground-water discharge because the water table is generally more than 15 ft below the land surface. Evapotranspiration processes are probably ineffective below about 4 ft, the depth of the corn root zone.

Table 17. Simulated steady-state ground-water budget from the ground-water flow model of Field-Site 2

Ground-water inflow	Cubic feet per second	Inches per year (rounded to nearest inch)	Percentage of total inflow
Recharge from precipitation	0.145	23.0	84
Flow across western boundary	.028	4.0	16
Total	.174	27.0	100

Ground-water outflow	Cubic feet per second	Inches per year (rounded to nearest inch)	Percentage of total outflow
Flow across eastern boundary	0.036	6.0	22
Flow across northern boundary	.042	7.0	26
Flow across southern boundary	.091	14.0	52
Total	.169	27.0	100

Quality

Seven wells and the spring were sampled regularly for specific conductance and nutrients. Samples were separated into nonrecharge and recharge groups. Samples were grouped as nonrecharge if they were collected at least 2 weeks after a significant ground-water recharge event. A significant recharge event was defined as being a rise of 0.6 ft in the high-yielding wells and a rise of 1.0 ft in the low-yielding wells (processes discussed by Keith and others, 1983). A uniform rise could not be used for all wells because the low-yielding wells show lower water-level rises than the higher-yielding wells during the same storm event. Water samples collected from well LN 1667 and spring LN SP61 could not be classified as nonrecharge or recharge because there were no water-level recorders at these sites. The nonrecharge and recharge groups were used to assess (1) how the ground-water quality varied during short recharge periods compared to baseline or nonrecharge periods, and (2) to what extent does recharge water, which may be directly influenced by agricultural activities and climatic factors, affect the overall quality of water in the system.

Nonrecharge ground-water samples were collected monthly and recharge samples were collected during selected storms. Approximately 300 ground-water samples were collected during the study period. Of these, 83 were collected during nonrecharge conditions and 95 were collected during recharge conditions; the remainder of samples collected at the sites could not be classified.

Description of Data

Ground-water samples collected on May 13, 1986, at eight wells and the spring were analyzed to characterize ground-water quality at the site (table 18). The ground-water chemistry in all samples closely reflects the carbonate mineralogy of the aquifer. The ground water is slightly alkaline (180 to 300 mg/L as $CaCO_3$) and is characteristic of a limestone or dolomite aquifer. The molar calcium/magnesium ratio ranges from 1.2 at spring LN SP61 to 2.2 at well LN 1670, which indicates the presence of dolomitic rock (Hem, 1985). Concentrations of dissolved oxygen range from 2.8 mg/L at well LN 1682 to 10.5 mg/L at well LN 1673; the median was 7.2 for the eight wells tested, which indicates that the aquifer is open to the atmosphere.

Ground-water samples collected during the rest of the study period were analyzed for concentrations of dissolved phosphorus, ammonia plus organic nitrogen, nitrite, and nitrate plus nitrite to assess changes in ground-water quality related to changes in agricultural activities; the specific conductance of each ground-water sample also was measured.

Specific conductances ranged from 450 to 1,560 µS/cm in nonrecharge samples and from 485 to 1,500 µS/cm for recharge samples during the study period (fig. 33). Ground-water samples with greater specific conductances correlate with analyses containing greater concentrations of calcium, chloride, and nitrate (table 18).

Table 18. Characteristics and concentrations of dissolved chemical constituents of ground-water samples from eight wells and a spring at Field-Site 2 on May 13, 1986

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

Well number	Specific conductance (μS/cm)	pH (units)	Temperature (°C)	Oxygen, dissolved (mg/L)	Acidity (mg/L as $CaCO_3$)	Calcium (mg/L)	Magnesium (mg/L)	Sodium (mg/L)	Potassium (mg/L)	Alkalinity (mg/L as $CaCO_3$)	Sulfate (mg/L)	Chloride (mg/L)	Dissolved solids (mg/L)	Nitrite (mg/L as N)	Nitrate plus nitrite (mg/L as N)	Ammonia (mg/L)	Ammonia plus organic nitrogen (mg/L as N)	Phosphorus (mg/L as P)	Orthophosphorus (mg/L as P)
LN SP61	715	7.3	11.0	7.7	27	97	28	7.3	11	240	31	21	460	<0.01	21	<0.01	0.58	0.02	<0.01
LN 1667	685	7.4	12.0	7.9	29	110	32	7.8	10	250	34	21	438	<.01	21	<.01	<.2	.04	<.01
LN 1669	631	7.8	12.0	5.7	12	83	39	5.0	10	225	55	14	402	.04	16	.02	.52	.01	<.01
LN 1670	1,380	7.6	13.0	4.7	16	190	52	20	9	180	97	30	1,100	.02	96	.06	.88	.03	<.01
LN 1673	1,200	7.8	11.5	10.5	12	140	44	10	12	240	57	26	680	.01	53	<.01	.60	.02	<.01
LN 1676	1,510	7.1	11.5	7.0	63	180	63	38	10	300	74	32	974	.01	70	.02	.62	--	--
LN 1677	805	7.3	12.0	8.8	22	100	42	5.2	10	260	39	15	484	<.01	30	<.01	.64	.02	.01
LN 1679	770	7.1	11.5	7.4	39	110	34	7.7	10	260	27	22	494	<.01	24	.01	.68	.03	<.01
LN 1682	622	7.6	12.0	2.8	15	88	43	6.7	17	230	83	14	490	.03	12	.02	.50	.02	<.01

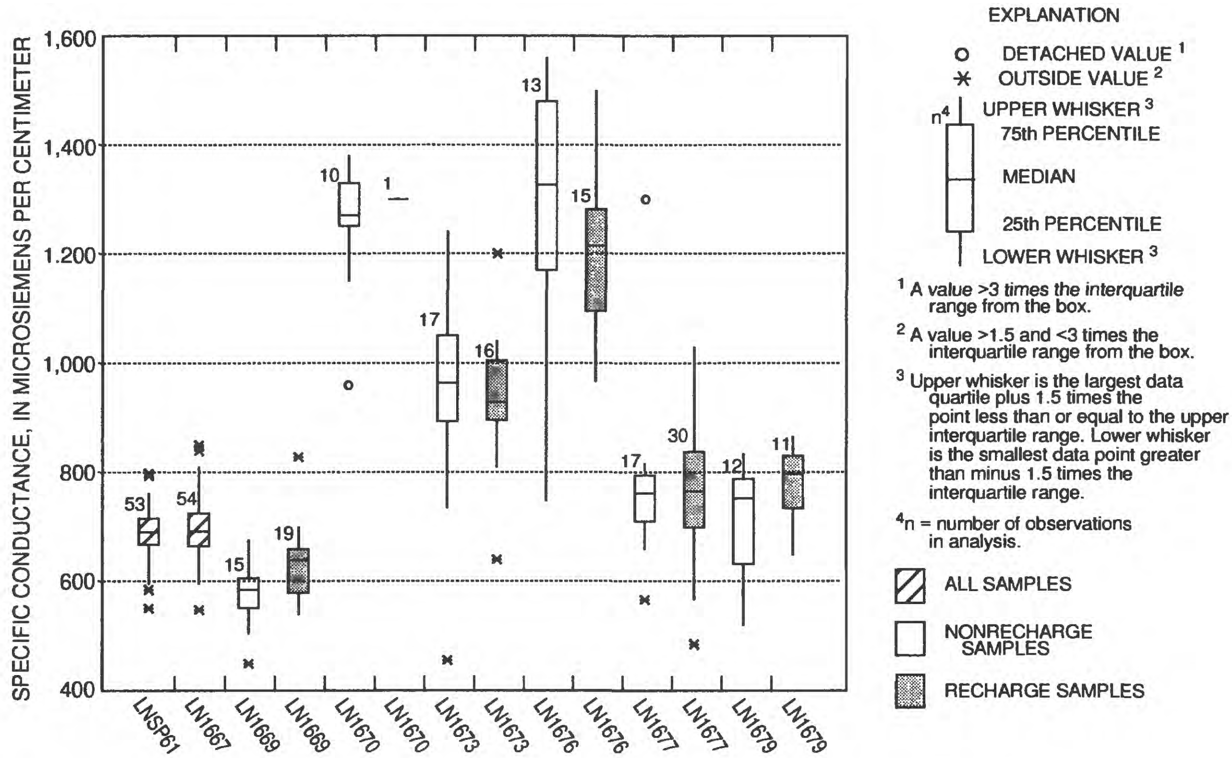


Figure 33. Distribution of specific conductances in ground-water samples from wells at Field-Site 2, October 1984 through September 1986.

Concentrations of dissolved phosphorus in all ground-water samples collected during the study period ranged from below the detection limit of 0.01 mg/L at several sites to a maximum of 0.51 mg/L in a sample collected at well LN 1673. The distribution of concentrations of dissolved phosphorus in all samples collected are shown in figure 34. Median concentrations of dissolved phosphorus ranged from 0.01 mg/L in samples from well LN 1679 to 0.06 mg/L in samples from well LN 1677. Phosphorus is essentially unavailable for leaching to the ground water because it rapidly sorbs to soil particles at the land surface and in the unsaturated zone (Graves, 1986b). It is possible that the largest (outlying) phosphorus concentrations (fig. 34) were caused by the use of a well-developer solution (containing phosphorus compounds) that was used on site wells during well completion.

Concentrations of ammonia plus organic nitrogen in ground-water samples collected during the study period were generally less than 0.3 mg/L (fig. 34). Under field conditions, organic nitrogen oxidizes to ammonia, which can then sorb to unsaturated zone clay particles in the form of ammonium. Sorbed ammonium oxidizes to soluble nitrate that moves easily with infiltration water through the soil column to the water table.

Nitrite concentrations ranged from below the detection limit of 0.01 mg/L in samples from several wells to 0.19 mg/L in samples from well LN 1677. Nitrite represents a short-lived intermediate oxidation product in the nitrification of ammonia to nitrate. Median concentrations of nitrite ranged from 0.01 to 0.04 mg/L for water samples from seven wells and spring (fig. 34).

All of the ground-water samples were analyzed for dissolved nitrate plus nitrite (fig. 35); most samples were analyzed for dissolved nitrite (fig. 34). Because concentrations of dissolved nitrite were consistently very small compared to nitrate concentrations, the analyzed concentrations of nitrate plus nitrite are referred to as dissolved nitrate for all samples. Dissolved nitrate accounted for over 90 percent of the total nitrogen in the ground water at the site.

Nitrate concentrations in the ground water commonly exceeded the U.S. Environmental Protection Agency (USEPA) drinking water maximum contaminant level of 10 mg/L as nitrogen (U.S. Environmental Protection Agency, 1989). Minimum and maximum concentrations of dissolved nitrate in water samples from all wells, except well LN 1670 (which probably was affected by an ammonia spill at the site prior to the study period), were 7.4 and 100 mg/L, respectively (fig. 35).

Relation to Agricultural Activities and Recharge

The transport of nutrients contained in manure and commercial fertilizer to the water table is a highly dynamic and complex system at the site. Bacterial processes that convert organic nitrogen to ammonium, nitrate, and nitrogen gases are affected in unknown ways by changes in temperature, moisture, and herbicide and pesticide applications (Stevenson, 1982).

Because animal operations at Field-Site 2 generate approximately twice the amount of manure-nitrogen that is recommended for application to site crops, application of the animal manure generated at the site causes excessive amounts of nitrogen to become available to accumulate in soils and to leach to the ground-water system (Graves, 1986a and 1986b; Roth and Fox, 1990).

