EVALUATION OF AGRICULTURAL BEST-MANAGEMENT PRACTICES IN THE CONESTOGA RIVER HEADWATERS, PENNSYLVANIA: Effects of Nutrient Management on Quality of Surface Runoff and Ground Water at a Small Carbonate-Rock Site Near Ephrata, Pennsylvania, 1984-90

WATER-QUALITY STUDY OF THE CONESTOGA RIVER HEADWATERS, PENNSYLVANIA

by David W. Hall, Patricia L. Lietman, and Edward H. Koerkle

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CONVERSION FACTORS, VERTICAL DATUM, 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 meter
acre	0.4047	square hectometer
	Temperature	
degree Fahrenheit (°F)	°C = 5/9 (°F-32)	degree Celsius

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

Abbreviated water quality units used in report:
- milligrams per liter (mg/L)
- microsiemens per centimeter (μS/cm)

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Effects of Nutrient Management on Quality of Surface Runoff and Ground Water at a Small Carbonate-Rock Site Near Ephrata, Pennsylvania, 1984-90

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ABSTRACT

The U.S. Geological Survey and the Pennsylvania Department of Environmental Protection[1] conducted a study from 1984 to 1990 to determine the effects of the implementation and practice of nutrient management [an agricultural best-management practice (BMP)] on the quality of surface runoff and ground water at a 55-acre crop and livestock farm in carbonate terrain near Ephrata, Pa.

Implementation of nutrient management at Field-Site 2 resulted in application decreases of 33 percent for nitrogen and 29 percent for phosphorus. There were no significant changes in nitrogen or phosphorus loads for a given amount of runoff from the pre-BMP to the post-BMP periods. However, less than 2 percent of the applied nutrients were discharged with runoff throughout the study period. After the implementation of nutrient management, statistically significant decreases in concentrations of nitrate in ground-water samples occurred at three of the four wells monitored throughout the pre- and post-BMP periods. The largest decreases in nitrate concentrations occurred at wells where samples had the largest nitrate concentrations prior to nutrient management. Changes in nitrogen applications to the contributing areas of five wells were correlated with nitrate concentrations of the well water. The correlations between the timing and amount of applied nitrogen and changes in ground-water quality met the four conditions that are characteristic of a cause-effect relation: an association, consistency, responsiveness, and a mechanism. Changes in ground-water nitrate concentrations lagged behind changes in loading of nitrogen fertilizers (primarily manure) by approximately 4 to 19 months.

INTRODUCTION

The U.S. Geological Survey (USGS), in cooperation with the Pennsylvania Department of Environmental Protection (PaDEP)[1], conducted a study from 1984 to 1990 to determine the effects of agricultural best-management practices (BMP's) on surface- and ground-water quality in the headwaters of the Conestoga River Basin. This study was 1 of 20 projects in the U.S. Department of Agriculture's (USDA) Rural Clean Water Program (RCWP) (Chichester, 1988; Little, 1989; Gunsalus, 1992).

The RCWP was authorized by the Agriculture, Rural Development, and Related Agencies Appropriations Act of 1980, Public Law 96-108 (Regulations 7CFR, Part 700), November 9, 1979. The Conestoga Headwaters RCWP Project was approved by the National RCWP Committee in July 1981 as a comprehensive monitoring and evaluation project. The Conestoga Headwaters area was chosen because it had previously been designated in Pennsylvania's Agricultural 208 Plan (Schueller, 1983) as a top-priority watershed for the study of agricultural nonpoint-source contamination of surface and ground water.

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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).

The Conestoga Headwaters Project area contained 132 mi^2 of streams that were used and had the potential for use for water supply, livestock watering, fish and wildlife habitat, and recreation. Ground water was a significant source of water for private water supplies in the project area. The primary RCWP objective was to improve surface- and ground-water quality in agricultural areas nationwide through the voluntary implementation of agricultural BMP's. Funding was provided to the RCWP projects to initiate the implementation of BMP's and to monitor the effects of BMP implementation on water quality.

The BMP's selected to improve water quality in the Conestoga Headwaters RCWP project area included terracing of sloped fields to reduce soil erosion, nutrient-management planning, construction of manure-storage facilities, pesticide-management programs, and installation of grassed waterways on or below hillslopes to slow runoff and thereby decrease soil erosion from agricultural fields. The Conestoga Project was designed to monitor water quality at three scales: regional, small watershed, and field. Pre-BMP and post-BMP monitoring was conducted at three scales to determine the effects of BMP's on surface- and ground-water quality. A detailed study description was published in "Evaluation of Agricultural Best-Management Practices in the Conestoga River Headwaters, Pennsylvania: Methods of Data Collection and Analysis and Description of Study Areas," by Chichester (1988).

Nutrients from manure and commercial fertilizer, pesticides, and eroded sediments were the nonpoint-source contaminants of interest in the project area. Disposal of large quantities of manure on land at intensive animal-production farms, such as those in the Conestoga Headwaters Basin (part of the Chesapeake Bay watershed), may lead to elevated concentrations of nutrients in surface and ground water (Gillham and Webber, 1969; McCalla and others, 1970; Pionke and Urban, 1985; Crowder and Young, 1988). Elevated concentrations of nutrients in water supplies are problematic for several reasons. About 2,000 cases of infant methemoglobinemia, a serious and potentially fatal health condition that results from the consumption of water with elevated concentrations of nitrites and nitrates, have been reported worldwide during 1945-72 (Shuval and Gruener, 1972). Many pesticides, including aldicarb, atrazine, carbofuran, and simazine, are known to react with nitrite at low pH to form N-nitroso compounds that are known to be potent animal carcinogens (Murdock, 1988). Elevated levels of nitrate in drinking water in Australia have been associated with increased human birth defects (Dyer and others, 1984). In the United States, nitrogen and phosphorus-induced algal blooms in the Chesapeake Bay have been linked to critically low dissolved-oxygen concentrations, decreased numbers of aquatic animals, and decreased survival of submerged aquatic vegetation (Ryther and Dunstan, 1971; Officer and others, 1984; Fisher, 1989).

Purpose and Scope

This report describes research conducted at Field-Site 2, which was located in the Conestoga Headwaters Basin immediately north of Ephrata in Lancaster County, Pa. (Hall, 1992; Hall and Risser, 1992; Unangst, 1992). The site was in a karstic, carbonate terrain and was used for intensive animal and crop production. A detailed characterization of Field-Site 2 and description of methods used in the study was published in Koerkle and others (1996). The primary objective of this report is to compare land-use and water-quality data from the period before nutrient-management practices were implemented (pre-BMP period—water years 1985-86[2], October 1, 1984, through September 30, 1986), with data from the period after implementation of nutrient-management practices (post-BMP period—water years 1987-90, October 1, 1986, through September 30, 1990) to determine the effects of nutrient management on water quality. A second objective of this report is to evaluate the movement of nitrogen at the site.

Site Description

Field-Site 2 was a 55-acre plot of a 67-acre crop and animal-production farm near Ephrata in Lancaster County, Pa. (fig. 1). Twelve acres of the farm are adjacent to the east bank of Indian Run and were not part of the study area. The site was located within the Conestoga Section of the Piedmont Physiographic Province.

[2] Water year is the 12-month period beginning on October 1 and ending on September 30; it is designated by the calendar year in which it ends.

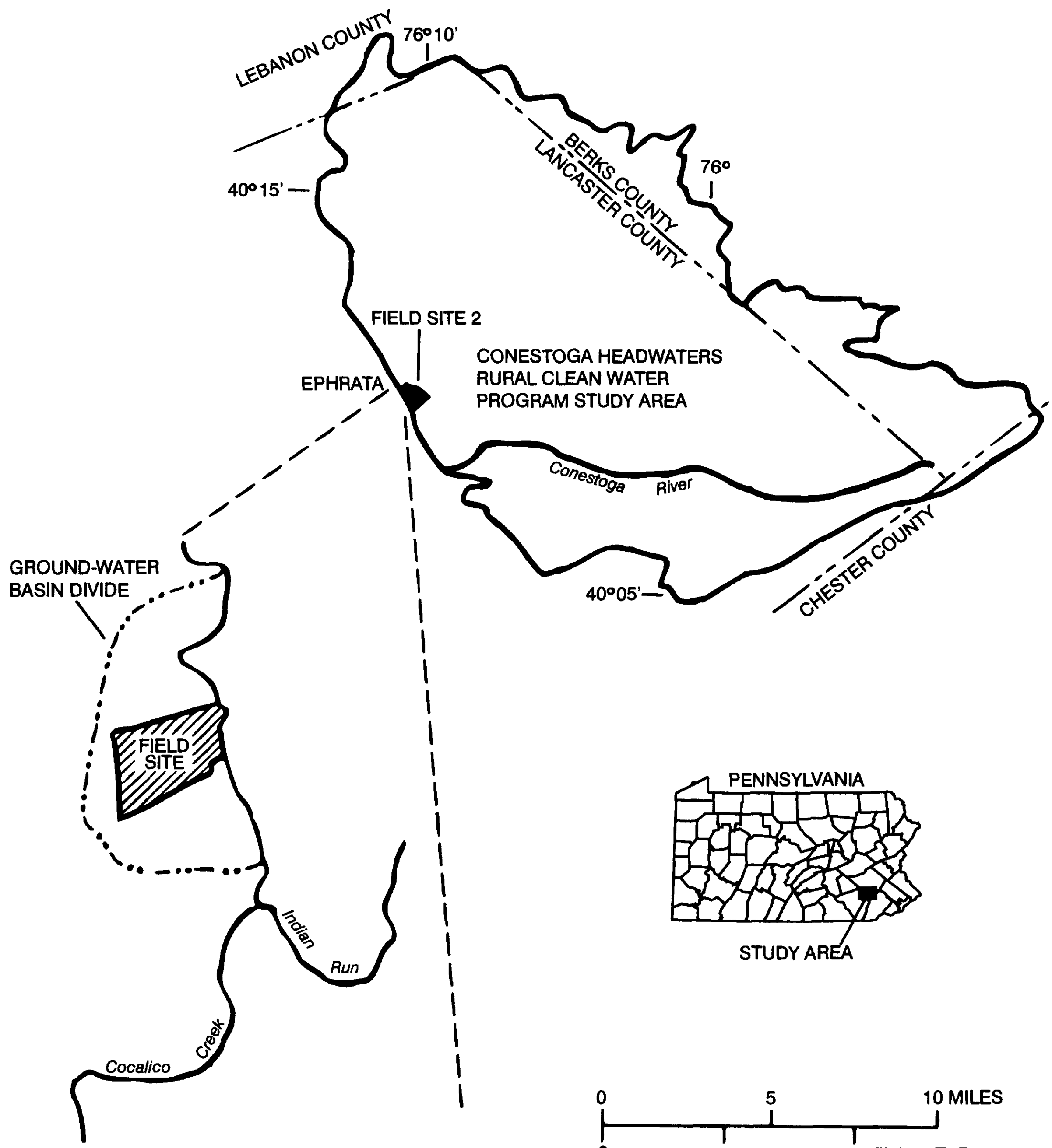

Figure 1. Location of the Conestoga Headwaters Rural Clean Water Program study area and Field-Site 2 near Ephrata, Pa.

A detailed site characterization is presented in Koerkle and others (1996). Low hills surrounding the site define a small surface-water and a shallow, unconfined ground-water basin that is drained by Indian Run (fig. 2). Surface altitude ranges from approximately 431 ft in the southwestern corner to 342 ft in the southeastern corner (fig. 3). Approximately half of the site is underlain by interbedded limestone and dolomite of the Millbach Formation, and the other half is underlain by dolomite of the Snitz Creek Formation, both of Cambrian age (Meisler and Becher, 1971). The fractured bedrock is overlain by a shallow mantle of soil and weathered rocks that ranges from about 5 to 30 ft in thickness. Depth to the water table has a similar but noncoincident range of about 5 to 30 ft below land surface. Soils at the site are Typic Hapludulf and Hagerstown silt and silt-clay loams (Custer, 1985).

In 1965, eight terraces were installed at the site to reduce soil erosion. The terraces were restructured in 1981 when a pipe-outlet drainage system was installed to further reduce soil erosion. The pipe-outlet system drained approximately 27 acres of the site (fig. 3). Terrace alignment followed the land-surface contours, and each terrace was constructed with a slight variable slope that slowed runoff and diverted water to perforated 6-in. inside-diameter plastic inlets to subsurface drain pipes. The runoff gage was installed at the outlet of the main terrace drain in the southeastern part of the site near Indian Run (fig. 3). A narrow grassed waterway extending west to east across the center of the site also reduced soil erosion by slowing runoff velocity.

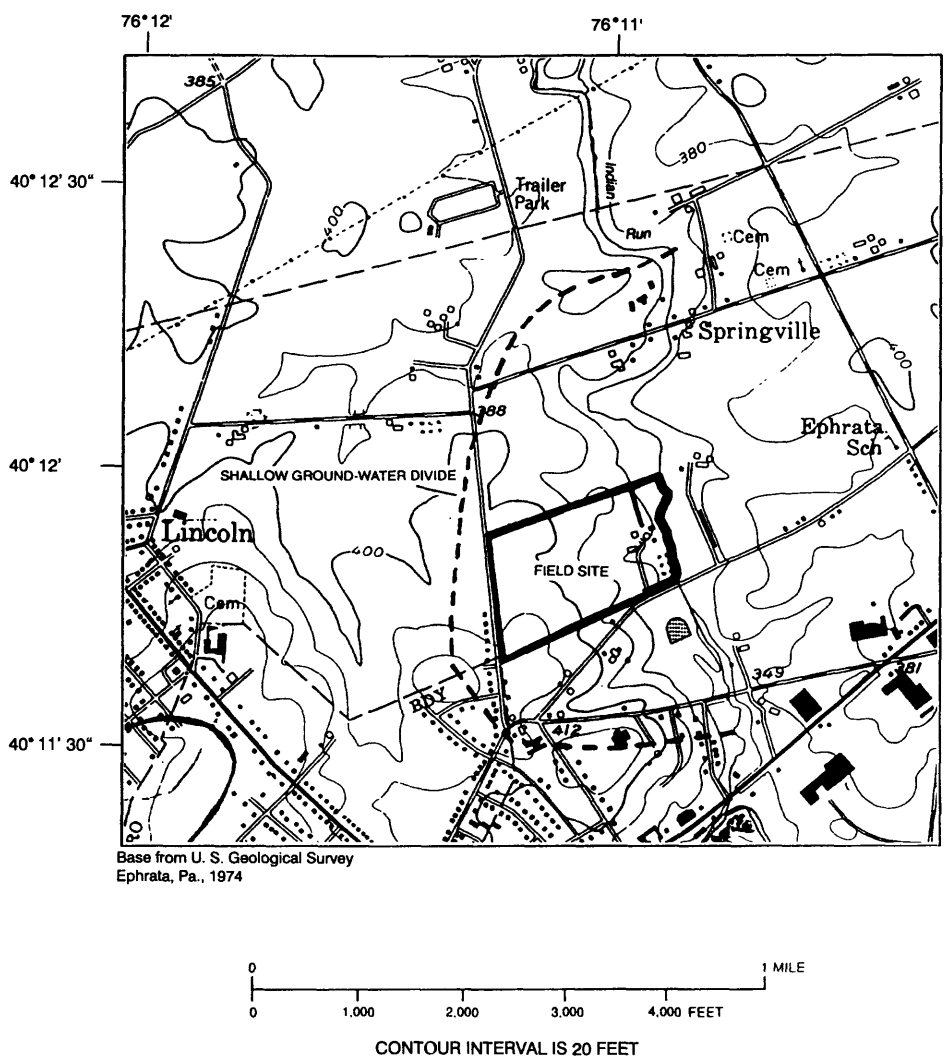

Figure 2. Location of Field-Site 2 in relation to the shallow ground-water basin and area topography.

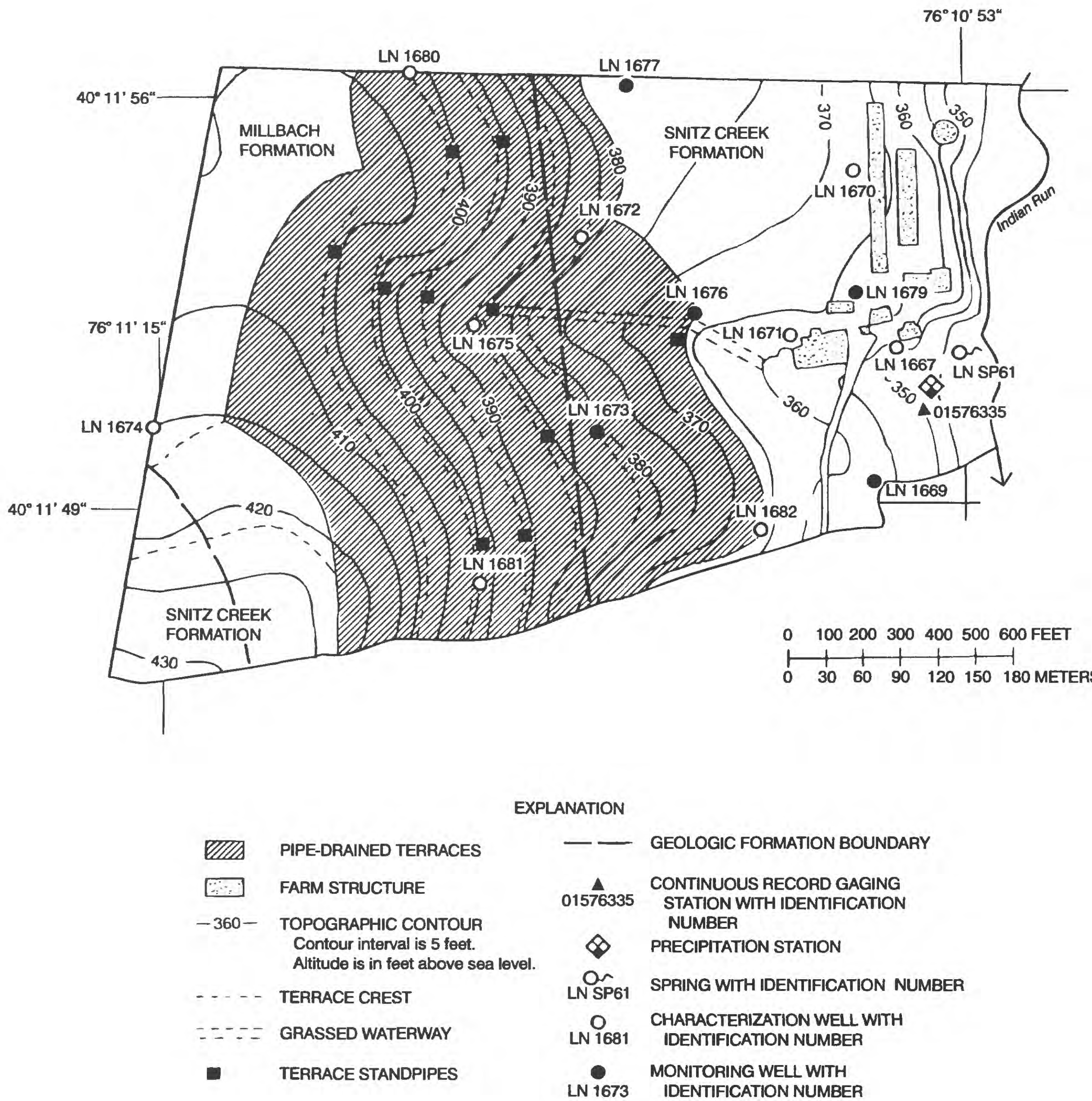

Figure 3. Data-collection locations and geologic units at Field-Site 2. (from Chichester, 1988, fig. 6)

Best-Management Practices

The BMP implemented at Field-Site 2 was nutrient management. Nutrient management is a BMP in which quantities of manure and commercial fertilizer that are applied to cropped land are limited to quantities that meet crop nutrient requirements (Graves, 1986a, 1986b; Pennsylvania State University, 1989; Lanyon and Beegle, 1989; Cronce and others, 1990; Bacon and others, 1990; Anderson, 1992; Ressler, 1992). This limitation of nutrient applications supports maximum expected crop yields while minimizing nutrient transport to surface runoff and ground water. A nutrient-management plan for the site was developed by personnel from the Pennsylvania State University Cooperative Extension Service. The nutrient-management plan was developed from crop-yield goals on the basis of soil type, methods of manure and commercial-fertilizer application, nitrogen concentrations in soils, nitrogen concentrations in manure samples, and crop rotations. Nutrient management was implemented in October 1986 and remained in effect through the remaining 4 years of the study period. Under nutrient management, the farmer exported all animal manure produced at the site in excess of the amounts specified for application to cropped land by the nutrient-management plan.

Acknowledgments

The authors acknowledge the dedicated efforts of many people who made this study possible. Mr. John Hauenstein and Mr. Gary Lesher of the Susquehanna River Basin Commission surveyed and mapped the site. Mr. Aaron Stauffer and Mr. Clark Stauffer allowed the use of their farm for the study site, cooperated in the implementation of nutrient management at the farm, and provided data on agricultural activities to project personnel. The U.S. Department of Agriculture, Natural Resources Conservation Service[3], provided planning and technical assistance for the installation and implementation of agricultural BMP's. Personnel from Pennsylvania State University and Pennsylvania State University Cooperative Extension Service collected and analyzed soil-nutrient samples and assisted with the development and implementation of nutrient-management practices.

[3] Prior to 1995, the Natural Resources Conservation Service was known as the Soil Conservation Service.

