Effects of Streambank Fencing of Pasture Land on Benthic Macroinvertebrates and the Quality of Surface Water and Shallow Ground Water in the Big Spring Run Basin of Mill Creek Watershed, Lancaster County, Pennsylvania, 1993-2001

By Daniel G. Galeone, Robin A. Brightbill, Dennis J. Low, and David L. O'Brien

In cooperation with the Pennsylvania Department of Environmental Protection

Scientific Investigations Report 2006–5141

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
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Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	4,047	square meter (m^2)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
gallon (gal)	3.785	liter (L)
cubic foot (ft^3)	0.02832	cubic meter (m^3)
	Flow rate	
foot per second (ft/s)	0.3048	meter per second (m/s)
foot per day (ft/d)	0.3048	meter per day (m/d)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
cubic foot per second per square mile [(ft^3/s)/mi^2]	0.01093	cubic meter per second per square kilometer [(m^3/s)/km^2]
gallon per minute (gal/min)	0.06309	liter per second (L/s)
	Mass	
pound, avoirdupois (lb)	0.4536	kilogram (kg)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

°F=(1.8×°C)+32

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

°C=(°F-32)/1.8

Vertical coordinate information is referenced to the National Geodetic Vertical Datum of 1929 (NGVD 29).

Horizontal coordinate information is referenced to the North American Datum of 1988 (NAD 88)."

Altitude, as used in this report, refers to distance above the vertical datum.

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25 °C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

ABBREVIATED UNITS OF MEASURE	
col/100 mL	colonies per 100 milliliters
lb/mi^2	pounds per square mile
Mft^3	millions of cubic feet
ml	milliliter
μg/m^2	micrograms per square meter

MISCELLANEOUS ABBREVIATIONS	
ANCOVA	Analysis of covariance
AUTO/DI	Automatic versus depth integrated
BMP	Best-management practice
BSAC	Bottom substrate available cover
BSD	Bottom scouring and deposition
BVS	Bank vegetative stability
CCA	Canonical correspondence analysis
CFC	Chlorofluorocarbons
DKN	Dissolved ammonia plus organic nitrogen
DI	Depth integrated
DO	Dissolved oxygen
EPT	Ephemeroptera, Plecoptera, and Trichoptera
FHBI	Hilsenhoff biotic index, family level
HBI	Hilsenhoff biotic index
^{3}H/^{3}He	Tritium/helium
LCCD	Lancaster County Conservation District
LOWESS	LOcally WEighted Scatterplot Smoothing
MAI	Macroinvertebrate aggregated index
MCL	Maximum contaminant level
N	Nitrogen
NA	Not available
NAWQA	United States Geological Survey National Water-Quality Assessment Program
NMP	National monitoring program
NPS	Non-point source
NRCS	Natural Resource Conservation Service
NS	No significant relation
NWQL	United States Geological Survey National Water Quality Laboratory
P	Phosphorus
PaDEP	Pennsylvania Department of Environmental Protection
PCA	Principal component analysis
PDTF	Percent dominant taxa, family level
PDTG	Percent dominant taxa, generic level
PEPT	Percentage of Ephemeroptera, Plecoptera, and Trichoptera
PRRBR	Pool/riffle, run/bend ratio
QC	Quality control
RBP	Rapid bioassessment protocol
SC	Specific conductance
SFR	Scrapers to filterers ratio
SF6	Sulfur hexafluoride
SHRED	Percentage of shredders to total taxa
TKN	Total ammonia plus organic nitrogen
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
VDR	Velocity to depth ratio
WY	Water year (October 1 - September 30)

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Abstract

Streambank fencing along stream channels in pastured areas and the exclusion of pasture animals from the channel are best-management practices designed to reduce nutrient and suspended-sediment yields from drainage basins. Establishment of vegetation in the fenced area helps to stabilize streambanks and provides better habitat for wildlife in and near the stream. This study documented the effectiveness of a 5- to 12-foot-wide buffer strip on the quality of surface water and near-stream ground water in a 1.42- mi^2 treatment basin in Lancaster County, Pa. Two miles of stream were fenced in the basin in 1997 following a 3- to 4-year pre-treatment period of monitoring surface- and ground-water variables in the treatment and control basins. Changes in surface- and ground-water quality were monitored for about 4 years after fence installation.

To alleviate problems in result interpretation associated with climatic and hydrologic variation over the study period, a nested experimental design including paired-basin and upstream/downstream components was used to study the effects of fencing on surface-water quality and benthic-macroinvertebrate communities. Five surface-water sites, one at the outlet of a 1.77-mi^2 control basin (C-1), two sites in the treatment basin (T-3 and T-4) that were above any fence installation, and two sites (one at an upstream tributary site (T-2) and one at the outlet (T-1)) that were treated, were sampled intensively. Low-flow samples were collected at each sites (approximately 25-30 per year at each site), and stormflow was sampled with automatic samplers at all sites except T-3. For each site where stormflow was sampled, from 35 to 60 percent of the storm events were sampled over the entire study period. Surface-water sites were sampled for analyses of nutrients, suspended sediment, and fecal streptococcus (only low-flow samples), with field parameters (only low-flow samples) measured during sample collection. Benthic-macroinvertebrate samples were collected in May and September of each year; samples were collected at the outlet of the control and treatment basins and at three upstream sites, two in the treatment basin and one in the control basin. For each benthic-macroinvertebrate sample: Stream riffles and pools were sampled using the kick-net method; habitat was characterized using Rapid Bioassessment Protocols (RBP); water-quality samples were collected for nutrients and suspended sediment; stream field parameters were measured; and multiple biological metrics were calculated.

The experimental design to study the effects of fencing on the quality of near-stream shallow ground water involved a nested well approach. Two well nests were in the treatment basin, one each at surface-water sites T-1 and T-2. Within each well nest, the data from one deep well and three shallow wells (no greater than 12 ft deep) were used for regional characterization of ground-water quality. At each site, two of the shallow wells were inside the eventual fence (treated wells); the other shallow well was outside the eventual fence (control well). The wells were sampled monthly, primarily during periods with little to no recharge, for laboratory analysis of nutrients and fecal streptococcus; field parameters of water quality also were measured.

Ancillary data collected during the study included precipitation amounts, inorganic and organic nutrient applications in both basins, and the number of cows in both basins. Precipitation during the pre-treatment period averaged about 5 in. more per year than during the post-treatment period; streamflow was about 56-63 percent less during the post-treatment period relative to the calibration period. Agricultural activity did show some changes from the pre- to post-treatment period. The estimated amount of nitrogen (N) and phosphorus (P) applied to the land as inorganic and organic fertilizers decreased 27 and

33 percent, respectively, from the pre- to the post-treatment period in the treatment basin. Over the same period, estimated N and P applications in the control basin decreased by 3 percent and increased by 7 percent, respectively. The number of cows decreased from the pre- to post-treatment period, primarily during the latter part of the study. The control basin showed an approximate 50-percent decrease in cow numbers over the last 2 years; the treatment basin showed a similar decrease during the last year of the study.

Improvements relative to control or untreated sites in surface-water quality (nutrients and suspended sediment) during the post-treatment period were evident at the outlet (T-1) of the treatment basin; however, a tributary site (T-2) (0.36 mi^2 drainage) showed reductions only in suspended sediment. N species at the outlet showed reductions of 18 percent (dissolved nitrate) to 36 percent (dissolved ammonia); yields of total P were reduced by 14 percent. Conversely, the tributary site showed increases in N species of 10 percent (dissolved ammonia) to 43 percent (total ammonia plus organic N), and a 51-percent increase in yield of total P. The average reduction in suspended-sediment yield for the treated sites was about 40 percent.