Soil-test data at Field-Site 2 indicate that up to 400 lb per acre of soluble nitrate are stored in the top 4 ft of soils after crop harvest. Test data were for soluble nitrate only and did not include potentially greater amounts of ammonium and organic nitrogen in the soils.

Nitrate from manure and commercial fertilizer applications is transported with recharge to the ground water throughout the year. The Penn State Agronomy Guide (Pennsylvania State University, 1989) indicates that only 15 to 75 percent of the nitrogen in manure is available at the time of application as soluble nitrate for crop use. After spring or fall nitrogen applications, some residual nitrogen becomes available for crop use or leaching to the ground water as the organic nitrogen decays to ammonium that eventually decays to nitrate. Because these bacterially mediated processes occur throughout the year as a function of soil temperature and conditions, substantial loads of nitrate may become available for leaching to the ground water during the fall, winter, and early spring seasons, when no crops are utilizing the nutrients, and nitrate is also transported to ground water in the summer below the crop root zone. Winter cover crops such as rye may fix some of this nitrate from the root zone, however nitrate can still leach

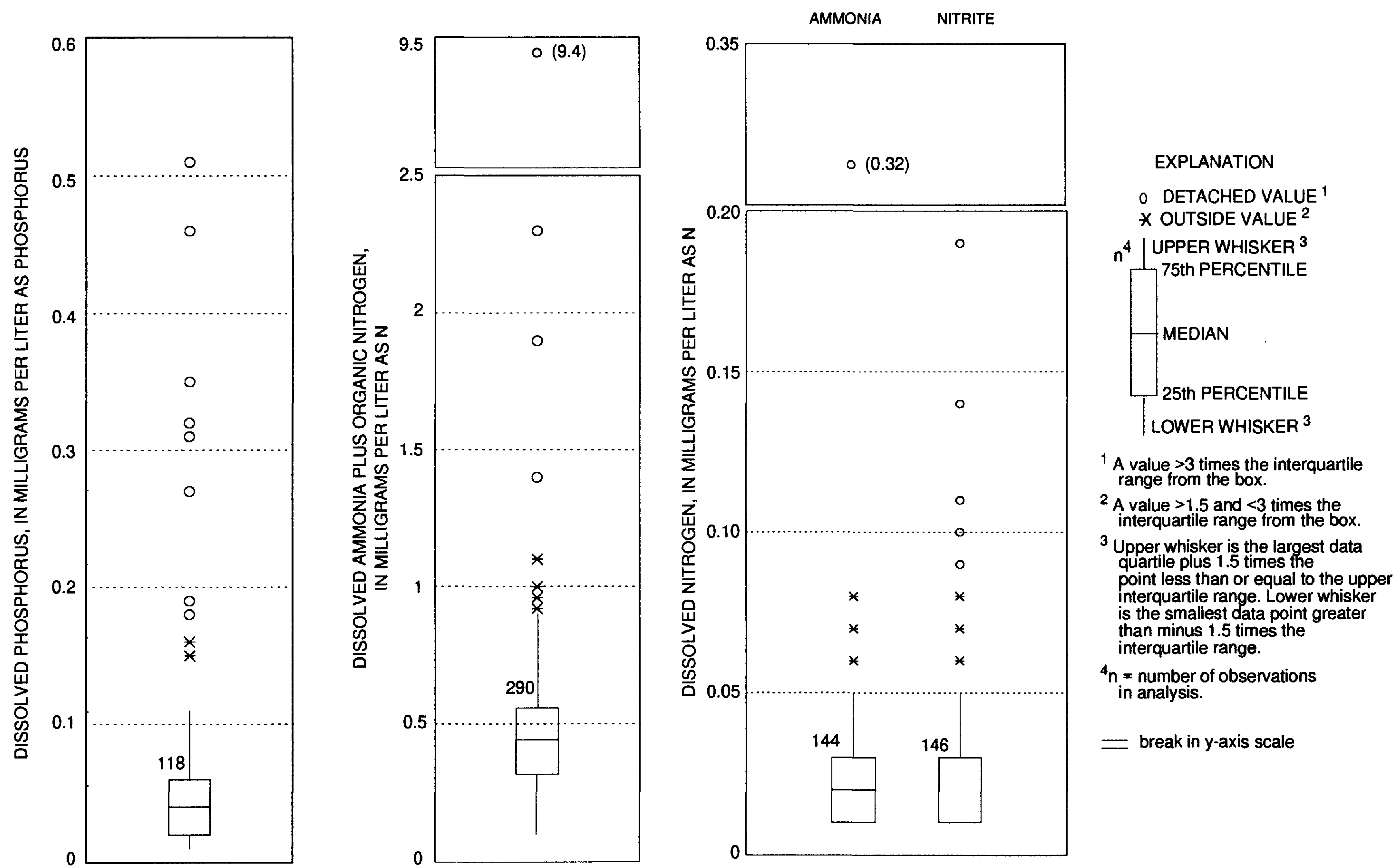


Figure 34. Distribution of dissolved-nutrient concentrations in ground-water samples from Field-Site 2, October 1984 through September 1986.

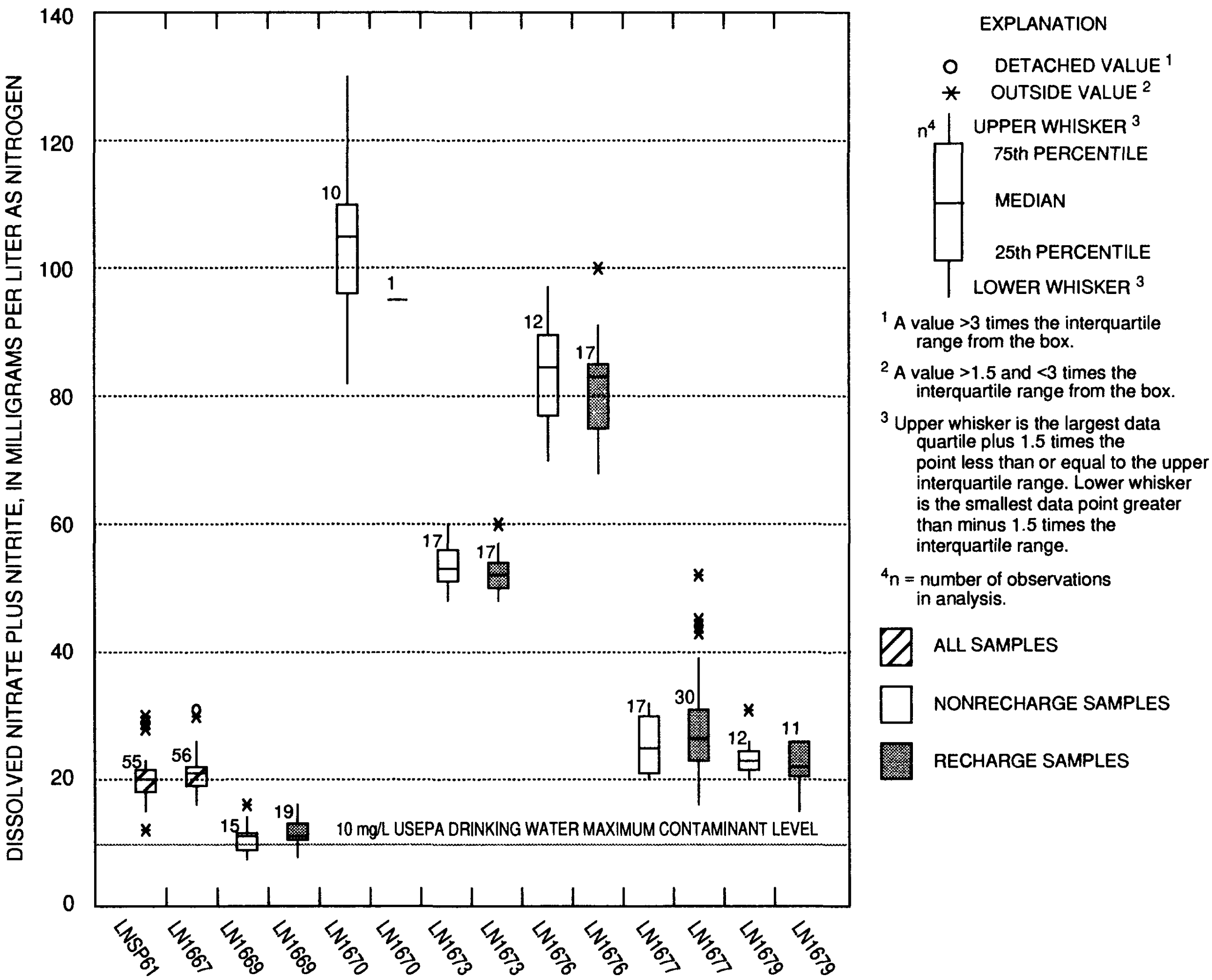


Figure 35. Distribution of dissolved nitrate plus nitrite concentration in ground-water samples from Field-Site 2, October 1984 through September 1986.

freely to the ground water beneath the root zone. Scientists have traditionally been interested in root-zone nutrient processes related to crop yields, and little research has been done to document nutrient transport through the "intermediate vadose zone" (Spalding and Kitchen, 1988).

Contributing areas (fig. 36) to seven wells and the spring were estimated in an effort to relate applications of nitrogen to farm fields to concentrations of nitrate in ground water. For the purpose of contributing area definition, soil- and ground-water flow at the site are assumed to resemble isotropic flow in a homogenous-porous media. The contributing areas were determined by first locating a primary flow line upgradient from each well and the spring, taking into account heads suggested by the slope of the water-table surface. The flow lines were then expanded by use of a 2:1 (roughly 25 degree) ratio of longitudinal to lateral dispersion (general transport processes discussed in Bouwer, 1978, and Memon and Prohic, 1989). The true dispersion ratio of this aquifer is unknown, because no tracer tests were conducted on the site. Some degree of dispersion undoubtedly occurs as materials infiltrate down through the unsaturated zone and additional dispersion (possibly anisotropic) probably occurs in the aquifer. While nutrient applications made far upgradient from a well can potentially contribute nitrogen to that well, applications made closer to the well could contribute greater concentrations of nitrogen to the well, because less dispersion occurs along shorter flow paths. Therefore, contributing areas were arbitrarily cut off at a distance of approximately 1,000 ft upgradient from each well or spring in order to determine an area of maximum influence.

Contributing area acreage, nitrogen applications, and median nitrate concentrations in ground water for the study period are shown in table 19. It is not known if the nitrogen applications listed on table 19 produced the measured concentrations of nitrogen in ground water. It is possible that concentrations of nitrate in ground water were affected by nitrogen applications made prior to the start of the study or other undefined factors.

