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EFFECTS OF AGRICULTURAL NUTRIENT MANAGEMENT ON NITROGEN FATE AND TRANSPORT IN LANCASTER COUNTY, PENNSYLVANIA[1]

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ABSTRACT: Nitrogen inputs to, and outputs from, a 55-acre site in Lancaster County, Pennsylvania, were estimated to determine the pathways and relative magnitude of loads of nitrogen entering and leaving the site, and to compare the loads of nitrogen before and after the implementation of nutrient management.

Inputs of nitrogen to the site were manure fertilizer, commercial fertilizer, nitrogen in precipitation, and nitrogen in ground-water inflow; and these sources 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 discharge, which averaged 37, less than 1, 25, and 38 percent of average annual nitrogen removals from the site, respectively. Virtually all of the nitrogen leaving the site that was not removed in harvested crops or by volatilization was discharged in the ground water.

Applications of manure and fertilizer nitrogen to 47.5 acres of cropped fields decreased about 33 percent, from an average of 22,700 pounds per year (480 pounds per acre per year) before nutrient management to 15,175 pounds of nitrogen per year (320 pounds per acre per year) after the implementation of nutrient management practices. Nitrogen loads in ground-water discharged from the site decreased about 30 percent, from an average of 292 pounds of nitrogen per million gallons of ground water before nutrient management to an average of 203 pounds of nitrogen per million gallons as a result of the decreased manure and commercial fertilizer applications. Reductions in manure and commercial fertilizer applications caused a reduction of approximately 11,000 pounds (3,760 pounds per year; 70 pounds per acre per year) in the load of nitrogen discharged in ground water from the 55-acre site during the three-year period 1987-1990.

(KEY TERMS: agricultural hydrology; ground water hydrology; nonpoint source pollution; nitrogen; best management practice; nutrient management; Pennsylvania; water quality.)

INTRODUCTION

Intensification of agricultural activities on Pennsylvania farms during the past three decades has beneficially increased production of farm products, but has also contributed to nonpoint source impairment of water resources. The growth of agricultural production in southeastern Pennsylvania is reflected in the numbers of farm animals raised in Lancaster County, Pennsylvania. Anderson (1992) used data from the Pennsylvania Crop Reporting Service (1960; 1970, 1980) and the Pennsylvania Agricultural Statistics Service (1990) to compare numbers of farm animals reported in Lancaster County, Pennsylvania, from 1960 to 1989. During the 29-year period, numbers of dairy cattle increased 52 percent, hogs increased 714 percent, poultry layers and pullets increased 190 percent, and broilers increased 640 percent. At the time that animal densities were increasing, levels of nutrients and herbicides became elevated in Pennsylvania soils, and surface and ground water (Mooney, 1984; Pionke and Urban, 1984; Fishel and Lietman, 1986; Pionke and Glotfelty, 1989; Roth and Fox, 1990; Lietman and Hall, 1991).

Disposal of large quantities of animal manure on land may lead to elevated concentrations of nitrogen in surface and ground water (Gilham and Webber, 1969; McCalla *et al.*, 1970; Pionke and Urban, 1985; Crowder and Young, 1988). Elevated concentrations of nitrogen 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 1945-1972 (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

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nitrate in drinking water in Australia have been associated with increased human birth defects (Dyer *et al.*, 1984). Nitrogen-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 *et al.*, 1984; Fisher, 1989).

The objective of this paper is to determine the fate and transport of nitrogen at fields of a 55-acre farm near Ephrata, Pennsylvania, and to determine if loads of nitrogen discharged from the farm in ground water decreased as the result of the implementation of nutrient management. The study was conducted from 1985-1990, and was part of the Conestoga Headwaters Rural Clean Water Program Project (Chichester, 1988; Little, 1989), and was conducted in cooperation with the Pennsylvania Department of Environmental Resources under direction of the U.S. Department of Agriculture.

Nutrient management is an agricultural Best-Management Practice (BMP) whereby quantities of commercial fertilizer and manure applied to cropped land are limited to quantities that meet crop nutrient requirements, so that leaching of nutrients to surface and ground water is minimized (Graves, 1986a, 1986b; Pennsylvania State University, 1989; Lanyon and Beegle, 1989; Cronce *et al.*, 1990; Bacon *et al.*, 1990). A nutrient management plan was designed by the Pennsylvania State University Cooperative Extension Service for a 55-acre crop and animal production farm near Ephrata, Pennsylvania. The nutrient management plan was developed from crop-yield goals, based on soil type, methods of manure of commercial fertilizer application, nitrogen concentration in soils, nitrogen concentrations in manure samples collected at the site, and crop rotations. Nutrient management was implemented at the farm in October 1986, and remained in effect through the remaining four years of the study period. After October 1986, the farmer exported all animal manure produced at the site in excess of the amounts specified by the nutrient management plan.

One technique to evaluate the effectiveness of a nutrient management plan on controlling nitrogen transport is to estimate quantities of nitrogen input to, or output from, farm fields. Difficulties encountered in the estimation of nitrogen inputs and outputs from basins and study areas are well documented (Kohl *et al.*, 1978; Miller and Wolfe, 1978; Pionke and Urban, 1985; Viets, 1978). Large errors in estimates can overwhelm detection of fluxes of nitrogen through the system, and may make a complete accounting, or nitrogen balance, impossible to estimate. However, even relatively incomplete estimates of inputs and outputs may be used to draw significant conclusions (Kohl *et al.*, 1978). Reduced amounts of nitrogen applied to cropped fields as part of nutrient management, and associated reductions in loads of nitrogen discharged in ground water, are the respective input and output of interest in this paper. Water years discussed in this article are defined as the one-year period that begins on October 1 and ends on September 30. The water year is designated by the calendar year in which it ends.

Site Description

The 55-acre site is part of a larger watershed near Ephrata, in northern Lancaster County, Pennsylvania (Figures 1 and 2), in the Conestoga Valley section of the Piedmont physiographic province. The farm is underlain by karstic, carbonate rocks of the Millbach and Snitz-Creek Formations (Meisler and Becher, 1971). Indian Run (creek) flows from north to south along the eastern site boundary. Bedrock at the farm is overlain by 5 to 30 feet of weathered regolith. Farm soils are classified as Hagerstown and Duffield series silt loams and silt-clay loams (Typic Hapludulfs) (Fox and Piekielek, 1983; U.S. Department of Agriculture, 1985), and were severely eroded on some hillslopes.

Surface elevation at the site ranged from 431 feet at the southwestern corner to 342 feet in the southeastern corner (Figure 3). The land surface had a median slope of about 5 percent, with a range from 2 to 9 percent. Terraces and a grass-covered waterway (Figure 2) were constructed in 1965 to reduce soil erosion. The terraces were restructed in 1981 (before the study period) when a pipe-drainage system was installed beneath approximately 27 acres of the site. Water collected on the terraces discharged through a central drainpipe to a point near Indian Run on the eastern boundary of the farm. The terraces and grass-covered waterway, in combination with permeable soils, facilitated the infiltration of precipitation to ground water; thus, surface runoff occurred only during intense rainstorms.

The water table at the site was about 5 to 30 feet below the land surface (noncoincident with the bedrock surface), and roughly replicated the surface topography. The median water-table elevations for the prenutrient management period (1985-1986) are shown on Figure 4. Sections A-A′, and B-B′ in Figure 5 illustrate the relations between the land surface, depth to bedrock, and the water table. Water recharged regolith and the underlying limestones and dolomites, where ground water flowed through a complex network of fractures, joints, and bedding planes, which had been enlarged to varying degrees by dissolution of the carbonate rocks. Well construction and pumping data, as well as water-level and water-

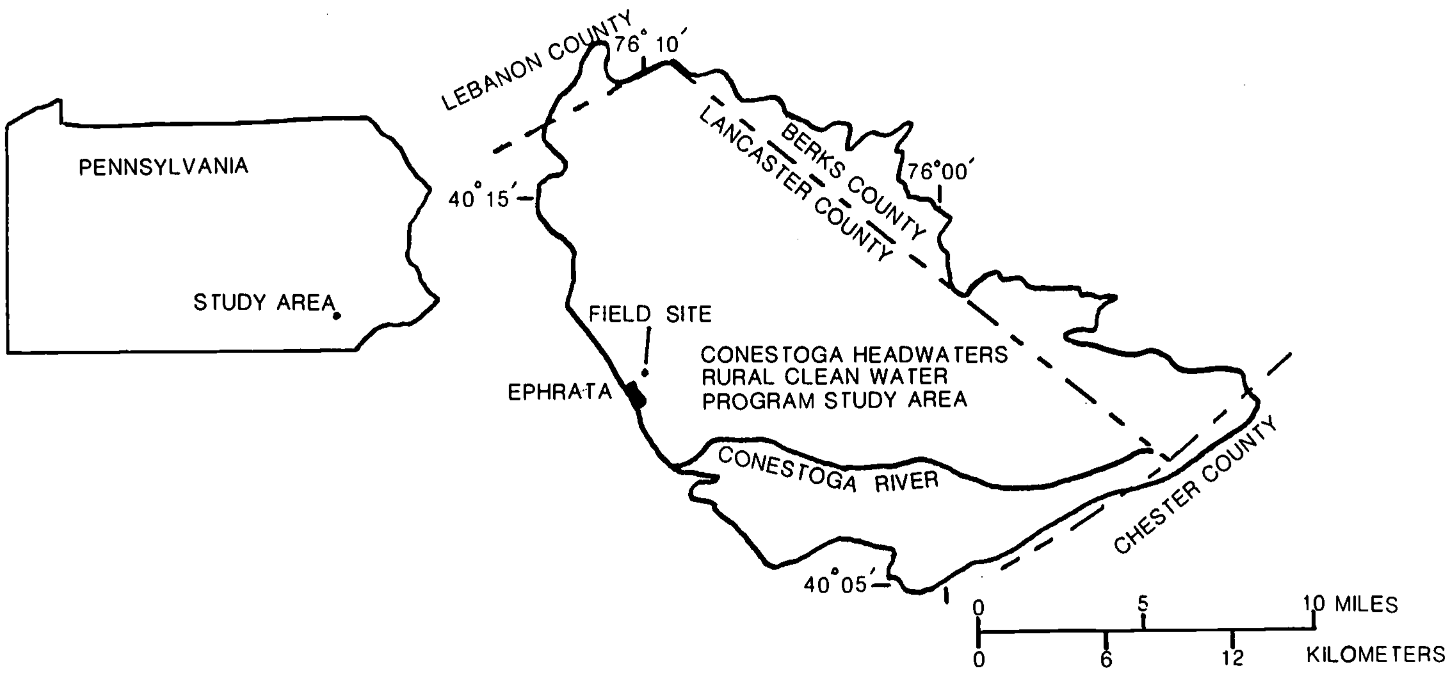

Figure 1. Location of the Conestoga Headwaters Rural Clean Water Program Study Area and Field Site Near Ephrata, Pennsylvania.

quality data, suggest that aquifer properties, such as specific yield and transmissivity, differed widely at the farm site. Such variability is common in fractured, karstic, carbonate-rock aquifers (Meisler and Becher, 1971; Parizek *et al.*, 1971).

Average annual precipitation at the site was about 43.5 inches, based on 30 years of record (1951-1980) for the nearby National Oceanic and Atmospheric Administration (NOAA) precipitation gage in Ephrata, Pennsylvania (National Oceanic and Atmospheric Administration, 1982). Monthly precipitation recorded by a raingage at the site for the study period (1985-1990) is shown on Table 1, with a comparison of annual precipitation measured at the site to the NOAA 30-year average precipitation.