METHODS OF DATA COLLECTION AND ANALYSIS

Detailed information on methods of data collection and analysis at Field-Site 2 are presented in Chichester (1988) and Koerkle and others (1996). Modifications of procedures described in these reports and used for sample and data analyses during the post-BMP period are described in this report. Factors affecting water quality at the site were examined to provide a valid basis for comparison of data between the pre-BMP and post-BMP periods and to assess the effects of BMP implementation on water quality. Precipitation quantity was continuously monitored, and published estimates of precipitation quality in the area of the site were used to estimate nitrogen loading from precipitation. Agricultural activities were monitored, and changes in cropping, tillage, and fertilization were documented. Statistical analyses of the relation of surface- and ground-water quality to agricultural activities were performed to determine if changes in water quality that occurred during the study period were attributable to agricultural activities. Nitrogen movement through the site was estimated by use of measured data and estimates of nitrogen inputs and outputs.

Data-Collection Network

The site data-collection network is summarized on tables 1 and 2. Precipitation-quantity data were collected at 5-minute intervals for comparison to surface-runoff and ground-water-quantity and -quality data.

Table 1. Data-collection network at Field-Site 2 during the post-Best-Management Practice period

Medium	Number of locations	Data-collection frequency	Analyses performed or data collected
Precipitation	1	5-minute intervals during storms	Volume
Agricultural activities	Entire site	Monthly	Nutrient timing and rates; planting, plowing, and harvesting locations and dates, and crop yields
Manure	Varied	At selected major applications	Nutrient concentration
Soil	Varied	Spring, fall	Nutrient concentration
Runoff	1	All runoff events	Volume
		Most runoff events	Suspended-sediment and nutrient concentration
Ground water	6 wells	Continuous	Water level
	5 wells; 1 spring	Monthly plus selected recharge events	Specific conductance and nutrient concentration
	1 well	Quarterly	Nutrient concentration

Table 2. Monitoring well locations, descriptions, and sampling depths at Field-Site 2

[All depths shown in feet below land surface; (gal/min)/ft, gallon per minute per foot of drawdown; M, monthly; Q, quarterly; C, continuous; <, less than]

Well number	Latitude	Longitude	Total depth of well	Depth to bottom of casing	Depth to bedrock	Bedrock altitude (feet)	Sampling depth	Estimated specific capacity [(gal/min)/ft]	Data collected: Nutrients	Data collected: Water level
LN 1669	40°11'49"	76°10'55"	100	11	6.5	352	85	<1	M	C
LN 1670	40°11'56"	76°10'57"	75	9.8	5.5	361	65	<1	Q	C
LN 1673	40°11'48"	76°11'03"	46	14	12	368	35	<1	M	C
LN 1676	40°11'52"	76°11'01"	40	8.8	11	356	35	<1	M	C
LN 1677	40°11'56"	76°11'05"	50	30	28	349	35	20	M	C
LN 1679	40°11'52"	76°10'57"	60	13	10	354	35	20	M	C

Agricultural activities in this report refer to cultivation and other practices associated with crop production. Manure- or commercial-fertilizer application data included types of fertilizer and dates, areas, amounts, and methods of application. Tillage types, dates, and areas, as well as planting and harvesting dates, were recorded. Manure samples were collected at various times during the study period. Four manure sources were sampled: manure/bedding mix from a steer pen, hog manure from a storage tank, hog manure from a pit below the hog finishing facility, and poultry manure from the floor of a poultry house.

Soil samples were collected by use of a hydraulic press. Each sample was a composite of three cores collected within a 25-ft radius. The 2-in. diameter cores were divided into composited depth increments for analysis.

Surface runoff from the 27 acres of pipe-outlet terraces was monitored at a gage (fig. 3) located at the outlet of the main drain pipe that collects discharge from all the terrace stand pipes. Stage in a 6-in. Parshall[4] flume was recorded on a continuous graphic recorder and an automatic data recorder. Runoff samples were collected by a modified automatic pumping sampler triggered by a stage-operated float switch. Runoff samples were chilled to approximately 4°C by a refrigeration unit until retrieval, usually within 24 hours.

Ground-water data were collected at 14 wells and the spring during site characterization in the pre-BMP period (1985-86). Five wells were selected for monitoring during the post-BMP period on the basis of successful well development, well specific capacity, and land use upgradient of sampling points. During the post-BMP period, all wells except LN 1677 were pumped prior to sampling until approximately three borehole volumes had been evacuated or until the wells were pumped dry. Well LN 1677 was not pumped prior to sampling because access to the well was restricted during the growing season and because sample analyses demonstrated little change in water quality during preliminary testing before and after pumping.

Data Analysis

Water-quality samples collected during the study were analyzed by the PaDEP, Bureau of Laboratories, in Harrisburg, Pa. Characteristics and chemical constituents for which water samples were analyzed during the post-BMP period, minimum reporting levels[5], and U.S. Environmental Protection Agency (USEPA) Primary Drinking-Water Regulations (U.S. Environmental Protection Agency, 1993) are shown in table 3. Suspended-sediment samples were analyzed by the USGS Sediment Laboratory in Lemoyne, Pa. All soil samples were analyzed at The Pennsylvania State University, Soils and Environmental Chemistry Laboratory, in University Park, Pa. Manure samples were analyzed for nutrients at the A&L Eastern Agricultural Laboratories, Inc., in Richmond, Va. The sampling schedule for nutrients and suspended sediment in surface and ground water is shown in table 4. Water-quality data collected during this study (table 4) were published in the annual USGS Water-Data Reports (1986-91) and are stored in the USGS WATSTORE data base. All project data are on file at the USGS, Water Resources Division, 840 Market Street, Lemoyne, Pa. Types of data collected were runoff volumes and concentrations of sediment, nitrogen, and phosphorus. Ground-water analyses characterized quantities of recharge and discharge, and nitrogen data from ground-water samples were analyzed to determine if changes in the concentrations and loads of nitrogen in ground water occurred as a result of the implementation of nutrient-management practices.

Data were estimated to supplement analyses when measured data were absent and there was a valid basis to perform the estimation. When equipment malfunction caused periods of missing record, total daily precipitation was estimated by use of records from a National Oceanographic and Atmospheric Administration (NOAA) weather station located less than 2 mi from the site in Ephrata, Pa.

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[4] Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Geological Survey.

[5] The smallest measured concentration of a constituent that may be reliably reported by the use of a given analytical method.

Table 3. Primary characteristics and chemical constituents for which precipitation, surface-runoff, or ground-water samples from Field-Site 2 were analyzed during the post-Best-Management Practice period

[°C, degrees Celsius; µS/cm, microsiemens per centimeter at 25 degrees Celsius; mg/L, milligram per liter; --, no data]

Characteristic or constituent	Laboratory minimum reporting levels[1]	U.S. Environmental Protection Agency Primary Drinking-Water Regulation
Temperature (field measurements)	Measured to nearest 0.5 °C	--
Specific conductance (field measurements)	1 to 10 µS/cm, dependent on value	--
Suspended sediment	1 mg/L	1 mg/L
Total and dissolved nutrients:		
Ammonium[2] plus organic nitrogen	.2 mg/L	--
Ammonium[2]	.02 mg/L	[3] 30 mg/L
Organic nitrogen (calculated)	.2 mg/L	--
Nitrate plus nitrite	.04 mg/L	[4] 10 mg/L
Nitrite	.01 mg/L	[4] 1 mg/L
Nitrate (calculated)	.04 mg/L	[4] 10 mg/L
Phosphorus	.02 mg/L	--

[1] The smallest measured concentration of a constituent that may be reliable reported by the use of a given analytical method.

[2] Ammonium in this report represents ammonia plus ammonium.

[3] Lifetime Health Advisory Level, as ammonia (U.S. Environmental Protection Agency, 1993).

[4] Maximum Contaminant Level (U.S. Environmental Protection Agency, 1993).

Table 4. Sampling schedule for nutrients and suspended sediment at Field-Site 2, October 1984 through September 1990

[Y, most major storms throughout year; S, selected storms; E, every 3 weeks; M, monthly; Q, quarterly; T, total; D, dissolved; X, suspended; --, no data]

Constituent	Water year 1985			Water year 1986			Water year 1987			Water year 1988			Water year 1989			Water year 1990		
	Number of sites	Frequency of collection	Phase of constituent	Number of sites	Frequency of collection	Phase of constituent	Number of sites	Frequency of collection	Phase of constituent	Number of sites	Frequency of collection	Phase of constituent	Number of sites	Frequency of collection	Phase of constituent	Number of sites	Frequency of collection	Phase of constituent
Surface Runoff																		
Nitrate + nitrite	1	Y	T	1	Y	T	1	Y	T	1	Y	T	--	--	--	--	--	--
	1	S	D	1	S	D	1	S	D	1	S	D	--	--	--	--	--	--
Nitrite	1	Y	T	1	Y	T	1	S	T	1	Y	T	--	--	--	--	--	--
	1	S	D	1	S	D	1	S	D	1	S	D	--	--	--	--	--	--
Ammonium	1	Y	T	1	Y	T	1	S	T	1	Y	T	--	--	--	--	--	--
	1	S	D	1	S	D	1	S	D	1	S	D	--	--	--	--	--	--
Ammonium + organic nitrogen	1	Y	T	1	Y	T	1	Y	T	1	Y	T	--	--	--	--	--	--
	1	S	D	1	S	D	1	S	D	1	S	D	--	--	--	--	--	--
Phosphorus	1	Y	T	1	Y	T	1	Y	T	1	Y	T	--	--	--	--	--	--
	1	S	D	1	S	D	1	S	D	1	S	D	--	--	--	--	--	--
Sediment	1	S	X	1	S	X	1	S	X	1	S	X	--	--	--	--	--	--
Ground Water																		
Nitrate + nitrite	6	E	D	7	E	D	7	E	D	7	M	D	5	M	D	5	M	D
	1	Q	D	1	Q	D	1	Q	D	1	Q	D	2	Q	D	2	Q	D
	7	S	T, D	8	S	D	7	S	D	7	S	D	--	--	--	--	--	--
Nitrite	1	Q	D	--	--	--	8	Q	D	8	Q	D	--	--	--	--	--	--
	7	S	T, D	8	S	D	--	--	--	--	--	--	--	--	--	--	--	--
Ammonium	1	Q	D	1	Q	D	8	Q	D	8	Q	D	--	--	--	--	--	--
	7	S	T, D	7	S	D	--	--	--	--	--	--	--	--	--	--	--	--
Ammonium + organic nitrogen	6	E	D	7	E	D	7	E	D	7	M	D	7	S	D	--	--	--
	1	Q	D	1	Q	D	1	Q	D	1	Q	D	--	--	--	--	--	--
	7	S	T, D	8	S	D	--	--	--	--	--	--	--	--	--	--	--	--
Phosphorus	6	E	D	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
	1	Q	D	--	--	--	8	Q	D	8	Q	D	--	--	--	--	--	--
	7	S	T, D	8	S	D	--	--	--	--	--	--	--	--	--	--	--	--

Estimates of annual nitrogen loads leaving the site were obtained from summations of monthly loads estimated to cross each of the northern, eastern, and southern site boundaries. Loads crossing each site boundary in each month were calculated by use of the following formula:

$$(A+B) \times C \times D \times E \times F = G, \qquad (1)$$

where A is the volume of ground water estimated to enter the site annually across the western boundary, in liters;

B is the volume of ground-water recharge entering the site annually from precipitation, in liters;

C is the proportional percentage of ground water estimated (by the use of model output) to discharge annually across a site boundary, in percent;

D is a unitless proportional monthly fraction of annual discharge (calculated from well water-level records);

E is the concentration of nitrate of a sample (or an average of multiple samples) collected monthly to characterize water quality of discharge across a site boundary, in milligrams per liter;

F is a milligram to pound conversion factor (2.205×10^6); and

G is the monthly nitrogen load discharged across a site boundary, in pounds.

Quantities of ground water discharged annually across each site boundary were computed by multiplying total annual discharge from the site (annual recharge plus flow into the site across the western boundary) by percentages of annual flow estimated to cross each site boundary as indicated by output from a two-dimensional ground-water flow model (McDonald and Harbaugh, 1988; a complete description of the model, as well as a more detailed explanation of loading calculations, was published in Koerkle and others, 1996). Annual discharge across each site boundary was then apportioned among months by use of water-level hydrograph rises for each month divided by the total annual water-level rise to obtain the proportional fraction of annual discharge occurring in each month.

Nitrogen loads discharged from the site in ground water were calculated from estimates of monthly ground-water discharge multiplied by measured concentrations of nitrate in monthly ground-water samples. Because ground water discharged across the northern, eastern, and southern site boundaries and concentrations of nitrate in ground-water samples collected in different parts of the site varied spatially and temporally, separate nitrogen loads in ground water were computed for water that discharged across each of the northern, eastern, and southern site boundaries. Samples from wells located in different parts of the site were chosen to represent the quality of ground-water discharge from the part of the site in which they were located. Samples from well LN 1677 were chosen to represent the concentrations of nitrate in water discharged across the northern site boundary, samples from wells LN 1676 and LN 1679 were chosen to represent the concentrations of nitrate in water discharged across the eastern site boundary, and samples from wells LN 1673 and LN 1669 were chosen to represent concentrations of nitrate in water crossing the southern site boundary. Estimated monthly loads crossing each site boundary were then summed to obtain the total monthly nitrogen load discharged from the site, and total monthly site loads were then summed to obtain the annual loads of nitrogen discharged from the site in ground water.

Reported loads of nitrogen in ground water are approximations that may contain errors because of the assumptions used in the load calculations. Estimates of ground-water recharge and discharge are very sensitive to values of specific yield selected to characterize aquifer materials. Because nitrogen loads are computed by multiplication of yield-based estimates of discharge by concentrations of nitrogen in ground water at selected wells, determinations of specific yield could have a significant effect on the magnitude of the calculated loads, as discussed in Koerkle and others (1996). This method of estimating loads of nitrogen in ground water is based on an assumption that recharge to ground water during a given month equals discharge during the same month. This is not always true, especially if large storms occur late in the month. The method would underestimate discharge from the site in dry months when there was no recharge to ground water. Additional errors may be present in reported loads if water samples from wells selected to represent the quality of water discharged across site boundaries were not representative or if model-based estimates of the percentages of ground-water flow across site boundaries were inaccurate.

Statistical procedures were used for summarizing data, making statistically supported inferences about the data, and defining explanatory relations between the data. Data summarization was accomplished with descriptive statistics, such as means, medians, ranges, and percentiles. Statistical inferences about data normality, differences between data set means or medians, correlation between variables, linear-regression analyses, and analysis of covariance were based on the results of hypothesis testing. All hypothesis tests were conducted at the 95-percent confidence level except where noted. Data normality was tested by use of the Shapiro-Wilk method. Differences between central values of data sets were tested by use of the nonparametric Mann-Whitney Rank Sum test. Complete discussions of descriptive, parametric, and nonparametric procedures can be found in Iman and Conover (1983) and Helsel and Hirsch (1992). Statistical procedures were run on software from the Statistical Analysis System (SAS) Institute, Inc. (1982), P-STAT, Inc. (1986), and SYSTAT, Inc. (Wilkinson, 1987).

For runoff, explanatory relations between hydrologic, climatic, and agricultural-activities data were explored by the use of linear regression and analysis of covariance. The Mallow's Cp statistic and adjusted R^2 values (adjusted for degrees of freedom and number of independent variables) were used to select the best regression models. F-tests were used to evaluate overall regression significance, and t-tests were used to evaluate the significance of regression coefficients. Additionally, plots of regression residuals and diagnostics for outliers, influence, and leverage were evaluated for each regression.

For ground water, cross-correlation functions were used to select the most significant lag times for lagged correlations between applied nitrogen and concentrations of nitrate in ground water. The population correlation coefficient (rho) and significance (p) of the strongest lagged correlations indicated in the cross-correlation procedure were determined by use of the nonparametric Spearman Rank Sum test.

Ground-water samples were separated into nonrecharge and recharge influenced groups to assess how ground-water quality varied during short recharge periods as compared to nonrecharge periods. Samples were empirically grouped as recharge influenced if they were collected less than 2 weeks after a significant recharge event. A significant recharge event was defined, by use of the continuous water-level records of each well, as a water-level rise of 0.6 ft in wells located in parts of the aquifer with large specific yields and a water-level rise of 1.0 ft for wells located in parts of the aquifer having small specific yields. Nonrecharge concentrations of nitrate from monthly samples were used in the cross-correlation analysis to eliminate short-term variations in ground water caused by transient recharge and to improve the validity of analyses by use of data collected during similar hydrologic conditions.

Quality Assurance

A quality-assurance (QA) plan for nutrient water-quality analyses was developed to check the quality of sample collection and analytical results. QA data pooled from all Conestoga Headwaters RCWP water-quality studies and from the USGS National Water Quality Laboratory (NWQL), Standard Reference Sample Program were used to evaluate the quality of the data.

Protocol for the QA plan called for routine submission of QA samples. Three types of QA samples were used: preservation blank, reference, and field-split duplicates. Preservation blanks, consisting of distilled water and preserved in the same manner as field samples, were used to evaluate sample contamination and the laboratory's baseline analytical capabilities near minimum reporting levels. Reference samples prepared from USEPA Quality Control samples were used to determine analytical accuracy. Field-split duplicate samples were used in the evaluation of analytical precision. QA data were monitored during the project, and corrective steps were taken if the data indicated analytical process problems. QA-sample data for the October 1, 1986, through September 30, 1990, post-BMP period are summarized in table 5.

Table 5. Summary statistics for quality-assurance analyses at Field-Site 2, October 1986 to September 1990

[Detection limits, ranges, and median differences in milligrams per liter as N or P; n, number of observations; <, less than; --, no data]

Constituent	Detection limit	Blanks			Reference samples				Duplicates		
		n	Range of reported concentrations	Median difference from detection limit	n	Range of known concentrations	Absolute value of difference between known and reported concentrations: Maximum	Absolute value of difference between known and reported concentrations: Median	n	Range of reported concentrations	Absolute value of differences between pairs[1] (Maximum)
Total nitrate + nitrite	0.04	31	<0.04/<0.04	0.00	--	--	--	--	67	1.0/18	4.5
Dissolved nitrate + nitrite	.04	22	<.04/.04	.00	--	--	--	--	161	1.1/74	10
Total nitrite	.01	31	<.01/.01	.00	--	--	--	--	64	<.01/.50	.20
Dissolved nitrite	.01	22	<.01/<.01	.00	--	--	--	--	54	<.01/.41	.18
Total nitrate	--	--	--	--	41	0.14/7.9	2.4	0.07	--	--	--
Dissolved nitrate	--	--	--	--	20	.14/6	.6	.04	--	--	--
Total ammonia + organic nitrogen	.20	30	<.20/1.9	.12	37	.33/21	7.7	.52	66	<.20/21	5.5
Dissolved ammonia + organic nitrogen	.20	21	<.20/1.9	.11	19	.66/20	2.1	.70	111	<.20/6.1	1.8
Total ammonia	.02	31	<.02/.09	.00	37	.07/7.9	4.0	.05	64	<.02/1.8	.18
Dissolved ammonia	.02	22	<.02/.06	.01	20	.28/7.6	1.2	.10	54	<.02/1.5	.77
Total phosphorus	.02	30	<.02/.03	.00	45	.02/6.1	1.7	.03	66	<.02/9.8	4.8
Dissolved phosphorus	.02	21	<.02/.02	.00	19	.20/6.1	.40	.07	54	<.02/1.9	.14

[1] Difference between pairs was determined by subtracting concentration value for blind duplicate from concentration value for sample.

Preservation blanks were prepared in the same manner as the nutrient samples and analyzed for all total- and dissolved-nutrient species listed in table 5. Ideally, measured concentrations of nutrients in blank samples should report as less than the minimum reporting level, and results reporting measurable concentrations are indicative of sample contamination, limitations of the analytical procedures, or both. Acceptable results, however, will report within two times the minimum reporting level. Total and dissolved ammonia plus organic nitrogen and total and dissolved ammonia were the only constituents with more than 5 percent unacceptable blank results (table 6). The large number of total and dissolved ammonia plus organic nitrogen blank analysis not within acceptable limits was probably because of known limitations with the analytical procedures. The large number of unacceptable total and dissolved ammonia results was not expected.

Table 6. Results of blank sample analyses at Field-Site 2, October 1986 through September 1990

[MRL, minimum reporting level; --, no data]

Constituent	Number of observations	Percent of samples within limit	
		MRL	2 × MRL
Total nitrate + nitrite	31	100	--
Dissolved nitrate + nitrite	22	100	--
Total nitrite	31	100	--
Dissolved nitrite	22	100	--
Total ammonia + organic nitrogen	30	63	93
Dissolved ammonia + organic nitrogen	21	62	81
Total ammonia	31	84	87
Dissolved ammonia	22	77	95
Total phosphorus	30	97	100
Dissolved phosphorus	21	100	--

A signed-rank comparison of median concentrations of total and dissolved ammonium in preservation blanks and ground-water samples showed a significant ($p<0.05$) 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 minimum reporting level, a bias because of inherent analytical problems was unlikely. Further investigation during a similar study (M.J. Langland, U.S. Geological Survey, written commun., 1992) determined that surface-water samples containing measurable concentrations of ammonium, transported in close proximity to blank water samples, could be a source of ammonia contamination.

Reference samples were analyzed for concentrations of total and dissolved nitrate, ammonia plus organic nitrogen, ammonium, and phosphorus. Results from the reference samples and sample data from the NWQL Standard Reference Sample Program were combined and evaluated as a group. A Wilcoxon Signed-Rank test showed a significant positive bias between the measured and expected concentrations for dissolved nitrate, total ammonia plus organic nitrogen, and total and dissolved phosphorus. A significant negative bias was found for dissolved ammonia. Except for total ammonia plus organic nitrogen, the median difference between known and reported concentrations was 0.10 mg/L or less. For reference samples, a relative percent difference (RPD) was calculated for each measured concentration and expected concentration pair. The RPD was calculated as follows:

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

The RPD's indicated that overall analytical accuracy varied considerably. A Wilcoxon Signed-Ranks test indicated significant bias for all constituents except dissolved ammonia plus organic nitrogen and dissolved ammonium. All significant biases were positive except for the total ammonia. For ground-water data, estimated biases in concentrations of reported total and dissolved ammonia plus organic nitrogen and total and dissolved phosphorus represent a large source of error, and caution was used in their interpretation. The estimated bias for concentrations of total and dissolved nitrate represented 2 percent or less of the nitrate concentrations measured in ground-water samples.