The results indicated that effects on suspended sediment were fairly consistent in the treatment basin, but this was not true for nutrients. The cumulative effect of 2 miles of fencing in the treatment basin helped to reduce nutrient yields at the outlet; in the upper parts of the treatment basin, however, other factors affected measurable water-quality improvements. Two factors were evident at T-2 that helped to overshadow any positive effects of fencing on nutrient yields. One was the increased concentration of dissolved P in shallow ground water. This influx of P through the ground-water system partially helped to increase P yield during the post-treatment period at T-2. This indicates that nutrient management in a basin is critical to reducing P yields, and that streambank fencing with small buffer widths cannot compensate for increased dissolved P moving to the stream system through shallow subsurface zones. Another factor that appeared to affect water quality at T-2 was that the cattle crossings were embedded in the stream, which was necessary for a drinking-water supply for the cattle and was less costly than installation of culverts and raising the crossing above the stream. Cattle excretions at the crossings appeared to increase concentrations of dissolved ammonia plus organic N and dissolved P. This factor would be one reason to install crossings using culverts if at all possible, but an alternative water supply would need to be provided for the animals.

After the fencing was installed, the treated sites sampled for benthic macroinvertebrates showed improvement relative to control sites in riparian and instream habitat as assessed through Rapid Bioassessment Protocols (RBP III). Habitat characteristics such as bank stability, bottom substrate available cover, and bottom scouring and deposition all showed relative improvements at the outlet and upstream sites in the treatment basin.These improvements were attributable to the fence keeping the cows out of the stream and allowing the vegetation to establish itself and stabilize the banks. Water-quality data collected during the benthic-macroinvertebrate sampling, along with data collected for the surface-water aspect of this study, indicated suspended-sediment loads decreased at treated sites relative to control sites during the post-treatment period. This suspended-sediment reduction helped to cause some of the habitat improvements detected in the treatment basin.

Using the macroinvertebrate metric data at the generic- and family-identification levels also showed improvement at treated sites relative to control sites during the post-treatment period. The treatment sites showed a relative increase in taxa richness and in the Ephemeroptera, Plecoptera, and Trichoptera (EPT) index, and a decrease in the percent oligochaetes during the post-treatment period. Responses were varied in other biological metrics, such as the Hilsenhoff Biotic Index (HBI), which showed improvement at the outlet of the treatment basin, but not at the upstream sites. Overall, slightly more improvement in structure of the benthic-macroinvertebrate community was detected at the outlet of the treatment basin relative to upstream sites. More detected improvement at the outlet could have been because of more overall area to habitat because the outlet sites had a larger stream width and deeper pools and riffles than the upstream sites.

Ground-water data for the shallow wells in the treatment basin showed markedly different flow patterns. The shallow ground-water flow system appeared to be controlled by bedrock geology, and the shallow and deep ground-water flow systems were not well-connected. Shallow ground-water flow at the nest at T-2 showed ground water contributing to the flow of the stream; at the T-1 well nest, however, the stream was actually losing water to the shallow ground-water system.

The difference in shallow ground-water flow patterns between the two well nests caused water-quality improvements during the post-treatment period to be mainly evident only at the T-2 well nest. This site, where shallow ground water was contributing to streamflow, showed relative improvements in water temperature, dissolved oxygen, N species, and counts of fecal streptococcus for treated wells during the post-treatment period. Concentrations of dissolved P in these wells did not show improvement during the post-treatment period, primarily because of an upland source of P from an agricultural field affecting these wells during the post-treatment period. Nevertheless, the relative improvements for the shallow wells at T-2 indicated that, even though the buffer width was small, there was still a noticeable improvement in the quality of shallow ground water. Improvements to the quality of shallow ground water because of streambank fencing, however, appeared to be dependent on the flow paths of that water.

Given the small buffer width within the fenced area (5 to 12 ft), it was unclear from this study to what extent water-quality changes would occur. Results of the study indicated that even a small buffer width can have a positive influence on surface-water quality, benthic macroinvertebrates, and near-stream shallow ground-water quality. Results do show, however, that streambank fencing in itself cannot alleviate excessive nutrient inputs that may be transported through subsurface zones into the stream system. Overland runoff processes that move suspended sediment to the stream can be controlled (or reduced) to some extent by the vegetative buffer established inside the fenced area.

Introduction

Nonpoint-source (NPS) contamination of water resources used for public and private drinking-water supplies, livestock watering, and aquatic and wildlife habitat has been documented in studies in carbonate rock, agricultural areas of the lower Susquehanna River Basin (Lietman and others, 1983; Ward, 1987). Agriculture is the predominant land use in the Mill Creek Basin of Lancaster County, Pa. Areas used to pasture animals in the Mill Creek Basin are commonly adjacent to streams so the animals have a readily available supply of drinking water. Streambank fencing to exclude animal access is a best-management practice (BMP) targeted to reduce suspended-sediment and nutrient inputs to streams by reducing direct nutrient inputs to streams and stopping streambank trampling. Streambank fencing also promotes revegetation of the banks and the development of a riparian zone by eliminating animal access to the fenced area.

Pastured areas have been identified as nonpoint sources of suspended sediment and nutrients to streams (McLeod and Hegg, 1984; Edwards and others, 1996). Livestock trampling of streambanks increases bank erosion (Kauffman and others, 1983). Livestock also can change physical soil properties in grazed areas by increasing soil compaction (Alderfer and Robinson, 1949; Orr, 1960; Bryant and others, 1972), which causes decreases in soil-infiltration rates and subsequent increases in overland flow (Rauzi and Hanson, 1966). Development of a vegetative buffer along each side of the stream is used to stabilize streambanks, thereby reducing bank erosion (Rogers and Schumm, 1991) and potentially reducing the input of nutrients to the stream channel by filtration of overland flow (Pearce and others, 1997) and through the retention of nutrients in the subsurface of the riparian zone (Jacobs and Gilliam, 1985; Lowrance, 1992; Nelson and others, 1995). Buffer strips along stream channels also can help reduce sediment loss from agricultural watersheds (Cooper and others, 1987; Dillaha and others, 1989; Parsons and others, 1994a).

Minimal buffer-width recommendations for optimum nitrogen (N) reduction in stream waters range from 15 ft (Osmond and Gilliam, 2002) to 65 ft (Fennessey and Cronk, 1997). For buffer widths beyond 65-80 ft, there is no effective increase in N reduction (Fennessey and Cronk, 1997). Kansas State University (Barden, 2001), however, recommends a buffer width of no less than 30 ft with optimal being 150 ft. The riparian-zone width for pastures is recommended to be a minimum of 15 ft, which keeps cows out of streams and reduces streambank degradation and nutrient deposition from cattle excretions (Osmond and Gilliam, 2002). Parsons and others (1994b) showed that a 13-ft grassed buffer strip could reduce phosphorus (P) loads during storm events by as much as 50 percent; others found grass buffers (widths ranging from 13 to 30 ft) could reduce P loads during storm events by 61 to 83 percent (Young and others, 1980; Dillaha and others, 1989). Parsons and others (1994b) also found that 13-ft grassed buffer strips were sufficient in most cases to reduce sediment loads during storms by greater than 50 percent. Others found total solids reduced from 67 to 84 percent by installation of grassy buffer strips (widths of 13 to 30 ft) downgradient of agricultural practices (Young and others, 1980; Dillaha and others, 1989). The recommended buffer widths are for optimum conditions when buffer widths can be expanded without significantly inhibiting available pasture. Pasture land in Lancaster County along stream channels is limited; therefore, the buffer widths usually can not meet recommended optimum widths. For this study, buffer width ranged from 5 to 12 ft.