Crops were grown in 47.5 acres of the 55-acre site. Annual crop acreage during 1985-1990 is shown on Table 2.

A continuous-record gage with an automatic sampler was used to monitor surface-runoff quantity and quality from the pipe-drained terraces from 1984 to 1988. A detailed discussion of surface-runoff sampling and analyses is beyond the scope of this report, because surface runoff transported a small percentage of nitrogen from the site relative to other outputs.

Thirteen wells were drilled to depths ranging from 40 to 350 feet to determine the hydrogeology of the site (Table 3; Chichester, 1988). An existing hand-dug well (LN 1667) and spring were also used to provide data for site characterization. The wells were air-rotary drilled, cased to bedrock with 6-inch steel casing, and were finished as open holes in the fractured bedrock. Annular spaces around the well casings were grouted with cement, and bentonite clay caps were installed as casing seals at the land surface. Well casings were pumped dry if possible, or three borehole volumes of water were pumped from the wells prior to sampling.

Ground-water samples were analyzed for ammonia plus organic nitrogen, ammonia, nitrite, and nitrite plus nitrate. Dissolved nitrate accounted for over 90 percent of all dissolved nitrogen in ground water. Nitrate concentrations at all sampling points commonly exceeded the U.S. Environmental Protection Agency (1990) maximum contaminant level for drinking water of 10 mg/L (milligrams per liter) nitrate as nitrogen.

Relations between applications of nitrogen to farm fields and concentrations of nitrate in ground water were reported in Hall (1992). After nutrient applications to farm fields were reduced under nutrient management, statistically significant decreases in median nitrate concentrations occurred in ground-water samples collected at four of five shallow ground-water monitoring wells (Figure 6; Table 4). Decreases in median concentrations of nitrate in ground-water samples ranged from 8 to 32 percent of the median concentrations prior to nutrient management and corresponded to nitrogen application decreases of 39 to 67 percent in contributing areas that were defined upgradient of these wells.

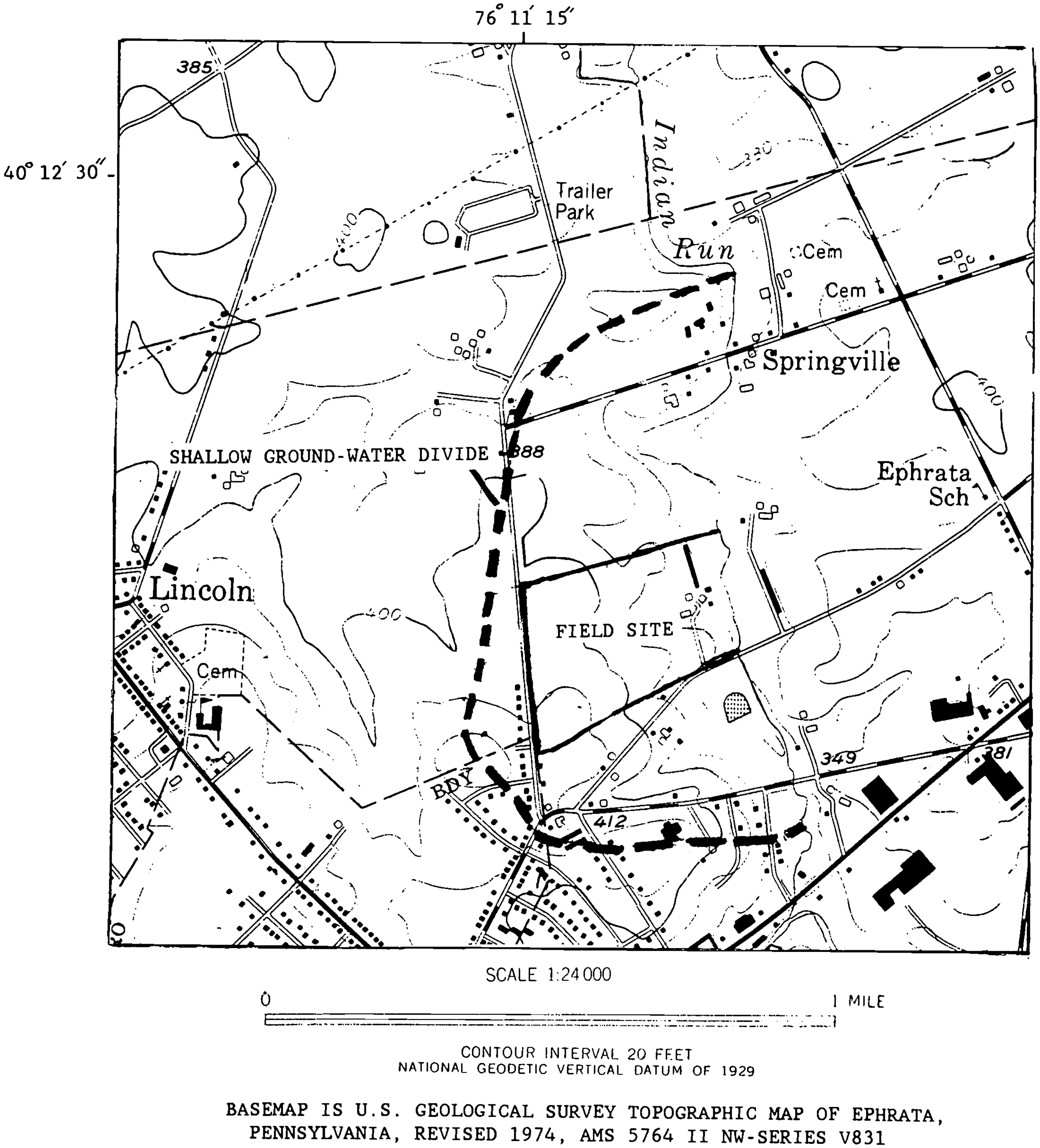

Figure 2. Location of the Site in Relation to the Shallow Ground-Water Basin and Area Topography.

Changes in nitrogen applications to the contributing areas of five wells were significantly correlated (using the Spearman Rank-Sum test) with nitrate concentrations of well water. Changes in concentrations of nitrate in ground water lagged behind the changes in applied nitrogen fertilizers (primarily

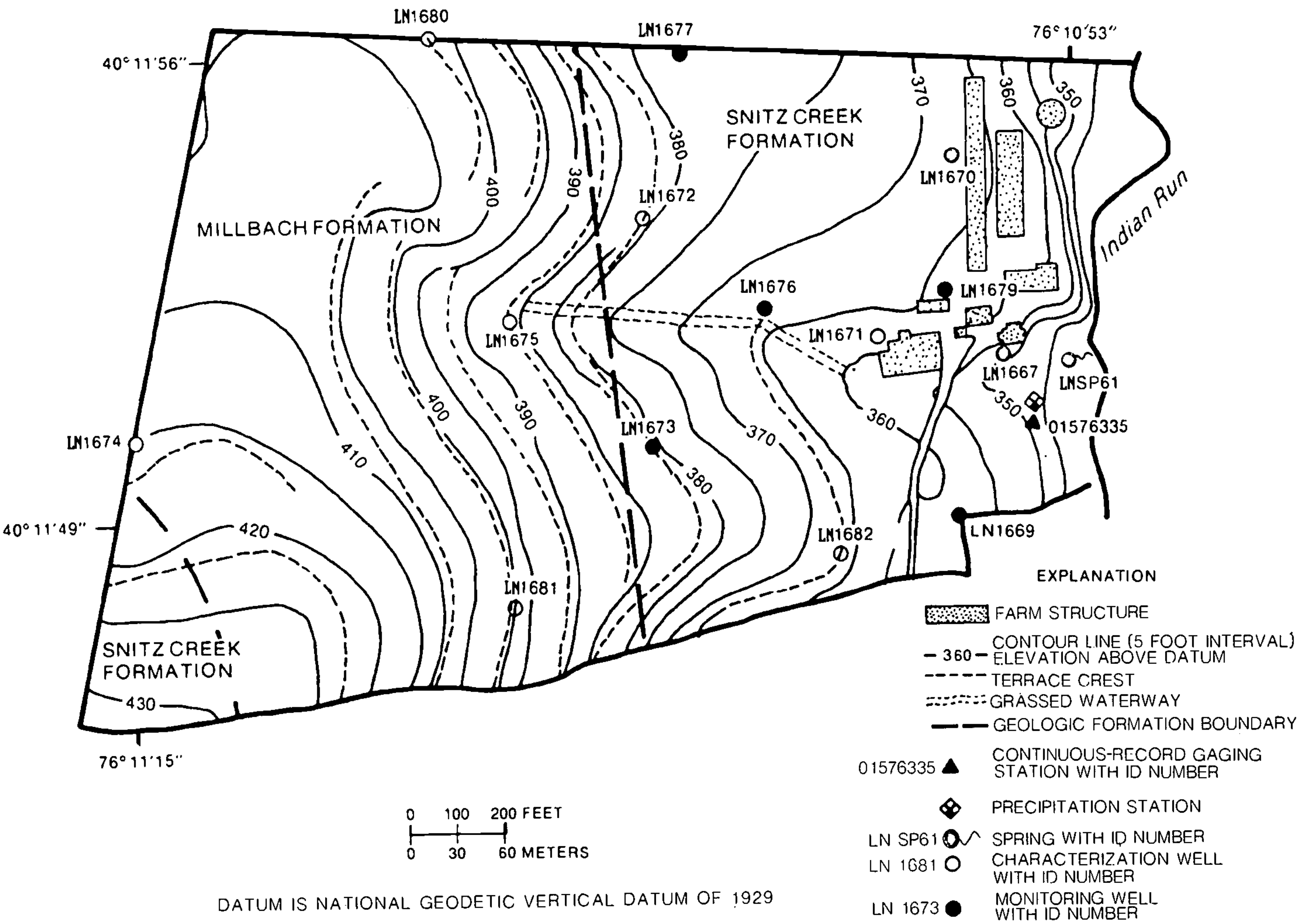

Figure 3. Data Collection Locations and Geologic Units (from Chichester, 1988, Figure 6).

manure) by approximately 4 to 19 months (Figure 7). The Spearman Rank-Sum correlations were performed at the 90 percent confidence level, and produced rho values indicating the strength of the correlations, and p values indicating the significance of the correlations. The Spearman rho and p values at well LN 1669 were rho = 0.401, p = 0.002; at well LN 1673 rho = 0.684, p = 0.001; at well LN 1676 rho = 0.583, p = 0.001; at well LN 1677 rho = 0.496, p = 0.001; and at well LN 1679 rho = 0.215, p = 0.067. A complete discussion of the Wilcoxon tests, the well contributing areas, and the cause-effect nature of the statistical correlations is contained in Hall (1992).

WATER BUDGET

Estimation of nitrogen inputs to, and outputs of nitrogen from, the site required the calculation of loads of nitrogen in surface and ground water. An accurate water budget of the site was essential to the accuracy of the nitrogen load estimations. Precipitation, runoff, ground-water recharge, ground-water inflow and outflow across site boundaries, and evapotranspiration were the major components of the water budget (Table 5). Equipment (Chichester, 1988) and methods used to measure or estimate the individual water budget components are described below.