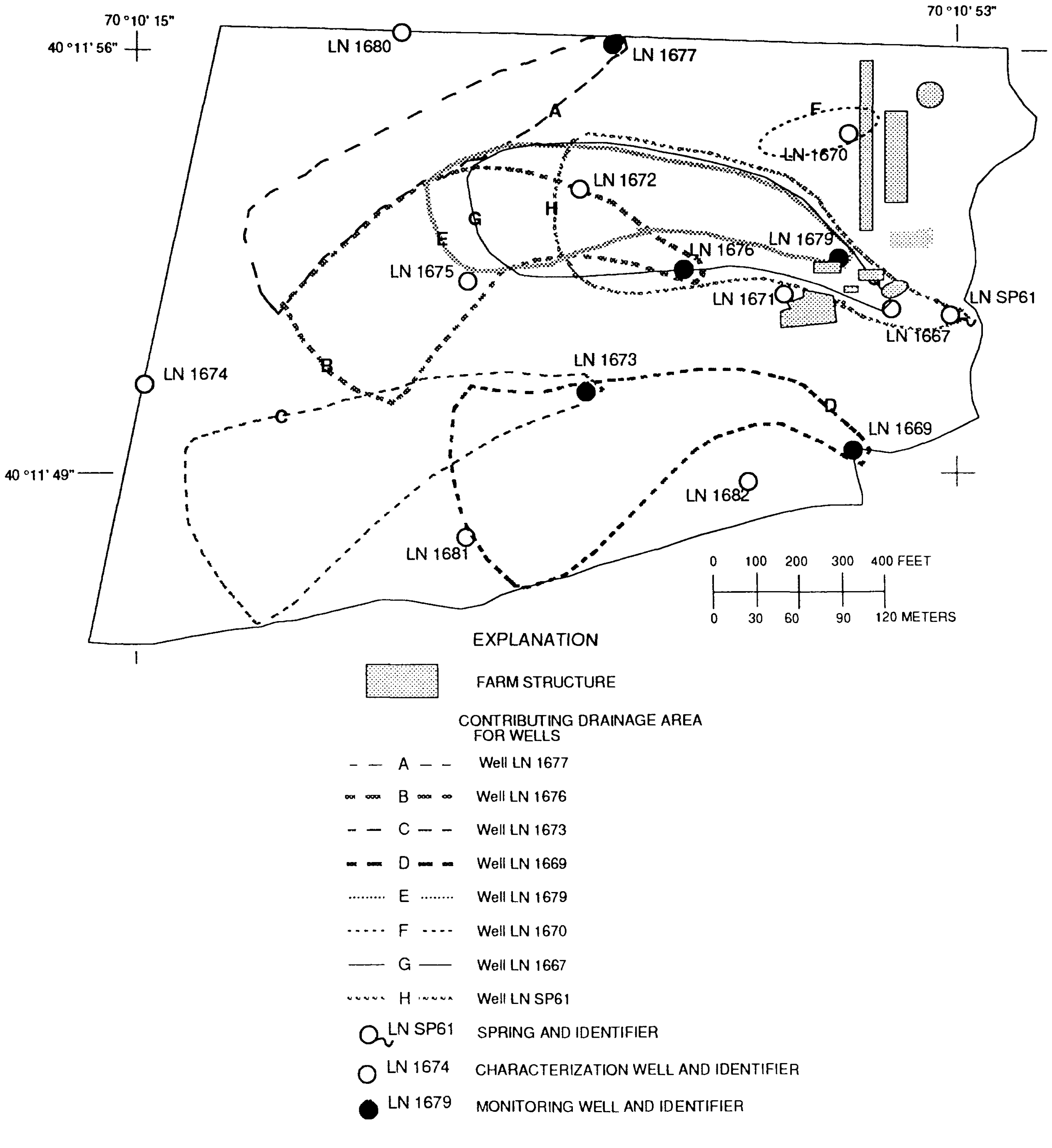


Figure 36. Estimated contributing areas for seven wells and the spring at Field-Site 2.

Table 19. Contributing area acreage, nitrogen applications, and median concentrations of nitrate in ground water for seven wells and the spring at Field-Site 2, October 1984 through September 1986

Well or spring	Contributing area acreage	Nitrogen applied in well contributing areas, in pounds per acre per year	Median nitrate concentration for all samples, in milligrams per liter
LN SP61 (spring)	4.8	640	20
LN 1667	4.8	940	21
LN 1669	5.6	480	12
LN 1670[1]	4.6	310	100
LN 1673	5.6	580	53
LN 1676	5.5	500	83
LN 1677	3.4	510	26
LN 1679	4.1	630	22

[1] Nitrate concentrations in ground water at well LN 1670 were probably affected by an ammonia spill that occurred before the study began and nitrogen loading from nearby animal shelters. Therefore, nitrate concentrations in water from this well did not originate only from field-applied nitrogen.

Shuford and others (1977) have determined that many limestone soils in Pennsylvania used for agriculture are highly structured and may contain both micropores and macropores in the soil matrix (White, 1985). Micropores are comprised of the small voids between soil granules; macropores are larger voids and passages in the soil tilth developed by processes such as root growth and worm tunnelling. Once the development of a macropore passage has been initiated, further development may occur through subsurface erosion associated with rapidly moving infiltration water. Infiltration is thought to proceed to the water table slowly in the form of wetting fronts through the micropore system and very rapidly as direct recharge through the macropore system. The theory is that precipitation infiltrates into the micropores of the soil until the infiltration capacity of the soil is exceeded, and then water is diverted into the soil macropore system.

Dual porosity in Pennsylvania soils is thought to have a pronounced effect on ground-water nitrate concentrations (Gerhart, 1986; Shuford and others, 1977). Recharge water that infiltrates slowly through the micropore system has sufficient time to oxidize ammonium to nitrate and transport soluble nitrogen to the water table; recharge occurring rapidly as macropore flow has little time to dissolve or leach nitrate to the water table. Therefore, rain that falls at slow rates over a long period (several days or longer) may lead to increases in ground-water nitrate levels because of recharge transport by micropores (if nitrate is available in the soils and regolith); an equal amount of rain falling at a rapid rate in a short period (1 or 2 days) may lead to dilution of nitrate concentrations in ground water because of recharge transport through macropores.

Data collected at the site may provide qualitative evidence in support of the theory that dual porosity in site soils may affect nitrate concentrations in ground water (figs. 37, 38, and 39). A series of storms in late November 1985 slowly deposited about 4 in. of rain at the site. Recharge from the storms caused the water table to rise about 2 ft. Because the rain fell over about a 1-week period, water could have slowly infiltrated to the water table through the soil micropore system. Concentrations of nitrate in water from well LN 1677 increased significantly during this period (fig. 37). However, when about 4 in. of rain occurred on July 26, 1985 (a 1-day period), and the water table rose about 2 ft in well LN 1677, concentrations of nitrate in ground water decreased (fig. 38). Recharge during this storm may have moved through the soil macropore system, bypassing available nitrate that may have been stored in soil micropores. Another example of the possible diluting effect of macropore flow is illustrated in figure 39. About 3 in. of rain fell during a 48-hour period from April 1-3, 1985. Water levels in well LN 1673 rose about 1.5 ft in response to recharge during this storm, and nitrate concentrations in ground water decreased approximately 5 mg/L.

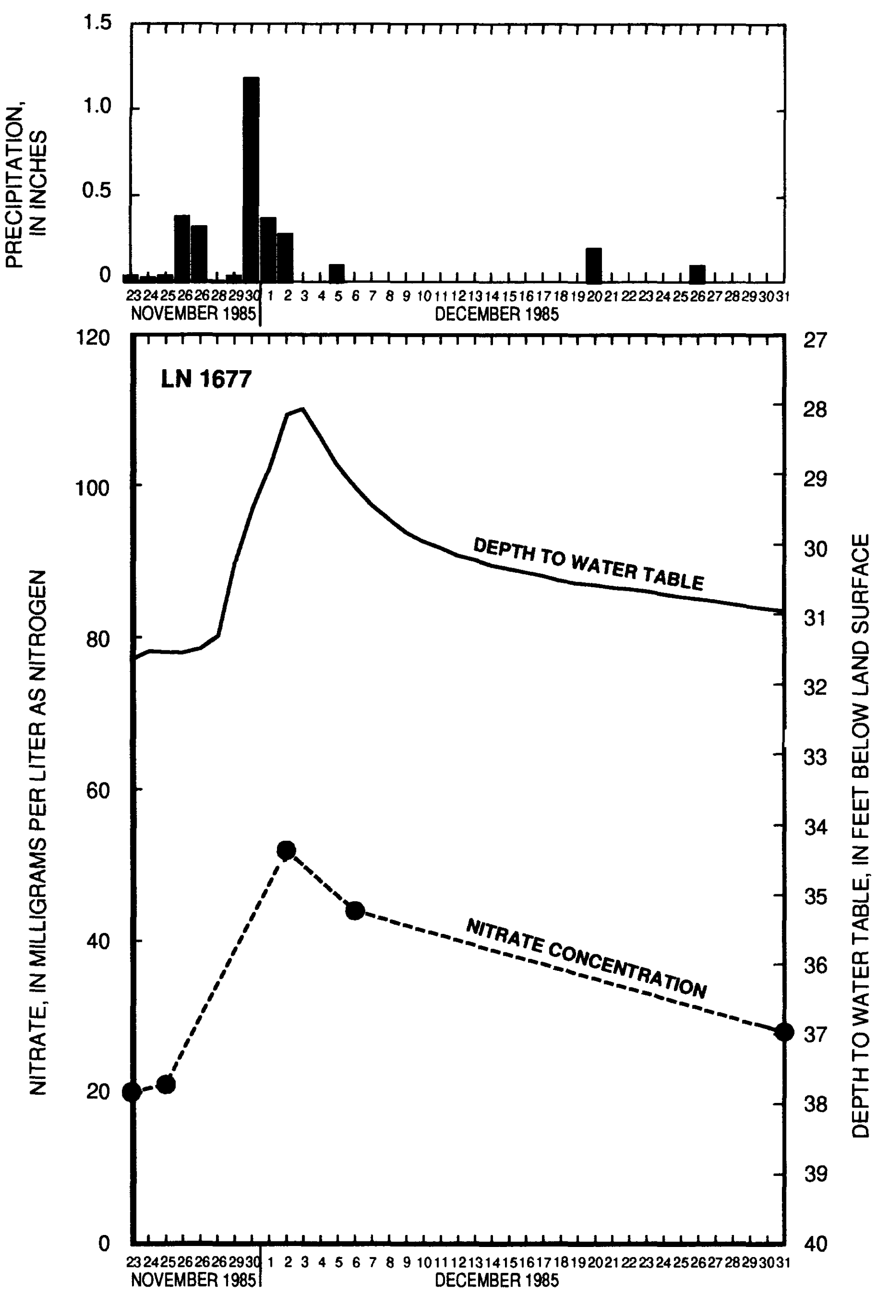


Figure 37. Precipitation, depth to water table, and nitrate concentrations at well LN 1677, November 23 through December 31, 1985, at Field-Site 2.