Not only buffer width but buffer type plays a large role in pollutant reduction into surface waters. Forested buffers are more efficient at retaining agricultural chemicals and N; shrubby and grassy vegetation are better for P removal (Fennessey and Cronk, 1997; Tjaden and Weber, 1998; Connecticut River Joint Commissions, 1998) and sediment trapping (Parsons and others, 1994a, 1994b). Revegetation inside fenced areas for this study area consisted of herbaceous vegetation.

The water-quality effects of specific BMP implementation, such as streambank fencing, on a basin scale are not well documented because land uses within a basin typically are mixed (Mostaghimi and others, 1989) and commonly several BMPs are implemented on a basinwide scale. The water-quality effects of multiple BMP implementation on a basin scale have been studied (Walker and others, 1995; Edwards and others, 1996), and some studies have quantified the effects of specific BMP implementation, such as pipe-outlet terracing (Lietman and others, 1997) and nutrient management (Koerkle and Gustafson-Minnich, 1997).

The quantification of BMP effects on water quality is critical to agencies or programs concerned with water resources. For example, the Chesapeake Bay Program has developed a basin model that requires data on the effectiveness of BMP implementation on reducing nutrient loads to receiving waters (Chesapeake Bay Program, 1992). This model has led to the development of a tributary nutrient-reduction strategy in states within the Chesapeake Bay Basin. Pennsylvania's nutrient-reduction strategy was published in 2002 (Pennsylvania Department of Environmental Protection, 2002). Tributary strategies from each state need quantifiable results from BMP implementations to determine the percentage reduction that could be realized with the development of farm-management plans within the basin.

Studies quantifying the effectiveness of BMPs have typically not had ideal study designs, primarily because of the lack of experimental controls at the basin scale. The ability to regulate agricultural practices over an extended time period on a basinwide scale is very difficult. The basins for this study were chosen because of their similarities in hydrology and geology and because of the apparent presence of a stable agricultural community that historically had not deviated from year-to-year farming practices. This relative constancy is critical to the study, because major changes in agricultural activities could make it difficult to detect changes in water quality caused by streambank fencing.

The paired-basin monitoring design has been shown to be one of the more reliable methods of potentially documenting BMP effectiveness in improving water quality (Clausen and Spooner, 1993; Clausen and others, 1996). This approach requires the use of two relatively similar basins with one basin used as a control and a second basin in which treatment is applied. A concurrent calibration period in the two basins is required so that hydrologic and water-quality relations between the basins can be documented prior to any BMP implementation. For this study, data were collected for approximately 8 years (1993-2001) in two small, paired basins to determine the water-quality effects of streambank fencing in pastureland within the treatment basin.

This project described in this report was a cooperative effort between the U.S. Geological Survey (USGS) and the Pennsylvania Department of Environmental Protection (PaDEP). Funds from the USGS were available through the USGS Federal-State Cooperative Water-Resources Program, which is a program designed to provide information that forms the foundation for many of the Nation's water-resources management and planning activities. The project was funded by PaDEP through the National Monitoring Program (NMP) of the U.S. Environmental Protection Agency (USEPA). The NMP stems from Section 319 of the 1987 amendment to the Clean Water Act. The NMP was developed to document the effects of NPS pollution-control measures and associated land-use modifications on water quality (Osmond and others, 1995).

Purpose and Scope

This report describes the effects of streambank fencing in an agricultural basin on the chemical, physical, and biological components of the surface-water system and also the effects on the chemical and bacteriological quality of the near-stream shallow ground-water system. The study area included a control and treatment basin within the Mill Creek Basin of Lancaster County, Pa. The report discusses the pre-treatment relation developed from October 1993 through mid-July 1997 for surface water in the control and treatment watersheds and compares that pre-treatment relation to the post-treatment data collected from mid-July 1997 through June 2001. Data on land use, hydrology, and water quality (chemical, physical, and biological) are presented to determine effects of streambank fencing relative to paired-basin and upstream-downstream analyses. Changes from the pre- to post-treatment periods in the concentrations and yields of nutrients and suspended sediment during low flow and storm events at treated relative to control sites were quantified using analysis of covariance (ANCOVA). Field water-quality properties were measured and counts of fecal streptococcus were made during low-flow sampling, and any changes in these characteristics in the treatment basin during the post-treatment period were quantified using ANCOVA. The effects of fencing on the benthic-macroinvertebrate community were determined by comparing sites in the treatment and control basins using canonical correspondence analysis (CCA). Explanatory variables used in the CCA included physical (such as stream cover) and chemical (such as nutrients) stream characteristics. Benthic-macroinvertebrate indices also were used to assess the effects of streambank fencing. Effects on the shallow ground-water system were limited to those identified in the treatment basin, because no wells were available for sampling in the control basin. Shallow wells were installed near the stream (treatment area) and away from the stream (outside the fenced area) at two locations in the treatment basin. Changes from the pre- to post-treatment periods in the concentrations of nutrients, fecal-streptococcus colonies, and field characteristics at treated wells compared to control wells were quantified using ANCOVA.

Study Area Description

The two adjacent study basins, similar in land use, were within the Big Spring Run Basin, a subbasin of Mill Creek Watershed of Lancaster County (fig. 1). The control basin was 1.77 mi^2 with 2.7 total stream miles and 1.9 mi of stream running through open pasture. The treatment basin was 1.42 mi^2 with 2.8 total stream miles and 2.0 mi of stream running through open pasture. The elevation at the outlet of the study basins was approximately 290 ft the highest elevations in the control and treatment basins were about 490 and 460 ft, respectively. The stream gradient was similar in both basins. In the lower part of the basins near the outlets, stream elevation decreased about 1 ft for every 100 ft of channel. In the upper parts of the basins, the gradient approached 1.5-2 ft of elevation change for every 100 ft of channel.

The climate of the study basins is typical of a temperate-zone climate. The average precipitation recorded at a long-term meteorological site 2 mi to the northeast of the study area was 41 in. The average temperature was about 52 °F (National Oceanic and Atmospheric Administration, 1994).

Land Use

Land use in the basins was predominantly agricultural during the study period. Geographic information system (GIS) data from 1993 indicated land use in the treatment basin was 89 percent agricultural, 7 percent residential/commercial, and 4 percent forested. Land use in the control basin was 81 percent agricultural, 12 percent residential/commercial, and 7 percent forested (fig. 1). The agricultural land in both basins