Precipitation. Precipitation was measured at the site using a rain gage equipped to record rainfall every 5 minutes. Precipitation measured at the site averaged 41.7 inches from 1985-1990 (Table 1, Figure 8A), and was slightly less than the average annual precipitation of 43.5 inches based on 30 years of record (1951-1980) from the weather station at Ephrata (National Oceanic and Atmospheric Administration, 1982).

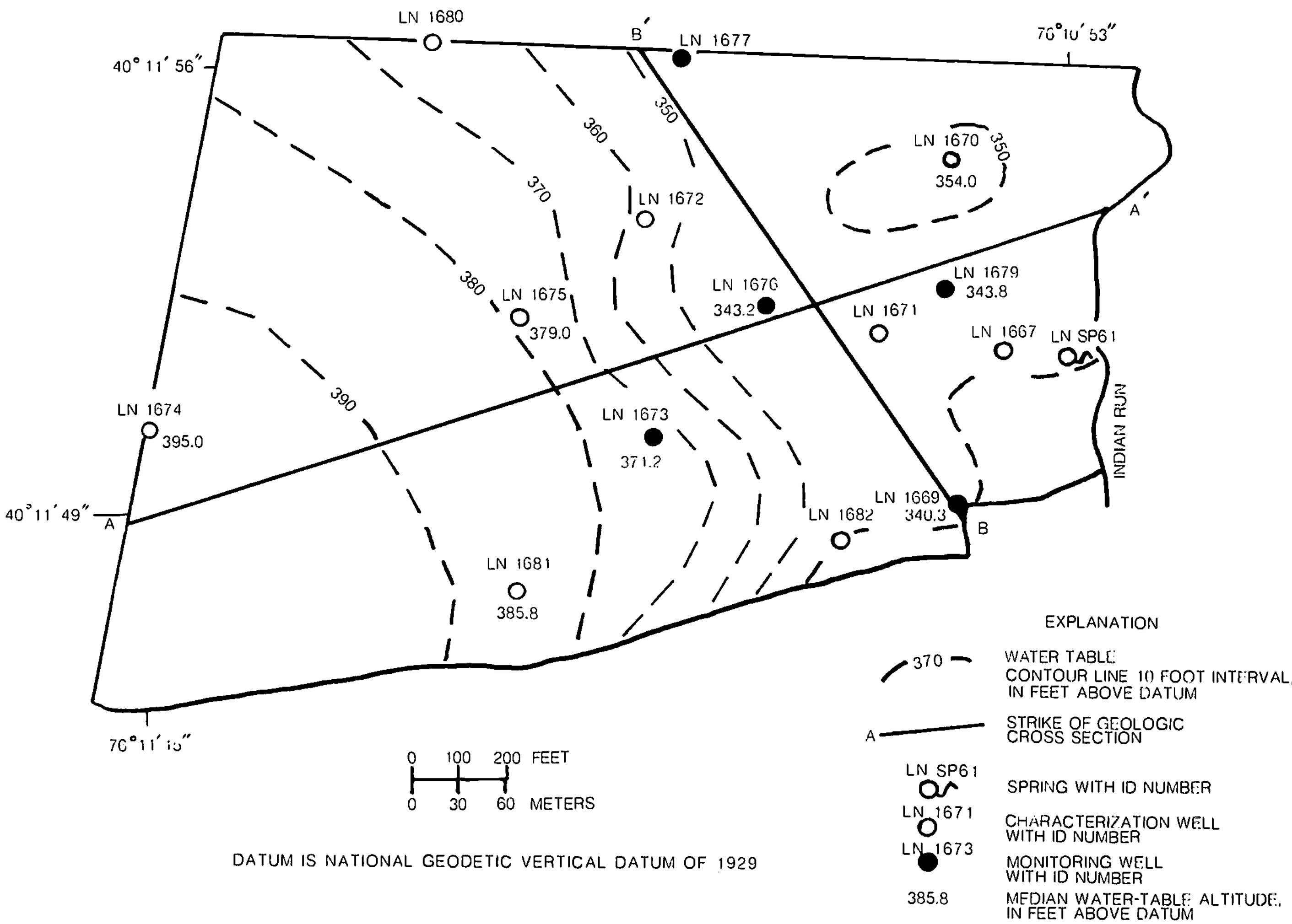

Figure 4. Estimated Water-Table Altitude Map (Sections A-A′ and B-B′ are on Figure 5).

Runoff. Surface runoff was recorded from 27 terraced acres of the farm fields from 1985 to 1988. Total runoff volumes from the 27 terraced acres were doubled to obtain a gross estimate of surface runoff from the entire 55-acre farm, based on visual observations of storm runoff discharging immediately north of the gage flume that were approximately equal to runoff through the gage. The estimated runoff averaged approximately 3 percent of annual precipitation recorded at the site during 1985-1988. Therefore, runoff was assumed to be 3 percent of annual precipitation during 1989-1990, when the gaging station was not operational.

Ground Water Recharge. Recharge to ground water from infiltration of precipitation was estimated from the water-level rise recorded in wells in response to storms as described by Gerhart (1986, p. 487). Rapid water-level rises following storms indicated rapid infiltration of water (Figure 8B). Therefore, the water level rise in a well multiplied by the specific yield of the geologic material in the vicinity of that well should provide a reasonable estimate of the ground-water recharge from that storm.

Recharge was computed using water-level observations (Figure 8B) and estimated specific yields at wells LN 1673 and LN 1677. These wells were selected to be representative of the entire site because aquifer yields were near the site average. The combined available water-level data from both wells was needed to close a complete record spanning the study period. The specific yields of the aquifer at wells LN 1673 and LN 1677 were determined using a method described in Gerhart (1986, p. 487). Precipitation, runoff, and water-level rises were measured for four storms that occurred from November 1985 through March 1986. The specific yield of the aquifer was then computed as

$$\text{Specific Yield} = (P - R) / WLR \tag{1}$$

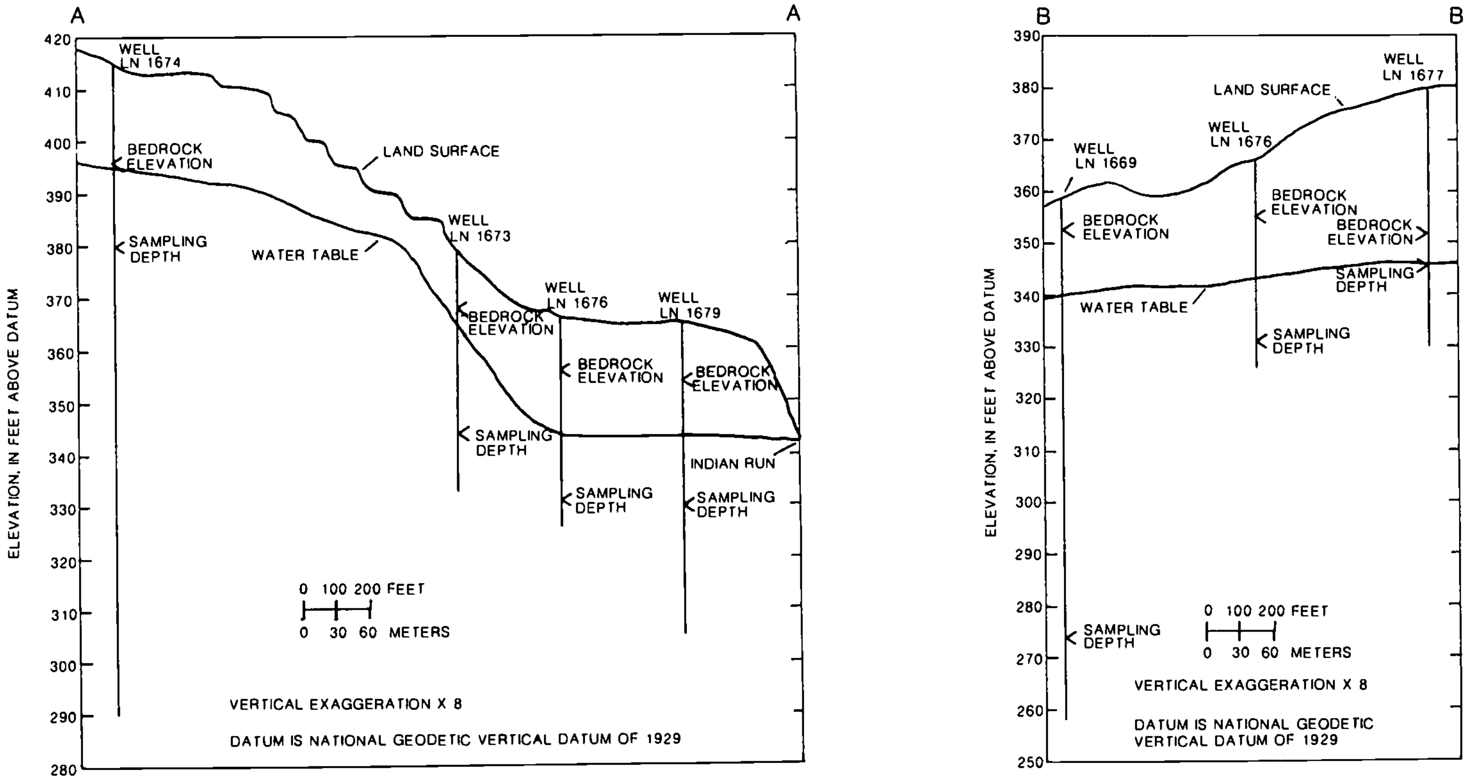

Figure 5. Sections A-A′ and B-B′ (strikes of sections shown on Figure 4).

TABLE 1. Comparison of Annual Precipitation at the Site, Long-Term Average for Ephrata, Pennsylvania, and Percent Deviation from Long-Term Average.

Period	Precipitation (inches)	Long-Term Average*	Percent Deviation from Long-Term Average
October 1984 through September 1985	35.8	43.5	–18
October 1985 through September 1986	38.8	43.5	–11
October 1986 through September 1987	45.0	43.5	+03
October 1987 through September 1988	40.4	43.5	–07
October 1988 through September 1989	46.6	43.5	+07
October 1989 through September 1990	43.6	43.5	+00

*Long-term average precipitation based on 30 years (1951-1980) of record from the National Oceanic and Atmospheric Administration weather station at Ephrata, Pennsylvania.

where P is precipitation measured at the site, in inches; R is runoff measured at the site, in inches; and WLR is water-level rise in a well, in inches.

Use of this method produced an estimated specific yield of 0.07 at well LN 1673 and 0.06 at well LN 1677. Therefore, for the period October 1987 through May 1985, the water-level rise in observation well LN 1673 was multiplied by a specific yield of 0.07 to estimate recharge. Recharge from later storms was computed from the water-level rise and specific yield (0.06) of well LN 1677, which was the well with the most complete water-level record for the period after May 1985. Recharge estimates are sensitive to the value of specific yield used in this computation. Regardless of the specific yield value used, proportional differences in total recharge between years should be significant for comparative purposes.

Recharge was computed for each storm and summed by month and year. Annual recharge to ground water was estimated to average about 44 percent of precipitation measured at the site during the years 1985-1990 (Table 4). If recharge were evenly distributed throughout the entire study period, 1.54 inches would be recharged monthly. From Figure 9, the importance of months of extreme recharge is evident. In the 1989 water year for example, 43 percent of the annual recharge occurred in May.