For duplicate data, RPD's were calculated for each duplicate pair. Determination of acceptable analytical repeatability was made by comparing the RPD for each duplicate pair with RPD goals. RPD goals ranged from 0 percent of the concentrations at the detection limit to 10 percent for concentrations equal to or greater than 20 times the detection limit (table 7). The RPD's for all constituents, with the exception of total ammonium plus organic nitrogen, total and dissolved nitrate plus nitrite, and total phosphorus, were within RPD goals for 90 percent or more of the samples analyzed. Total phosphorus samples commonly exceeded the RPD goal.

Table 7. Relative percentage difference goals for analytical results from duplicate samples at Field-Site 2

Sample concentration range, in minimum reporting levels	Relative percent difference goal, in minimum reporting levels or percent
0-5	1
5-20	2 or 20 percent[1]
20 or greater	10 percent

[1] Whichever is greater.

Results from the QA program indicate that bias and accuracy limitations existed for many of the constituents. In terms of accuracy, the water-quality data for nutrients should be interpreted with caution when concentrations are near the detection limit. However, for those constituents that constituted the primary nutrient sources in surface water and ground water at Field-Site 2, the bias and accuracy limitations are not detrimental.

SELECTED FACTORS THAT AFFECTED WATER QUALITY

Physiography and Geology

Site physiography and geology affected surface water and ground water. The terraced land surface had a median slope of about 5 percent and ranged from 2 to 9 percent. The farmer reported that soil erosion was significantly reduced after terrace construction. Soils at the site were, in general, very permeable, and runoff typically occurred only during the most intense rainstorms. It is unknown if terracing altered the water-table configuration and (or) the percentages of precipitation that infiltrated to ground water at Field-Site 2. The terraces were constructed in 1965, and pipe-outlet drains were added in 1981 before the Field-Site 2 investigation began in October 1984.

A 5- to 30-ft thick mantle of soil and weathered rock formed over moderately fractured carbonate bedrock, forming a dual porosity ground-water flow system at the site. Although the effective depth of ground-water flow through fractures at Field-Site 2 was unknown, drilling logs from site wells suggest that fracture occurrence decreased with increasing depth and that most ground water probably flowed through solutionally-developed fractures near the bedrock surface. The water table in the limestone of the Millbach Formation in the western part of Field-Site 2 (figs. 3, 4, and 5) was very near the bedrock surface and fluctuated with changing hydrologic conditions between the bedrock and the porous granular media at the regolith base. Well-pumping-test data indicate that the dolomite of the Snitz Creek Formation was more fractured than the limestone of the Millbach Formation. The water table in the Snitz Creek Formation beneath the eastern part of the site (figs. 3, 4, and 5) was primarily in the fractured bedrock. Detailed descriptions of site hydrogeology were published in Koerkle and others (1996).

Hydrology

Identification and description of relations between the implementation of BMP's and changes in water quality were facilitated through an understanding of site hydrology. Precipitation, runoff, recharge to ground water, discharge from ground water, and evapotranspiration were the major hydrologic inputs and outputs at the site. This section contains brief discussions of each hydrologic input and output, followed by an annual summary of site hydrologic conditions in a water budget.

Precipitation

Precipitation measured at the site was compared to the long-term (1951-80) average of 43.5 in/yr as measured at the Ephrata, Pa., NOAA gage (table 8). Although precipitation is well distributed throughout the average year, temporal variability in precipitation is typically large in this region, and both the pre-BMP and post-BMP periods contained months that were significantly above or below normal averages. Overall, precipitation during the 24-month pre-BMP period was 14.5 percent below the long-term average, and precipitation in the 48-month post-BMP period was 1.5 percent above average.

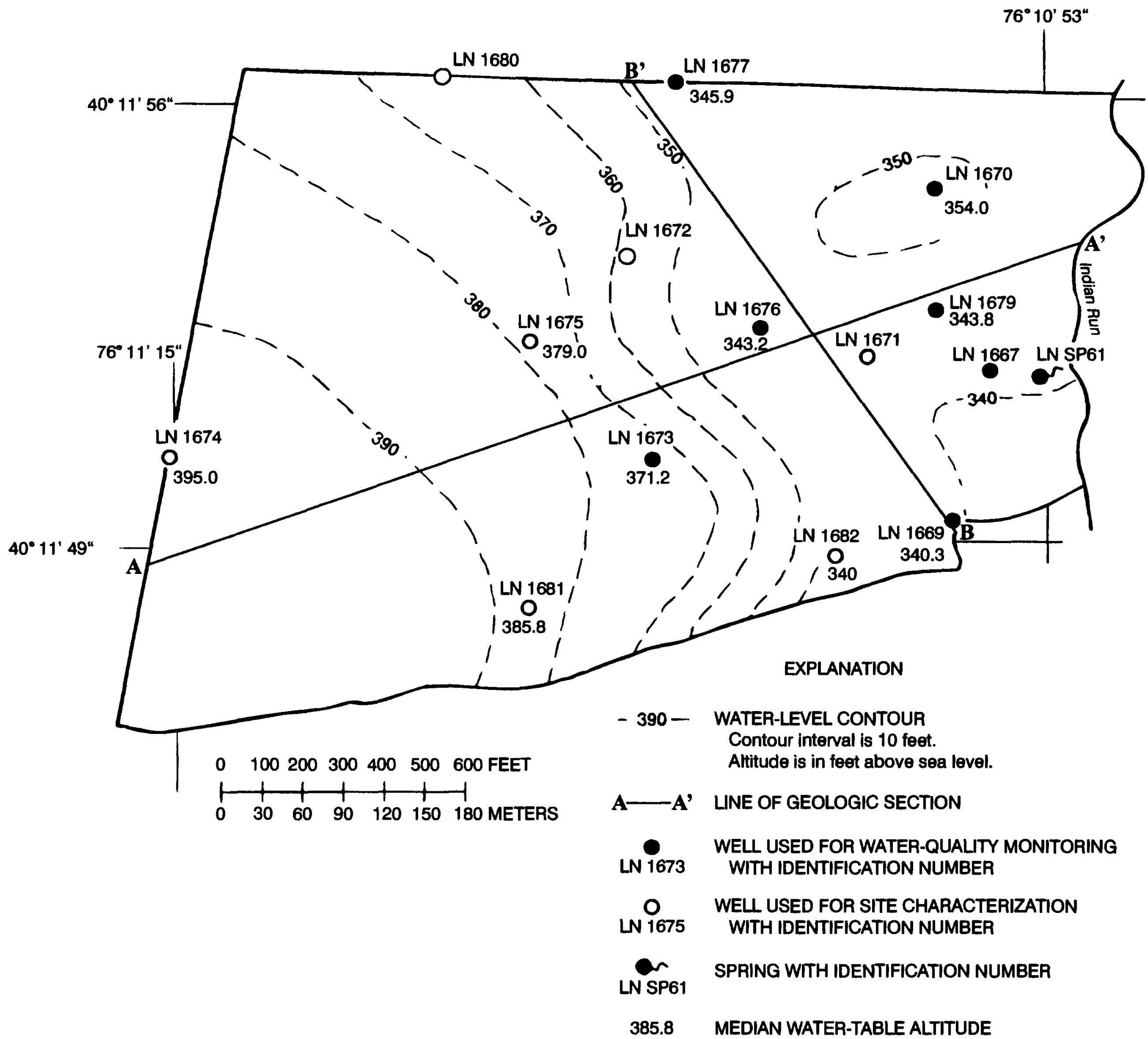

Figure 4. Estimated water-table altitude at Field-Site 2 (sections A-A' and B-B' are on figure 5).

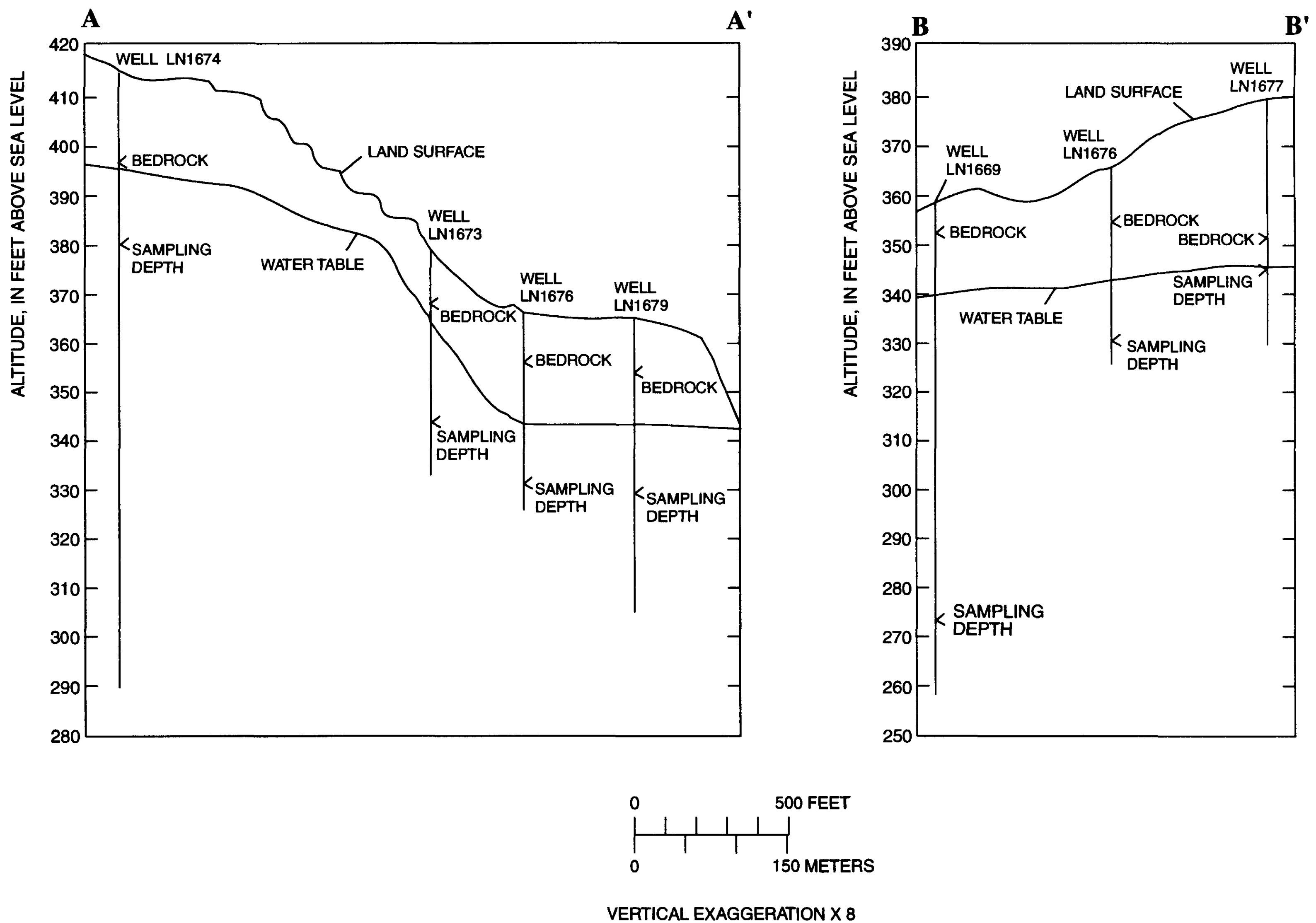

Figure 5. Sections A-A' and B-B' at Field-Site 2 (orientations of sections are shown on figure 4).

Table 8. Annual precipitation at Field-Site 2, long-term average for Ephrata, Pa., and percentage deviation from long-term average, 1985-90

[pre-Best-Management Practice (pre-BMP) period is October 1984 through September 1986; post-Best-Management Practice (post-BMP) period is October 1986 through September 1990]

Period	Water years	Precipitation, in inches	Long-term average[1]	Percentage deviation from long-term average
Pre-BMP	1985	35.8	43.5	-18
Pre-BMP	1986	38.8	43.5	-11
Post-BMP	1987	45.0	43.5	+3
Post-BMP	1988	40.4	43.5	-7
Post-BMP	1989	46.6	43.5	+7
Post-BMP	1990	43.6	43.5	+0

[1] Long-term average precipitation on the basis of 30 years (1951-80) of record from the National Oceanic and Atmospheric Administration weather station at Ephrata, Pa.

Surface Runoff

Surface runoff from 27 terraced acres of Field-Site 2, drained by a pipe-drainage system, was monitored for 2 years prior to BMP implementation (October 1984 through September 1986) and for 2 years after BMP implementation (October 1986 through September 1988). Total annual runoff for the 1985, 1986, 1987, and 1988 water years of the study period was 60,900, 29,800, 28,800, and 130,000 ft^3, respectively, and represented from 0.7 to 3.3 percent of annual precipitation. The greatest monthly runoff generally occurred in the winter during frozen-ground conditions and in the summer during thunderstorms (fig. 6).

During the first 3 years of the study, more than 75 percent of the runoff for each year was produced by five storms or less. In the 1985 water year, 79 percent of the annual runoff was recorded during one 0.87-in. storm that occurred on frozen soil and snowcover. In the 1986 water year, five storms that occurred when soils were frozen and snow covered produced 46 percent of the annual runoff, and one thunderstorm produced another 42 percent of the annual runoff. In the 1987 water year, two summer storms produced 40 and 22 percent of the annual runoff. Three storms when soils were frozen produced an additional 14 percent of the annual runoff.

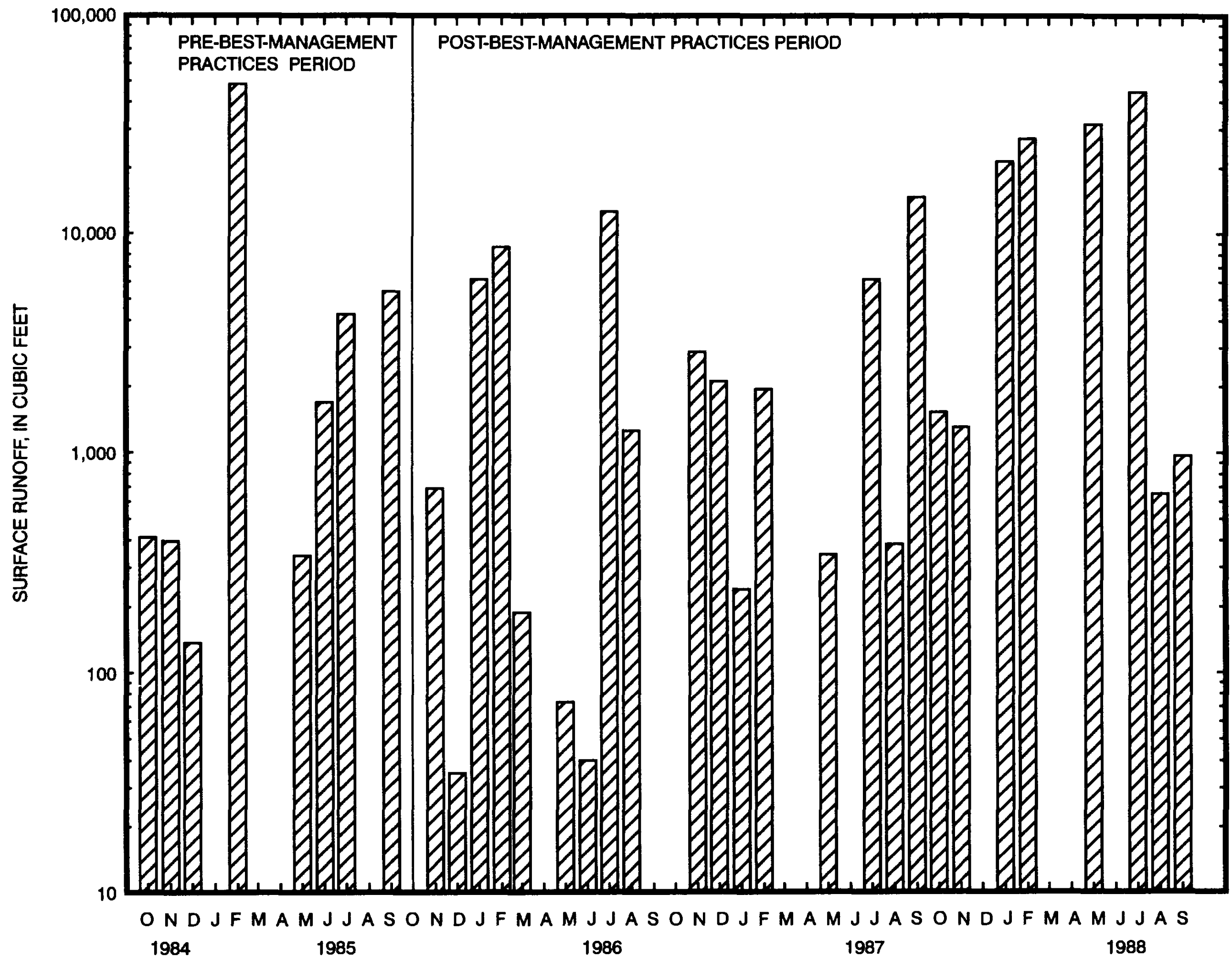

Figure 6. Monthly runoff from 27 terraced acres at Field-Site 2.

In the 1988 water year, the runoff was more evenly distributed among storms. One storm on May 19 produced 17 percent of the annual runoff, and four storms in July produced an additional 31 percent of the annual runoff. Eight storms during frozen-ground conditions in January and February produced 38 percent of the annual runoff. During the entire 4-year study period, each storm on frozen ground produced less than 1.0 in. of precipitation.

The amount of runoff produced by a given amount of precipitation was quite sensitive to frozen soil conditions (fig. 7). Regression equations describing the relation between precipitation and runoff (table 9) indicate that the increase in total runoff with increasing precipitation was about 400 times greater on frozen soil than on thawed soil. Thus, one large storm on frozen soil could produce a substantial part of the total annual runoff.

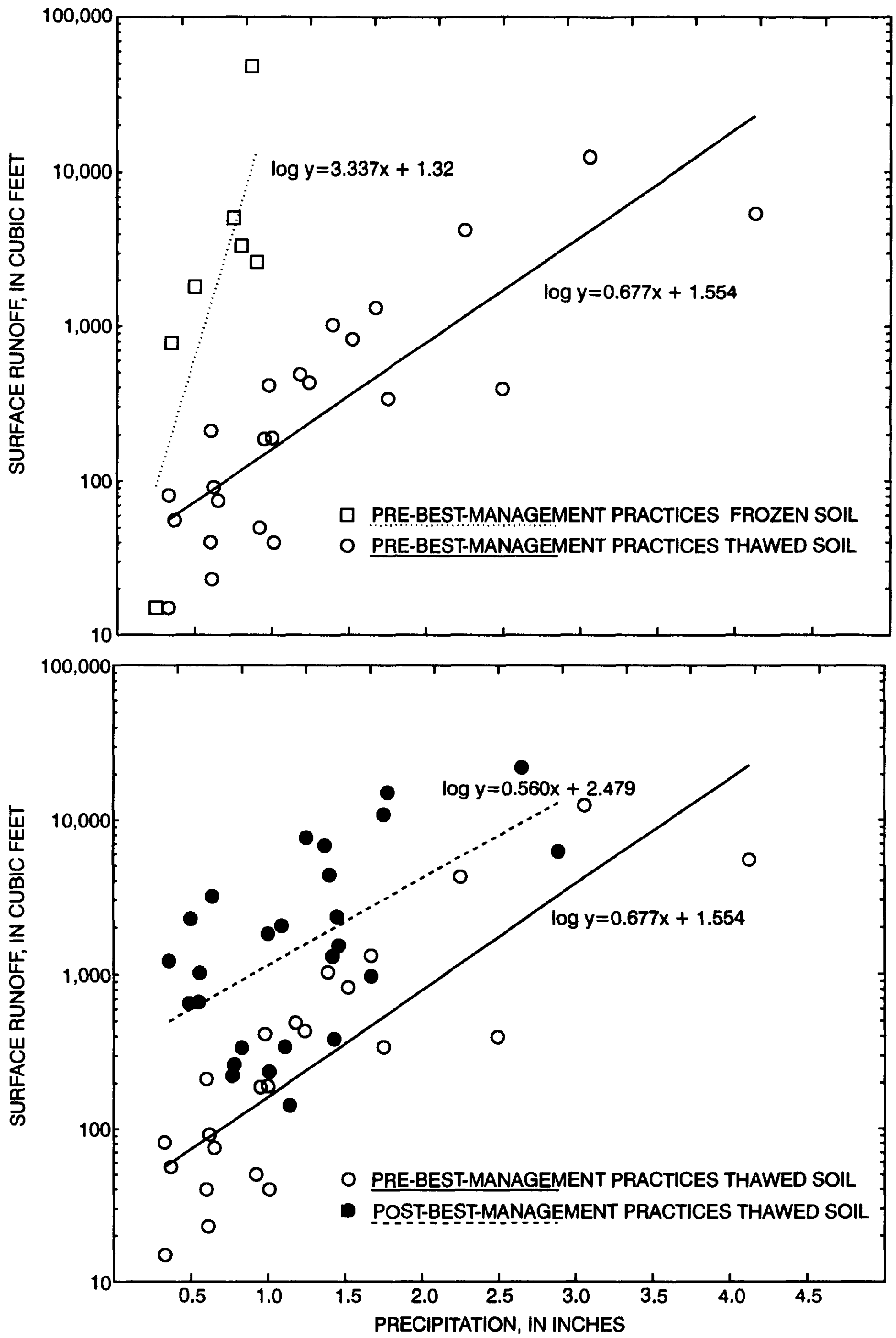

Figure 7. Relation between pipe discharge of surface runoff and precipitation for storms occurring on thawed and frozen soil during the pre-Best-Management Practice period (water years 1985-86), excluding discharge from snowmelt only, and for storms occurring on thawed soil during the pre-Best-Management Practice (water years 1985-86) and post-Best-Management Practice (water years 1987-88) periods, excluding discharge from snowmelt only.