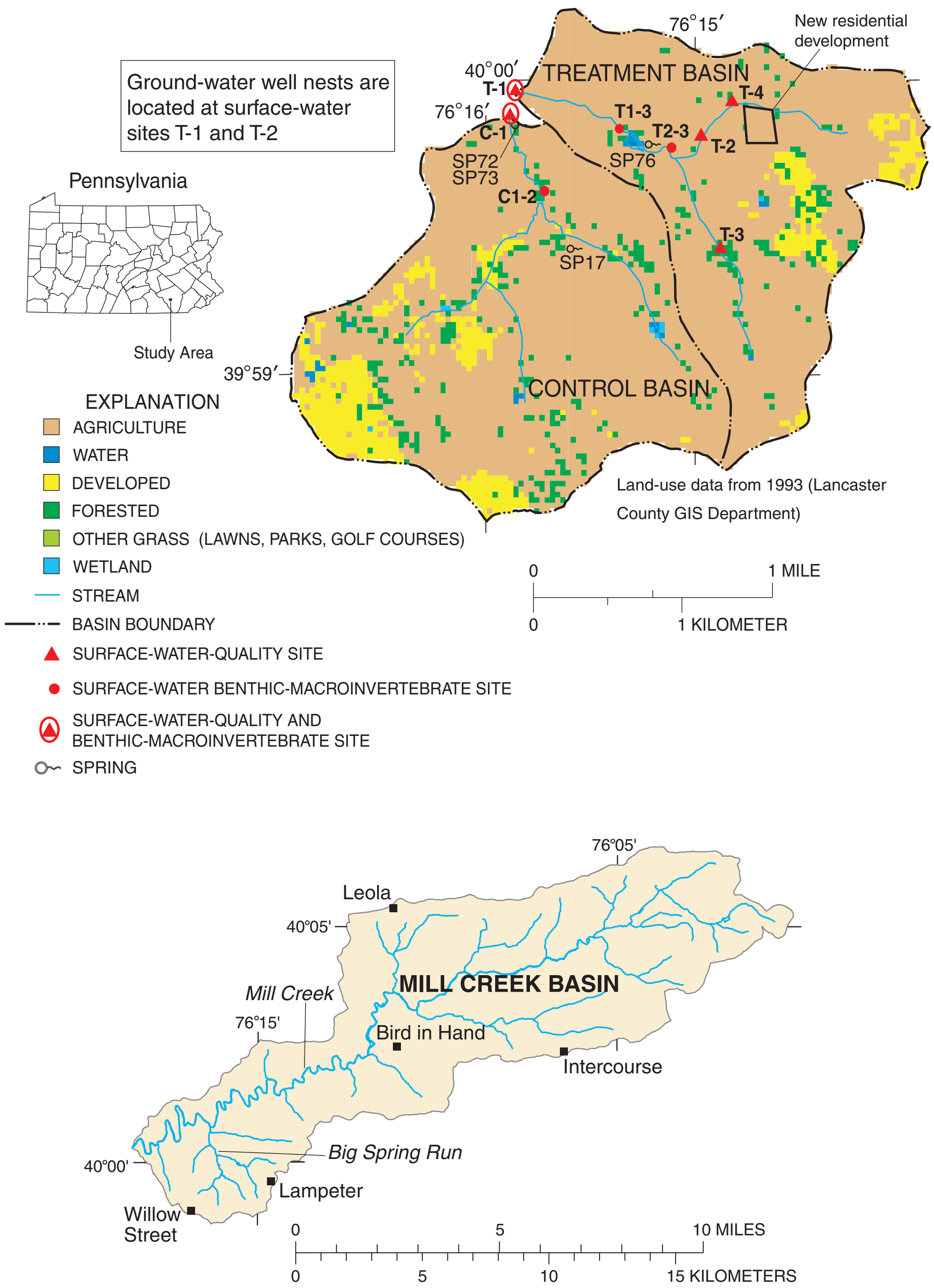


Figure 1. Land-use map of study area and location of surface-water sites, ground-water well nests, and selected springs in the Big Spring Run Basin, Lancaster County, Pa.

was predominantly row crop (about 60 percent); the remaining part was primarily pasture or hay fields. Agriculture in the two basins consisted of about 14 major farming operations that had corn and alfalfa as the primary row crops. Dairy-cattle husbandry was the predominant form of animal agriculture in the basins. A few horses were pastured in the treatment basin (fewer than 10) and there was one chicken-raising operation for a few years in the upper part of the treatment basin, but dairy cattle made up over 95 percent of the total animal units in both basins. Each basin had about 50-55 acres of pasture. This pasture land accounted for about 7 percent of the total amount of agricultural land in both basins.

Hydrologic Setting

The Mill Creek Basin lies within the Susquehanna River Basin. The broad valleys in northern Lancaster County are drained by an elaborate, branched network of meandering streams. Although the Vintage, Antietam, and Harper Formations form a small ridge within the study area, Big Spring Run in the control basin and an unnamed tributary to Big Spring Run in the treatment basin bisect the ridge with little or no deviation in their flow direction.

Geology

The rocks that underlie the study area consist of dolomite (Vintage Formation), limestone (Conestoga Formation), quartzite and schist, or phyllite (Antietam and Harpers Formations) (Berg and others, 1980) (table 1). The topography consists of broad rolling hills and valleys with a low to moderate relief.

Geohydrology

The Big Spring Run Basin is underlain by a sequence of carbonate and siliciclastic rocks of Cambrian age covered by a thin layer of soil and a mantle of regolith derived from weathered bedrock (table 1). The ground-water/surface-water system that has developed is complex. This system is controlled by the bedrock geology but is driven by the timing, duration, and intensity of precipitation events. Although the ground-water flow system is well connected to the surface-water system, the exact flow paths and traveltimes remain relatively unknown.

The ground-water system in the study area was characterized on the basis of water levels, flow directions, age dating, and chemical quality. The three-dimensional configuration of the ground-water system was interpreted from well logs and the results of previous studies. Water-level data from 8 wells and 17 piezometers were used to characterize ground-water flow directions in local reaches of unnamed tributaries to Big Spring Run. Results of water-quality analyses were used to relate agricultural activities (manure application and animal grazing) and recharge events to ground-water quality before and after the establishment of a riparian buffer through the use of fencing.

Structural Framework

Approximately 90 percent of the study area (control and treatment basin) is underlain by limestone of the Conestoga Formation (fig. 2). The dolomite of the Vintage Formation, and the siliciclastics of the Antietam and Harpers Formations form a narrow finger that extends east to west into the central part of the basin.

The study area lies within the southern part of the Lancaster belt. The Lancaster belt is a structurally complex zone of repeated deformation, faulting, and steep isoclinal folding. Poth (1977, p. 23) noted that the Vintage Formation is generally cut or bounded by faults. Meisler and Becher (1971, figs. 3 and 4) found that the folding is characterized by steeply dipping cleavages that strike east or northeast.

Table 1. Hydrogeologic-stratigraphic column, Mill Creek Watershed, Lancaster County, Pennsylvania

(Data from Berg and others, 1980).

Age	Geologic Formation	Geologic description	Quality of water
Lower and Middle Cambrian	Conestoga Formation	Generally a gray finely to coarsely crystalline limestone with clay laminae, thin graphitic and micaceous beds. Conglomeratic at base.	Very hard. Water type - calcium bicarbonate.
Lower Cambrian	Vintage Formation	Consists dominantly of gray, thick-bedded to massive, finely crystalline dolomite. Contains siliceous laminae or thin shale interbeds.	Hard. Water type - calcium magnesium bicarbonate.
Lower Cambrian	Antietam Formation	Light-gray quartzitic sandstone with a calcareous cement.	Soft.
Lower Cambrian	Harpers Formation	Dark-bluish-gray phyllite that may grade laterally into schist.	Soft.