Evapotranspiration. Water lost from the site in a year as evapotranspiration was computed as the residual term in the water budget. Evapotranspiration ranged from 46 percent of precipitation in 1987 to 70 percent of precipitation in 1985.

TABLE 2. Annual Crop Acreage.

Year	Growing Season	Crop Type	Acreage
1985	Summer	Corn	43.5
	Summer	Tobacco	4.0
	Winter (1984-1985)	Rye	22.5
1986	Summer	Corn	43.5
	Summer	Tobacco	4.0
	Winter (1985-1986)	Rye	25.0
1987	Summer	Corn	42
	Summer	Tobacco	5.5
	Winter (1986-1987)	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	3.0
	Summer	Fruit and Vegetables	10
1990	Summer	Corn	37
	Summer	Tobacco	4.0
	Summer	Fruit and Vegetables	6.5

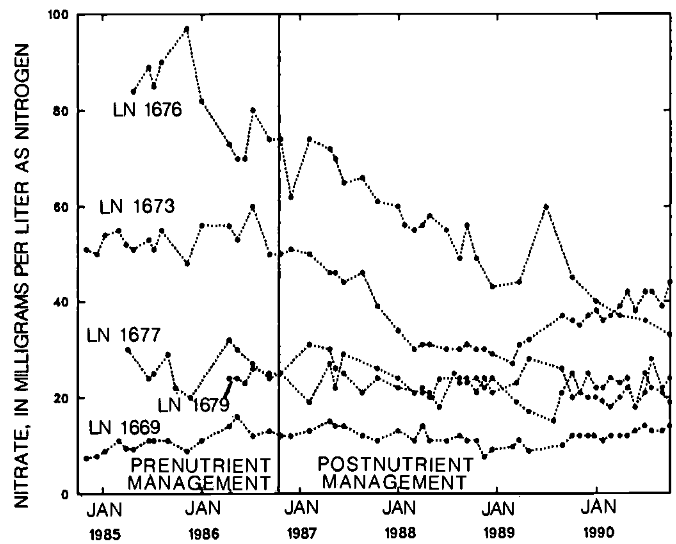

Figure 6. Nitrate Concentrations in Ground-Water Samples Collected at Wells LN 1669, LN 1673, LN 1676, LN 1677, and LN 1679, During the Prenutrient-Management and Postnutrient-Management Periods (modified from Hall, 1992, Figure 9C).

TABLE 3. Monitor Well Locations and Descriptions, and Sampling Depths [all depths shown in feet below land surface; (gal/min)/ft, gallon per minute per foot of drawdown; >, less than; –, no data].

Well Number	Latitude	Longitude	Total Depth of Well	Depth to Bottom of Casing (overburden thickness)	Depth to Bedrock	Bedrock Elevation (feet)	Sampling Depth	Estimated Specific Capacity [(gal/min)/ft]	Data Collected: Nutrients*	Data Collected: Water Level**
LN 1667	401152	761055	15	–	–	–	12	–	O	N/A
LN 1669	401149	761055	100	11	6.5	352	85	< 1	M	C
LN 1670	401156	761057	75	9.8	5.5	361	65	< 1	Q	C
LN 1671	401152	761058	28	18.8	13	342	–	< 1	O	I
LN 1672	401152	761105	100	10.9	10	370	–	< 1	O	I
LN 1673	401148	761103	46	13.8	12	368	35	< 1	M	C
LN 1674	401145	761115	125	25.2	19	396	–	< 1	O	I
LN 1675	401150	761107	55	17.2	14	374	–	< 1	O	I
LN 1676	401152	761101	40	8.8	11	356	35	< 1	M	C
LN 1677	401156	761105	50	30.0	28	349	35	20	M	C
LN 1679	401152	761057	60	13.4	10	354	35	20	M	C
LN 1680	401156	761109	60	7.8	7	375	–	< 1	O	I
LN 1681	401147	761108	60	8.8	8	400	35	< 1	O	I
LN 1682	401148	761059	350	18.6	18	350	35	< 1	O	I

*M, monthly; Q, quarterly; O, occasionally.
**C, continuous; I, intermittent.

TABLE 4. Average Depth to Water Table, Lag Time, Significance of Wilcoxon-Mann-Whitney Test, Pre-Nutrient Management, and Post-Nutrient Management Nitrate Concentrations, Percent Change in Median Nitrate Concentrations, and Percent Change in Nitrogen Applications to Contributing Areas at Five Wells (from Hall, 1992, Table 2) (mg/L, milligram per liter).

Well Number	Average Depth to Water in Feet, From Land Surface	Lag Time (months)	Sampling Depth, in Feet Below Water-Table Surface	Wilcoxon-Mann-Whitney Test, Significant Pre- to Post-Increase or Decrease?	Pre-Nutrient Management Median Nitrate Concentration (mg/L)	Post-Nutrient Management Median Nitrate Concentration (mg/L)	Change From Pre- to Post-Period in Median Nitrate Concentration (percent)	Change From Pre- to Post-Period in Applications (percent)
LN 1669	18	19	66	Yes (+)	11	12	+8	–67
LN 1673	10	16	23	Yes (–)	53	37	–30	–53
LN 1676	24	9	10	Yes (–)	82	56	–32	–67
LN 1677	30	18	3	Yes (–)	26	23	–12	–39
LN 1679	20	4	13	Yes (–)	24	22	–8	–60

Ground-Water Flow Across Boundaries. A ground-water flow model of the hillslope on which the farm site is situated was constructed to help estimate the magnitude of ground-water inflow and outflow across site boundaries. The hillslope was simulated as a two-dimensional, steady-state flow system using the finite-difference model of McDonald and Harbaugh (1988). The finite difference grid, aquifer properties, and boundary conditions used in the model are shown in Figure 10. Site transmissivities input to the model were based on the water-table configuration (Figure 4) in conjunction with estimates of transmissivity from Darcy's law as described as Driscoll (1986).

The simulated water table is contoured and measured water levels are shown at observation well locations for comparison (Figure 11). Although the simulated surface does not fit the observed water levels exactly, the shape of the water table surface has been reproduced. The water budget computed by the model is based on the long-term average recharge from precipitation of 19.1 inches per year, estimated as 44 percent of the long-term precipitation of 43.5 inches per year recorded at Ephrata (National Oceanic and Atmospheric Administration, 1982). Ground-water inflow from 1985 to 1990 was estimated as 16 percent of total inflow of water to the site as indicated by modeling simulations (Table 6).

Ground-water recharge and inflow across site boundaries was balanced by outflow across site boundaries and discharge to Indian Run. The boundaries of the ground-water flow system at the site were poorly defined. Because the site occupied only a part of a larger watershed, the northern, southern, and western boundaries of the site did not correspond to real physical boundaries. The water table configuration (Figure 4) indicated that ground water flowed across boundaries in the amounts shown in Table 6.

FATE AND TRANSPORT OF NITROGEN

Ideally, all sources of nitrogen input to, and output from, and all nitrogen storage at a site could be accurately quantified, and all mechanisms of nitrogen transport could be described (Stevenson, 1982, Chapter 1). An equation describing the fate and transport of nitrogen through a farm field could be (inputs = output, ± change 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 – change in soil storage of nitrogen.

To the extent possible, inputs and outputs of nitrogen from the site were calculated for each year of the study period (Table 7; Hall and Risser, 1992). Annual inputs of nitrogen to the site cannot be expected to quantitatively balance annual outputs on Table 7. 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. Additional time may be required for soluble nitrate to infiltrate from the land surface to ground water. Inputs of nitrogen were

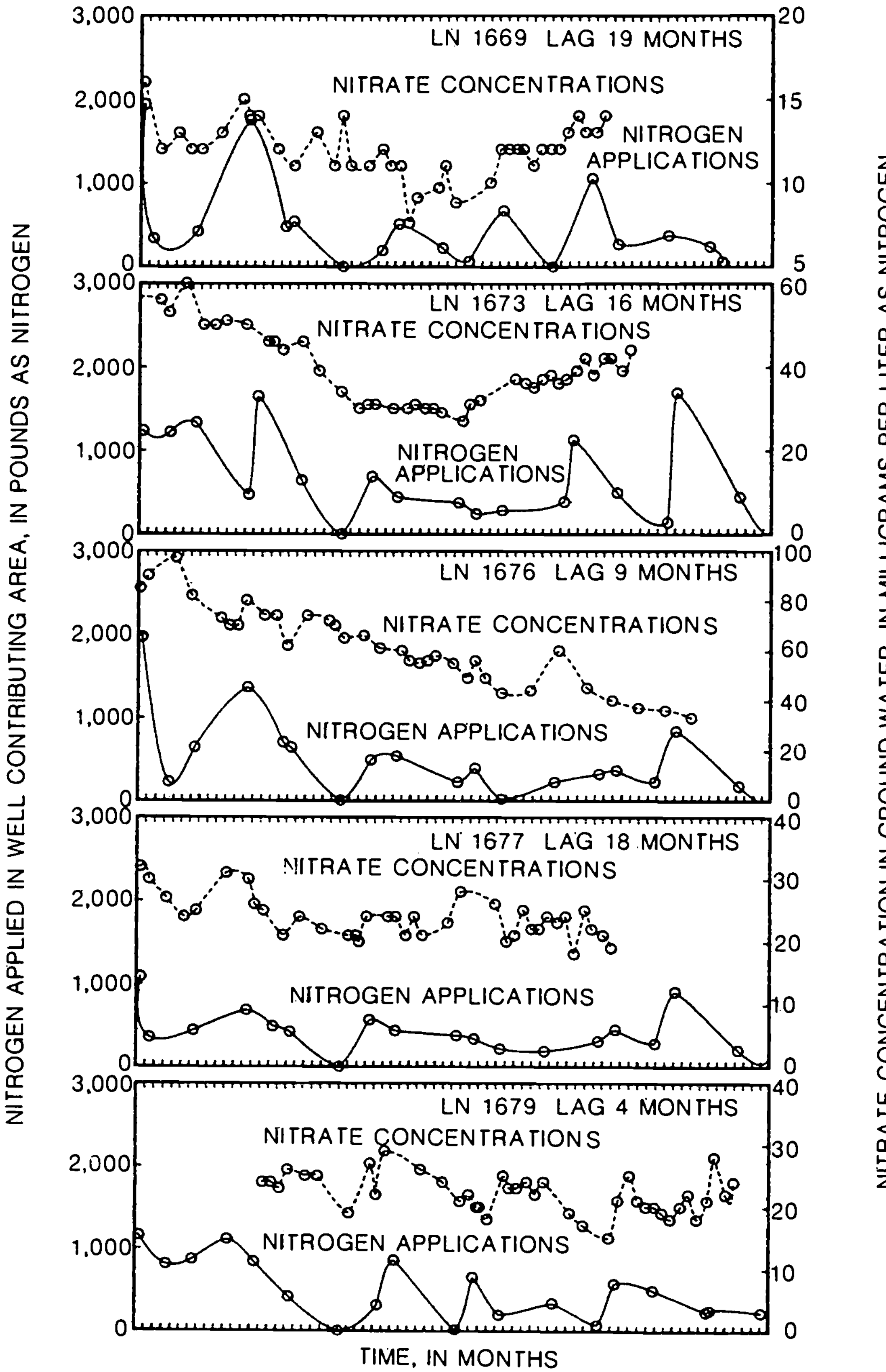

Figure 7. Ground-Water Nitrate Concentrations and Nitrogen Applied to Contributing Areas of Wells LN 1669, LN 1673, LN 1676, LN 1677, and LN 1679 (to illustrate correlations, application curves have been moved to the right to match resultant concentrations; to illustrate real-time relations, move the application curves to the left by the indicated time lags; from Hall, 1992, Figure 10).