Table 9. Regression statistics for the log of total storm runoff as a function of storm precipitation for Field-Site 2
[Runoff is in cubic feet and storm precipitation is in inches; pre-Best-Management Practice (pre-BMP) period is October 1984 through September 1986; post-Best-Management Practice (post-BMP) period is October 1986 through September 1988; n, number of storms; <, less than]

Dependent variable	Variable	n	Regression coefficient			Intercept	Coefficient of determination (adj. R^2)[1]	Standard error		
			Storm precipitation	t-test	p-value			Log units	Percent[2] Plus	Percent[2] Minus
Pre-BMP runoff	Thawed soil	23	0.677	7.128	<0.001	1.554	0.69	0.42	163	62
Post-BMP runoff	Thawed soil	26	.560	3.378	.002	2.479	.29	.51	224	69
Pre-BMP runoff	Frozen soil	7	3.337	3.238	.023	1.132	.61	.66	357	78
Pre-BMP and post-BMP runoff	Frozen soil	17	1.719	2.267	.039	2.257	.21	.71	413	69

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

During the post-BMP period, the amount of runoff per amount of precipitation increased for storms occurring on thawed soil (fig. 7). Analysis of covariance confirmed a significant difference between pre-BMP and post-BMP period regressions of the relation between precipitation and runoff. The relation was not significantly different between periods when the soil was frozen.

Overall, runoff increased significantly from a median of 0.66 percent of the precipitation for all storms occurring during the pre-BMP period to 1.9 percent in the post-BMP period (fig. 8). Because the distribution of precipitation quantities and intensities did not change significantly from the pre-BMP to the post-BMP period, the change in tillage practices, from no-till with continuous cover to more conventional tillage with a fallow period, was the likely cause of the increased runoff. The increase in runoff percentage is important because it would result in larger total loads of nutrients leaving the site if nutrient concentrations in runoff remained constant.

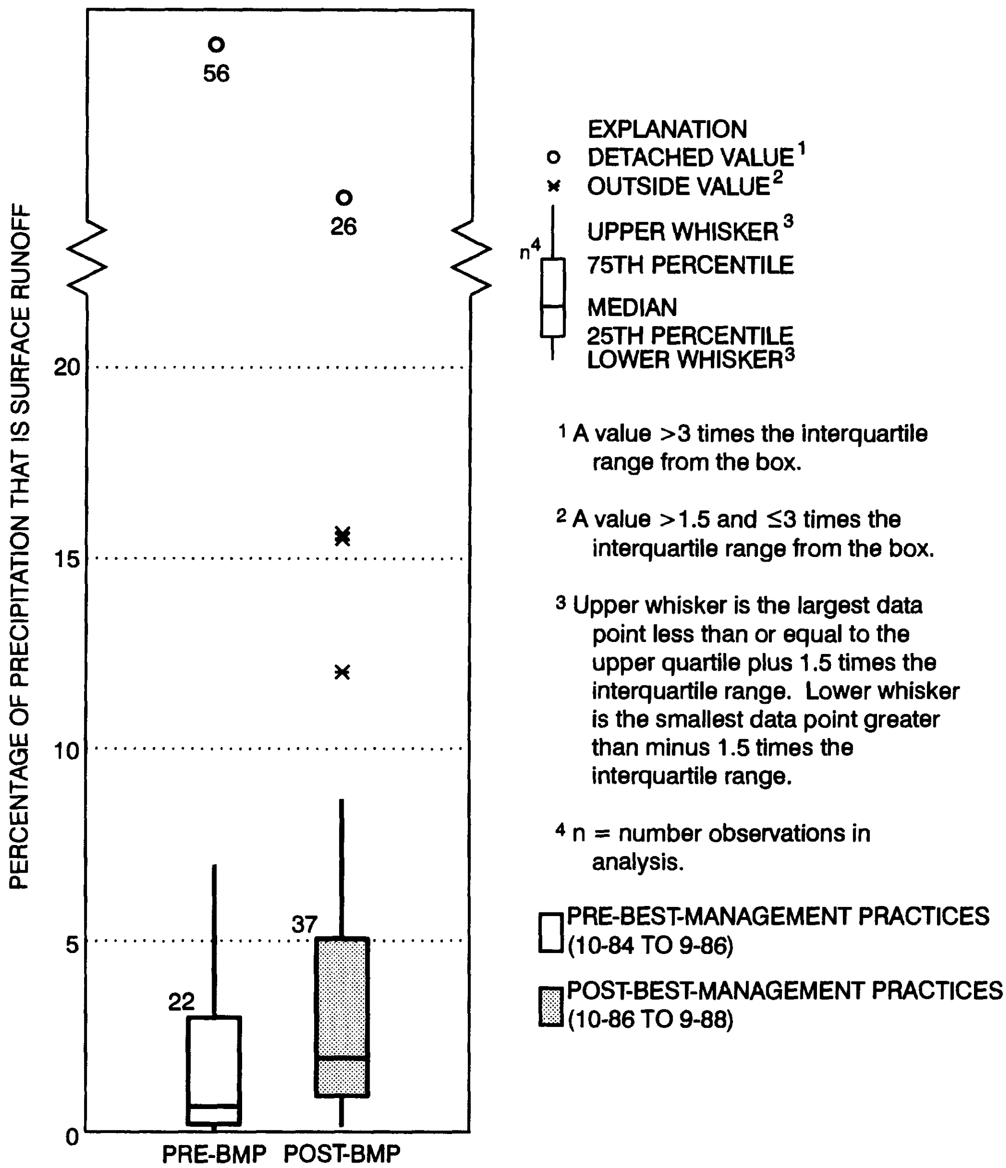

Figure 8. Percentage of precipitation from storms that was discharged as surface runoff from the 27 terraced acres at Field-Site 2.

Ground Water

Configuration of the ground-water table and geologic sections through the site are on figures 4 and 5. The water-table altitude in this karstic terrain responded to recharge within 1 to 3 days following the onset of precipitation. Shallow ground water moved rapidly through porous regolith and fractured bedrock to Indian Run (fig. 3).

A simulated steady-state ground-water budget (table 10) (Koerkle and others, 1996) shows fluxes of water that occurred through the site ground-water system during an average year. Ground-water inflow to the site beneath the western boundary and recharge to ground water were assumed to equal the sum of ground water discharging across the eastern, northern, and southern site boundaries on an annual basis. Ground-water inflow across the western boundary of the field site was estimated as 16 percent of total annual ground-water input to the site, with recharge from infiltration of precipitation falling directly on the site accounting for the remaining 84 percent. Approximately 16 percent of annual discharge flowed beneath the eastern site boundary, 24 percent flowed beneath the northern site boundary, and 60 percent flowed beneath the southern site boundary.

Contributing areas to the wells (areas where applied nitrogen in manure and commercial fertilizer can significantly influence the water quality of water samples collected from a well) are shown in figure 9. Methods used in the estimation of well contributing areas at the site are discussed in Koerkle and others (1996), Hall (1992), Hall and Risser (1992), and Unangst (1992).

Table 10. Simulated steady-state ground-water budget for Field-Site 2

	Inches per year	Percentage of total inflow or outflow
Ground-water inflow		
Recharge from precipitation	19.1	84
Flow across western boundary	3.7	16
Total	22.8	100
Ground-water outflow		
Flow across eastern boundary	3.7	16
Flow across northern boundary	5.4	24
Flow across southern boundary	13.7	60
Total	22.8	100

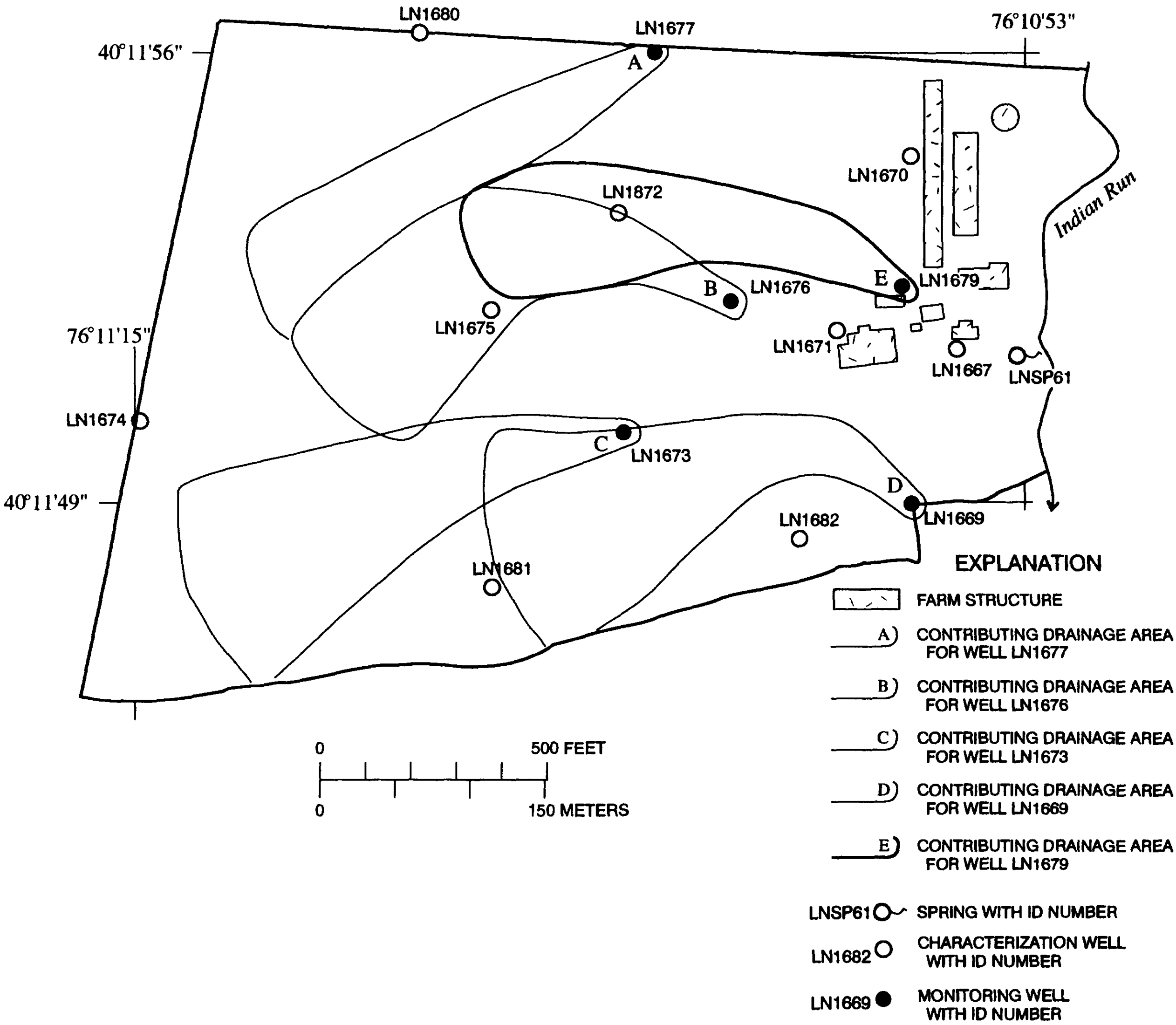

Figure 9. Estimated contributing areas for five wells at Field-Site 2.

Water Budget

Estimation of nitrogen inputs to, and outputs from, the site required calculation of loads of nitrogen in surface and ground water. Estimation of nitrogen loads required calculation of a site water budget in terms of precipitation, ground-water inflow, runoff, evapotranspiration, and ground-water outflow (table 11). Changes in storage of water were assumed to be zero on an annual basis.

The annual precipitation falling on the site (table 8) was measured at the on-site precipitation gage (fig. 3).

Ground-water inflow across site boundaries is, on average, 19 percent of the recharge from direct precipitation on the site on the basis of ground-water modeling simulations. Annual ground-water recharge from direct precipitation on the field site was estimated from the sum of water-level rises recorded in observation wells for all storms during a year (Koerkle and others, 1996). Ground-water outflow is the sum of the two ground-water recharge terms (infiltration from direct precipitation and inflow across site boundaries).

Surface runoff from the 27 terraced acres of the site was measured from 1985 to 1988. Total gaged runoff volumes from the 27 terraced acres were doubled to obtain a gross estimate of surface runoff from the entire 55-acre site. During storms, ungaged water was visually observed to discharge to Indian Run about 150 ft north of the gage flume and also near the northeastern corner of the site. Quantities of ungaged discharge were observed to be approximately equal in volume to the quantity of discharge measured at the site gage. A potentially large percentage of error in the estimation of surface runoff volume is acceptable for the purposes of this water budget because surface runoff accounted for a small percentage of water output from the site.

Water lost from the site by evapotranspiration was computed as the residual term in the water budget. An average of 21.9 in. of water was lost by evapotranspiration during the period 1985-90, from a minimum of 18.2 in. in 1986 to a maximum of 25 in. in 1985. Flippo (1982) estimated an annual evapotranspiration potential of about 40 in/yr for the Ephrata area, an estimate that is in reasonable agreement with an actual evapotranspiration of 21.9 in/yr.

Table 11. Annual water budget in terms of precipitation, ground-water inflow, runoff, evapotranspiration, and ground-water outflow at Field-Site 2, water years 1985-90

[All budget terms are in inches; numbers in parentheses are percentage of total outflow]

Water year	Inflow			=	Outflow				
	Precipitation	+	Ground-water inflow[1]	=	Runoff	+	Evapotranspiration	+	Ground-water outflow
1985	35.8		1.9		0.6 (2)		25.0 (66)		12.1 (32)
1986	38.8		3.5		.3 (1)		18.2 (43)		24.0 (56)
1987	45.0		4.1		2.9 (6)		20.5 (42)		25.7 (52)
1988	40.4		3.7		1.3 (3)		19.6 (44)		23.2 (53)
1989	46.6		4.1		[2]1.4 (3)		23.5 (46)		25.8 (51)
1990	43.6		3.4		[2]1.3 (3)		24.7 (52)		21.0 (45)
Average	41.7		3.5		1.3 (3)		21.9 (48)		22.0 (49)

[1] Ground-water inflow computed as 19 percent of recharge from direct precipitation on the field site. It also equals 16 percent of total ground-water outflow.

[2] Runoff estimated as 3 percent of precipitation on the basis of average of 1985-88 data.

Agricultural Activities

Water quality at the site was affected by agricultural activities, soil and aquifer characteristics, and climatic conditions. Because the objective of this investigation was to determine the effects of changes in agricultural-management practices on water quality, detailed agricultural-activity data were collected monthly from the farm managers. Agricultural-activity data included times of planting and harvest, tillage information, and areas and times of applications of manure and commercial fertilizer.

Approximately 100 steers, 1,500 swine (three groups - 500 at a time), and 110,000 chickens (five groups - 22,000 at a time) were raised at the site during each year of the study period. Nutrient-management planning was implemented in October 1986 and resulted in the export of approximately 33 percent of the nitrogen in manure generated by the site animal operations. Pounds of nitrogen and phosphorus applied monthly to farm fields are shown on figures 10 and 11, respectively. Annual applications of nitrogen and phosphorus are shown in table 12. Table 13 shows the pounds per acre of nitrogen that were applied to the contributing areas of five wells during the study period. Reductions in nitrogen applications from 1986 to 1987 resulted from the implementation of nutrient-management planning and the related export of excess manure. Nitrogen and phosphorus applications to farm fields potentially represent the primary nutrient source in surface runoff and ground water.

Crops were primarily fertilized by the use of manure, although some commercial fertilizer was applied. Three types of manure were applied: steer manure and bedding pack from the feedlot, hog manure from gestation and finishing operations, and poultry manure from the poultry shed. The steer manure mix and the poultry manure were applied to cropped fields by surface spreading. The hog manure liquid was injected into the soil 8 to 10 in. below the soil surface. During times of frozen soil, all manures were surface spread. Commercial fertilizers applied to the site included ammonium sulfate, broadcast before planting, and a nitrogen liquid that was co-applied with pre-emergent pesticides or added as a sidedress during the early stages of crop growth.

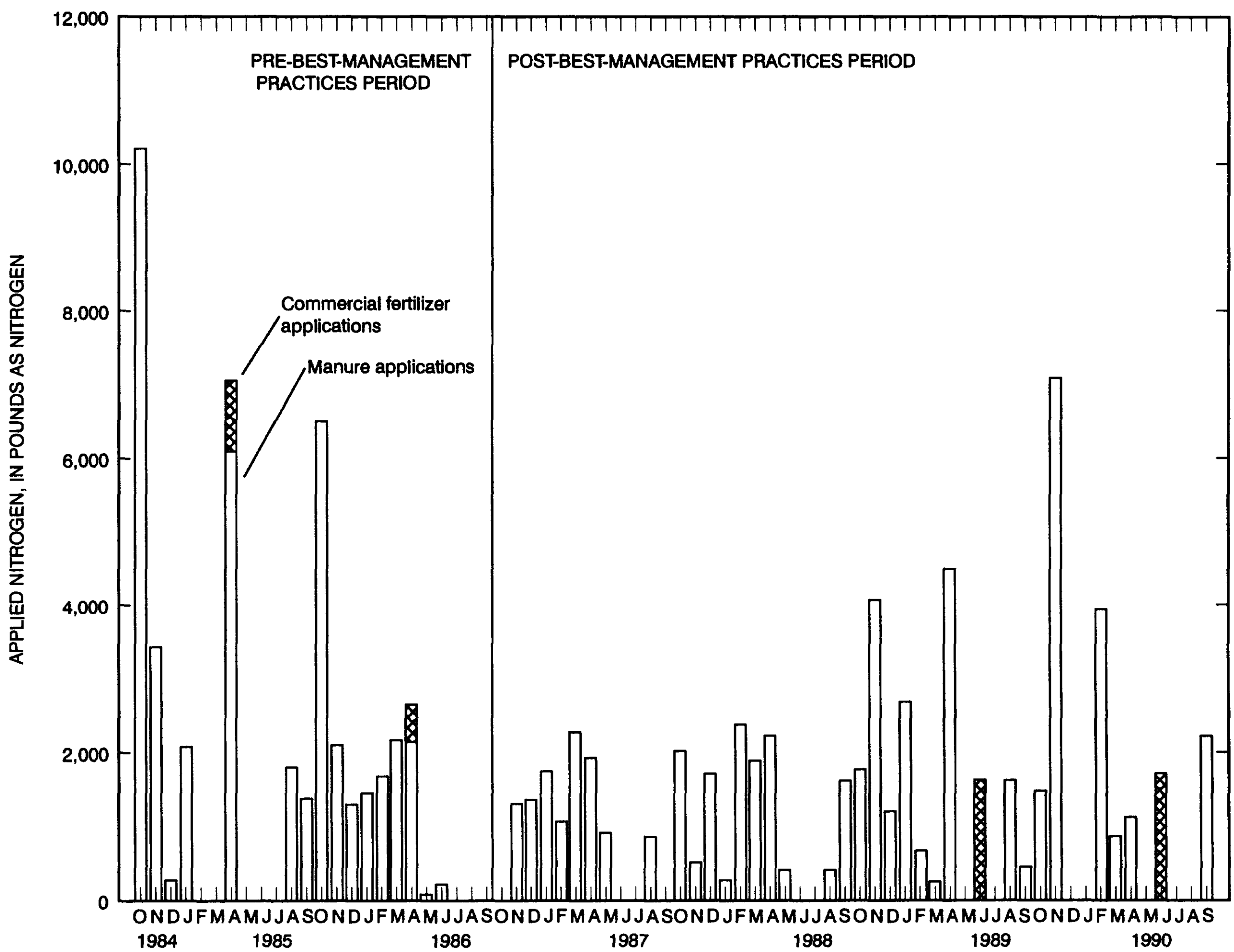

Figure 10. Monthly manure and commercial fertilizer nitrogen applications to Field-Site 2.