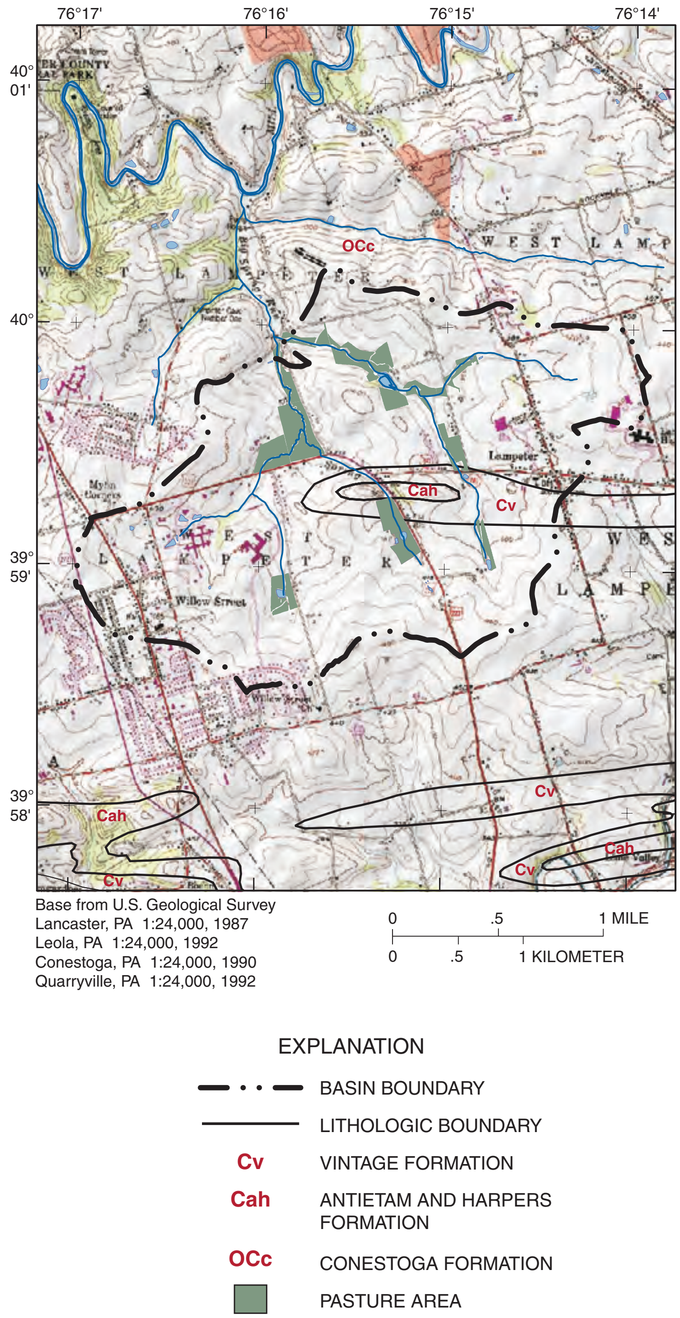


Figure 2. Map of study area, underlying lithology, and pasture locations along stream channels in the Big Spring Run Basin, Lancaster County, Pa.

Geohydrologic Framework

Geology determines ground-water flow paths by the development, orientation, transmissivity, and inter-connectedness of secondary openings (joints, faults, fractures, cleavage planes, bedding planes) that may be filled with silty or clay-rich material and variations in lithology (resistance to physical and chemical weathering). These differences will determine the subsequent formation of topographic features, soil and regolith thickness, and secondary porosity.

The primary source of ground-water recharge to the basin is precipitation. Much of the precipitation, however, returns to the atmosphere through evapotranspiration or reaches the streams as surface runoff or base flow. According to Gerhart and Lazorchick (1988, table 12), 30 to 35 percent of the total annual precipitation recharges the ground water in the immediate vicinity of the study basins. Areas underlain by the Conestoga Formation have higher recharge rates than areas underlain by the Vintage, Antietam, and Harpers Formations. The greater percentage of recharge in areas underlain by the Conestoga Formation is because of (1) lower relief and greater permeability, and (2) the presence of karst features, such as closed depressions, sinkholes, and fissures. Base flow of streams is maintained by ground-water discharge. To the north of the study basins and just west of the city of Lancaster, approximately 77 percent of the total streamflow of Little Conestoga Creek is base flow (Meisler and Becher, 1971, p. 55). Although the study area receives nearly equal amounts of precipitation throughout the year, evapotranspiration is greatest in late spring, summer, and early fall when plants are actively growing. Surface runoff, however, is greatest in the winter when the ground is frozen and lowest in late summer and early fall. The remaining precipitation infiltrates through the soil into the regolith and the underlying bedrock aquifer, flowing from areas of high relief such as hilltops (high hydraulic head) to areas of low relief such as valleys (low hydraulic head), through secondary features.

Soils

Soils in the two study basins are generally similar; six different soil series make up the majority of the soils classified (Custer, 1985). Soils along the ridges and adjacent side slopes are predominantly of the Conestoga series (fine-loamy, mesic Typic Hapludalf), followed by Penlaw (fine-silty, mixed, mesic, Aquic Fragiudalf) and Pequea (coarse-loamy, mixed, mesic Typic Eutrochrept) series. Soils of the Hollinger series (fine-loamy, mixed, mesic Typic Hapludalf) were identified only on the side slopes. The most common series identified in the basins was the Lehigh series (fine-loamy, mixed, mesic Aquic Hapludalf), which was along the lower and middle slopes. Gentle sloping terrain is the most common topography in the basin. The soils adjacent to the stream channel were identified as the Clarksburg series (fine-loamy, mixed, mesic Typic Fragiudalf). All the soils are deep and moderately to well drained, except for the Penlaw series, which is deep and poorly drained. The reported soil depths range from 50 to 75 in. (Custer, 1985). Slopes are low to moderate, ranging from 0 to 15 percent, but more commonly from 3 to 8 percent. Shuford and others (1977) have determined that many limestone soils in Pennsylvania used for agriculture are highly structured and may contain micropores (small voids between soil granules) and larger macropores (developed by root growth and worm tunnelling).

Regolith

The thickness and composition of the regolith depends on the composition of the bedrock that weathers to form the saprolite and the amount and type of material subsequently eroded and deposited as colluvium and alluvium. The thickness of regolith is greatest where the Conestoga Formation contains substantial amounts of sand, silt, or clay. Regolith is thinnest where the bedrock consists of the Antietam and Harpers Formations, where the carbonate rocks contain few impurities, or where bedrock crops out. Dissolution of carbonate rocks can result in the formation of pinnacles, swales, and cavities that can in turn lead to the development of sinkholes, springs, and other karst features. Colluvium is typically the thickest at the base of hills from which large amounts of sediments have eroded. In the study area, regolith varies from 0 ft at a bedrock outcrop to at least 20 ft. The porosity of the regolith typically exceeds that of the underlying fractured bedrock, permitting infiltration of precipitation and storage of large quantities of water. The intergranular pores then slowly release the unconfined water to wells, base flow in streams, and the underlying fractured bedrock.

Fractured Bedrock

The quantity of ground water stored in fractured bedrock is limited; however, the fractures may be recharged by seepage if the overlying regolith is saturated and the network of fractures extends to the saturated regolith. Primary porosity is negligible. Secondary porosity depends on the presence of secondary openings such as fractures, bedding planes, joints, faults, and solution cavities and the absence of weathered material that commonly infills these openings. Once secondary openings are formed in the siliciclastics rocks, such features change little. Solution enlargement of secondary openings is concentrated in areas where (1) carbonate material is dominant over noncarbonate material, and (2) water movement is relatively rapid and recharge water is acidic.

Study Design

The study was designed to document changes in surface- and ground-water systems that could result from streambank fencing. A paired-basin and upstream-downstream monitoring design were used. Chemical, physical, and biological water-quality samples were collected prior to and after streambank-fence installation.

Experimental Design

A nested experimental design was used to study streambank-fencing effects. The paired-basin monitoring design requires the use of two relatively similar basins with one basin used as a control and a second basin in which treatment is applied. The criteria for acceptable basins was proximity and similarity to each other and the presence of an agricultural community willing to participate in the project. Once adjacent basins were identified, it was assumed the physical characteristics were generally similar (and climatic variation between the adjacent basins would be minimal). A calibration period between the two basins is required so that water-quality relations between the basins can be documented prior to any BMP implementation. Secondary approaches to documenting surface-water-quality changes included collecting pre- and post-treatment data at sites within the treatment basin and monitoring of sites upstream and downstream of fence installation.