TABLE 5. Annual Water Budget in Terms of Precipitation, Ground-Water Inflow, Runoff, Evapotranspiration, and Ground-Water Outflow, in inches (numbers in parentheses are percentage of total inflow or outflow).

Year	Precipitation	+	Ground-Water Inflow	=	Runoff	+	Evapotranspiration	+	Ground-Water Outflow
1985	35.8 (84)		1.9 (16)		0.6 (2)		25 (66)		12.1 (32)
1986	38.8 (84)		3.5 (16)		0.3 (1)		18.2 (43)		24.0 (56)
1987	45.0 (84)		4.1 (16)		2.9 (6)		20.5 (42)		25.7 (52)
1988	40.4 (84)		3.7 (16)		1.3 (3)		19.6 (44)		23.2 (53)
1989	46.6 (84)		4.1 (16)		1.4* (3)		23.5 (46)		25.8 (51)
1990	43.6 (84)		3.4 (16)		1.3* (3)		24.7 (52)		21.0 (45)
Average	**41.7**		**3.5**		**1.3**		**21.9**		**22.0**

*Runoff estimated as 3 percent of precipitation based on average of 1985-1988 data.

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 small percentages of the nitrogen input to the site and were therefore omitted.

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 (< 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 at the farm were commonly near saturation, indicating that anaerobic denitrification of nitrogen or dissimilatory reduction to ammonium from shallow ground water were not steady state processes. However, intermittent denitrification or dissimilatory reduction may have occurred, especially in soils and regolith during saturated conditions.

Potential errors due to assumptions made in the calculation of loads may greatly influence the numbers reported in Table 7. A brief description of methods of calculation and errors associated with each nitrogen input and output term follows.

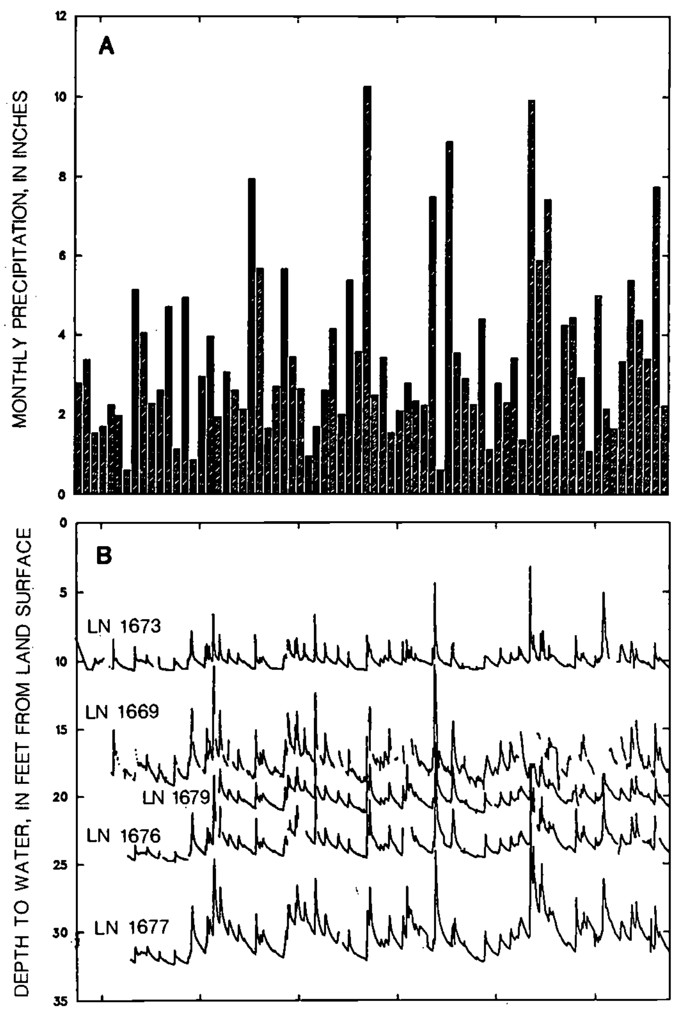

Figure 8. Monthly Precipitation (A), and Depth to Water from Land Surface (B) at Wells LN 1669,LN 1673, LN 1676, LN 1677, and LN 1679.

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 using application data supplied by the farmer and laboratory analysis of manure samples collected at the site during the study period.

Annual applications of manure nitrogen to the farm decreased after nutrient management was implemented in October 1986.

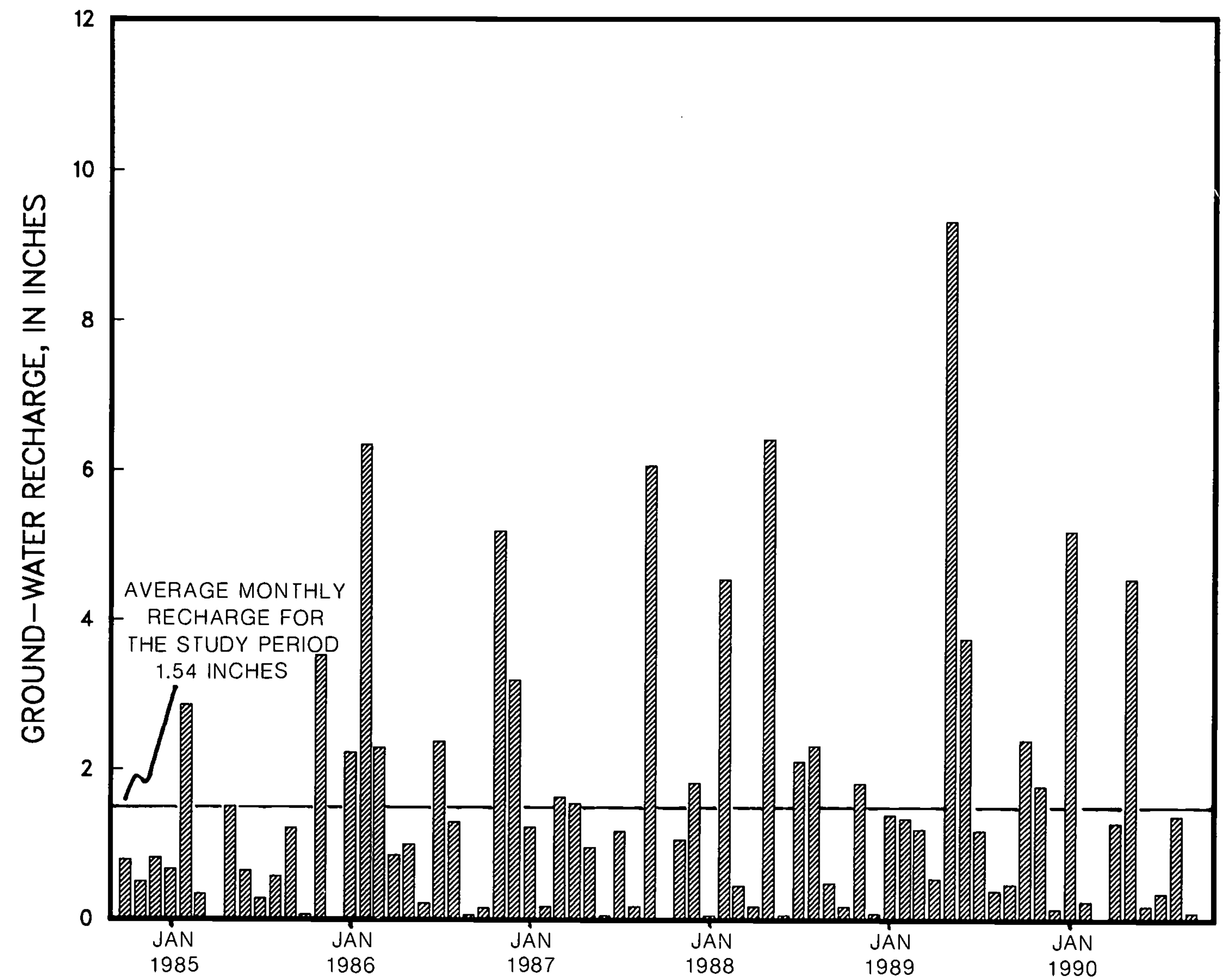

Figure 9. Estimated Monthly Ground-Water Recharge, October 1984 Through September 1990.

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, as information about the nitrogen content and quantity of the fertilizer were typically available when the fertilizer was purchased.

Precipitation Nitrogen. Nitrogen in precipitation is in the form of ammonium and nitrate ions. Loads of nitrogen entering the site in precipitation were calculated using volumes of precipitation estimated from rain gage data at the site multiplied by concentrations of the nitrogen content of ammonium and nitrate in precipitation samples in Pennsylvania reported by Lynch *et al.* (1986, 1987), Lynch *et al.* (1988), Lynch *et al* (1989; 1990; 1991, Table 8).

Errors in the measurement of precipitation at the site gage were small. A much greater error in the budget was possible from using the ammonium and nitrate values reported for the Lancaster County area by Lynch *et al.* (1986 to 1991) that were 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 easily have had an elevated precipitation-nitrogen load relative to those calculated using the regional ammonium and nitrate concentration estimates of Lynch and others.

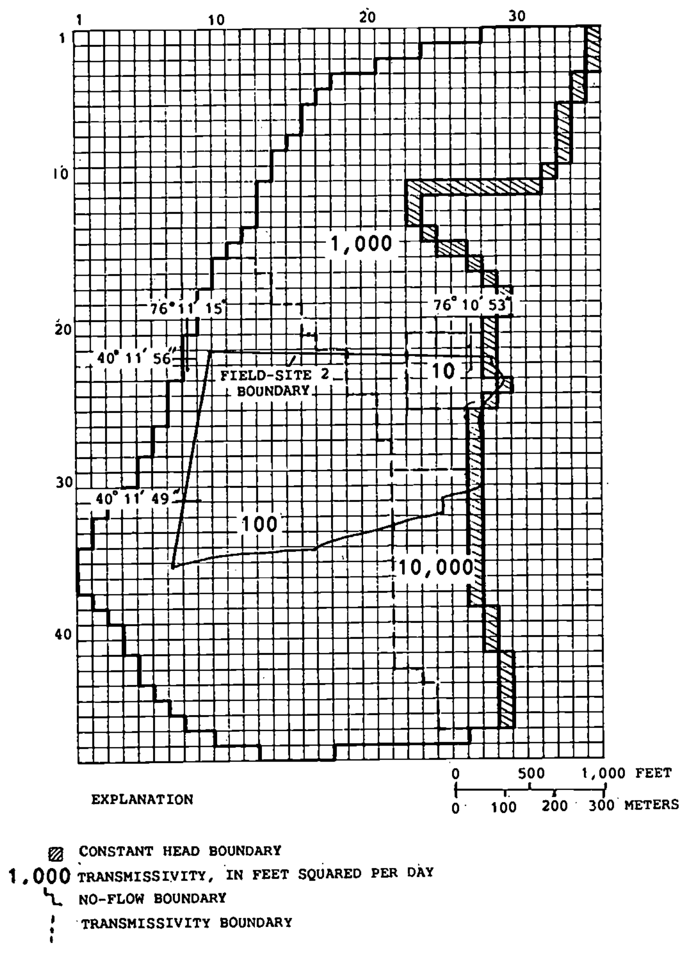

Figure 10. Finite-Difference Grid, Hydrologic Boundaries, and Aquifer Properties Used in the Ground-Water Flow Model.