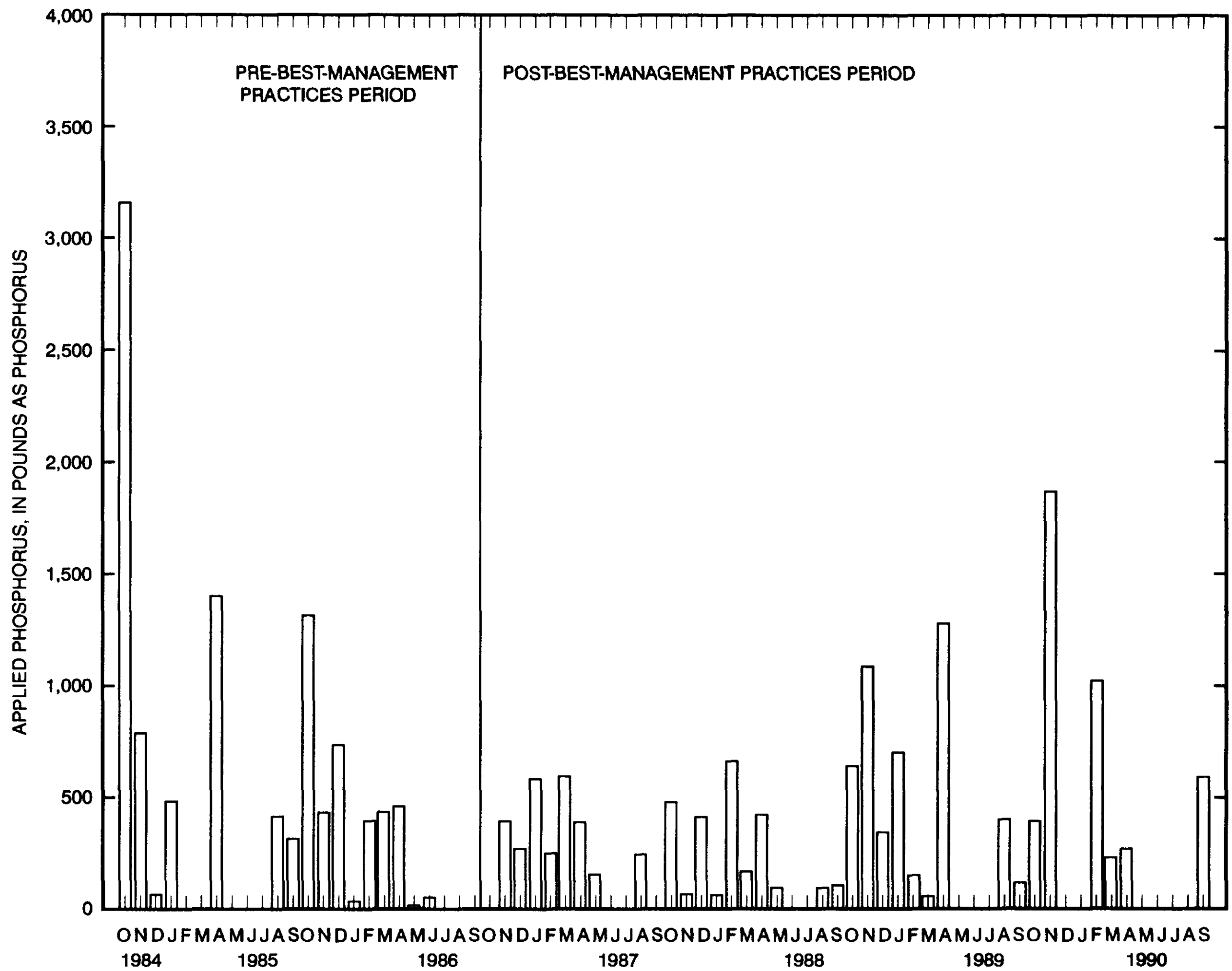

Figure 11. Monthly manure phosphorus applications to Field-Site 2.

Table 12. Annual nitrogen and phosphorus loads applied to Field-Site 2, water years 1985-90

[pre-Best-Management Practice (pre-BMP) period is October 1984 through September 1986; post-Best-Management Practice (post-BMP) period is October 1986 through September 1990]

Period	Water year	Nitrogen load, in pounds as nitrogen	Phosphorus load, in pounds as phosphorus
Pre-BMP	1985	26,500	6,600
Pre-BMP	[1]1986	19,000	3,700
Post-BMP	1987	11,500	2,900
Post-BMP	1988	13,500	2,600
Post-BMP	1989	18,800	4,800
Post-BMP	1990	18,400	4,400
	Average annual pre-BMP load (1985-86)	22,700	5,150
	Average annual post-BMP load (1987-90)	15,550	3,675

[1] 1986 manure applications were unusually small because of less manure disposal as the result of an outbreak of avian influenza in Lancaster County, Pa., which significantly reduced the number of chickens present on the farm.

Table 13. Nitrogen in manure and commercial fertilizer applied in the contributing areas of five wells at Field-Site 2, water years 1985-90

[pre-Best-Management Practice (pre-BMP) period is October 1984 through September 1986; post-Best-Management Practice (post-BMP) period is October 1986 through September 1990]

Period	Year	Nitrogen applied to well contributing areas, in pounds per acre per year				
		LN 1669 (5.6 acres)	LN 1673 (5.6 acres)	LN 1676 (5.5 acres)	LN 1677 (3.4 acres)	LN 1679 (4.1 acres)
Pre-BMP	1985	580	670	570	550	830
Pre-BMP	1986	390	490	440	470	430
Post-BMP	1987	160	200	180	290	290
Post-BMP	1988	130	160	150	310	290
Post-BMP	1989	230	360	150	310	270
Post-BMP	1990	120	380	180	340	160

Planting and harvesting were scheduled according to crop-growth requirements and weather conditions. Crop acreage for the years 1985-90 is listed on table 14. Field corn, the primary crop, was usually planted during the last 2 weeks of April and harvested between mid-September and early October. Tobacco requires a shorter, warmer season and was transplanted from starting beds to the field in mid-June and harvested in mid- to late August. During 1985 and 1986, a winter cover crop of rye was broadcast seeded after corn harvesting. The rye was mainly planted on the pipe-drained terraces. The rye crops were not harvested but were killed with herbicide before the planting of corn. In 1987, a winter cover crop of sudan grass was planted on approximately 5.5 acres of the site. Beginning in 1988, fruit and vegetables, in addition to sweet corn, were cultivated. In 1989 and 1990, more acreage was planted in fruit and vegetables and less in tobacco.

Table 14. Annual crop types and acreage at Field-Site 2

Water year	Growing season	Crop type	Acreage
1985	Summer	Corn	43.5
	Summer	Tobacco	4.0
	Winter (1984-85)	Rye	22.5
1986	Summer	Corn	43.5
	Summer	Tobacco	4.0
	Winter (1985-86)	Rye	25.0
1987	Summer	Corn	42
	Summer	Tobacco	5.5
	Winter (1986-87)	Sudan grass	5.5
1988	Summer	Corn	39.5
	Summer	Tobacco	5.0
	Summer	Fruit and vegetables	2.5
1989	Summer	Corn	35
	Summer	Tobacco	2.5
	Summer	Fruit and vegetables	10
1990	Summer	Corn	37
	Summer	Tobacco	4.0
	Summer	Fruit and vegetables	6.5

Two tillage practices were used during the pre-BMP period (table 15). Minimum-tillage (Pennsylvania State University, 1991) practices were used on approximately 18 acres that had low or no slopes. On steeper areas of the site, including the pipe-drained terraces, no-till practices were used. The minimum-till methods consisted of chisel plowing and disking before spring planting. During the post-BMP period, tillage practices changed several times, and in 1990, the entire site was moldboard plowed (table 15).

Table 15. Tillage practices used at Field-Site 2, water years 1985-90

Water year	Tillage
1985	No till - on 29.5 acres Minimum till - on 18 acres: chisel plowed[1] throughout nongrowing season and disked before spring planting
1986	No till - on 29.5 acres Minimum till - on 18 acres: chisel plowed throughout nongrowing season and disked before spring planting
1987	Minimum till - chisel plowed throughout nongrowing season, disked before spring planting
1988	Minimum till - approximately 15 acres moldboard plowed in spring, entire site chisel plowed throughout year and disked in spring
1989	Minimum till - approximately 15 acres moldboard plowed in winter, entire site chisel plowed throughout year and disked in spring
1990	Conventional till - entire site moldboard plowed in spring, entire site chisel plowed throughout year and disked in spring

[1] Hog manure injection causes soil tillage that is essentially identical to chisel plowing. Hog manure is injected throughout the nongrowing season.

Soils

Soils at the site are classified as Hagerstown and Duffield series silt loams and silt-clay loams (fine, mixed, mesic, Typic Hapludulfs) (Custer, 1985). Hagerstown and Duffield soils are typically deep and well drained and are present on low hills and in valleys. Soils were severely eroded on hillslopes and in the western upland area of the site. A clay hardpan, where present, retarded the downward movement of solutes in some areas of the site.

Site soils were regularly loaded with large quantities of nitrogen for crop fertilization, primarily in the form of organic nitrogen in animal manures and associated urea. Organic nitrogen mineralizes through several mechanisms that result in the production of nitrate, the soluble form of nitrogen (oxidation state +5) that is assimilated by plants from the soil (Ehrlich, 1990). Nitrate that is not assimilated by plants is available for transport to surface runoff and ground water.

Soils were sampled in seven areas of the site (fig. 12) for soluble nitrate-nitrogen and soluble phosphorus during the spring and fall of each year from 1984 to 1990. Summaries of soluble nitrate-nitrogen and soluble phosphorus content in soil samples from Field-Site 2 are given in tables 16 and 17.

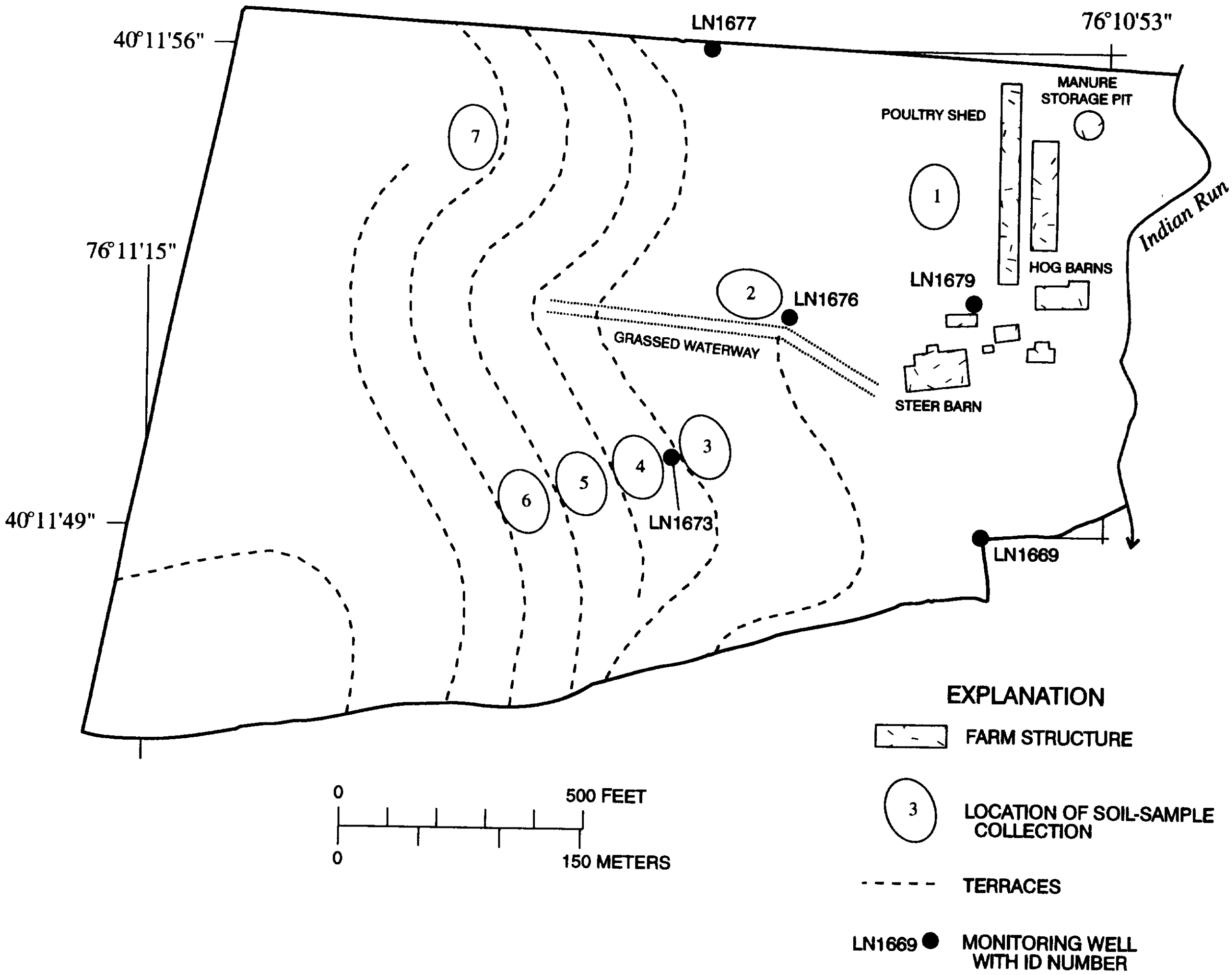

Figure 12. Areas of soil sample collection at Field-Site 2.

Table 16. Soluble nitrate and soluble phosphorus content in the top 0 to 4 feet of the soil profile at Field-Site 2

[pre-Best-Management Practice (pre-BMP) period is October 1984 through September 1986; post-Best-Management Practice (post-BMP) period is October 1986 through September 1990]

Period	Sampling date	Number of samples	Soluble nitrate in pounds per acre as nitrogen			Soluble phosphorus in pounds per acre as phosphorus		
			Mean	Minimum	Maximum	Mean	Minimum	Maximum
Pre-BMP	Fall 1984	2	224	177	272	6.1	3.9	7.8
Pre-BMP	Fall 1985	11	265	128	428	28	3.5	160
Pre-BMP	Spring 1986	14	291	77	614	33	4.4	77
Pre-BMP	Fall 1986	9	227	104	364	27	2.6	45
Post-BMP	Spring 1987	9	123	85	184	35	6.5	110
Post-BMP	Fall 1987	9	96	31	180	22	1.3	63
Post-BMP	Spring 1988	9	147	68	222	22	5.2	42
Post-BMP	Fall 1988	9	205	46	462	17	4.8	52
Post-BMP	Spring 1989	5	260	150	389	16	2.2	36
Post-BMP	Fall 1989	5	182	110	251	27	3.9	57
Post-BMP	Spring 1990	5	186	126	262	25	1.7	45
Post-BMP	Fall 1990	5	170	80	324	24	4.4	47

Table 17. Soluble nitrate content in the top 0 to 4 and 4 to 8 feet of the soil profile at Field-Site 2

[in pounds per acre as nitrogen]

Sampling date	Number of samples	0 to 4 feet			4 to 8 feet		
		Mean	Minimum	Maximum	Mean	Minimum	Maximum
Spring 1989	4	228	150	302	136	82	164
Fall 1989	4	200	130	251	251	85	499
Spring 1990	4	194	126	262	175	99	263
Fall 1990	4	194	80	324	201	134	277

The relation between the quantity and timing of nitrogen (primarily in the form of organic nitrogen contained in manure) applied at the land surface and the quantity of soluble nitrate in the top 4 ft of soils is shown in figure 13. Although it is difficult or impossible to relate long-term applications in a specific month to concentrations of soluble nitrate in soils by use of a plot of this type, a long-term qualitative relation is apparent. When large amounts of nitrogen were applied to the site during the fall of 1984 and the fall of 1985, loads of soluble nitrate in soils were high. When monthly applications were reduced under nutrient management during the period from the fall of 1986 through the fall of 1988, loads of soluble nitrate in soils also were reduced. When nitrogen applications increased during the period from the fall of 1988 to the fall of 1990, loads of soluble nitrate in soils increased.

It is estimated that from 40 to 75 percent of the total nitrogen remaining in most types of manure after storage is converted to soluble nitrate within the first year after application, despite differences in mineralization rates between types (Graves, 1986b; Stevenson, 1982). Some field-applied nitrogen may remain in organic form at the land surface for more than 1 year. A substantial quantity of the organic nitrogen that does ammonify could exist as ammonium or ammonia absorbed to minerals in the soil for periods of time ranging from several days to several years before oxidizing to nitrate. Therefore, some of the soluble nitrate detected by the soil tests early in the study period probably originated from nitrogen applied to the land surface before the study began (process described by Graves, 1986a; Roth and Fox, 1990; Cronce and others, 1990).

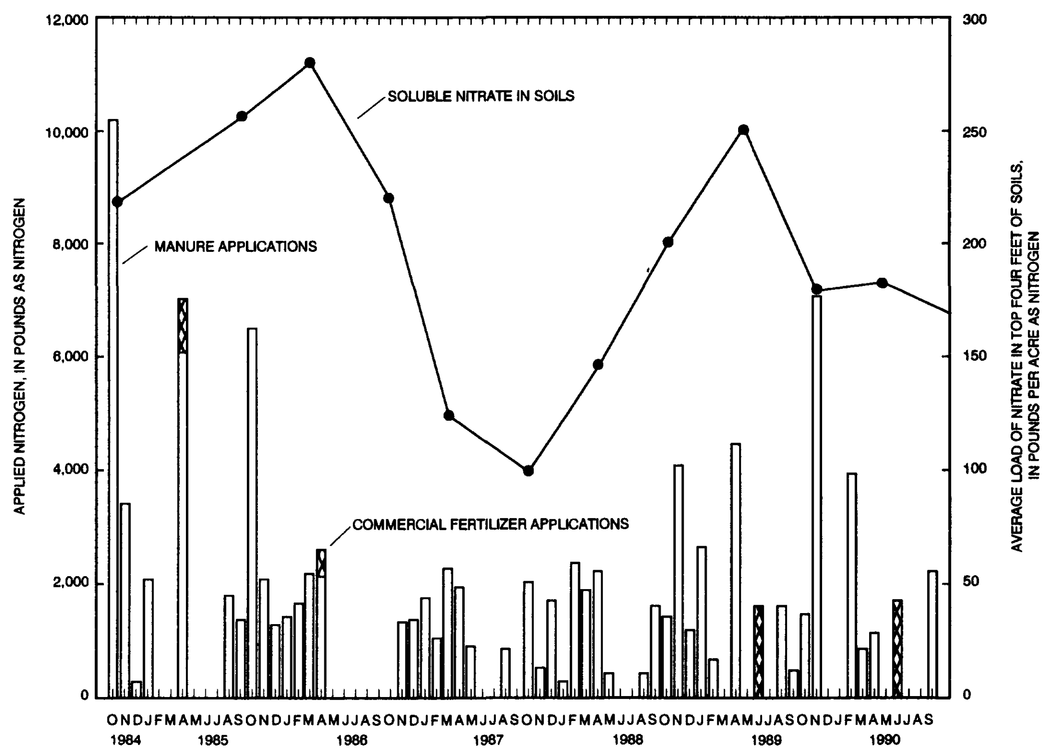

Figure 13. Monthly manure and commercial fertilizer nitrogen applications to 47.5 cropped acres and mean nitrogen concentrations of soil samples collected during each spring and fall.

Changes in the soluble-nitrate content of soils with depth below the land surface are shown in figure 14. Soil samples were collected in triplicate within several feet of one another in the areas illustrated on figure 12 and then composited by depth intervals for analysis. Samples were analyzed from depths of 0 to 8, 8 to 24, and 24 to 48 in. In 1989 and 1990, additional soil samples were collected at depths of 48 to 72 and 72 to 96 in. at the same sampling locations. Depth-normalized data are plotted on figure 14 at the midpoint of the sampled intervals; thus, the average soluble-nitrate concentration for the 0 to 8-in. interval is plotted at 4 in. Similarly, data from the 8 to 24-in. interval are plotted at 16 in.; data from the 24 to 48-in. interval are plotted at 36 in.; data from the 48 to 72-in. interval are plotted at 60 in.; and data from the 72 to 96-in. interval are plotted at 84 in. The data were normalized to pounds per acre per inch to facilitate comparisons between the uneven intervals that were sampled.

Agronomists and farmers are frequently concerned about nitrogen levels in the crop-root zone, which is a depth of approximately 4 ft for corn. Data presented on figure 14 can be used to calculate the amount of nitrogen (in pounds per acre) in the top 4 ft of soils at a sampled location as follows: multiply the value plotted at 4 in. by the 8 in. of sampled depth, add to the value at 16 in. multiplied by the 16 in. of sampled depth, and add to these the value plotted at 36 in. multiplied by 24 in. of sampled depth. Similarly, the pounds per acre of nitrogen in the 4 to 8-ft depth can be calculated by multiplying the value plotted at 60 in. by the 24 in. of sampled depth, and add the value plotted at 84 in. multiplied by 24 in. of sampled depth.

Quantities of nitrogen in the top 4 ft of soils varied significantly from spring to fall and from year to year (fig. 14). For example, at site 4, approximately 88 lb/acre of nitrogen was in the top 4 ft of soils during October 1987 and approximately 456 lb/acre of nitrogen was in the top 4 ft of soils during October 1988. At site 5 approximately 168 lb/acre of nitrogen was in the top 4 ft of soils during the March 1987 sampling but only 64 lb/acre of nitrogen was in the top 4 ft of soils during the October 1987 sampling. The spring and fall soil-sampling data presented do not represent the amount of nitrate in soils immediately before planting or residual nitrate in soils after crop harvest. Applications of commercial fertilizer were sometimes made to fields in the spring after the time of soil sampling, and manure nitrogen was frequently applied to fields after the spring soil sampling and before the fall soil sampling.

The average content of soluble phosphorus in the top 4 ft of soils ranged from 6.1 lb/acre in the fall of 1984 to a maximum of 35 lb/acre in the spring of 1987 (table 16). Virtually all of the soluble phosphorus in soil samples collected at the site was detected in the top 8 in. of soil. No apparent relations existed between the amount of phosphorus applied to the site (fig. 11) and the amount of soluble phosphorus in soils (table 16). Most of the phosphorus applied to soils probably forms relatively insoluble, inorganic calcium, iron, and aluminum phosphates (process described by Ehrlich, 1990; Kardos, 1955).