Both paired-basin and upstream-downstream monitoring designs help account for climatic and hydrologic variability when monitoring before and after a specific event (Spooner and others, 1985), in this case, streambank-fence installation. The design for this study incorporated multiple opportunities for comparisons to ensure that effects of fencing could be documented (table 2). It is not uncommon for land uses to change in agricultural areas of Lancaster County as residential development progresses. The ability to compare numerous sites in the treatment and control basin can alleviate problems associated with changes in land use. Eight surface-water locations were sampled; four were continuous-recording stations (C-1, T-1, T-2, and T-4) and four were intermittent stations (C1-2, T1-3, T2-3, and T-3) (fig. 1). An intermittent station is a station at which a discharge measurement must be made when water-quality samples are collected because there is no stage datum.

Surface-water station T-4 was operational by January 1995. The original study design did not include T-4. A new residential development began construction during 1994. This development was upgradient of station T-2. Station T-2 was in one of the most intensively grazed pastures in the treatment basin. In order to factor out any impacts of the development on water quality, a new continuous station was installed approximately 1,500 ft upstream from station T-2 and directly downstream of the residential development. Stormwater runoff from the development was discharged to the stream channel after passing through a sediment-retention pond. This discharge point was about 50 ft upstream of surface-water site T-4. The development was supplied with public water and all wastewater was transported out of the development to the local treatment plant.

Five surface-water sites were sampled for benthic macroinvertebrates (fig. 1), – three sites in the treatment basin and

Table 2. Description of surface-water sampling sites in the Big Spring Run Basin, Lancaster County, Pa., water samples collected at each site, and use of data in project design.

[bs, base-flow samples; st, stormflow samples; bn, benthic-macroinvertebrate samples]

Site	Drainage area, square miles	Description	Water samples collected	Data use
C-1	1.77	Outlet of control basin	bs, st, bn	Compare to T-1 and T-2 for paired-basin analysis
C1-2	1.62	Upstream site in control basin	bn	Benthic-macroinvertebrate sampling location for comparison with T1-3 and T2-3
T-1	1.42	Outlet of treatment basin	bs, st, bn	Compare to C-1 for paired-basin analysis and T-3 for upstream-downstream analysis
T1-3	1.21	Upstream site in treatment basin	bn	Benthic-macroinvertebrate sampling location for comparison with C1-2
T-2	.36	Visually degraded upstream tributary site in treatment basin	bs, st	Compare to C-1 for paired-basin analysis and T-4 for upstream-downstream analysis
T2-3	1.13	Upstream site in treatment basin	bn	Benthic-macroinvertebrate sampling location for comparison with C1-2
T-3	.33	Upstream site in treatment basin above most pasture land (approximately 1,000 feet of stream is fenced above T-3)	bs	Compare to T-1 for upstream-downstream analysis
T-4	.32	Upstream tributary site in treatment basin downstream of new residential development and above all pasture land	bs, st	Compare to T-2 for upstream-downstream analysis

two in the control basin. The outlets of the treatment basin (T-1) and the control basin (C-1) were sampled. The remaining sites were in the upper parts of the study basin (table 2). Differences in habitat based on stream size required that sites at the outlets be compared separately from sites in the upper parts of the basins.

Two nests of ground-water wells were installed in the treatment basin to document effects of riparian vegetation on shallow near-stream ground-water quality (fig. 1). These wells were placed at the outlet of the treatment basin (adjacent to surface-water station T-1) and upstream adjacent to surface-water station T-2. At each well nest, two shallow ground-water wells were placed near the stream channel (treatment wells) and one shallow well was at a distance away from the channel (control well) that was outside the fenced area (table 3). A deep well also was installed at each well-nest location so that any changes in deeper ground-water quality could be documented during the study period. A paired-comparison approach was used to determine if streambank fencing affected the wells inside the fence as opposed to the well outside the fenced area. In addition, results of analyses of samples from the wells were used to explain observed changes in low-flow stream quality before and after fencing.

In order to better understand shallow ground-water flow paths and characterize near-stream shallow ground-water quality, a network of piezometers was installed in the treatment and control basins (fig. 3). These piezometers were not installed until 1998, so no pre-treatment data were available. Four piezometers were near surface-water site T-1 and seven were near surface-water site T-2 in the treatment basin. Another six were near surface-water site C-1 (table 4). The piezometer network was within and near the stream channel at each site. Water-level measurements at the piezometers and shallow wells, along with the gage height from the surface-water sites, were used to determine the hydraulic potential for ground-water movement toward and away from the stream.

Implementation of Best Management Practices

Fence was installed in the treatment basin from May 1997 through July 1997. Approximately 2 mi of stream length were fenced to prevent cows from accessing the stream channel in pastured areas (fig. 2). All pasture areas in the treatment basin along the stream network were fenced. One- or two-strand high-tensile wire was used with an electrical current supplied by solar power. On either side of the stream, the distance between the streambank and the fence was anywhere from 5 to 12 ft. For each pasture fenced, approximately two cattle crossings were installed to allow the animals to access pasture and also to supply the cows with an area for water consumption. After fence installation, a variety of brushy, herbaceous vegetation was naturally established (fig. 4).

Table 3. Description of ground-water wells in the treatment basin of the Big Spring Run Basin, Lancaster County, Pa.

[NGVD29, National Geodetic Vertical Datum of 1929; DIfference between top and bottom of opening is equal to screened interval, except for LN-2039 and LN-2043, which were completed open hole]

U.S. Geological Survey identification number	Surface-water site location (and local identification number)	Well location relative to fence	Well depth (feet below land surface)	Casing length (feet)	Altitude of land surface (feet above NGVD29)	Altitude of opening (top/bottom) (feet above NGVD29)
LN-2037	T-2 (W1)	inside	6	3.4	347.97	344.57/343.47
LN-2038	T-2 (W2)	inside	6.6	3.9	347.84	343.94/342.84
LN-2039	T-2 (W3)	inside	63	19	348.18	329.18/285.18
LN-2040	T-2 (W4)	outside	7.6	5.0	348.26	343.26/342.06
LN-2041	T-1 (W5)	inside	6	3.0	294.78	291.78/290.68
LN-2042	T-1 (W6)	inside	12	9.2	294.75	285.55/284.55
LN-2043	T-1 (W7)	inside	100	17	295.59	278.59/195.59
LN-2044	T-1 (W8)	outside	8	4.4	296.84	292.44/291.34

Table 4. Physical characteristics of piezometers in the Big Spring Run Basin, Lancaster County, Pa.