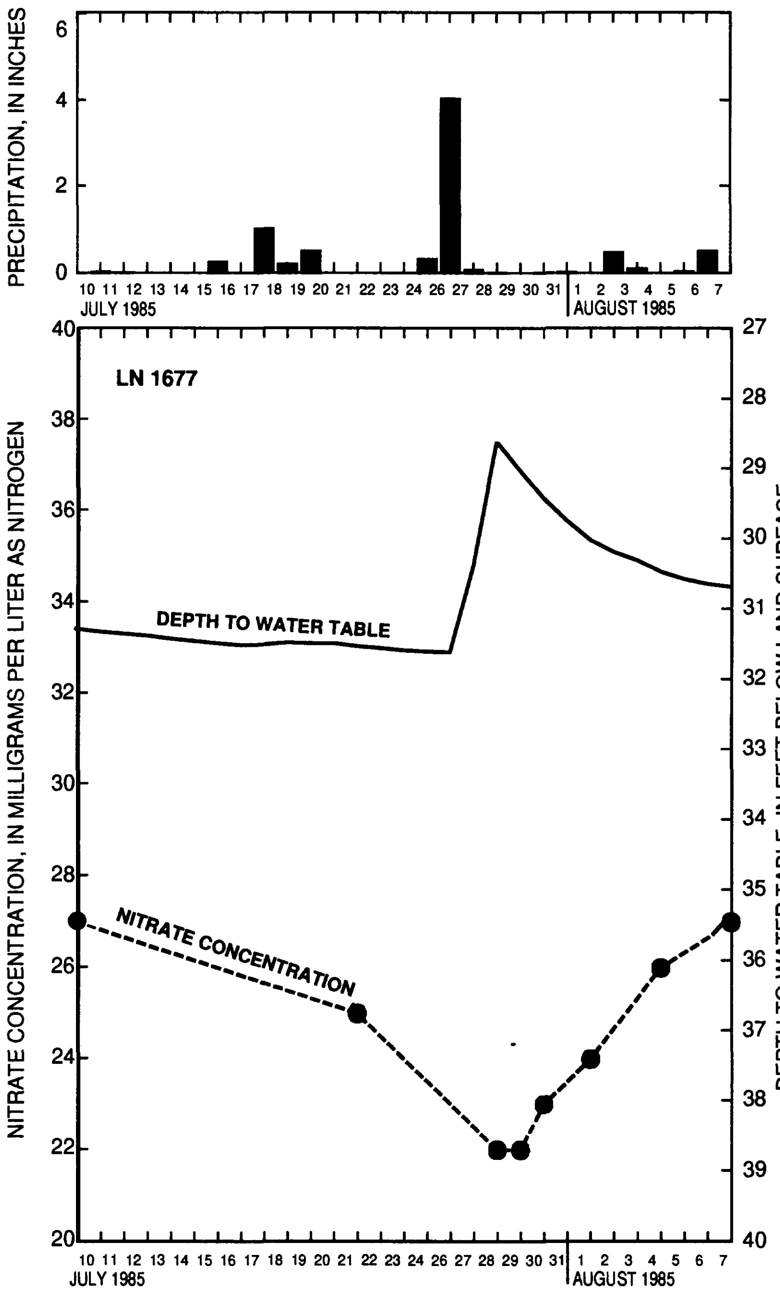


Figure 38. Precipitation, depth to water table, and nitrate concentrations at well LN 1677, July 10 through August 7, 1985, at Field-Site 2.

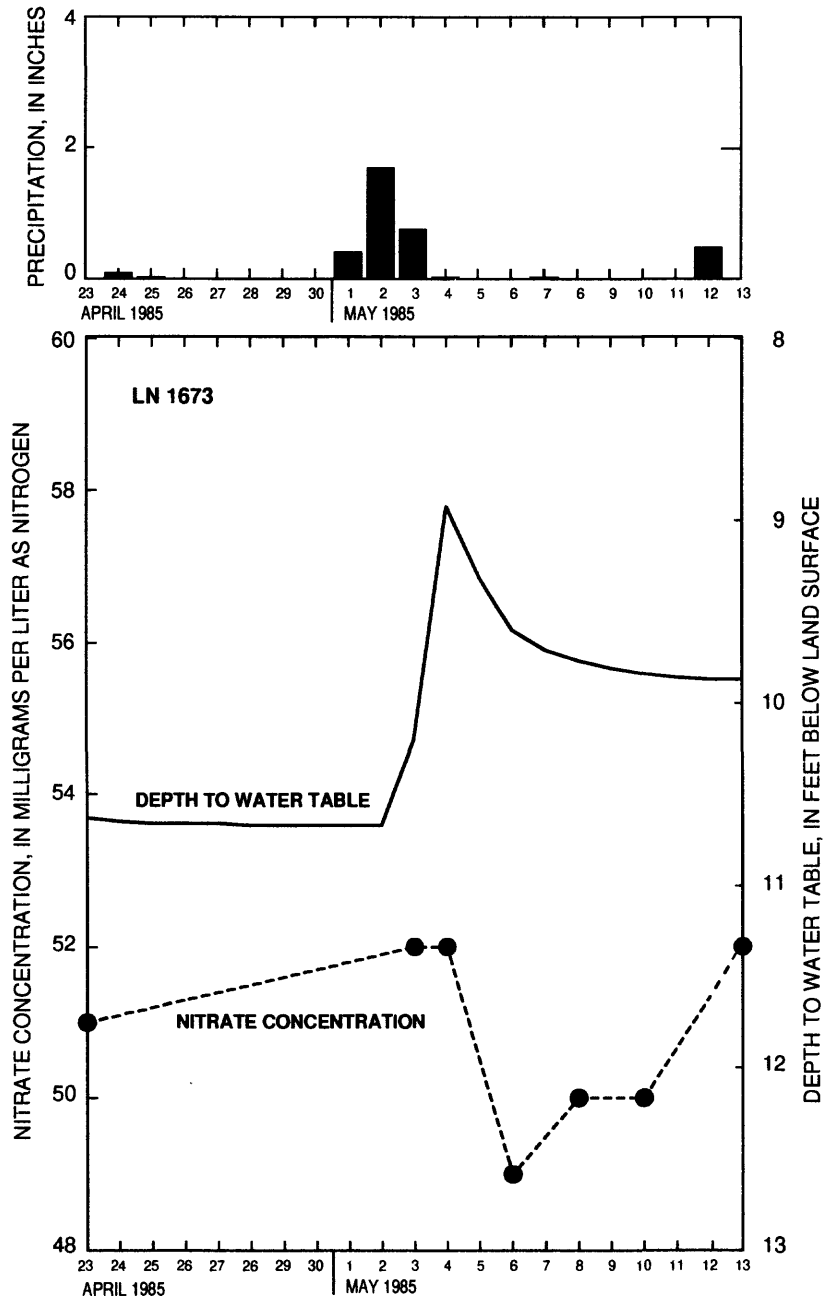


Figure 39. Precipitation, depth to water table, and nitrate concentrations at well LN 1673, April 23 through May 13, 1985, at Field-Site 2.

The limited amount of data presented in figures 37, 38, and 39 support the observations of Shuford and others (1977) regarding the diluting effects of macropore flow on nitrate concentrations of ground water beneath farm fields; however, the data are not conclusive. Other factors, including the amount of available nitrogen in the unsaturated zone, undoubtedly influence soil-water quality, and further research is needed to investigate macropore flow and its effects on ground-water quality.

Although nitrate concentrations in ground water are affected by storm recharge, storm-induced variations in concentration (enrichment or dilution) are of short duration. If no additional recharge occurs, nitrate concentrations in ground water return to prestorm levels in approximately 1 to 3 weeks (figs. 37, 38, and 39). Long-term changes in nitrate concentrations of ground water are probably determined by the slower-moving, micropore-transmitted recharge water, water that may effectively transport the nitrogen applied to farm fields through the unsaturated zone to ground water.

The altitude of the water table is slightly elevated in the vicinity of well LN 1670, which may cause this well to have a smaller contributing area than other wells at the site. A comparison of boxplots of nitrate-concentration data in water samples from all wells and the spring (fig. 34) show that the greatest variations and largest median concentrations were in water samples collected from well LN 1670. The contributing area to this well was evidently affected by a farm accident that occurred on September 11, 1983 (Pennsylvania Department of Environmental Resources, written commun., 1983). Temperatures of 95°F caused a tank of liquid anhydrous ammonia to fail. After approximately 50 gallons of ammonia had spilled onto the ground by the silo near well LN 1671 (fig. 35), the farmer moved the leaking tank to a land-surface depression downgradient of well LN 1677. Although wind dispersed the fumes early in the day, increasing calm allowed for the accumulation of fumes, and water was misted into the fumes late in the day and through the night until the tank was empty. The misting created a pool of ammonia-rich water that covered from 1 to 2 acres. A water sample collected from this pool was analyzed and found to contain 3,060 mg/L of ammonia, 0.612 mg/L of nitrite, and 12.65 mg/L of nitrate. Recharge from the misting may have produced a temporary ground-water mound that caused the contaminated water to flow in the direction of well LN 1670. The total land-surface area affected the spill, including the initial spill site, pool site, and drag trail, was about 4 acres. The farmer does not recall moving any of the contaminated soil and reported high corn yields from that field during the following year. Assuming a volatilization rate of 45 percent for the spilled ammonia, the spill is estimated to have contributed from 1,800 to 2,700 lb of nitrogen to the soils over a 4-acre area. The farmer frequently made manure applications of this magnitude to fields during the study period.

While it is probable that the spill affected soil and ground-water nitrate concentrations around well LN 1670, the magnitude of the presumed effect is unknown because no pre-spill ground-water-quality data exists (the spill occurred in 1983, before this study began). Nitrate concentrations in water samples collected at well LN 1670 declined at a rate of about 20 mg/L per year during the 2-year study period, from about 120 mg/L to about 80 mg/L (fig. 40).

Because the ammonia spill probably contributed to the elevated nitrate concentrations of ground water in well LN 1670, data from this well are not useful to test the effects of nutrient management on ground water.

Water-level elevation, precipitation, nitrate concentrations in ground-water samples, and nitrogen applications in the contributing area to well LN 1669 are shown in figure 41. This well was sampled at 85 ft, which is 50 ft deeper than most of the other wells, and sample analyses probably represent the ground-water chemistry of a deeper part of the aquifer than the sample analyses from the other sites. Possible explanations for the significantly low ground-water nitrate concentrations include (1) a constant or intermittent denitrification process may affect ground water at depth, but a dissolved-oxygen measurement of 5.7 mg/L made during the May 13, 1986, sampling is inconsistent with this explanation, because denitrification is usually considered to be an anaerobic process (intermittent processes could still be responsible); (2) the ground water at well LN 1669 may not be hydrologically connected to the shallower aquifer system, but this explanation is disputed by the fact that samples collected at well LN 1669 show rapid responses in nitrate concentrations to storm recharge as do the other wells; and (3) part of the area contributing water to this well may be outside the field-site boundary to the south and could, therefore, receive substantially less applied nitrogen than is calculated. However, because almost