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.

While there was undoubtedly error involved in using 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 due to the relatively small quantity of water that was estimated to enter the site across the western boundary.

Nitrogen in Soils

Soils at the site act as a sink for temporary storage of nitrogen. Nitrogen in soils exists in three phases: in soil air, bound to soil particles and organic compounds, and in soil water. A silt-loam soil in good condition for plant growth may be 20 to 30 percent (by volume) air, 20 to 30 percent water, 45 percent minerals, and 5 percent organic matter (Thibodeaux, 1979, p. 36). As with nitrogen movement through the site as a whole, nitrogen in the soil exists in states of unsteady-state, non-equilibrium conditions (MacKay, 1991, p. 29).

Complete discussions of nitrogen cycling in agricultural soils are contained in Stevenson (1982) and Ehrlich (1990), and are beyond the scope of this short article. At the study site, organic nitrogen enters soils primarily from applications of animal manure and associated urea. This organic nitrogen rapidly mineralizes to nitrate (oxidation state +5), a highly soluble form of nitrogen (Ehrlich, 1990). Ammonification is a first step in the mineralization process. Ammonium cations may sorb to soil particles (particularly clays) and become immobilized for periods of time in the unsaturated zone through cation exchange and other processes. The nitrogen contained in ammonium becomes available for use by plants (and for leaching to surface and ground water) through the process of nitrification, a conversion of ammonium to nitrate. Graves (1986b) and Stevenson (1982) have estimated that from 40 to 75 percent of the nitrogen remaining in most types of manure after storage becomes available (as nitrate) for crop use or leaching during the year of application to farm fields.

Because the organic nitrogen applied to the site results in the formation of nitrate in soils, the top four feet of soils at the site were sampled for soluble nitrate nitrogen during the spring and fall of each year from 1985-1990 (Table 9). Soil samples were collected as 2-inch diameter cores using a hydraulic soil press. Samples were collected upgradient of wells LN 1670, LN 1673, LN 1676, and LN 1677 (Figure 4). Each sample was a composite of three subsamples that were collected from areas measuring 9 feet-squared. Samples were analyzed for soluble nitrate at the Pennsylvania State University, College of

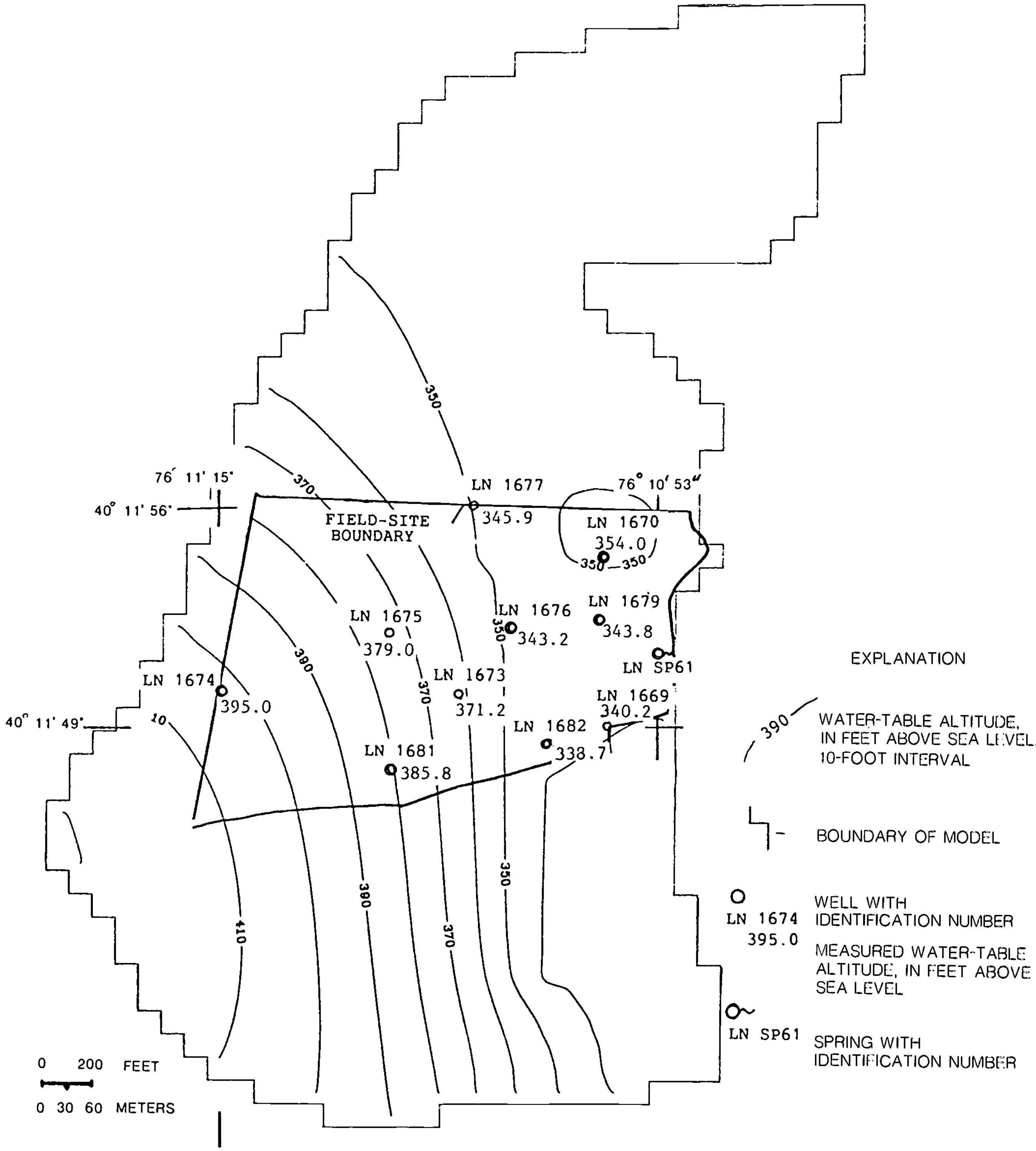

Figure 11. Simulated Water-Table Surface at the Site and Surrounding Areas.

Agriculture, Merkle Soil Laboratory in State College, Pennsylvania. If 40 to 75 percent of manure nitrogen applied to farm fields is transformed to nitrate during the first year following application as suggested by Stevenson (1982) and Graves (1986a), and if the bulk of the nitrogen applied to farm fields at the site leaches to ground water within a 4 to 19-month period (Figure 7), then annual nitrate loads in the top four of site soils could be expected to be correlated with annual nitrogen applications. A comparison of annual nitrogen application data (Table 7) and loads of nitrate in the top four feet of soils (Table 9) indicates that loads of nitrate nitrogen in soils decreased (1985-1987; 1989-1990) and increased (1987-1989) in response to annual changes in loads of nitrogen applied.

TABLE 6. Simulated Steady-State Ground-Water Budget for the Site.

	Inches Per Year	Percentage of Total Inflow
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**

TABLE 7. Estimated Inputs and Outputs of Nitrogen (numbers are in pounds as nitrogen).

	Estimated Nitrogen Inputs to Site				Estimated Nitrogen Outputs from Site			
Water Year	Nitrogen in Manure Fertilizer	Nitrogen in Commercial Fertilizer	Nitrogen in Precipitation	Nitrogen in Ground-Water Inflow	Nitrogen Consumed by Crops	Surface-Water Loads of Nitrogen	Volatilization	Ground-Water Loads of Nitrogen
1985[1]	25,500	1,000	290	90	8,500	120	7,300	5,100
1986[1]	18,500[2]	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	90[3]	5,500	7,700
1990	16,700	1,700	320	160	8,100	90[3]	4,600	6,400
Average 1985-1990	**17,150**	**800**	**330**	**160**	**7,700**	**90[3]**	**5,180**	**7,930**
Percent of Average Annual Inputs or Outputs	**93**	**4**	**2**	**1**	**37**	**<1**	**25**	**38**

[1]1985 and 1986 were pre-nutrient management.
[2]1986 manure applications were unusually small due to 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 1985-1988.

TABLE 8. Calculation of Annual Nitrogen Loads in Precipitation, by Use of Annual Nitrate and Ammonium Concentrations Reported by Lynch *et al.* (1986-1991), and Precipitation Quantity Data from the Site [mg/L, milligrams per liter; pounds of nitrogen in precipitation were calculated by multiplying total ammonium plus nitrate concentrations (as nitrogen) by annual precipitation (in liters) and a milligram to pound conversion factor (2.205 x 10^{-6} pounds per milligram)].

Water Years	Annual Dissolved Ammonium* Concentrations as N, in mg/L	Annual Dissolved Nitrate Concentrations as N, in mg/L	Total Ammonium Plus Nitrate as N, in mg/L	X Annual Precipitation in Liters	X $\left(\frac{1\,g}{1,000\,mg}\right)\left(\frac{2,205\,lb}{1,000\,g}\right)$ =	Annual Pounds Nitrogen in Precipitation
1985	0.20	0.45	0.65	202,624,000		290
1986	0.20	0.44	0.64	219,528,000		310
1987	0.21	0.50	0.71	254,637,000		400
1988	0.24	0.41	0.65	228,461,000		330
1989	0.20	0.38	0.58	263,230,000		340
1990	0.25	0.34	0.59	246,439,000		320

*Lynch and others incorrectly reported laboratory ammonia analyses results as ammonium in Lynch *et al* (1986-1990). Values reported here have been corrected. Ammonium values in Lynch *et al.* (1991) were published correctly (J. Lynch, personal communication, 1992).

TABLE 9. Soluble Nitrate-Nitrogen Content in the Top Four Feet of Soils (in pounds as nitrogen).

Sampling Date	Number of Samples	Soluble Nitrate Nitrogen		
		Mean	Minimum	Maximum
Fall 1985	11	265	128	428
Spring 1986	14	291	77	614
Fall 1986	9	227	104	364
Spring 1987	9	123	85	184
Fall 1987	9	96	31	180
Spring 1988	9	147	68	222
Fall 1988	9	205	46	462
Spring 1989	5	260	150	389
Fall 1989	5	182	110	251
Spring 1990	5	186	126	262
Fall 1990	5	170	80	324

Outputs of Nitrogen

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

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 based on information in Pionke and Urban (1985).

Quantification of the volatilization of nitrogen from manure and commercial fertilizer nitrogen was difficult. Rates of nitrogen volatilization are affected by air temperature, humidity, manure type, manure texture and moisture content, timing of incorporation into soil, soil composition, 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 due to volatilization of nitrogen in manure, reported by Graves (1986a) 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 using discharge-weighted mean-storm nitrogen concentrations for each storm multiplied by measured water discharge. Because there was relatively little surface runoff from the site, removals of nitrogen in surface runoff accounted for < 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 pounds per year from 1985-1990 (Table 7), which was about 38 percent of the total nitrogen load removed from the site.

Nitrogen loads in ground water were calculated from estimates of monthly ground-water discharge multiplied by an estimate of nitrate concentrations in monthly ground-water samples.

Because ground water discharged across the northern, eastern, and southern site boundaries, and nitrogen concentrations in samples collected at site wells 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. The separate estimated monthly loads were then summed to obtain the total monthly nitrogen load from the site, and total monthly site loads were then summed to obtain the annual nitrogen loads discharged from the site in ground water.