[NGVD29, National Geodetic Vertical Datum of 1929]

U.S. Geological Survey identification number	Surface-water site location (and local identification number)	Piezometer depth (feet below land surface)	Piezometer length (feet)	Altitude of land surface (feet above NGVD29)	Altitude of opening (top/bottom) (feet above NGVD29)
LN-2070	T-1 (P1)	2.92	1.92	291.91	289.99/288.99
LN-2071	T-1 (P2)	3.03	2.03	292.99	290.96/289.96
LN-2072	T-1 (P3)	4.01	3.01	294.09	291.08//292.08
LN-2073	T-1 (P4)	4.81	3.81	294.35	290.54/289.54
LN-2074	T-2 (P5)	4.00	3.00	345.18	342.18/341.18
LN-2075	T-2 (P6)	4.07	3.07	346.04	342.97/341.97
LN-2076	T-2 (P7)	4.22	3.22	347.83	344.61/343.61
LN-2077	T-2 (P8)	5.19	4.19	347.94	343.75/342.75
LN-2078	T-2 (P9)	4.24	3.24	345.22	341.98/340.98
LN-2079	T-2 (P10)	5.32	4.32	345.93	341.61/340.61
LN-2080	T-2 (P11)	6.23	5.23	345.35	340.12/339.12
LN-2090	C-1 (P20)	6.21	5.21	298.37	293.16/292.16
LN-2091	C-1 (P21)	4.19	3.19	294.90	291.71/290.71
LN-2092	C-1 (P22)	4.22	3.22	294.96	291.74/290.74
LN-2093	C-1 (P23)	5.05	4.05	295.76	291.71/290.71
LN-2094	C-1 (P24)	5.19	4.19	296.59	292.4/291.4
LN-2095	C-1 (P25)	6.24	5.24	294.93	289.69/288.69

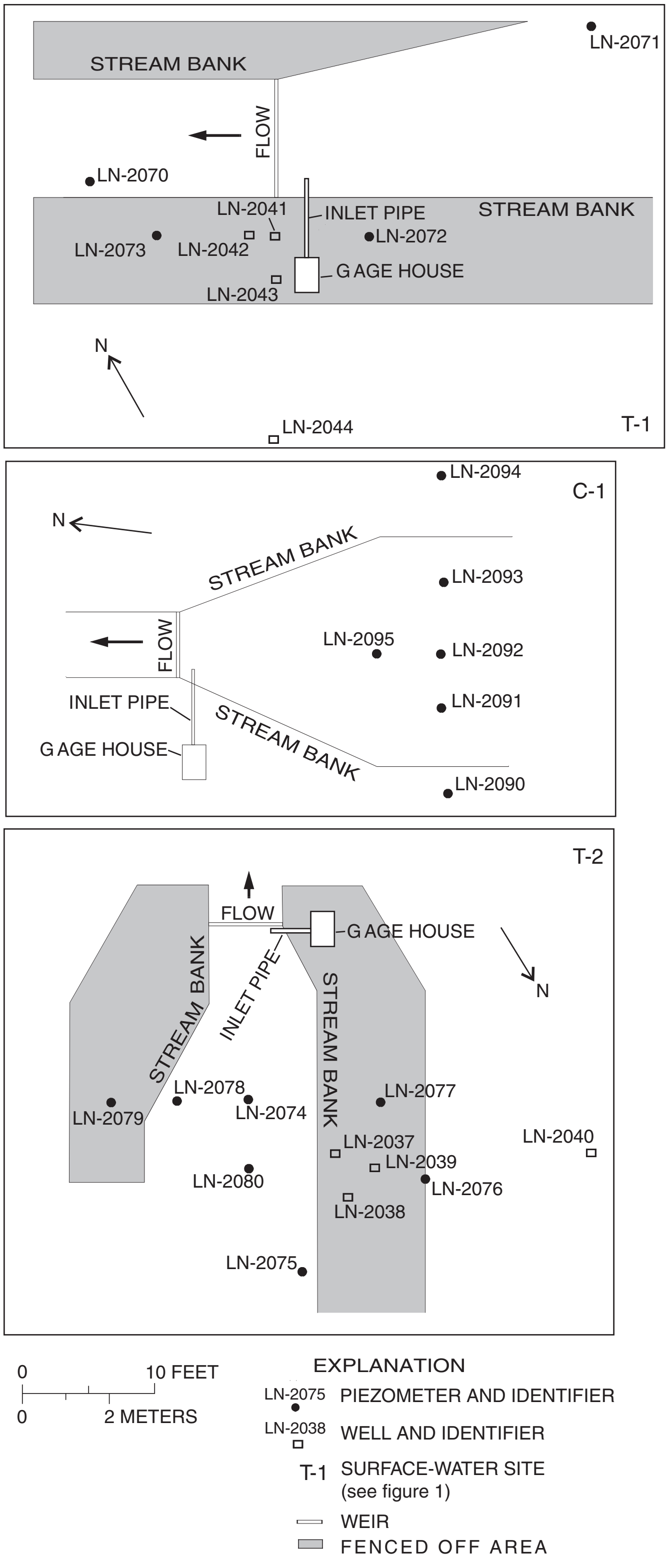


Figure 3. Maps of piezometer and well locations in the Big Spring Run Basin, Lancaster County, Pa.

Pre-treatment photo, May 1996

Post-treatment photo, May 1998

Figure 4. Tributary site (T-2) in treatment basin in the Big Spring Run Basin, Lancaster County, Pa., before (top) and after (bottom) fence installation.

Data Collection and Analysis

Pre-treatment data were collected primarily from October 1993 until July 15, 1997. Although some grab samples were collected in the study basins prior to October 1993, no storm samples were collected until October 1993. The other exception to this protocol was that benthic-macroinvertebrate samples were first collected in September 1993. Sampling in September was necessary in order to adhere to standard sampling windows for benthic-macroinvertebrate communities. Thus, the pre-treatment period lasted about 45.5 months. Post-treatment data collection was mostly discontinued by the end of June 2001, so the post-treatment period lasted about 47.5 months.

Ancillary Data

In addition to the collection of surface-water, benthic-macroinvertebrate, and ground-water samples, data also were collected for variables that affect the quantity and quality of the water and habitat sampled. The primary types of ancillary data collected in the study basins were precipitation and agricultural activity.

Precipitation

A precipitation gage was installed and operational by October 1993 at the outlet of the treatment basin (at surface-water site T-1). Precipitation was measured using a weighing-bucket rain gage and were electronically logged at 15-minute intervals during the entire study. The gage was calibrated on an annual basis in order to check data accuracy.

Agricultural Activity

Two types of agricultural-activity data were collected from each farm operator in the study basins. Farm operators provided data on the dairy-cow activity in the pastures and the loading of inorganic and organic fertilizers within the study area. Farmers with pastured animals provided monthly records of the number of cows out to pasture and the time that number of cows spent in each pasture in the treatment and control basins. Monthly estimates of manure and inorganic-fertilizer applications were provided on a field basis so that nutrient loadings could be estimated for the treatment and control basins. Some farms extended into both the treatment and control basins. Each farm operator reported manure (originating from bedding material) applications in “loads.” Prior to and during early parts of the study, each farmer had developed nutrient-management plans with the Lancaster County Conservation District (LCCD) and Natural Resource Conservation Service (NRCS). One aspect of the plan was to calibrate manure spreaders. The information from these calibrations was used to determine the weight of each “load” of manure applied by each farmer. Liquid manure was reported in gallons applied. The amount of manure applied, whether liquid or bedding material, was subsequently converted to pounds of N and P based on published values for concentrations of nutrients in different sources and forms of animal manure (Pennsylvania Department of Environmental Resources, 1986).

The dairy-cow pasture data were used to determine the density of cows in the pastures in both basins over time. The time that cows were in pasture was used to estimate the amount of waste excreted by the animals. These estimates were then added to manure-application data supplied by the farmers so that a total amount of N and P applied to the landscape could be estimated. The nutrient-application data were used to estimate the loading of N and P to both basins over time.

Surface Water

Surface-water samples were collected from eight surface-water sites (table 2). Surface-water samples were collected during low-flow and stormflow conditions. Any grab samples collected at intermittent stations required a discharge measurement at the time of sample collection. Most discharge measurements were made using a pygmy current meter.