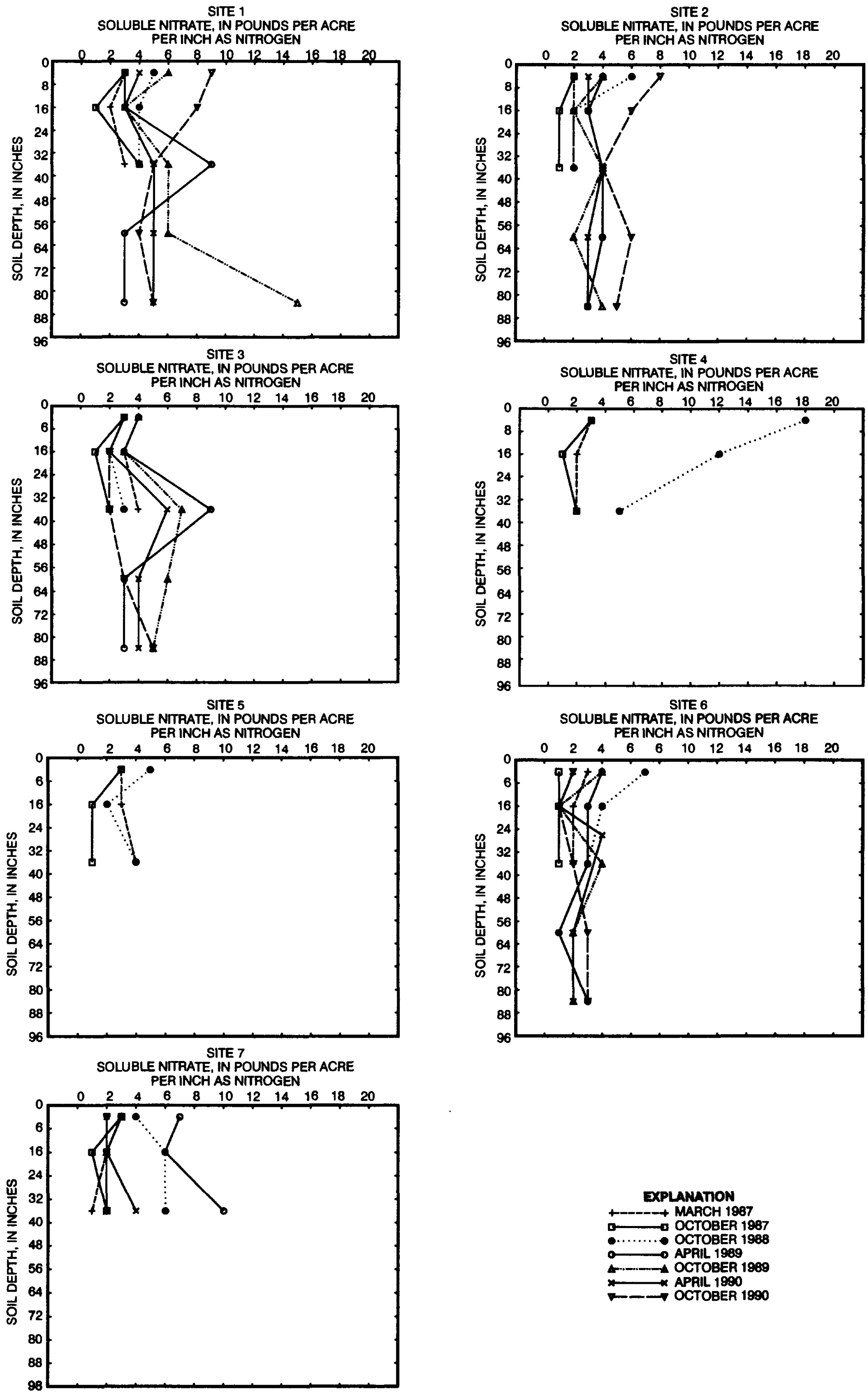

Figure 14. Profiles of soluble nitrate in the soil column for soil-sampling sites 1-7 at Field-Site 2.

EFFECTS OF NUTRIENT MANAGEMENT ON QUALITY OF SURFACE RUNOFF AND GROUND WATER

Surface Runoff

Water-quality samples of surface runoff discharged through the terrace pipe were collected during 22 of 36 runoff events in the pre-BMP period and during 26 of 38 runoff events during the post-BMP period. The surface-runoff samples were analyzed for total nutrients, and samples from 12 pre-BMP and 9 post-BMP events were analyzed for suspended sediment.

Estimated annual nutrient loads from the runoff of 27 terraced, pipe-drained acres are shown in table 18. For the study period, annual total-nitrogen loads in runoff ranged from 14 to 93 lb; of this, about half was ammonium plus organic nitrogen, and about half was nitrate plus nitrite nitrogen. The annual total phosphorus load ranged from 8.4 to 44 lb (table 18). The average annual nutrient loads from runoff for the study period were less than 2 percent of the nutrients applied to the 27 terraced acres.

Table 18. Estimated annual nutrient loads in runoff from 27 terraced acres at Field-Site 2, water years 1985-88

[pre-Best-Management Practice (pre-BMP) period is October 1984 through September 1986; post-Best-Management Practice (post-BMP) period is October 1986 through September 1988]

Period	Water year	Total annual nitrogen load, in pounds	Median percentage of total nitrogen load[1]		Total annual phosphorus load, in pounds
			Ammonium + organic nitrogen	Nitrite + nitrate	
Pre-BMP	1985	62	58	42	28
Pre-BMP	1986	25	47	56	9.1
Post-BMP	1987	14	43	57	8.4
Post-BMP	1988	93	66	31	44

[1] Median of measured storms only.

Suspended-sediment samples were collected only for selected storms. Therefore, annual sediment loads were not estimated. However, nutrient management, the only BMP change implemented for the study at this site, was not expected to substantially affect sediment concentrations or loads.

The distributions of mean storm concentrations and loads of nutrients and suspended sediment in runoff for the pre-BMP and the post-BMP periods are shown in figure 15. Comparison, by use of the Mann-Whitney test, showed that mean storm concentrations of nitrate plus nitrite decreased significantly (95-percent-confidence interval) between the pre-BMP and post-BMP periods. Concentrations of ammonium plus organic nitrogen or total phosphorus did not change significantly. The decrease in median concentrations of nitrate plus nitrite from 5.9 to 4.0 mg/L may have been caused by the change in tillage practices and crop cover, which may have decreased contact time between the surface runoff and highly soluble nitrate and the nutrient-rich soils. However, the change in nitrate concentrations was not large enough to change the total-nitrogen concentrations significantly. Median total-nitrogen loads increased from 0.97 to 1.8 lb, and median total phosphorus loads increased from 0.30 to 0.78 lb from the pre-BMP to the post-BMP period, respectively (fig. 15). Increases in ammonium plus organic nitrogen accounted for most of the increase in total-nitrogen loads.

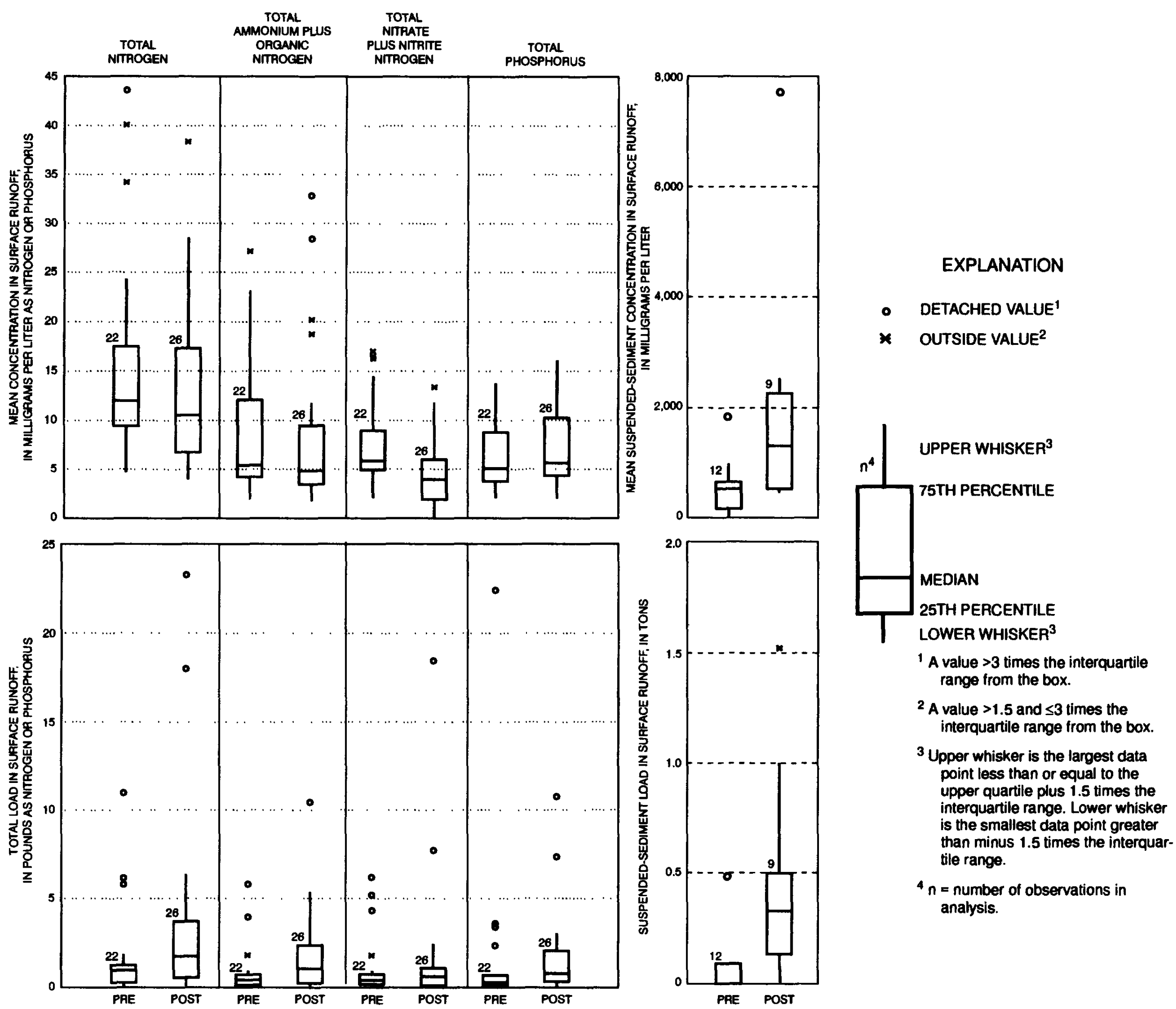

Figure 15. Mean concentrations from storms (top) and total storm loads (bottom) of nitrogen, phosphorus, and suspended sediment in surface runoff for the pre-Best-Management Practice (October 1984 through September 1986) and the post-Best-Management Practice (October 1986 through September 1988) study periods at Field-Site 2.

Median storm post-BMP loads of total nitrogen and total phosphorus were larger than the pre-BMP loads; however, no significant change was detected in the nitrogen and phosphorus loads for a storm of a given total runoff (fig. 16) (table 19). During the post-BMP period, more nitrogen, phosphorus, and sediment were discharged from the 27 terraced acres in 1988 because the amount of runoff was greater than during the pre-BMP period (table 11). The percentage of runoff was much greater in 1988 when conventional tillage was used on part of the site and no cover crop was planted. In this instance, tillage changes masked possible reductions in loads resulting from reduced nutrient applications in the post-BMP period.

In addition to the increase in nutrient loads during the post-BMP period, median suspended-sediment concentrations and loads (fig. 15) for sampled storms also increased significantly from the pre-BMP to post-BMP period (Mann-Whitney test). Again, the lack of winter cover crop and the change in tillage methods probably resulted in more soil being available for transport in runoff. During the entire 4-year monitoring period, suspended sediment was generally discharged as silt and clay with very little sand. All except 1 of the 14 samples analyzed for the silt/clay fraction contained greater than 93 percent silt and clay.

Runoff per unit of precipitation was greater during frozen-ground conditions than during thawed-soil conditions (fig. 7). Surface-applied nutrients on frozen ground were readily available for transport with runoff. In the pre-BMP period, runoff on frozen ground within 30 days after surface nutrient applications accounted for 76 percent of the total-nitrogen load in runoff and 75 percent of the total-phosphorus load in runoff. These loads were transported by 68 percent of the runoff (Koerkle and others, 1996). In the post-BMP period, runoff on frozen ground accounted for 35 percent of the total-nitrogen load in runoff and 40 percent of the total-phosphorus load in runoff. These loads were transported by 34 percent of the total post-BMP runoff. Differences between the pre-BMP and post-BMP period loads were probably caused by differences in the timing of precipitation rather than changes in BMP's on site conditions. Because large total-nutrient loads can be carried in runoff from storms during frozen-ground conditions, it is important that nutrients are not applied to frozen ground.

During the post-BMP period, nutrient loads in runoff from large summer storms (greater than 1.0 in. of precipitation) also contributed substantially to the total load. One storm in May 1988, which produced 2.65 in. of rain, discharged 22 percent of the total post-BMP nitrogen load and 14 percent of the total-phosphorus load. Two storms in July 1988 discharged another 10 percent of the total-nitrogen load and 8 percent of the total-phosphorus load.

Evaluation of effects of nutrient management on runoff water quality can be simplified if a quantitative relation between water quality and nutrient applications can be demonstrated. For runoff data from Field-Site 2 for the pre-BMP period, graphical and regression methods were used to identify factors that explained the variation in the mean-storm nutrient concentrations. Variables considered included precipitation quantity and duration, 5- and 7-day antecedent precipitation, nutrient applications, crop cover, and days since last nutrient application. These variables were used in multiple regression models to estimate a quantitative relation among mean-storm nutrient concentrations, climatic factors, and agricultural activities. Regressions were run on the complete data set and on the thawed- and the frozen-soil data subsets. Significant explanatory variables were not found for either the complete data set or the frozen-soil data subset. For the thawed-soil data subset, the relation between the number of days since the previous nutrient application and mean concentrations of total nitrogen and phosphorus in runoff was significant for the pre-BMP period. However, similar analysis for data from the post-BMP showed no significant relation between mean nutrient concentrations in runoff and days since the previous nutrient application. Hence, in general, regression techniques were not successful in identifying key relations between nutrient concentrations and the considered variables.

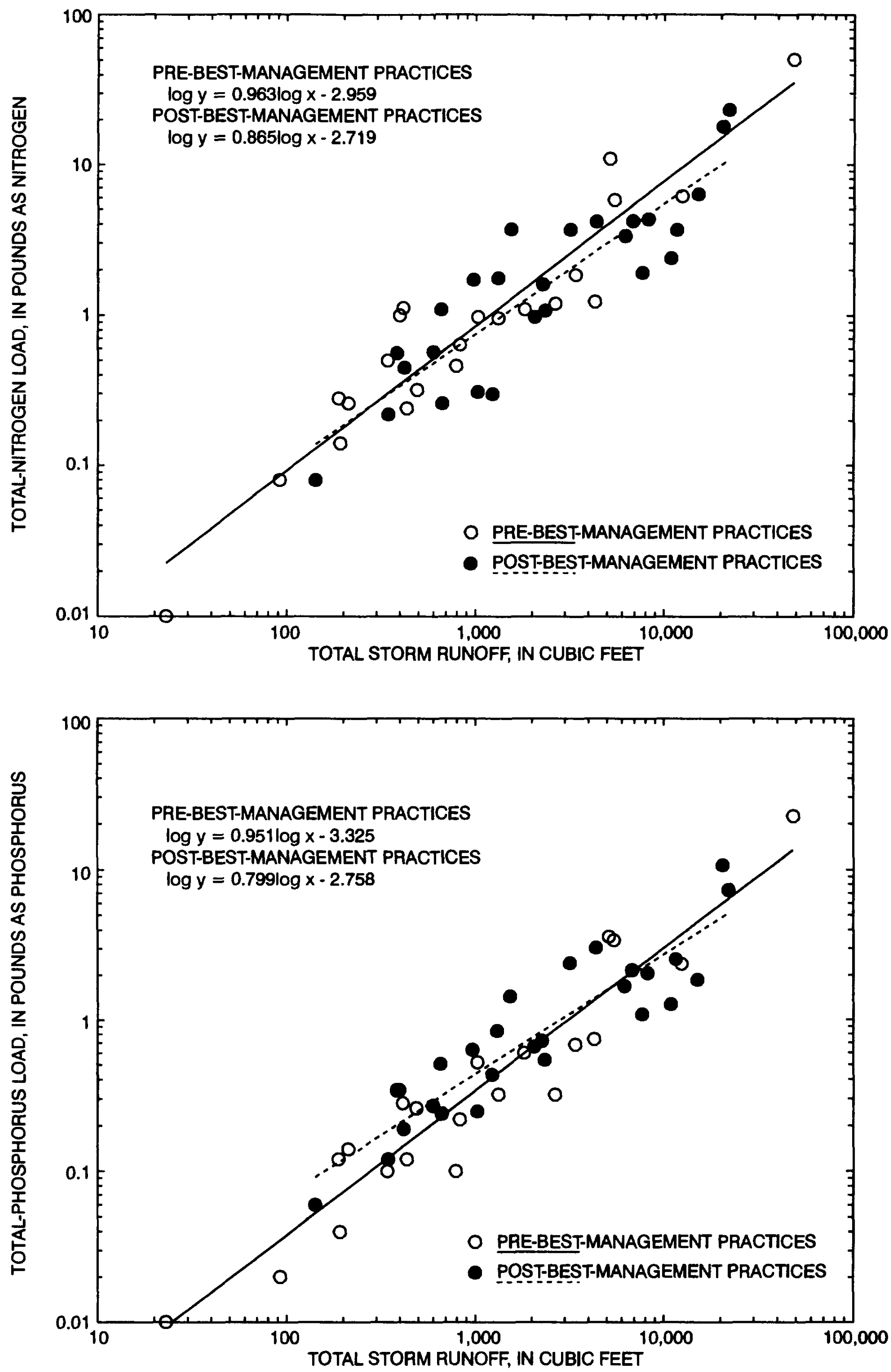

Figure 16. Total-nitrogen and total-phosphorus load as a function of runoff from storms at 27 terraced acres at Field-Site 2 for the pre-Best-Management Practice (October 1984 through September 1986) and the post-Best-Management Practice (October 1986 through September 1988) study periods. (Regression statistics are on table 19.)

Table 19. Regression statistics for the log of nutrient loads as a function of the log of total surface runoff at Field-Site 2

[pre-Best-Management Practice (pre-BMP) period is October 1984 through September 1986; post-Best-Management Practice (post-BMP) period is October 1986 through September 1990; nutrient loads are in pounds; surface runoff is in cubic feet; <, less than]

Dependent variable	Number of storms	Regression coefficient: Log of total surface 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
Pre-BMP total nitrogen load	22	0.963	12.248	<0.001	-2.959	0.89	0.260	82	45
Post-BMP total nitrogen load	26	.865	9.364	<.001	-2.719	.78	.281	91	48
Pre-BMP total phosphorus load	22	.951	12.651	<.001	-3.325	.88	.261	82	45
Post-BMP total phosphorus load	26	.799	10.785	<.001	-2.758	.82	.225	68	40

[1] Coefficient of determination (R^2) adjusted for degrees of freedom.

[2] Presented as described by G.D. Tasker, U.S. Geological Survey, written commun., 1978.

Ground Water

Ground-water samples were collected monthly from five wells from 1985 to 1990 and were analyzed to determine if reductions in the quantity of nitrogen loading to farm fields caused by the implementation of nutrient management in October 1986 reduced concentrations of nutrients in ground water at the site.

Median concentrations of dissolved phosphorus in water samples collected at each of the wells were less than 0.1 mg/L. Phosphorus is essentially unavailable for leaching to ground water because it rapidly sorbs to materials in the soils and unsaturated zone. Phosphorus concentrations in ground water were therefore, small, and phosphorus was determined to be a poor indicator of the effects of nutrient-management implementation on ground-water quality. Analyses for phosphorus concentrations in ground-water samples were discontinued in 1988 (table 4).

All ground-water samples were analyzed for dissolved nitrate plus nitrite, and most samples were analyzed for dissolved nitrite. Because 99.9 percent of the dissolved nitrate plus nitrite was nitrate, the analyzed concentrations of nitrate plus nitrite are referred to as dissolved nitrate for all ground-water samples. Dissolved nitrate accounted for over 90 percent of the total nitrogen in ground water. Because of the large percentage of total nitrogen that was nitrate and the relative stability and mobility of nitrate ions in the unsaturated and saturated zones, nitrate concentrations in ground water were selected as the most appropriate indicator to determine the effects of nutrient-management practices on nitrogen levels in ground water.

Concentrations of dissolved ammonium plus organic nitrogen and dissolved ammonium in ground-water samples collected during the study period were small, and these species were poor indicators of the effects of nutrient management on ground-water quality. Analyses for these species were discontinued in 1988 (table 4).

Median nitrate concentrations in ground-water samples from four wells ranged from 11 to 82 mg/L as N during the pre-BMP period (table 20). From the pre-BMP to the post-BMP periods, median nitrate concentrations decreased in water samples from three of the wells and increased in samples from one well. The largest decreases in nitrate concentrations occurred at wells where samples had the largest nitrate concentrations prior to nutrient management.

Table 20. Nitrate concentrations in wells before and after implementation of nutrient management at Field-Site 2

[pre-Best-Management Practice (pre-BMP) period is October 1984 through September 1986; post-Best-Management Practice (post-BMP) period is October 1986 through September 1990; mg/L, milligram per liter]

Well depth	Pre-BMP median nitrate concentration (mg/L as N)	Post-BMP median nitrate concentration (mg/L as N)	Wilcoxon-Mann-Whitney test, significant pre-BMP to post-BMP increase or decrease (p-value)
LN 1669	11	12	Significant increase (0.039)
LN 1673	53	37	Significant decrease (0.001)
LN 1676	82	56	Significant decrease (0.001)
LN 1677	26	23	Significant decrease (0.029)

Concentrations of nitrate in water samples from the five wells (fig. 17) were apparently unrelated to depth from the land surface to the water table. In water samples from well LN 1669, however, nitrate concentrations were lower than in water samples collected at the other wells. Because the lower 89 ft of this 100-ft well was an uncased, open hole in the fractured bedrock, it is possible that water in this well was a mixture of nitrate-rich shallow and nitrate-poor deeper waters.

Neither changes in quantity or timing of precipitation, associated ground-water recharge, nor changes in other climatic factors (within the variation that occurred during the study period) had predictable effects on concentrations of nitrate in ground water (fig. 18). Although the wells at the site were a maximum distance of only 1,150 ft apart and climatic influences (including precipitation quantity, quality, and intensity) were virtually identical across the site, concentrations of nitrate in ground-water samples from the five monitor wells demonstrated no climate-induced, long- or short-term trends or consistent seasonal variations. For example, a period of increased recharge to the aquifer upgradient of all five of the monitor wells occurred during the late spring and summer of 1989 (fig. 18). During this period of recharge, concentrations of nitrate in ground-water samples collected at wells LN 1673 and LN 1676 were increasing, concentrations in samples from wells LN 1677 and LN 1679 were decreasing, and concentrations in samples collected at well LN 1669 were remaining relatively constant (fig. 18).