The quantity of ground water discharged annually across each site boundary was computed by multiplying the total annual discharge from the site (annual recharge from precipitation plus flow into the site across the western boundary) by the percentage of annual flow estimated to cross each site boundary indicated by output from the ground-water model (Table 6). Annual discharge across each site boundary was then proportioned among months using water-level hydrograph rises for each month divided by the total annual water level rise to obtain the fraction of annual discharge occurring in each month.

Having thus obtained quantities of ground-water discharge crossing each site boundary during each month of the study period, samples from wells located in different parts of the site were chosen to characterize the water quality of the ground-water discharges from the part of the site that they were located in. Liters of ground-water discharge were multiplied by milligrams per liter of nitrate in ground-water samples to obtain loads of nitrogen. Samples from well LN 1677 were chosen to characterize the nitrogen concentrations of water discharged across the northern site boundary, samples from wells LN 1676 and LN 1679 were chosen to characterize the nitrogen concentrations of water discharged across the eastern site boundary, and samples from wells LN 1673 and LN 1669 were chosen to characterize the nitrogen concentration of water discharged across the southern site boundary. Samples from these wells selected to represent monthly nitrate concentrations in ground water were samples collected monthly during nonrecharge conditions or, if no nonrecharge-condition samples were collected, the median of all recharge-influenced samples collected during a month was used. Samples were empirically grouped as recharge influenced if they were collected less than two weeks after a significant recharge event. A significant recharge event was defined using the continuous water-level record of each well as being a water-level rise of 0.6 foot in wells located in parts of the aquifer with large specific yield, and a water-level rise of 1.0 foot in wells located in parts of the aquifer having small specific yield.

In summary, 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 using the formula:

$$(A+B) \times C \times D \times E \times F = G \qquad (2)$$

where the variables are defined as follows: A is volume of ground-water recharge entering the site annually across the western boundary; B is volume of ground-water recharge entering the site annually from precipitation; C is proportional percentage of ground water estimated (from model output) to discharge annually across a site boundary; D is monthly fraction of annual discharge; E is nitrate concentration, in milligrams per liter, of a sample (or of samples) collected to characterize water quality; F is milligram to pound conversion factor (2.205×10^{-6}); and G is monthly nitrogen load, in pounds, discharged across a site boundary.

Reported loads of nitrogen in ground water in Table 6 are approximations that may contain errors because of the assumptions used in load calculations. As previously noted, ground-water recharge and discharge quantity estimates are very sensitive to the values of specific yield used in calculations. Because nitrogen loads are computed by multiplication of yield-based estimations of discharge by concentrations of nitrogen in ground water collected at selected wells, determinations of specific yield for the site could have a large effect on the magnitude of calculated loads. 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 the model-based estimates of ground-water flow across site boundaries were inaccurate.

Virtually all of the nitrogen leaving the site that was not consumed by crops or lost to the atmosphere by volatilization was transported off site in ground water. Over 99 percent of the nitrogen discharged from the site in the surface and ground water was discharged in the ground water. Highly permeable soils, terraced hillslopes, and a grassed waterway may have facilitated the rapid infiltration of precipitation to ground water at the site.

EFFECTS OF NUTRIENT MANAGEMENT ON NITROGEN TRANSPORT

Nutrient management caused changes in the loads of nitrogen input to and output from the site. The major source of nitrogen, manure applications to the 47.5 cropped acres by the farmer, decreased from an average of 22,700 pounds per year (480 pounds per acre per year) to 15,175 pounds per year (320 pounds per acre per year) from the pre- to postnutrient-management periods (Table 7) because part of the manure produced at the site was exported to other farms during the nutrient-management period. Because less manure was applied to farm fields at the 55-acre site, it was estimated that volatilization losses decreased from an average of 5,950 to 4,650 pounds of nitrogen per year from the prenutrient to the postnutrient-management periods.

Loads of nitrogen in ground water decreased substantially after the implementation of nutrient management at the site at the beginning of the 1987 water year (Table 7). Because of the 4 to 19 month times required for nitrogen to oxidize to nitrate and then leach to ground water (Figure 7), loads of nitrogen discharged in ground water during 1985 were probably strongly influenced by (unrecorded) applications of nitrogen made to farm fields during 1984 (before the study began). Loads of nitrogen discharged in ground water during 1986 and 1987 were probably influenced by nitrogen applications made in 1985 and 1986, respectively. Thus, loads of nitrogen discharged in ground water during 1988-1990 provide the best indication of the effects of implementation of nutrient management.

Because loads of nitrogen in ground water were computed by multiplying concentrations of nitrate in ground water by discharge, it was necessary to calculate unit loads of nitrogen in ground-water discharge to quantify any reductions in the nitrogen loads that occurred as a result of nutrient management (Table 10; Figure 12). Loads of nitrogen discharged from the site in ground water decreased about 30 percent from the prenutrient-management period (an average of 292 pounds of nitrogen per million gallons) to the postnutrient-management period (an average of 203 pounds of nitrogen per million gallons).

TABLE 10. Loads of Nitrogen (in pounds) Per Inch Per Million Gallons of Ground-Water Discharge, 1985-1990.

Year	Period	Nitrogen Load in Ground Water in Pounds of Nitrogen Per Inch of Ground-Water Discharge	Nitrogen Load in Ground Water, in Pounds of Nitrogen Per Million Gallons of Ground-Water Discharge
1985	Prenutrient Management	421	282
1986	Prenutrient Management	450	302
1987	Transition Year	408	273
1988	Postnutrient Management	306	205
1989	Postnutrient Management	298	200
1990	Postnutrient Management	304	204

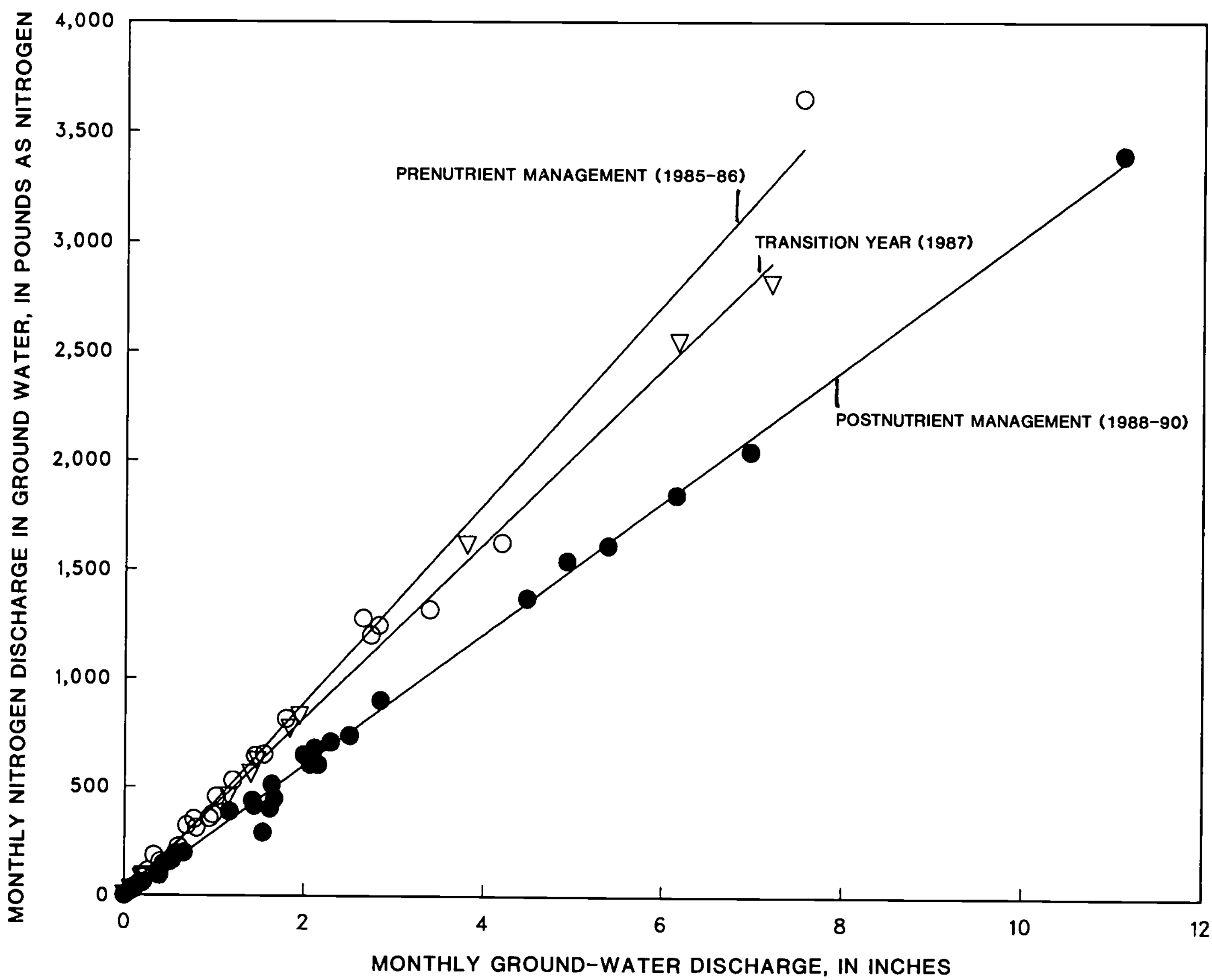

Figure 12. Relation Between Monthly Ground-Water Discharge and Monthly Nitrogen Discharge.

Reductions of the annual unit load of nitrogen discharged in ground water were not related to either the timing or quantity of ground-water recharge. There were no significant positive or negative correlations between concentrations of nitrate in ground water and the volumes of recharge to ground water (Figure 13). Concentrations of nitrate in ground water increased during some recharge events and decreased during others as a function of nitrogen availability in soils at this site (Hall, 1992) and at a similar site in Lancaster County, Pennsylvania (Gerhart, 1986). Reductions in unit loads of nitrogen discharged in ground water during the postnutrient-management period were caused by reductions in nitrogen applications which caused less nitrogen to be available for leaching.

Loads of nitrogen that would have been discharged in ground water from the site if nutrient management had not been implemented were estimated by entering the ground-water discharge values (MGWD) from the postnutrient-management period (Equation 2, Table 11) into the equation defining the discharge-nitrogen load relation from the prenutrient-management period (Equation 1, Table 11). When cumulative sums of these predicted monthly nitrogen loads in ground-water discharge were compared to cumulative sums of the calculated monthly nitrogen loads using a double-mass procedure (Figure 14) (Searcy and Hardison, 1960), it is apparent that nitrogen discharge in ground water decreased by approximately 11,000 pounds during the last three years (3,670 pounds per year) of the postnutrient-management period.

CONCLUSIONS

Inputs of nitrogen to a 55-acre site in Lancaster County, Pennsylvania, were manure fertilizer, commercial fertilizer, precipitation, and ground-water inflow, which averaged 93, 4, 2, and 1 percent of nitrogen added to the site, respectively. Outputs of nitrogen from the site were crops, surface-water runoff, volatilization, and ground-water outflow, which averaged 37, less than 1, 25, and 38 percent of nitrogen removed from the site, respectively. Virtually all of the nitrogen leaving the site that was not consumed by crops or lost to the atmosphere by volatilization was discharged in the ground water.