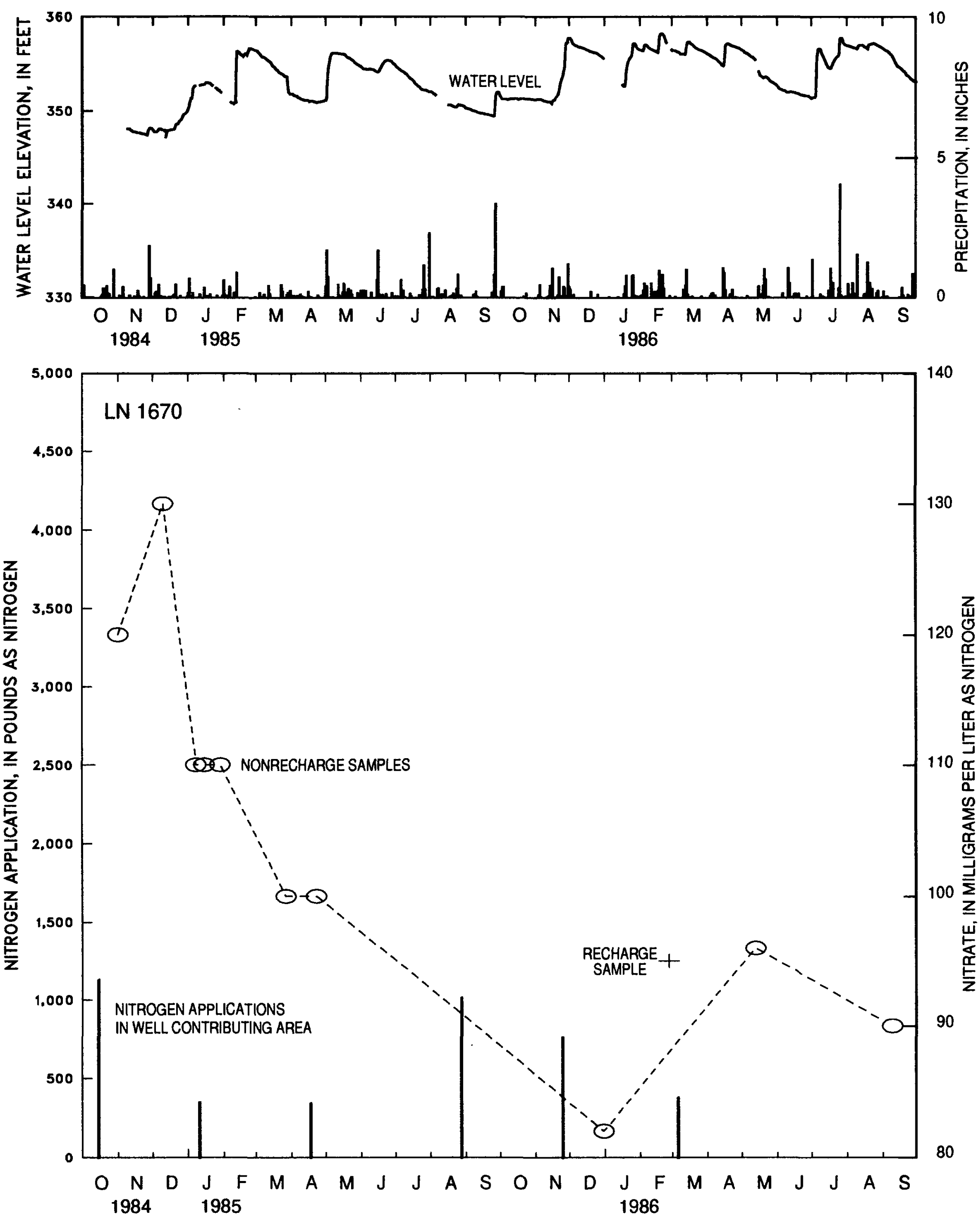


Figure 40. Water-level altitude, precipitation, nonrecharge and recharge nitrate concentrations, and nitrogen applications at well LN 1670, October 1984 through September 1986, at Field-Site 2.

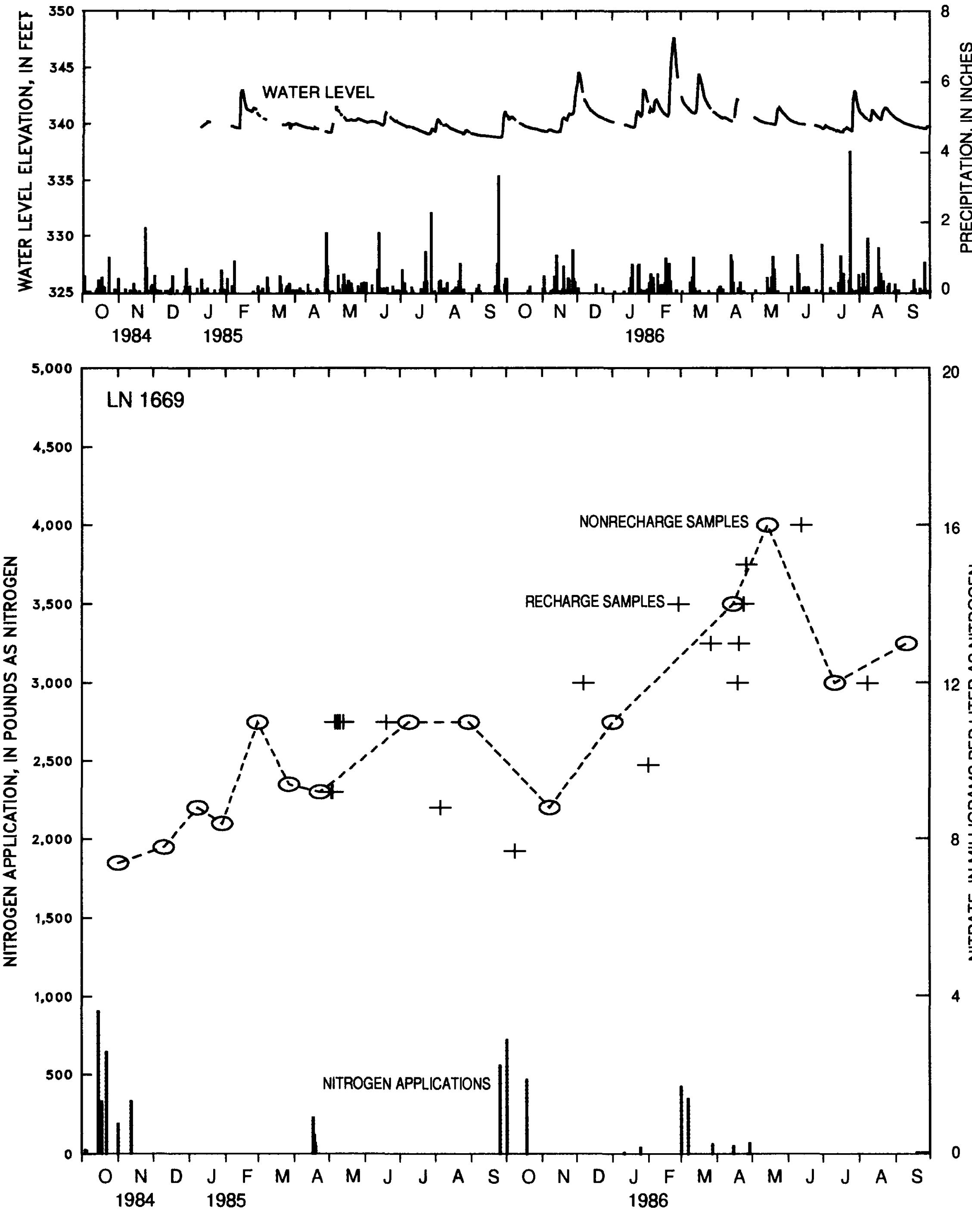


Figure 41. Water-level altitude, precipitation, nonrecharge and recharge nitrate concentrations, and nitrogen applications at well LN 1669, October 1984 through September 1986, at Field-Site 2.

any flow direction is possible in such a folded anisotropic carbonate aquifer, this explanation is disputed by observed ground-water levels, flow predicted by the ground-water model, and the fact that a creek is flowing in the opposite direction within 200 ft of the well.

Precipitation, nitrogen applications, and nitrate concentrations in water at well LN 1667 and spring LN SP61 are shown in figure 42. No water-level data were collected at this (hand-dug) well or spring. Concentrations of nitrate in ground water at these sampling sites are quite similar in magnitude and short-term trend. Well LN 1667 is located approximately 150 ft upgradient of spring LN SP61, and it is likely that they are connected by a fracture or other solutionally developed passage in the carbonate rock.

Water levels, precipitation, nitrogen applications, and concentrations of nitrate in ground water samples are shown for wells LN 1673, LN 1676, LN 1677, and LN 1679 on figures 43-46. Nitrate concentrations in water from these wells are not similar in magnitude or short-term trend, indicating that local variations in the physical properties of the site or different rates of nitrogen loading to farm fields may influence nitrate concentrations in different parts of the site.

Insufficient data exist to determine relations between nitrogen applications in sampling site contributing areas and concentrations of nitrate in ground water at the sampling site. Additional data collected during the postnutrient-management phase of the study may help to define relations that may exist between surface-applied nitrogen and concentrations of nitrate in ground water.

Nitrogen Loads

Monthly and annual nitrogen loads in ground-water discharge across each boundary from Field-Site 2 were computed and are shown in tables 20 and 21.

Calculated nitrogen loads in ground-water discharge totalled 3,200 lb in the 1985 water year and 7,600 lb in the 1986 water year. While actual numbers used in these calculations are estimations, comparisons of ground-water nitrogen discharge between years should be proportionally significant.

WATER BUDGET

Estimation of nitrogen inputs to, and outputs from, the site required the calculation of loads of nitrogen in surface and ground water. Estimation of an accurate water budget of the site was essential to the accuracy of the nitrogen load estimations.

A water budget during an average year at Field-Site 2 was estimated from measurements of runoff and estimates of ground-water recharge. The budget is based on the assumption that precipitation on the site is balanced by runoff, ground-water recharge, and evapotranspiration. Because of the rapid movement of ground water through the site, storage on an annual basis was assumed to be zero. The normal precipitation at Ephrata is 43.5 in/yr (National Oceanic and Atmospheric Administration, 1984). Runoff is a small percentage of the total precipitation. Measurements indicate that runoff was no greater than 4 percent of the precipitation recorded at the site. Recharge to ground water was estimated to be about 53 percent of precipitation for water year 1986. Withdrawals from well LN 1667 are not included in the water budget calculations because most of the water withdrawn is quickly returned to the aquifer. Water lost from the site as evapotranspiration is computed as the residual term in the balance; in this case about 43 percent of precipitation. The water balance is represented by the following equation:

$$\text{precipitation} = \text{runoff} + \text{ground-water recharge} + \text{evapotranspiration} \qquad (7)$$

$$43.5 \text{ inches} = 1.6 \text{ inches} + 23.0 \text{ inches} + 18.9 \text{ inches}$$

$$(100 \text{ percent}) = (4 \text{ percent}) + (53 \text{ percent}) + (43 \text{ percent})$$

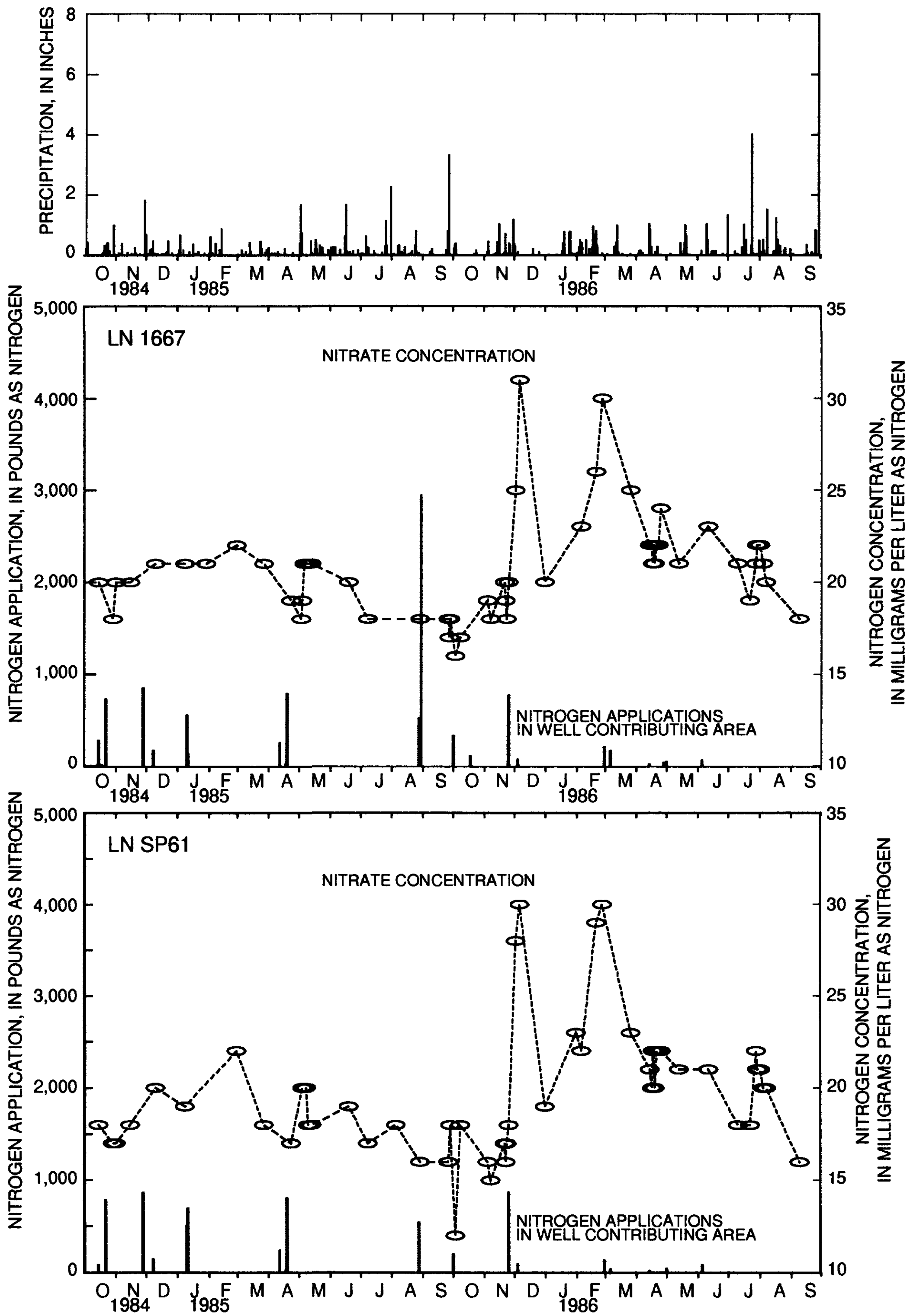


Figure 42. Precipitation, nitrate concentrations, and nitrogen applications at well LN 1667 and spring LN SP61, October 1984 through September 1986, at Field-Site 2.