There was no consistent response (dilution or enrichment) in concentrations of nitrate in ground water to monthly ground-water recharge (figs. 18 and 19). Evidence of the lack of any consistent response of concentrations of nitrate in ground water to recharge was provided by examination of precipitation, water level, and nitrate-concentration data collected from site wells during individual storms. For example, figure 20 shows the water-level and water-quality responses at well LN 1677 during storms that occurred in November 1985 and in July 1986. During each of the storms, about 4 in. of precipitation caused the water table to rise by about 2 ft, indicating that hydrologic conditions were roughly comparable. Changes in ground-water quality were not comparable during the two storms; concentrations of nitrate in ground-water samples increased by about 25 mg/L during the November 1985 storm and decreased by approximately 5 mg/L during the July 1986 storm. Similar data collected at this and other wells indicated that concentrations of nitrate in ground water were not consistently correlated in either a negative or positive relation to recharge.

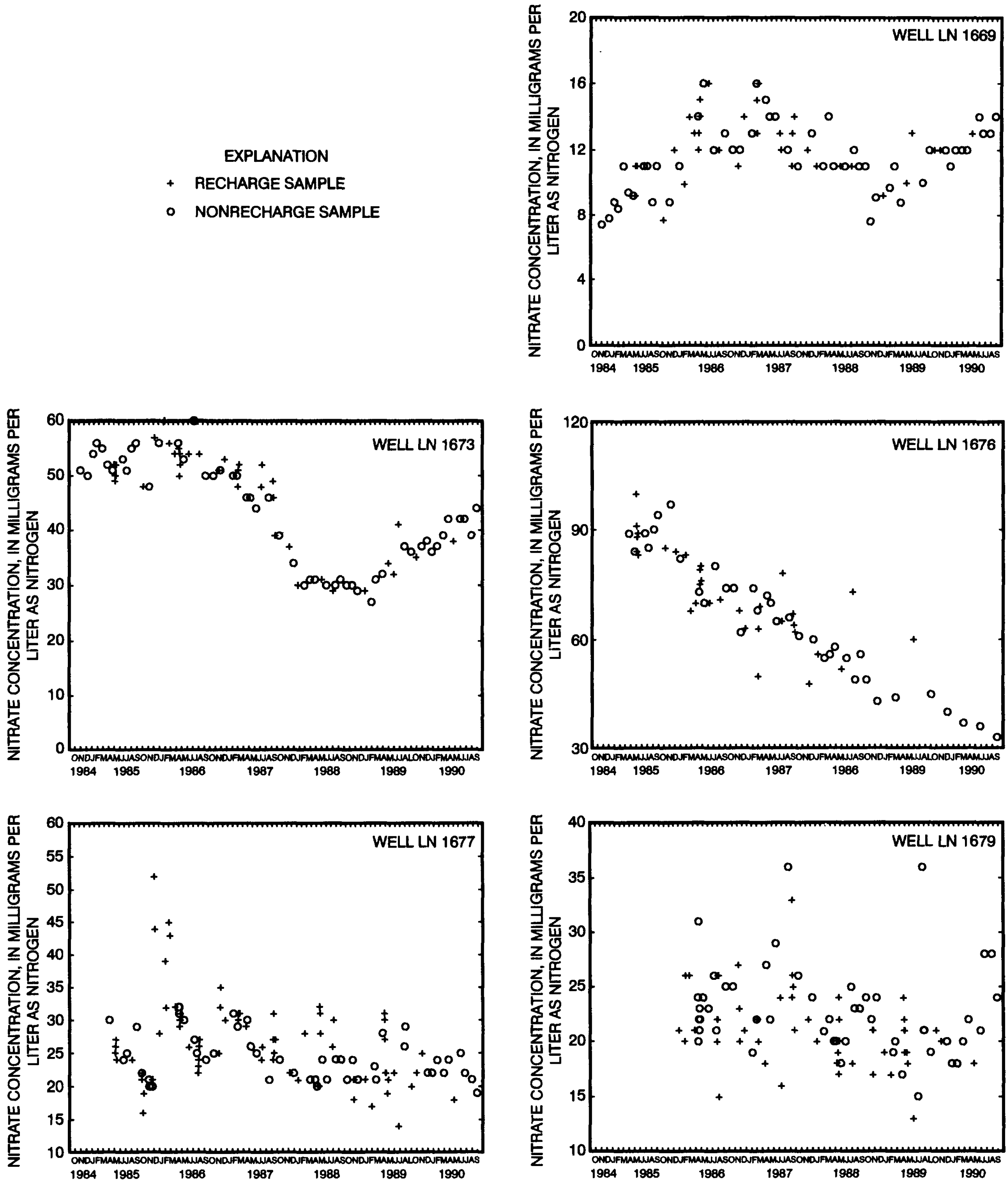

Figure 17. Concentrations of nitrate in recharge and nonrecharge ground-water samples collected at five wells at Field-Site 2.

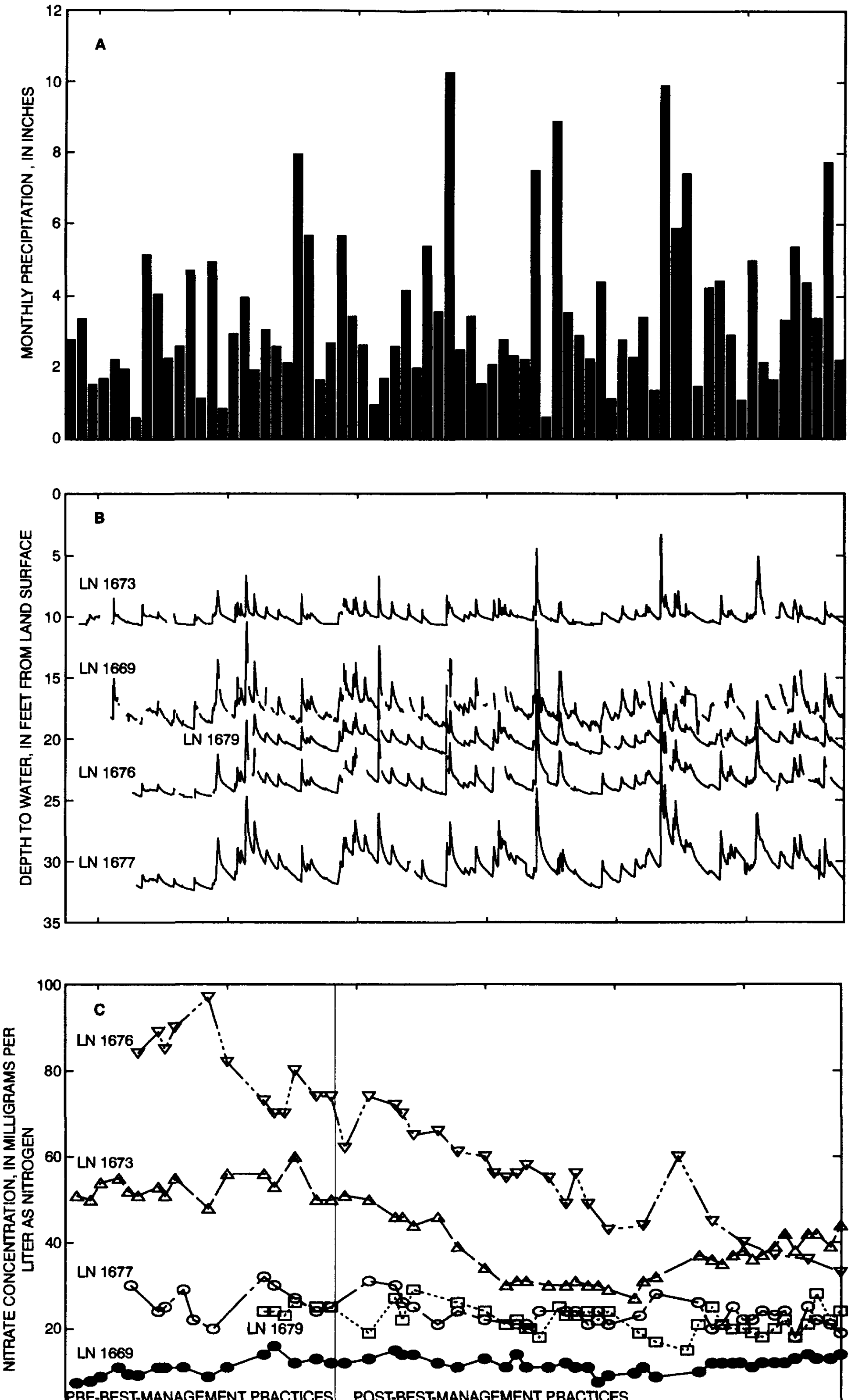

Figure 18. Monthly precipitation (A), depth to water from land surface (B), and concentrations of nitrate in nonrecharge ground-water samples (C) at Field-Site 2.

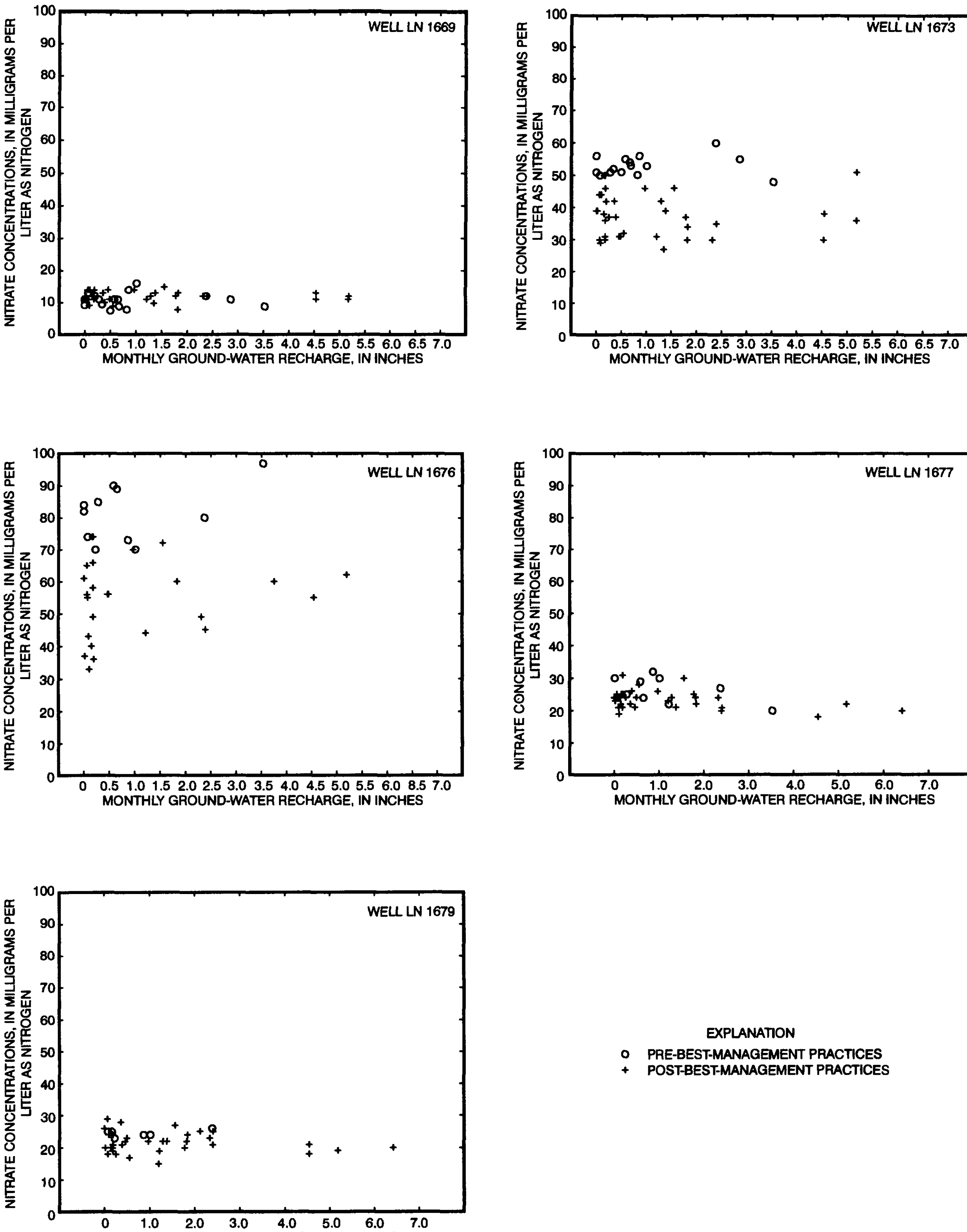

Figure 19. Relations between monthly ground-water recharge and monthly nitrogen concentrations of samples collected from wells LN 1669, LN 1673, LN 1676, LN 1677, and LN 1679 during the pre-Best-Management Practice and post-Best-Management Practice periods at Field-Site 2.

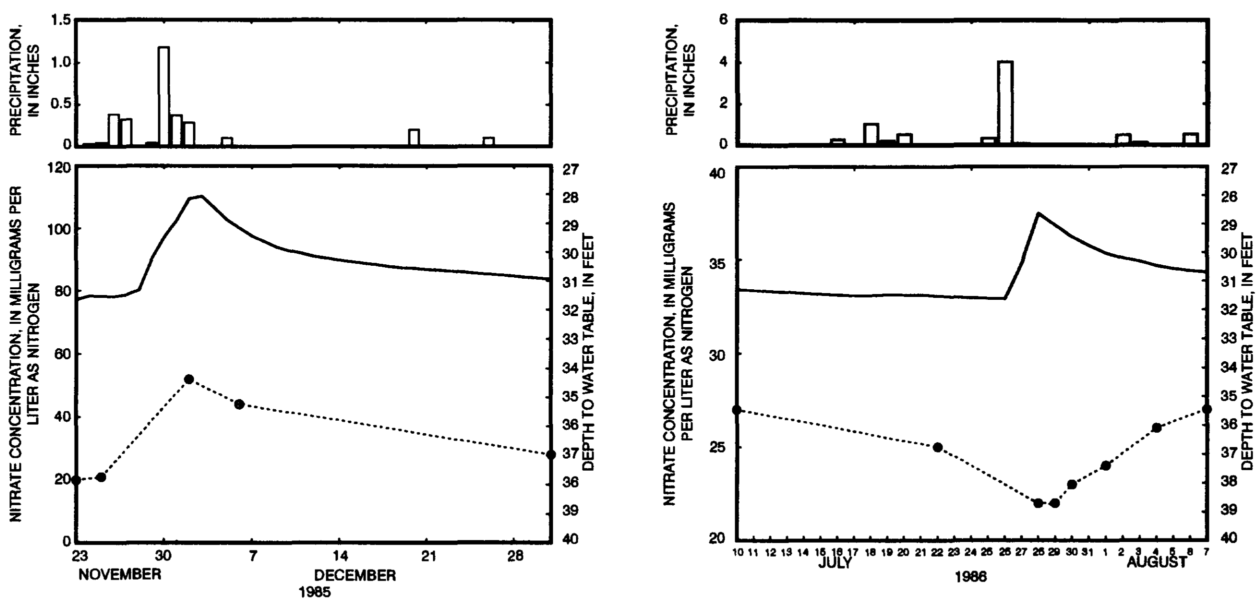

Figure 20. Precipitation, water levels, and nitrate concentrations in ground-water samples collected at Field-Site 2 from well LN 1677 during two storms in November 1985 and July 1986.

Correlation of Nitrogen Applications With Concentrations of Nitrate in Ground Water

Nitrogen applications made to the contributing area of a well and concentrations of nitrate in ground-water samples collected at that well were significantly correlated at each of the five site monitoring wells. The following procedures were used to test for time-lagged correlations:

1. Nitrogen loads applied to a well contributing area were plotted as a time series of points, with each point representing the sum of 4 months of nitrogen applications. Separate plots were constructed for each of five wells. Typically, one large and several small nitrogen applications were made to a well contributing area during a 4-month period, and the points were plotted on the day of the largest application made during the period. Although 2-, 4-, 6-, 8-, and 12-month summing of the application data were considered, the 4-month grouping was selected as the best time unit by which to portray applied nitrogen as a source of nitrate for leaching to ground water. Many 2-month summations of applications had zero values and were considered to be a poor data series to represent soluble nitrate at the land surface. Tests of soluble nitrate in soils indicate that soil nitrate levels rarely approach zero (fig. 14). The 6-, 8-, and 12-month groupings produced relatively flat curves that were considered poor data series to represent soluble nitrate available for leaching at the land surface. It is likely that most nitrogen applied to farm fields has leached to some depth in the soils before subsequent applications are made 6, 8, or 12 months later.
2. The 4-month nitrogen-application points were then connected by use of a curve-smoothing procedure (poly3) on Tellagraf (1984) software. The position of the curve connecting the nitrogen-application points was intended to provide a gross approximation of the amount of soluble nitrate available for leaching at the land surface as a function of the applied nitrogen.
3. Concentrations of nitrate in monthly ground-water samples were then plotted as a time series of points and connected using a simple line for data from each of the five wells.
4. A series of monthly application values and a separate series of monthly nitrate concentration values were constructed for each of the five wells by discretizing the curves into monthly increments.
5. On the assumption that nitrogen applied to farm fields could potentially take months or years to leach to ground water, a parametric cross-correlation procedure (Wilkinson, 1987) was used to identify significant time-lagged relations between the two series. No nonparametric cross-correlation software was available.
6. Significant correlations identified in the cross-correlation procedure were subsequently described using rho and p-values from the nonparametric Spearman Rank Sum test (P-STAT, Inc., 1986) because some ground-water nitrate data were nonnormally distributed.

Final results of the cross-correlation procedure are illustrated on figures 21 and 22. Figure 21 illustrates the nitrogen-application curve paired with the ground-water nitrogen-concentration line for each well. To illustrate correlations, the nitrogen-application series has been lagged (moved to the right) to match the resultant nitrate-concentration series. To view real-time relations, move the application curves to the left by the indicated time lag for each well. The relation between applied nitrogen and resultant concentrations of nitrogen in ground water is shown in figure 22. The rho values of 0.215 to 0.684 associated with correlations of data from the five wells are large given the climatic and other variability in the system, and all correlations were significant at the 90-percent confidence level as indicated by the p-values of 0.001 to 0.067.

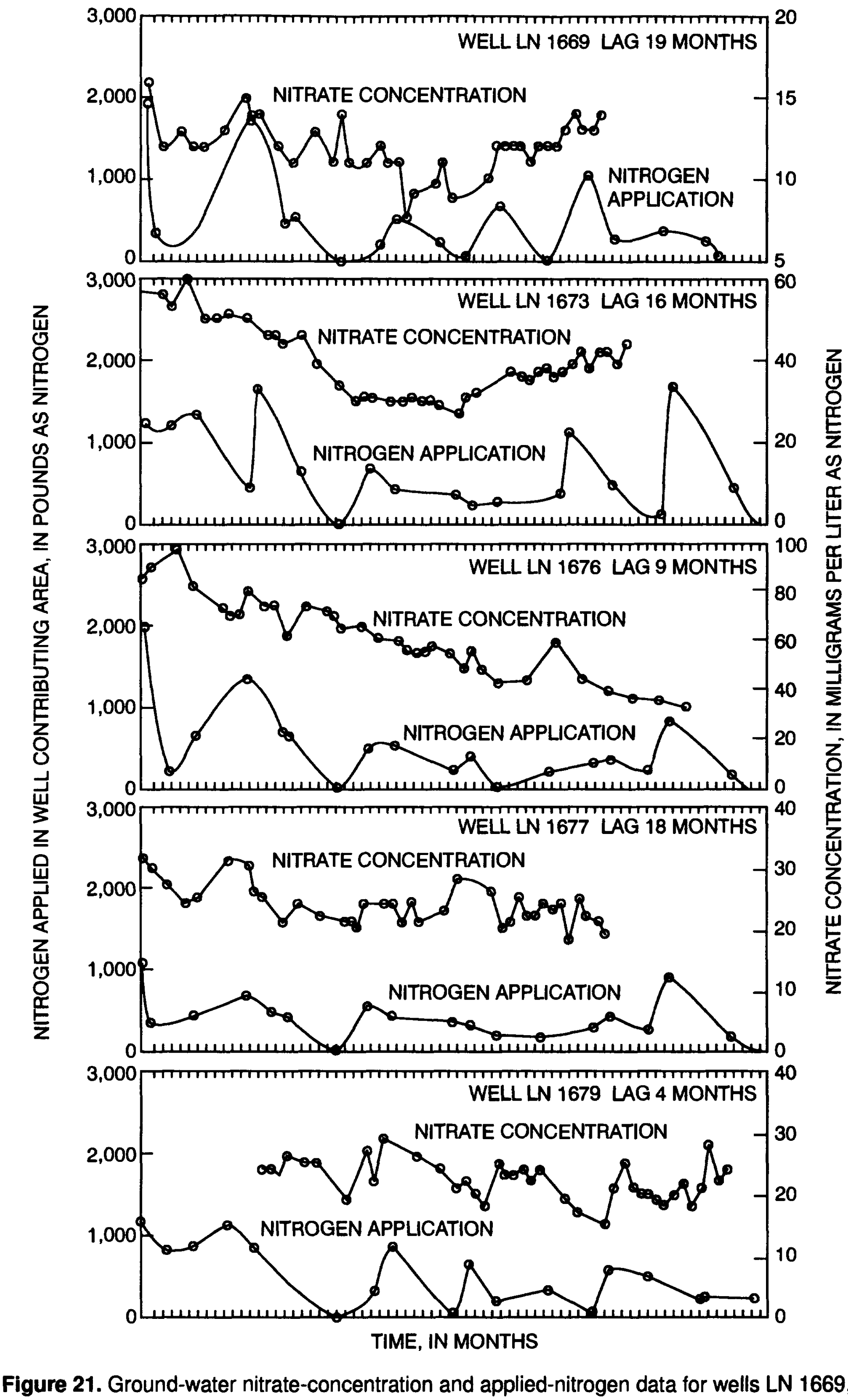

Figure 21. Ground-water nitrate-concentration and applied-nitrogen data for wells LN 1669, LN 1673, LN 1676, LN 1677, and LN 1679 at Field-Site 2. (To illustrate correlations, nitrogen applications have been moved to the right to match resultant ground-water nitrate concentrations; to get real-time relations, move the application curves to the left by the indicated time lag for each well.)