Applications of manure and fertilizer nitrogen to 47.5 acres of cropped fields decreased about 33 percent, from an average of 22,700 pounds per year (480 pounds per acre per year) before nutrient management to 15,175 pounds of nitrogen per year (320 pounds per acre per year) after the implementation of nutrient management practices. Nitrogen loads in ground-water discharged from the site decreased about 30 percent, from an average of 292 pounds of nitrogen per million gallons of ground water before nutrient management to an average of 203 pounds of nitrogen per million gallons as a result of the decreased manure and commercial fertilizer applications. Reductions in manure and commercial fertilizer applications caused a reduction of approximately 11,000 pounds (3,670 pounds per year; 70 pounds per acre per year) in the load of nitrogen discharged in ground water from the 55-acre site during the three-year period 1987-1990.

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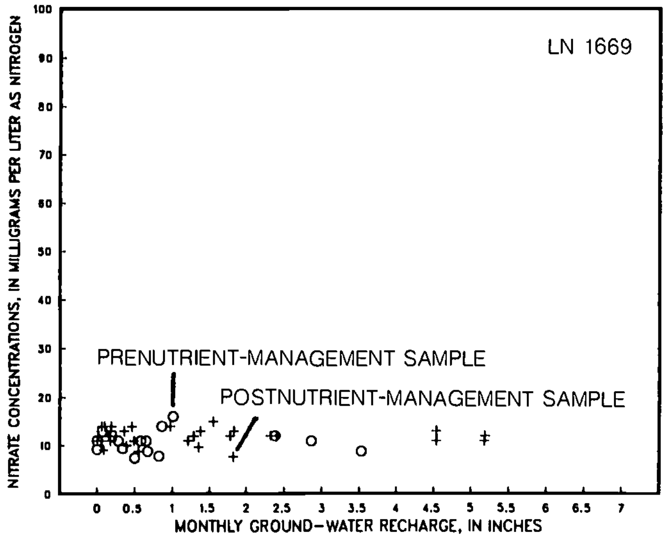

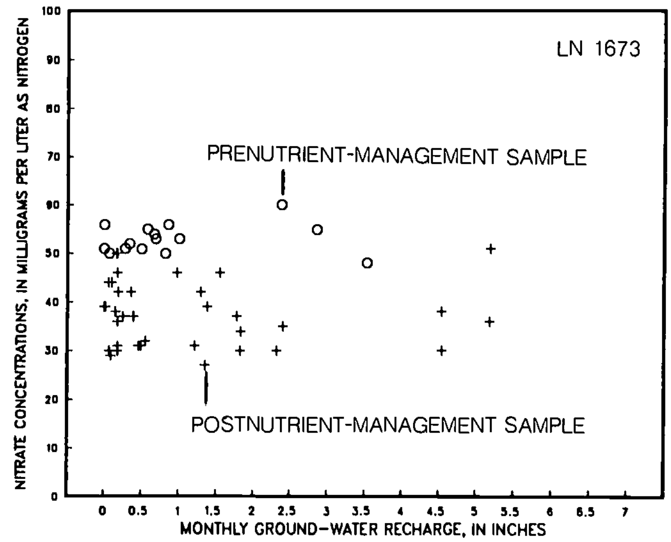

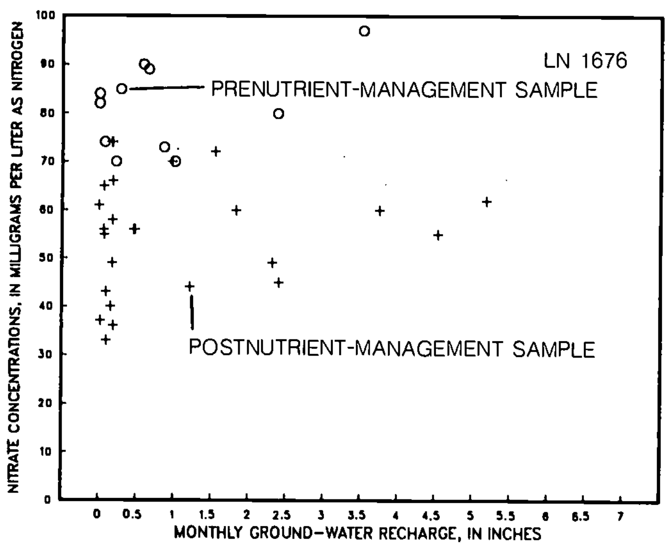

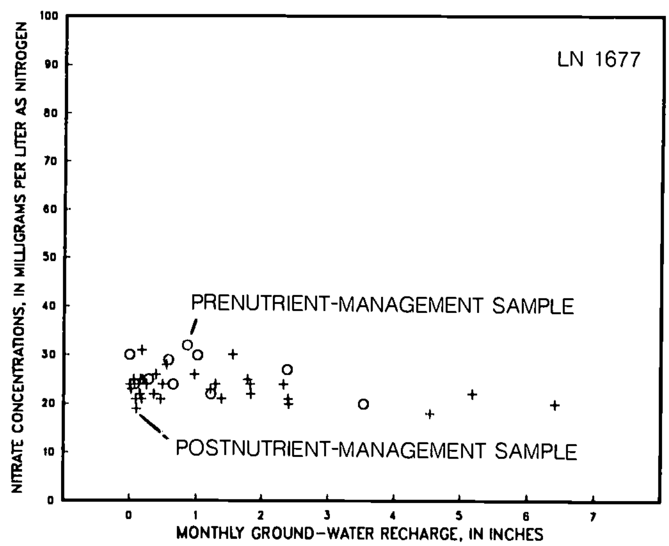

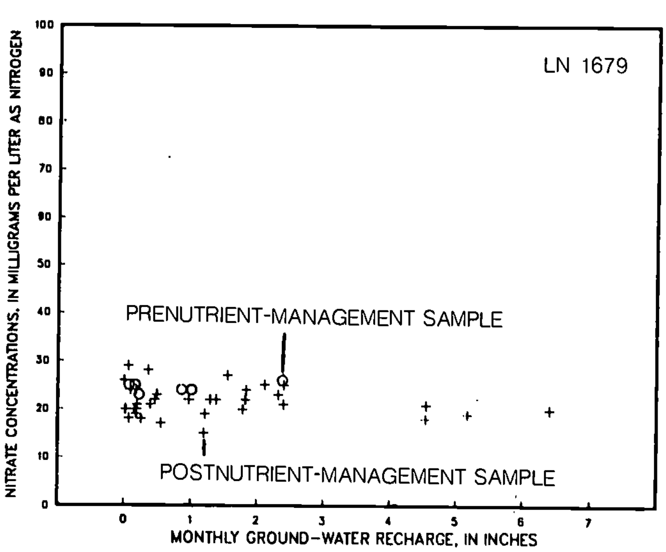

Figure 13. Relations Between Monthly Recharge to Ground Water and Monthly Concentrations of Nitrate in Ground-Water Samples Collected from Wells LN 1669, LN 1673, LN 1676, LN 1677, and LN 1679 During the Prenutrient-Management (1985-1986) and Postnutrient Management (1987-1990) Periods.

TABLE 11. Regressions of Monthly Ground-Water Discharge (MGWD) and Monthly Ground-Water Nitrogen (MGWN) Loads for the Prenutrient-Management and Postnutrient Management Periods.

Period	Equation	R^2	p-Value
Pre-BMP (1985-1986)	MGWN = 457.23 MGWD – 27.4 (1)	0.987	0.001
Post-BMP (1988-9010)	MGWN = 303.09 MGWD – 7.13 (2)	0.996	0.001

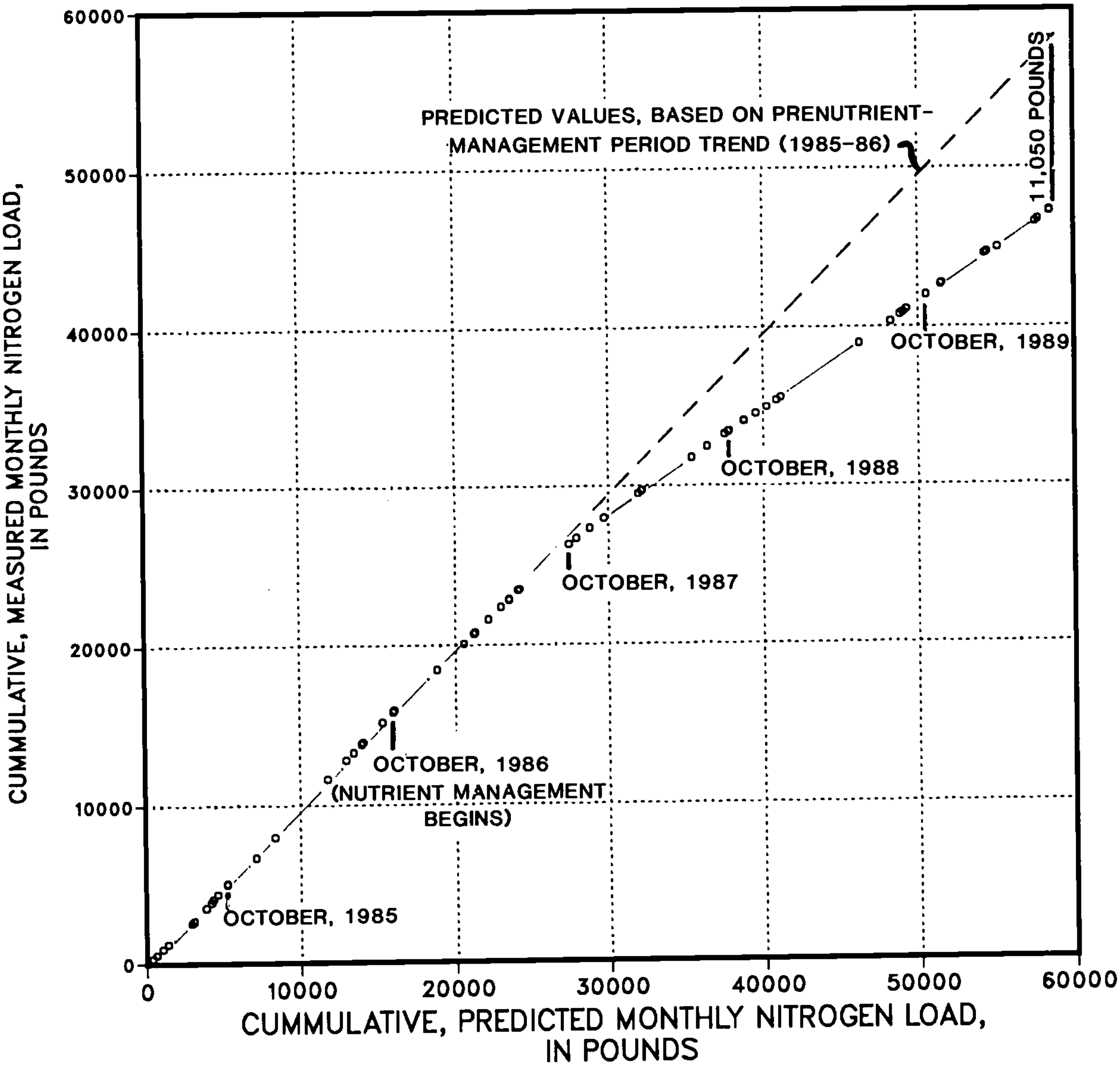

Figure 14. Comparison of Measured and Predicted Cumulative Monthly Loads of Nitrogen for Water Years 1985-1990.

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