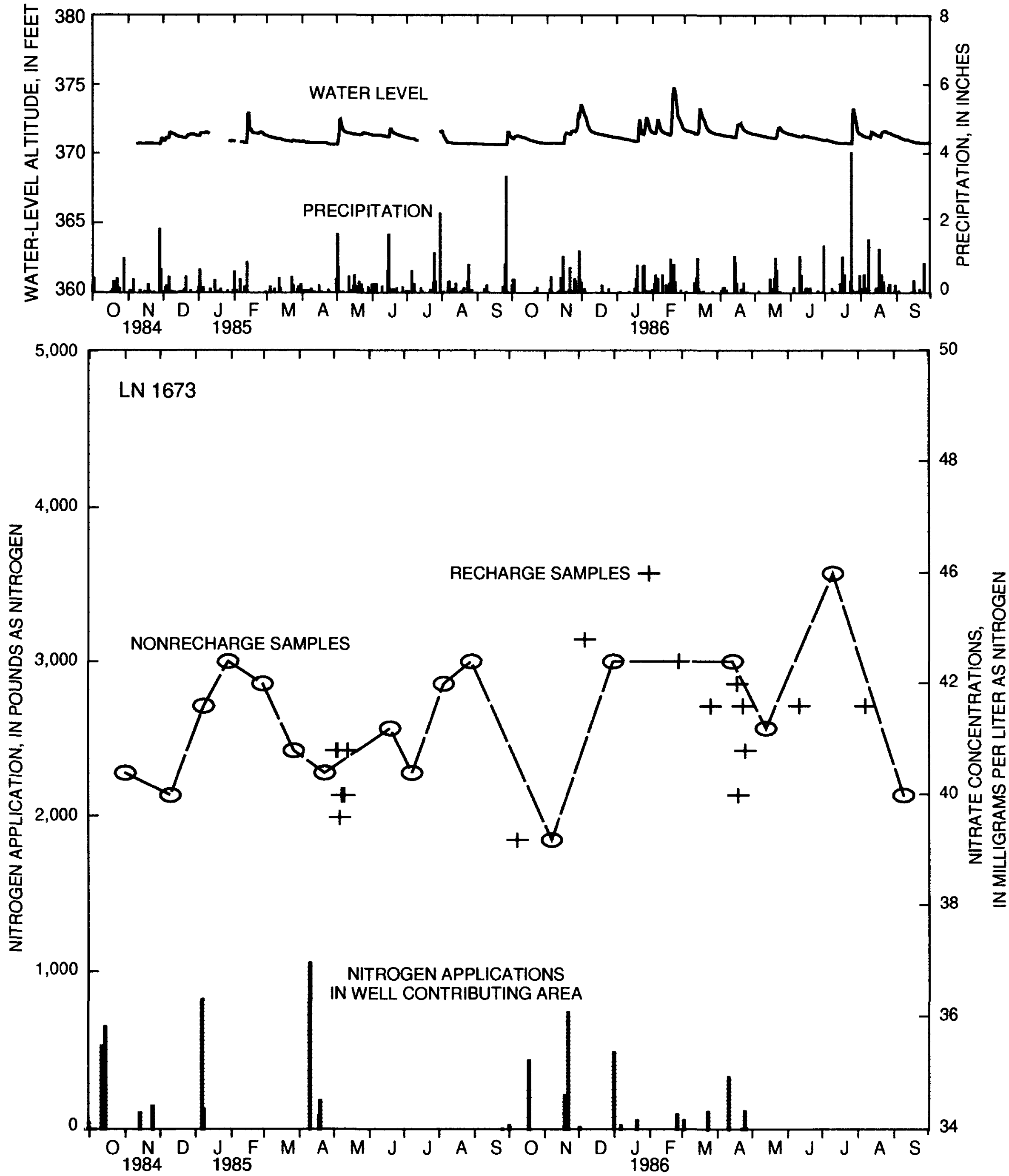


Figure 43. Water level, precipitation, nonrecharge and recharge nitrate concentrations, and nitrogen applications at well LN 1673, October 1984 through September 1986, at Field-Site 2.

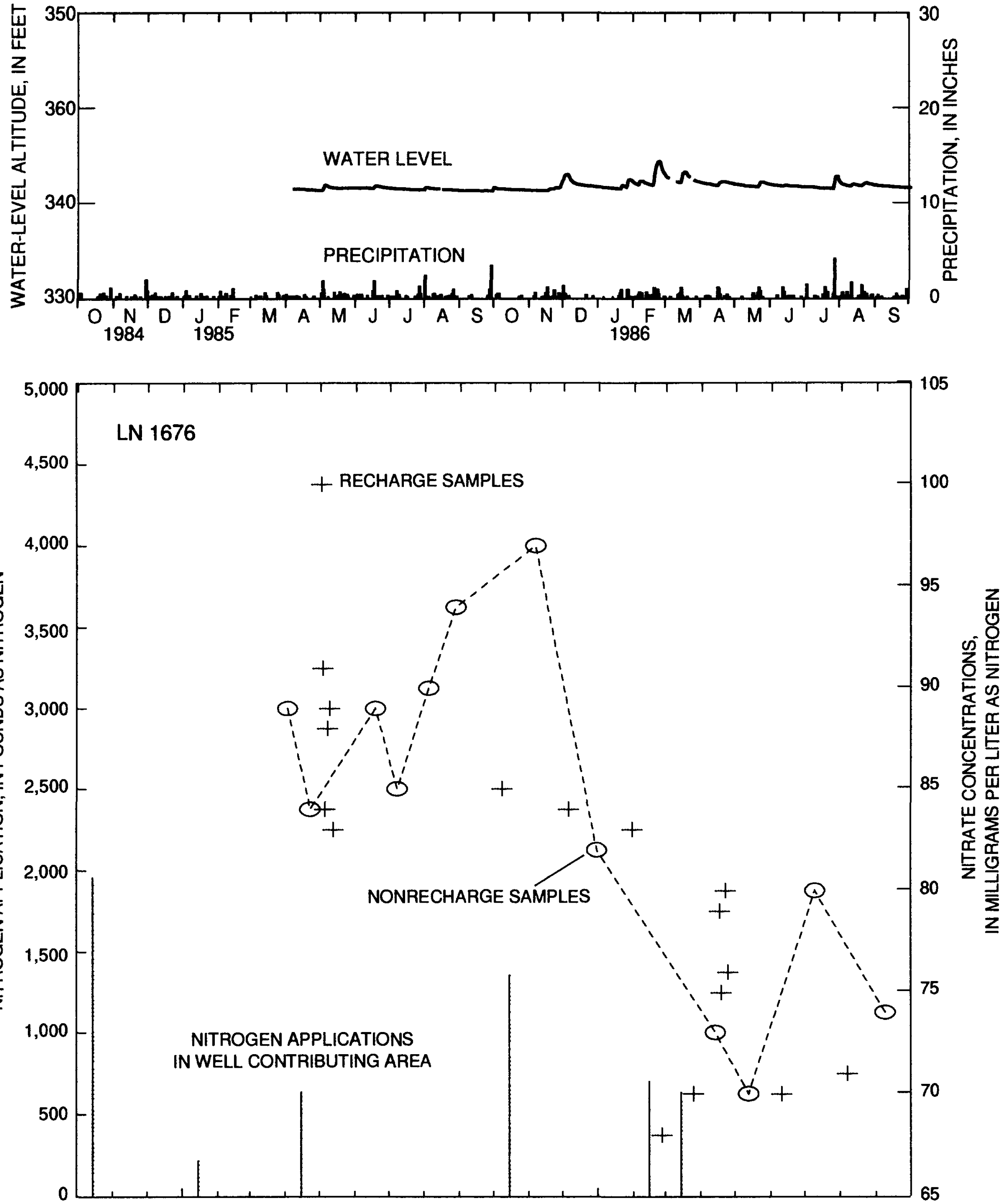


Figure 44. Water level, precipitation, nonrecharge and recharge nitrate concentrations, and nitrogen applications at well LN 1676, October 1984 through September 1986, at Field-Site 2.