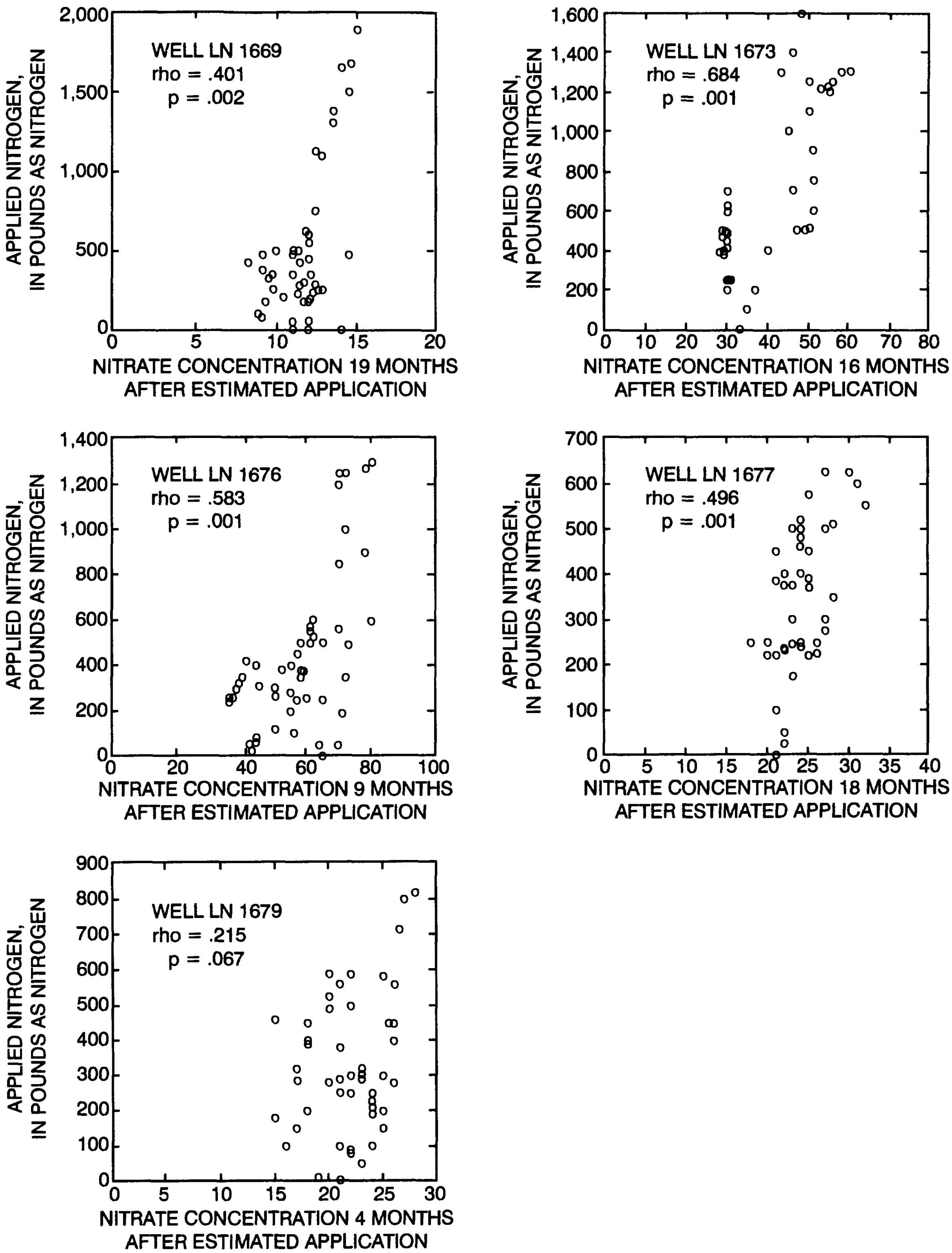

Figure 22. Relation between ground-water nitrate concentrations and applied nitrogen and results of Spearman Rank correlations from wells LN 1669, LN 1673, LN 1676, LN 1677, and LN 1679 at Field-Site 2.

In most cases, correlations between applied nitrogen and concentrations of nitrate in ground water were significant for several consecutive months. In each case, only the time lag with the most significant correlation (largest rho value) was reported (fig. 22). Results of the cross-correlation analyses suggest that, during the study, wetting fronts of nitrate-rich or nitrate-depleted water leached to ground water diffusely over 1- to 4-month periods.

The statistically-significant correlations indicate that different amounts of time, from roughly 4 months in the area upgradient of well LN 1679 to about 19 months in the area upgradient of well LN 1669, were required for surface-applied nitrogen to leach to the ground water upgradient of each of the wells. Long-term changes of nitrate concentrations in ground water were probably produced by the slow movement of recharge waters through the soil micropores under the influence of the pore-pressure gradient (Shuford and others, 1977; White, 1985). The time required for a nitrogen-rich or nitrogen-depleted "front" of water to travel from the land surface to the water table may vary in each contributing area as a function of decay of organic nitrogen to nitrate, quantities and rates of infiltration, unsaturated-zone hydraulic conductivity, unsaturated-zone clay content, the distance between the land surface and the water table, and flow through the carbonate bedrock. Solute transport may take longer during periods of drought and may occur faster during periods of above-normal precipitation because of precipitation-induced effects on the timing and quantity of recharge movement to ground water.

Correlation alone does not prove that a cause-effect relation exists between two data sets (Iman and Conover, 1983; Helsel and Hirsch, 1992). Four conditions must be satisfied to determine that a statistically-significant correlation results from a cause-effect relation (Mosteller and Tukey, 1977): an association, consistency, responsiveness, and a mechanism. The first condition, an association between changes in amounts and timing of applied nitrogen and changes in magnitude and timing of ground-water nitrate concentrations, was met, inasmuch as statistically-significant relations were demonstrable in the data collected at each well (figs. 21 and 22). The second condition, consistency, was satisfied by the fact that the correlation is significant not only for data from one well but for data from each of the five wells tested. Although data from each of the wells showed some responsiveness that satisfied the third condition, responsiveness was very pronounced in data collected at wells LN 1669 and LN 1673, wells where nitrogen applications followed a large-small-large pattern during the study period. Concentrations of nitrate in ground-water samples mirrored the changes in nitrogen applications and demonstrated the same large-small-large pattern at lag times of approximately 16 months for well LN 1673 and approximately 19 months at well LN 1669. Different degrees of responsiveness at the different wells could be caused by variable hydrodynamic dispersion along different flow paths in heterogenous terrains (Boggs and others, 1992).

The mechanism suggested to link the applied nitrogen and the nitrate concentrations in ground water, in order to satisfy the fourth condition needed to prove a cause and effect relation, was transport of the nitrogen through the micropore system of the unsaturated zone and through saturated fractured bedrock. Therefore, satisfaction of the four cause-effect conditions demonstrated that changes in rates of applied nitrogen caused changes in concentrations of nitrate in ground water at the site.

INPUTS AND OUTPUTS OF NITROGEN

Ideally, all sources of nitrogen input to, and output from, a site could be accurately quantified, and all mechanisms of nitrogen transport could be described (Stevenson, 1982, chapter 1); however, quantification of some terms is difficult, and all mechanisms are not well understood. An equation describing all nitrogen transport through a farm field could be (inputs = outputs plus or minus changes in storage):

Inputs of nitrogen:
Manure + commercial fertilizer + atmospheric deposition (wet, dry, and gaseous) +
ground-water inflow + surface-water inflow + any other inputs,
equals,

Outputs of nitrogen:
Crops harvested or grazed + surface-water outflow + volatilization + denitrification +
ground-water outflow + any other outputs,
+ or - soil storage of nitrogen.

For each year of the study period, inputs of nitrogen to, and outputs of nitrogen from, the site were calculated to the extent possible (table 21). Inputs of nitrogen were manure nitrogen (93 percent of average-annual inputs), commercial fertilizer nitrogen (4 percent of average-annual inputs), nitrogen in precipitation (2 percent of average-annual inputs), and nitrogen in ground water entering the site across the western boundary (1 percent of average-annual inputs). Inputs such as dry deposition from the atmosphere (Baker, 1991) and bacterial fixation of nitrogen in the soil (Hauck and Tanji, 1982) were probably a small percentage of the nitrogen budget and, therefore, were omitted.

Table 21. Estimated inputs and outputs of nitrogen at Field-Site 2, water years 1985-90
[Numbers are in pounds as nitrogen; <, less than]

Water year	Estimated nitrogen inputs to site				Estimated nitrogen outputs from site			
	Nitrogen in manure fertilizer	Nitrogen in commercial fertilizer	Nitrogen in precipitation	Nitrogen in ground-water inflow	Nitrogen consumed by crops	Nitrogen in surface-water outflow	Nitorgen volatilized	Nitrogen in ground-water outflow
[1]1985	25,500	1,000	290	90	8,500	120	7,300	5,100
[1]1986	[2]18,500	500	310	180	8,700	50	4,600	10,800
1987	11,500	0	400	190	6,900	20	4,400	10,500
1988	13,500	0	330	170	6,500	160	4,700	7,100
1989	17,200	1,600	340	190	7,400	[3]90	5,500	7,700
1990	16,700	1,700	320	160	8,100	[3]90	4,600	6,400
Average 1985-90	17,150	800	330	160	7,700	90	5,180	7,930
Percent of average annual inputs or outputs	93	4	2	1	37	<1	25	38

[1] 1985 and 1986 were during the pre-Best-Management Practices period.

[2] 1986 manure applications were unusually small because of less manure disposal as the result of an outbreak of avian influenza in Lancaster County, which significantly reduced the number of chickens present on the farm.

[3] Estimated as average of surface-runoff loads calculated for years 1985-88.

Nitrogen was removed in harvested crops (38 percent of average-annual outputs), in ground-water discharge (38 percent of average-annual outputs), in volatilization gases (24 percent of average-annual outputs), and in surface runoff (less than 1 percent of average-annual outputs). No data were collected to estimate the loads of nitrogen that were denitrified from the site. Concentrations of oxygen in ground water were commonly near saturation, indicating that anaerobic denitrification of nitrogen from shallow ground water was not a steady-state process. However, intermittent denitrification may have occurred, especially in soils and weathered rock during saturated conditions and at depth below the water table (Korom, 1992).

Annual inputs of nitrogen to the site do not balance annual outputs on table 21. From 10 to 60 percent of the nitrogen applied to site fields may not oxidize to soluble nitrate within the year of application (Stevenson, 1982). This nitrogen may remain temporarily stored in site soils as organic nitrogen or ammonium, and additional time may be required for soluble nitrate to infiltrate from the land surface to ground water.

Potential errors because of assumptions made in the calculation of loads may greatly influence the numbers reported in table 21. A brief description of methods of calculation and errors associated with each nitrogen input and output term follows.

Inputs of Nitrogen:

Manure Nitrogen: The major input of nitrogen to the site was manure from cattle, swine, and poultry operations. Loads of nitrogen in manure were estimated by the use of application data supplied by the farmer and laboratory analysis of manure samples collected at the site during the study period. Applications of manure to farm fields decreased significantly after the implementation of nutrient management in October 1986 (fig. 10). Despite moderate increases in applications in 1989-90 relative to 1987-88, post-BMP applications of manure nitrogen never approached levels applied in 1985, a typical pre-nutrient management year. Although 1986 is a pre-BMP study year, 1986 manure applications were unusually small because of less manure disposal as the result of an outbreak of avian influenza that significantly reduced poultry numbers.

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. 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.

Commercial Fertilizer Nitrogen: Applications of commercial fertilizer to the site were small relative to the large amounts of manure nitrogen applied. Most of the commercial fertilizer used at the site was either starter fertilizer applied at the time of planting or a sidedress of nitrogen to crops made early in the crop-growing season. Reported applications of commercial fertilizer nitrogen were probably accurate because information about the nitrogen content and quantity of the fertilizer is typically available when the fertilizer is purchased.

Precipitation Nitrogen: Nitrogen in precipitation is in the form of ammonia and nitrate ions. Loads of nitrogen entering the site in precipitation were calculated by the use of volumes of precipitation estimated from site rain-gage data multiplied by concentrations of the nitrogen content of ammonia and nitrate in precipitation samples in southeastern region of Pennsylvania reported by Lynch, Corbett, and Grim (1988), Lynch, Corbett, and Kostelnik (1986, 1987), and Lynch, Grimm, and Corbett (1989, 1990).

Errors in the measurement of precipitation were small. A much greater error in the budget was possible from the use of the regional ammonia and nitrate concentrations used for the calculation of loads of nitrogen in precipitation. Precipitation in farm areas with large animal operations are sites of the active volatilization of manure nitrogen (Langland, 1992). Because the farm has a manure-storage facility and was the site of a concentrated animal population, the site could have had an elevated precipitation-nitrogen load relative to those calculated by use of the regional ammonia and nitrate concentration estimates.

Nitrogen in Ground-Water Inflow: 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 in March 1985 and April 1988 from well LN 1674 on the western site boundary.

Although there was 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 of the relatively small quantity of water that was estimated to enter the site across the western boundary.

Outputs of Nitrogen:

Nitrogen in Harvested Crops: Nitrogen removed from the site in harvested corn, tobacco, rye, and Sudan grass was calculated by use of the yield-based estimates of nutrient consumption by crops (Robert Anderson, Pennsylvania State University Cooperative Extension, written commun., 1989). Nitrogen removed from the site in fruits and vegetables was estimated on the basis of information in *Knott's Handbook for Vegetable Growers* (Knott, 1962).

Crop yields did not change significantly when manure-nitrogen fertilizer applications were reduced under nutrient management (Leon Ressler, Penn State Cooperative Extension, written commun., 1987-90). This term is subject to errors in estimates of crop nitrogen content and errors in estimates of crop yields.

Volatilization of Nitrogen: Volatilization of nitrogen from the site was estimated as a percentage of nitrogen applied. 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 information in the Pennsylvania Department of Environmental Resources *Field Application of Manure* manual (Graves, 1986a). Commercial fertilizer was estimated to volatilize at a rate of 15 percent of applications, an estimate made on the basis of information in Pionke and Urban (1985).

Exact quantification of the volatilization of nitrogen from manure and commercial fertilizer nitrogen was not possible. The rate of nitrogen volatilization is affected by air temperature, humidity, manure type, manure texture and moisture content, timing of incorporation into soil, soil pH, and any factor influencing bacterial activity (Stevenson, 1982) associated with volatilization. The percentage of nitrogen volatilized at the site could therefore have been expected to vary greatly during short time periods. 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.

Nitrogen in Runoff: Loads of nitrogen discharged from the site in surface runoff were calculated by the use of discharge-weighted mean nitrogen concentrations in surface runoff 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.

Nitrogen in Ground-Water Outflow: Nitrogen in ground-water outflow averaged 7,930 lb per year from 1985 to 1990 (table 21), which was about 38 percent of the load of total nitrogen removed from the site. Virtually all of the nitrogen leaving the site that was not consumed by crops or lost to the atmosphere by volatilization was transported off the site in ground water.

SUMMARY AND CONCLUSIONS

The U.S. Geological Survey and the Pennsylvania Department of Environmental Protection conducted a study from 1984 to 1990 to determine the effects of the implementation and practice of nutrient management [an agricultural best-management practice (BMP)] on surface- and ground-water quality at Field-Site 2, a 55-acre crop and livestock farm in carbonate terrain near Ephrata, Pa.

Surface runoff that was pipe discharge from 27 terraced acres of the site was monitored for 2 years prior to BMP implementation (October 1984 through September 1986) and for 2 years after BMP implementation (October 1986 through September 1988). Total-annual runoff during the 4-year period ranged from 28,800 to 130,000 ft^3 and represented 0.7 to 3.3 percent of the annual precipitation. The amount of runoff produced by a given amount of precipitation was quite sensitive to frozen-soil conditions. Regression equations describing the relation between precipitation and runoff indicated that increases in total runoff with increasing precipitation were about 400 times greater on frozen soil than on thawed soil. Thus, one large storm on frozen soil could produce a substantial part of the total annual runoff.

Overall, runoff during conditions of thawed soil increased significantly from a median of 0.66 percent of the precipitation in the pre-BMP period to 1.9 percent in the post-BMP period. Because the distribution of precipitation quantities and intensities did not change significantly from the pre-BMP to the post-BMP period, the change in tillage practices from no-till with continuous cover to more conventional tillage with a fallow period was the likely cause of the increased runoff.

Although the median post-BMP loads of total nitrogen and total phosphorus in surface runoff were larger than the pre-BMP loads, there was no statistically significant change in the nitrogen and phosphorus loads for a storm of a given total discharge. For the period, annual loads of total nitrogen in runoff ranged from 14 to 93 lb, of which about half was ammonium plus organic nitrogen, and half was nitrate plus nitrite nitrogen. Annual loads of total phosphorus ranged from 8.4 to 44 lb. The annual nutrient loads from runoff were less than 2 percent of the nutrients applied to the 27 terraced acres. Median loads of total nitrogen and total phosphorus in surface runoff increased in the post-BMP period from 0.97 to 1.8 lb and 0.30 to 0.78 lb, respectively.

Mean surface runoff concentrations of nitrate plus nitrite decreased significantly from the pre-BMP to the post-BMP periods. Ammonium plus organic nitrogen and total phosphorus concentrations did not change significantly. The decrease in median concentrations of nitrate plus nitrite from 5.9 to 4.0 mg/L may have been caused by the change in tillage practices and crop cover, which may have decreased the contact time between surface runoff and the nutrient-rich soils. Reductions in the median concentrations of nitrate plus nitrite were not large enough to significantly change the total nitrogen concentrations in runoff. Runoff constituted a relatively small percentage (average less than 1 percent) of total nitrogen export from the site. Therefore, under normal climatic conditions, changes in runoff nitrogen would result in only insignificant total nitrogen export from the site.

Runoff per unit precipitation was much greater during frozen-soil conditions than during thawed-soil conditions. Surface-applied nutrients on frozen ground were readily available for transport in runoff. In the pre-BMP period, runoff from frozen ground within 30 days after surface applications of manure accounted for 76 percent of the annual total-nitrogen load and 75 percent of the annual phosphorus load. These loads were transported by 68 percent of the pre-BMP runoff. During the post-BMP period, runoff from frozen ground accounted for 35 percent of the annual total-nitrogen load and 40 percent of the total-phosphorus load. These loads were transported by 34 percent of the total post-BMP runoff.

Graphical and regression methods were used to identify variables to explain variations in mean-surface nutrient concentrations. Variables considered included precipitation quantity and duration, 5- and 7-day antecedent precipitation, nutrient applications, crop cover, and days since last nutrient application. These variables were entered into multiple regression models to estimate a quantitative relation among the mean-surface nutrient concentrations, climatic factors, and agricultural activities. Significant explanatory variables were not found for either a complete data set or a frozen-ground subset. Therefore, no method was developed to quantitatively determine the effects of BMP's on runoff-water quality.

Statistically significant correlations exist between the timing and amount of nitrogen applied to farm fields upgradient of five monitoring wells and changes in nitrate concentrations of ground-water samples collected at those wells. Different amounts of time, from roughly 4 months in the area upgradient of well LN 1679 to about 19 months in the area upgradient of well LN 1669 were required for significant quantities of applied nitrogen to leach to the water table upgradient of each of the five wells used for ground-water monitoring. The statistically significant correlations between the timing and amount of applied nitrogen and changes in ground-water quality met the conditions of an association, consistency, responsiveness, and a mechanism that are characteristic of a cause-effect relation.

Quantities of nitrogen input to, and output from, farm fields were estimated to evaluate nitrogen transport through the site. Inputs of nitrogen to the site were manure fertilizer, commercial fertilizer, nitrogen in precipitation, and nitrogen in ground-water inflow, which averaged 93, 4, 2, and 1 percent of average annual nitrogen additions, respectively. Outputs of nitrogen from the site were nitrogen in harvested crops, loads of nitrogen in surface runoff, volatilization of nitrogen, and loads of nitrogen in ground-water outflow, which averaged 37, less than 1, 25, and 38 percent of average annual nitrogen removals from the site, respectively. Applications of manure and commercial fertilizer nitrogen to cropped fields decreased about 33 percent, from an average of 22,700 lb/yr before nutrient management to 15,175 lb/yr of nitrogen after the implementation of nutrient-management practices. Virtually all of the nitrogen leaving the site that was not removed in harvested crops or by volatilization was discharged in the ground water.

Implementation of nutrient management at Field-Site 2 resulted in application decreases of 33 percent for nitrogen and 29 percent for phosphorus. There were no significant changes in nitrogen or phosphorus loads for a given amount of runoff from the pre-BMP to the post-BMP periods. However, less than 2 percent of the applied nutrients were discharged with runoff throughout the study period. After the implementation of nutrient management, statistically significant decreases in concentrations of nitrate in ground-water samples occurred at three of the four wells monitored throughout the entire pre- and post-BMP periods. The largest decreases in nitrate concentrations occurred at wells where samples had the largest nitrate concentrations prior to nutrient management. Changes in nitrogen applications to the contributing areas of five wells were correlated with nitrate concentrations of the well water. Changes in ground-water nitrate concentrations lagged behind changes in loading of nitrogen fertilizers (primarily manure) by approximately 4 to 19 months.

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