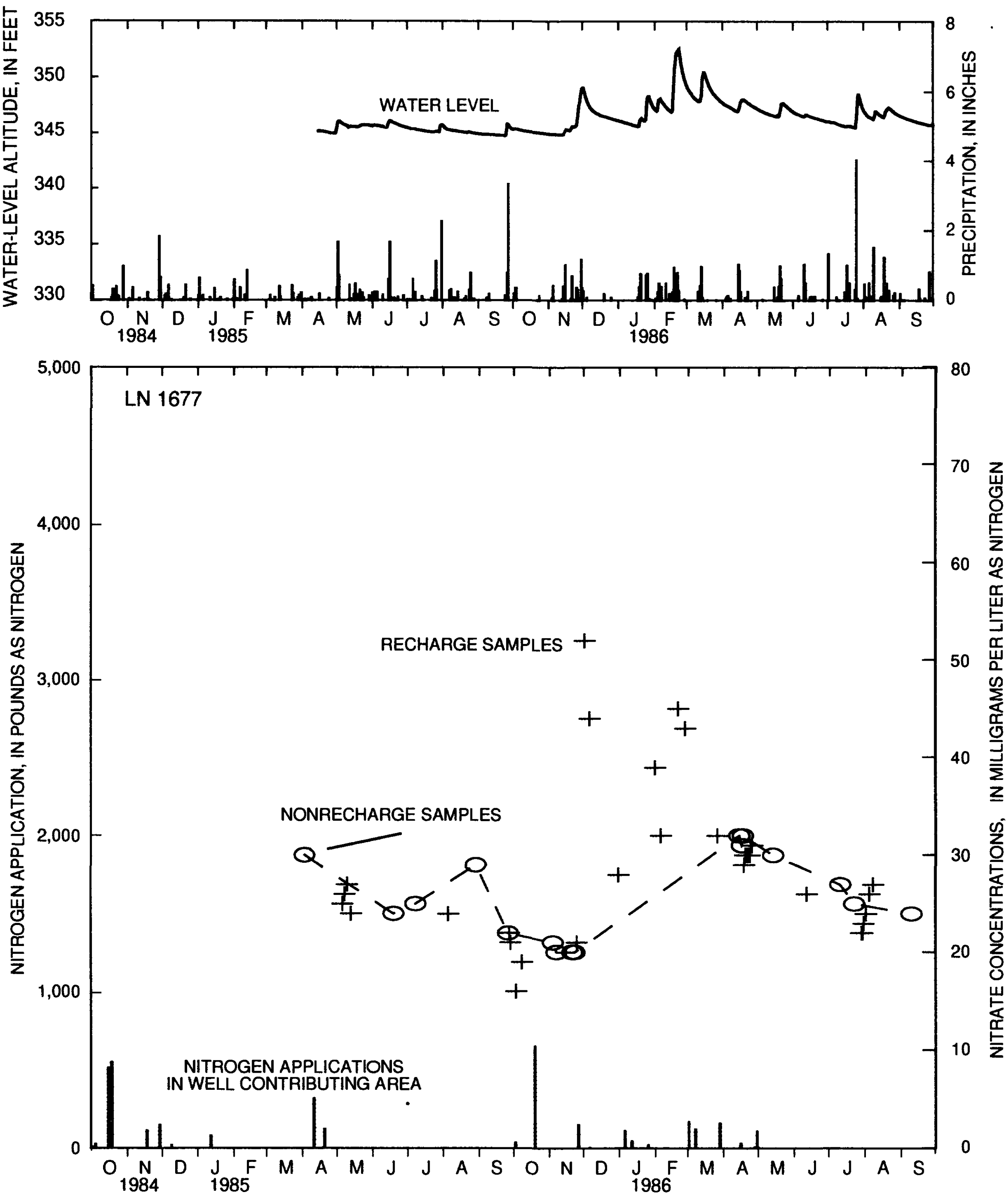


Figure 45. Water level, precipitation, nonrecharge and recharge nitrate concentrations, and nitrogen applications at well LN 1677, October 1984 through September 1986, at Field-Site 2.

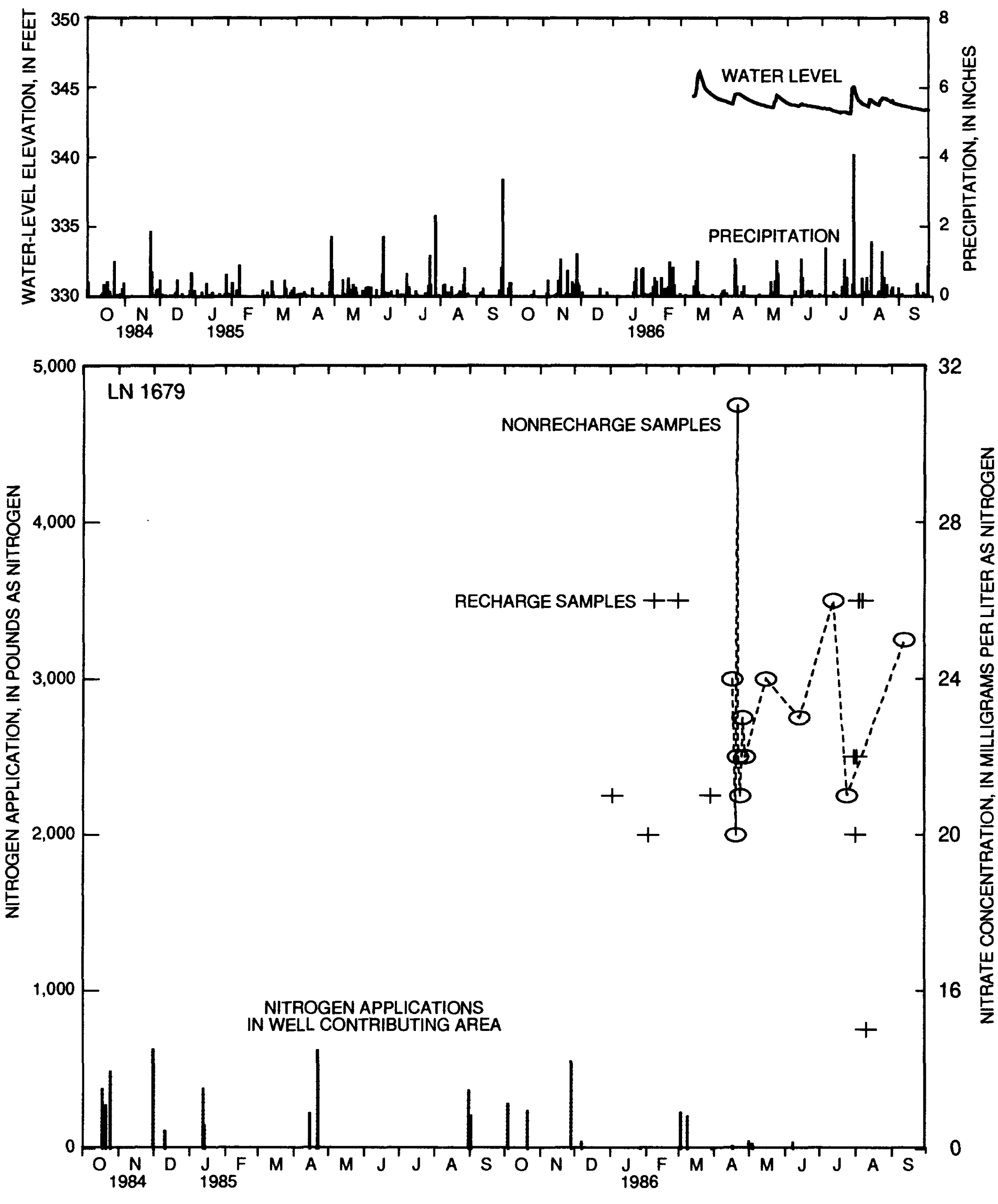


Figure 46. Water level, precipitation, nonrecharge and recharge nitrate concentrations, and nitrogen applications at well LN 1679, October 1984 through September 1986, at Field-Site 2.

Table 20. Calculation of nitrgen load discharged across the northern boundary of Field-Site 2 during the 1985-86 water years

[A, volume of ground-water recharge entering the site across the western boundary, calculated as 16 percent of water year ground-water recharge from precipitation, in liters; B, volume of ground-water recharge entering the site from precipitation, calculated by use of methods discussed in recharge section; C, milligram-to-pound conversion; D, percentage of ground water estimated by the model to discharge across a site boundary; E, monthly fraction of annual discharge; F, median monthly nitrate concentration, in milligrams per liter as nitrogen; G, monthly nitrogen load, discharged across the northern site boundary, in pounds]

(A + B)	×	(C)	×	(D)	×		(E)	×	(F)	=	(G)
				LN 1677 - Water Year 1985							
68,800,000		2.21×10^{-6}		0.26		OCT.	0.08		[1]25.3		80
						NOV.	.05		[1]25.3		50
						DEC.	.08		[1]25.3		80
						JAN.	.07		[1]25.3		70
						FEB.	.28		[1]25.3		280
						MAR.	.03		[1]25.3		30
						APR.	.0		30.0		0
						MAY	.15		25.5		151
						JUNE	.06		24.0		57
						JULY	.03		25.0		30
						AUG.	.06		26.5		63
						SEPT.	.12		22.0		104
									TOTAL		995 ≈ 1,000
				LN 1677 - Water Year 1986							
137,000,000		2.21×10^{-6}		0.26		OCT.	0.003		17.5		4
						NOV.	.17		20.0		268
						DEC.	.0		44.0		0
						JAN.	.11		39.0		338
						FEB.	.31		43.0		1,049
						MAR.	.11		32.0		277
						APR.	.042		30.5		101
						MAY	.050		30.0		118
						JUNE	.01		26.0		22
						JULY	.12		23.0		217
						AUG.	.064		26.0		131
						SEPT.	.003		24.0		6
									TOTAL		2,531 ≈ 2,500

[1] Well LN 1677 was not drilled until April 1986. Therefore, nitrate concentrations used in this calculation were estimated by use of analyses from well LN 1667, a nearby well.

Table 21. Calculation of nitrogen load discharged across the eastern and southern boundaries of Field-Site 2 during the 1985-86 water years

[A, volume of ground-water recharge entering the site across the western boundary, calculated as 16 percent of water year ground-water recharge from precipitation, in liters; B, volume of ground-water recharge entering the site from precipitation, calculated by use of methods discussed in recharge section; C, milligram-to-pound conversion; D, percentage of ground water estimated by the model to discharge across a site boundary; E, monthly fraction of annual discharge; F, median monthly nitrate concentration, in milligrams per liter as nitrogen; G, monthly nitrogen load, discharged across the northern site boundary, in pounds]

(A + B)	×	(C)	×	(D)	×	(E)	×	(F)	=	(G)
				LN SP61 - Water Year 1985						
68,800,000		2.21×10^{-6}		0.74	OCT.	0.08		17.5		158
					NOV.	.05		17.5		98
					DEC.	.08		20.0		180
					JAN.	.07		19.0		150
					FEB.	.28		22.0		693
					MAR.	.03		18.0		61
					APR.	.0		17.0		0
					MAY	.15		19.0		321
					JUNE	.06		19.0		128
					JULY	.03		17.0		57
					AUG.	.06		17.0		115
					SEPT.	.12		17.0		230
								TOTAL		2,191 ≈ 2,200
				LN SP61 - Water Year 1986						
137,000,000		2.21×10^{-6}		0.74	OCT.	0.0		15.0		0
					NOV.	.17		16.5		628
					DEC.	.0		28.0		0
					JAN.	.11		23.0		567
					FEB.	.31		29.0		2,014
					MAR.	.11		23.0		567
					APR.	.042		22.0		207
					MAY	.050		21.0		235
					JUNE	.01		21.0		52
					JULY	.12		21.0		565
					AUG.	.064		20.0		287
					SEPT.	.0		16.0		0
								TOTAL		5,122 ≈ 5,100

ADDITIONS AND REMOVALS OF NITROGEN

Additions and removals of nitrogen at Field-Site 2 were calculated for 1985 and 1986. Annual amounts are shown in table 22 in addition to average quantities for the study period. Nitrogen was added to the site in manure (95 percent of average-annual additions), commercial fertilizer (3 percent of average-annual additions), precipitation (1 percent of average-annual additions), and ground water entering the site across the western boundary (1 percent of average-annual additions). Nitrogen was removed in harvested crops (43 percent of total removals), ground-water discharge (27 percent of all removals), volatilization gases (30 percent of removals), and surface runoff (less than 1 percent of removals).

Table 22. Estimated additions and removals of nitrogen at Field-Site 2, October 1984 through September 1986

[Numbers are in pounds as nitrogen; <, less than]

Water year	Additions				Removals			
	Nitrogen in manure fertilizer	Nitrogen in commercial fertilizer	Nitrogen in precipitation	Nitrogen in ground-water inflow	Nitrogen consumed by crops	Surface-water loads of nitrogen	Ground-water loads of nitrogen	Volatilization
1985	25,500	1,000	290	90	8,500	120	3,200	7,300
1986	18,500	500	300	180	8,700	50	7,600	4,600
Average annual additions or removals 1985 - 86								
	22,000	750	295	135	8,600	85	5,400	5,950
Percentage of average annual additions or removals								
	95	3	1	1	43	<1	27	30

More than 99 percent of the nitrogen leaving the site and not consumed by crops or lost to the atmosphere was discharged in the ground water. Highly permeable soils, terraced hillslopes, and a grassed waterway facilitate the leaching of soluble nitrate from the land surface to ground water.

Potential errors could greatly influence the numbers reported in table 22, which should be read as a conceptual rather than quantitative estimate of nitrogen additions to and removals from the site. A brief description of methods of calculation and errors associated with each nitrogen addition and removal term follows.

Additions

Most of the nitrogen added to the site comes from manure from the farm cattle, swine, and poultry operations. Loads of nitrogen in manure were estimated from application data supplied by the farmer and laboratory analysis of manure samples collected at the site during the study period.

Potential errors in the calculation of the quantities of manure nitrogen applied include inaccuracy and variability of the reported quantities applied and variability in the nitrogen content of the manure. Manure amounts are typically reported as numbers of loads in a spreader. A load may mean the spreader was filled to overflowing or may mean it was filled nearly to capacity.

Variability in the nitrogen content of manure causes additional error. The nitrogen content of each manure was based on the average of analyses from several samples collected at different points in the animal confinement areas (table 8).

Applications of commercial fertilizer to the site were small relative to the large amounts of manure nitrogen applied (table 22). Most of the commercial fertilizer used at the site was starter fertilizer applied at the time of planting.

Reported applications of commercial fertilizer nitrogen are probably reasonably accurate because information about the nitrogen content and quantity of the fertilizer are typically available when the fertilizer is purchased.

Loads of nitrogen added to the site in precipitation are reported in tables and are discussed in the Precipitation Quality section of this report.

Nitrogen in ground-water inflow was estimated from the volume of water estimated by the ground-water model to enter the site across the western boundary during an average year multiplied by the mean nitrate concentrations of two ground-water samples collected from well LN 1674 on the western site boundary.

Because there is undoubtedly error involved in the use of only two samples to estimate the nitrate concentrations of ground water entering the site across the western boundary, this budget term would remain small even if nitrate concentrations were considerably larger because the relatively small quantity of water that is estimated to enter the site across the western boundary.

Removals

Nitrogen removed from the site in corn, tobacco, rye, and Sudan grass were calculated by use of yield-based estimates of nutrient consumption by crops supplied by the Pennsylvania State University Cooperative Extension (Robert Anderson, written commun., 1989). Nitrogen removed from the site in fruits and vegetables were estimated on the basis of discussions contained in *Knott's Handbook for Vegetable Growers* (Knott, 1962). Estimated annual nitrogen removals in harvested crops are shown in table 22. This term is subject to errors from nonrepresentative determinations of crop nitrogen content and nonrepresentative determinations of crop yields.

Loads of nitrogen discharged from the site in surface runoff were calculated by use of discharge-weighted mean-storm concentrations of nitrogen for each storm multiplied by measured water discharge. Because there is relatively little surface runoff from the site, removals of nitrogen in surface runoff account for less than 1 percent of nitrogen removed from the site. Probable amounts of error associated with calculation of nitrogen loads in runoff would have a small effect on the magnitude of this term.

Loads of nitrogen discharged with ground water from the site during the 1985 and 1986 water years were 3,200 and 7,600 lb, respectively. Detailed discussions describing the calculation of these loads is contained in the Nitrogen Loads section of this report under "Ground Water."

Volatilization of nitrogen from the site was estimated as a percentage of nitrogen applied (table 22). For surface-applied manure applications, 40 percent of the nitrogen was assumed to be lost to volatilization. For injected manure applications, 20 percent of the nitrogen was assumed to be lost to volatilization. These estimates were based on discussions contained in the Pennsylvania Department of Environmental Resources *Field Application of Manure* manual (Graves, 1986). Commercial fertilizer was estimated to volatilize at a rate of 15 percent of applications, an estimate that is based on discussions by Pionke and Urban (1985).

Quantification of the volatilization of nitrogen in manure and commercial fertilizer nitrogen is difficult. Losses of nitrogen because of volatilization are affected by air temperature, humidity, manure type, manure texture and moisture content, timing of incorporation into soil, and any factor influencing bacterial activity associated with volatilization. The percentage of nitrogen volatilized at the site could therefore be expected to vary greatly. Estimated losses of nitrogen caused by volatilization of manure, reported by Graves (1986b) for manure treatment, handling, and field application, range from 10 to 90 percent.

SUMMARY AND CONCLUSIONS

This report characterizes the surface-water and ground-water quality from October 1984 through September 1986 at a 47.5-acre agricultural site, Field-Site 2, in Lancaster County, Pa., prior to implementation of nutrient-management practices. The report is one in a series of reports documenting the results of the 10-year Conestoga River Headwaters study to determine the effects of agricultural BMP's on surface-water and ground-water quality by monitoring at the regional, small-watershed, and field-site levels.

Field-Site 2 is part of a 55-acre farm near Ephrata, Lancaster County, Pa. The site, underlain by carbonate rock, is agricultural cropland with slopes that range from 2 to 9 percent. Twenty-seven acres are in pipe-drained terraces under no-till practices, and the remaining acres are under minimum-till practices. Most of the manure produced by the farm's beef cattle, hogs, and poultry is applied to the site, which is planted in corn (92 percent) and tobacco.

Measured annual precipitation at the site was 35.9 in. in 1985 and 38.8 in. in 1986, a 17 and 11 percent deficit, respectively, from the 30-year average at a nearby NOAA precipitation station. Fifty-eight storms occurred in 1985 and 59 occurred in 1986; these storms produced precipitation that ranged from 0.10 to 1.3 in. Ammonium and nitrate in precipitation contributed an estimated 140 and 107 lb/yr, respectively, to the site.

Cattle and poultry manure were applied to the surface, and hog manure was injected 8-10 in. below the surface. Approximately 26,000 lb of nitrogen and 6,600 lb of phosphorus were applied to the 47.5 acres in 1985, and 19,000 lb of nitrogen and 3,700 lb of phosphorus were applied in 1986. The area in pipe-drained terraces received 64 percent of the nitrogen and phosphorus in 1985 and about 75 percent of the nitrogen and phosphorus in 1986.

Soils at the site are 5 to 30 ft deep silt loams and silty-clay loams. Median concentrations of soluble nitrate as nitrogen and soluble phosphorus as phosphorus in the soil ranged from 177 to 329 lb/acre and 8.5 to 35 lb/acre, respectively, for the top 4-ft profile.

Runoff discharge, consisting of surface runoff and subsurface flow from the terrace drain pipes, was measurable during 36 storms over the study period. Discharge ranged from 10 to 48,000 ft^3 and accounted for 1.7 and 0.8 percent of the 1985 and 1986 annual precipitation, respectively. One storm on snow-covered, frozen ground in February 1985 produced 79 percent of the 1985 annual runoff. Five storms under similar conditions in January and February 1986 produced 46 percent of the 1986 annual runoff. An intense thunderstorm in July 1986 produced another 42 percent of the 1986 annual runoff. Graphical and regression analysis of data showed that storm discharge was primarily dependent on storm precipitation, but the relation between discharge and precipitation was different for thawed- and frozen-soil conditions. A secondary factor in explaining storm discharge for storms on thawed ground was antecedent 7-day precipitation.

During the study period, a total of 90,700 ft^3 of water discharged from the pipe-drained terraces, carrying with it 87 lb of nitrogen and 37 lb of phosphorus. Two storms on snow-covered, frozen ground in February 1985 accounted for 58 percent of the load of total nitrogen and 61 percent of the load of total phosphorus in runoff during the study period. During the study period, less than 1 percent of nitrogen and phosphorus applied to the site was discharged with runoff.

Graphical and regression analyses showed that variations in mean storm concentrations of total nitrogen species and total phosphorus were explained primarily by the time elapsed since the previous nutrient application. Mean storm concentrations decreased as the number of days since the last nutrient application increased. For mean total phosphorus, this decrease was asymptotic; after several months mean concentrations of total phosphorus approached a "baseline" concentration estimated to be 2 to 5 mg/L.

An unconfined limestone and dolomite aquifer underlies the site, with depth to bedrock ranging from 5 to 30 ft below land surface. Estimates of specific yield made from water-level records of eight wells in different parts of the aquifer range from 0.05 to 0.10. Specific capacities of 15 wells range over an order of magnitude, from less than 1 to 20 (gal/min)/ft. Estimates of transmissivity made by use of water-level recession slope analysis and a two-dimensional finite difference model, range over four orders of magnitude from about 10 ft^2/d to about 10,000 ft^2/d. The wide range of values in aquifer properties is a consequence of the heterogeneity of bedding planes, fractures, faults, and irregular weathering of carbonate materials in the saturated zone.

Specific-capacity and transmissivity data indicate the existence of two contrasting aquifer regimes at the site. Wells drilled in the eastern part of the site are underlain by the Snitz Creek Formation. These wells have large specific capacities [averaging 20 (gal/min)/ft of drawdown] relative to the specific capacities [averaging less than 1 (gal/min)/ft of drawdown] of wells drilled in the Millbach Formation in the western part of the site. Estimates of transmissivity made by use of data from site wells indicate that aquifer materials in the Snitz Creek Formation are at least an order of magnitude more transmissive than those in the Millbach Formation.

Ground-water recharge, calculated from water-level rise and estimated specific yield, was 10.2 in. for 1985 and 20.3 in. for 1986, or 29 and 52 percent of the total annual precipitation. By use of the 1986 recharge data for the site and the 30-year normal precipitation from a nearby weather station, the long-term ground-water recharge at Field-Site 2 would average about 23 in/yr.

Volume of ground-water discharge was estimated by assuming steady-state conditions between recharge and discharge on an annual basis. Directions of ground-water flow were estimated by use of topography, measured water levels, and a two-dimensional finite-difference model. Ground-water discharge was 26 percent across the northern field-site boundary, 22 percent across the eastern boundary, and 52 percent across the southern boundary, with all discharge eventually reaching Indian Run, which flows along the eastern boundary of the site.

Ground-water chemistry at Field-Site 2 closely reflects the carbonate mineralogy of the aquifer. The ground water is slightly alkaline (180 to 330 mg/L as $CaCO_3$) and is characteristic of a limestone or dolomite aquifer. Specific conductances ranged from 450 to 1,560 μS/cm in nonrecharge ground-water samples and from 485 to 1,500 μS/cm in recharge ground-water samples during the study period.

Dissolved nitrate accounted for over 90 percent of all dissolved nitrogen in ground-water samples, and concentrations at all sampling sites frequently exceed the U.S. Environmental Protection Agency maximum contaminant level for drinking water of 10 mg/L nitrate as nitrogen. Minimum and maximum concentrations of nitrate for all sampling sites, except well LN 1670, which was probably affected by an ammonia spill, were 7.4 and 100 mg/L, respectively. Concentrations of dissolved phosphorus ranged from below the detection limit of 0.01 to 0.51 mg/L.

Contributing areas where surface-applied materials can have a substantial effect on ground-water quality at a sampling site were defined for eight wells and the spring. Agricultural-activity data were used to estimate the nitrogen loading in these contributing areas for all sites except well LN 1670.

Calculated nitrogen load in ground-water discharge totaled 3,200 lb in the 1985 water year, and 7,600 lb in the 1986 water year. Average annual additions and removals of nitrogen to the site were estimated from data collected during the 1985-86 water years. Additions of nitrogen came from manure (95 percent), commercial fertilizer (3 percent), precipitation (1 percent), and ground water entering the site across the western site boundary (1 percent). Nitrogen was removed from the site in harvested crops (43 percent), ground-water discharge (27 percent), by volatilization (30 percent), and in surface runoff (less than 1 percent).

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