Streamflow

A pipe well was installed at each continuous-record surface-water site and data loggers and automatic storm samplers were located within a gage house (fig. 5). The pipe well was dug into the bank along the stream channel with upper and lower intakes protruding from the pipe well into the stream channel. Stream stage height was monitored continuously at the four continuous-record surface-water sites by use of shaft encoders wired to data loggers recording data at either 5-minute or 15-minute intervals. Discharge at each of the four sites was measured through a wide range of stages to develop a relation (a rating curve) between stage and discharge. Most discharge measurements were made using a pygmy current meter, but Price-AA meters also were used when stream stage exceeded 1.5 ft. Changes in channel characteristics during the course of the study necessitated the modification of stage-discharge relations for each continuous station. Data were reviewed at the end of each water year (water years begin October 1 and end September 30 of the next year) in order to determine if the rating curve changed, and if so, shifts to the relation were developed, or a complete new rating was developed.

Water Quality

Water samples for analyses of nutrients and suspended sediment were collected at a fixed-time interval and during storm events. Samples were analyzed for dissolved forms of ammonia, nitrite, ammonia plus organic N (hereinafter referred to as DKN), nitrite plus nitrate, P, and orthophosphate. Analyses also included total forms of ammonia plus organic N

Figure 5. Photo of gage house located at outlet of control basin (C-1) in the Big Spring Run Basin, Lancaster County, Pa.

(hereinafter referred to as TKN) and P, and suspended sediment.

Fixed-time interval (grab) samples were collected every 10 days (regardless of flow conditions) from April through November and on a monthly basis during a low-flow period from December through March. These fixed-time samples were collected at four sites in the treatment basin (T-1, T-2, T-3, and T-4) and one site in the control basin (C-1) (table 2). The more intensive sampling from April through November coincided with the typical period when cows are pastured in south-central Pennsylvania. Fixed-time samples were collected by hand at the downstream side of the weir used to control flow. At the time of sample collection, pH, water temperature, dissolved oxygen (DO), and specific conductance (SC) were recorded by use of a multi-parameter probe placed at the upstream side of the weir after the grab samples were collected. Grab samples were filtered in the field through a 0.45 micron pore-size filter pre-rinsed with deionized water. Samples to quantify the abundance of fecal streptococcus were collected once per month during low-flow conditions, and enumeration analysis was performed according to techniques described by Ehlke and others (1987) for the membrane-filter method and immediate incubation test. This method involves processing the bacteria on KF agar. Three replicates were processed for each fecal streptococcus sample collected in the field.

Storm samples were collected with an automated sampler having either a 72-bottle or 24-bottle capacity. The larger capacity samplers were placed at the outlets of the treatment (T-1) and control (C-1) basins. The 24-bottle samplers were placed at surface-water sites T-2 and T-4. The larger capacity samplers were placed at the outlets of the basins because storm hydrographs at the outlets were of longer duration than storm hydrographs in the upper parts of the treatment basin. Sample collection during a storm event at the outlets was initiated by a float switch that turned the samplers on at a specific stage. This stage height was periodically adjusted by 0.01 to 0.05 ft on the basis of current flow conditions. After initialization, samples were collected every 15 minutes until either the 72 bottles were filled or the stage dropped below the point at which initialization occurred. Sample collection during storm events at T-2 and T-4 was initiated by a data-logger program that sent a pulse to begin sampling if the change in stage exceeded a set threshold over a time period. After sample initiation at T-2 and T-4, samples were collected anywhere from 5 to 30 minutes apart. If stage was rising rapidly, samples were collected every 5 to 15 minutes; otherwise, samples usually were collected every 30 minutes. Sample collection at T-2 and T-4 ended when the automatic sampler was full or if a threshold stage was reached that would stop sampling. Storm samples were retrieved within a day of the completion of the event and chilled prior to sample processing. No preservatives were placed into storm sample bottles prior to sample collection. After defining the storm interval so that similar time intervals and parts of the hydrograph were used for samples collected from each site, the samples from the storms were composited into one storm sample per site per event. Aliquots pipeted from the bottles were flow weighted so the composite sample represented the mean conditions for the storm event. Chemical and suspended-sediment analyses were conducted on the composited samples.

Chilled samples were shipped to the USGS National Water Quality Laboratory (NWQL) in Arvada, Colo., for nutrient analysis. Analyses were performed according to techniques described in Fishman and Friedman (1989). Suspended-sediment concentration analyses were conducted by the USGS Sediment Laboratory in Pennsylvania through water year 1995 and thereafter at the USGS Sediment Laboratory in Kentucky. Both sediment laboratories used procedures described by Guy (1969) to determine suspended-sediment concentrations. The water-quality and sediment laboratories have published quality-assurance plans (Friedman and Erdmann, 1982; Pritt and Raese, 1995; Sholar and Shreve, 1998).

Nutrient and suspended-sediment yields for low-flow and stormflow samples were determined for each sample collected so that pre- and post-treatment comparisons could be conducted. The yield for low-flow samples was determined by multiplying the measured concentration by the daily discharge value for that site, then dividing by the drainage area. This was called an instantaneous yield for brevity. Instantaneous yield data were estimated in units of pounds per day per square mile. This instantaneous yield, in general, is equal to a daily yield because it was assumed the daily discharge for that day did not change. Stormflow yields were determined for each storm in units of pounds per square mile. Stormflow yields were estimated by multiplying the sampled concentration for that storm by the mean discharge for the storm, then multiplying by the total storm length (in days) and dividing by the drainage area.

Data Analysis

Annual yields (loads per unit area) of nutrients and suspended sediment for the four surface-water sites (C-1, T-1, T-2, and T-4) were estimated by combining the non-storm (hereinafter referred to as low flow) and stormflow loads. Low-flow and stormflow loads were computed using a multiple regression technique that included explanatory variables such as discharge, season, and time to estimate concentrations (and subsequently loads). This method was similar to that used by Andrews (1978) and Lystrom and others (1978). Regressions were developed separately for low-flow and stormflow periods, and for both low flow and stormflow, separate models were generated for the pre- and post-treatment periods for each site. Models were selected on the basis of the highest adjusted R^2, residuals plots to detect trends, and all F-values had to exceed the value for the F distribution for the appropriate degrees of freedom and an alpha equal to 0.05.

Prior to load estimation for low-flow and stormflow periods, the continuous discharge data for the four sites had to be separated into low-flow and stormflow periods. Criteria for a "storm event" were specific to each site. Sampled storms were reviewed to determine the typical rate of stage-height increase that initiated storm sampling. The recession and subsequent completion of storm sampling was also reviewed to determine the typical termination of storm sampling at each of the four sites. This information was used to go through the 5- or 15-minute stage data and manually cut the storms from the continuous record. Sampled and unsampled storms were cut from the continuous record and placed into the stormflow discharge data set; the remaining data were placed in the low-flow data set.

For low-flow periods, a subset of the grab-sample data was used to develop the relation between constituent concentrations and explanatory variables. Prior to using the grab-sample data, the cumulative frequency distribution (Viessman and others, 1977) for each site was determined using the continuous discharge data for the entire period of record. Grab samples collected at flows above the 97th percentile (flows above the 97th percentile are the flows that exceed 97 percent of the sampled flows for the period of record) were deleted prior to load analysis. With these higher flows deleted, the relation between constituent concentrations and explanatory variables was developed. The low-flow constituent concentrations were estimated on a daily basis using the daily-mean discharge data for low-flow periods. The estimated concentrations were multiplied by the daily-mean discharge to estimate daily loads.

Stormflow loads for nutrients and suspended sediment were estimated by use of the mean discharge and mean constituent concentration for sampled storms. The mean discharge-concentration relation developed for sampled storms using regression analysis was used to predict the concentrations for unsampled storms. The mean discharge was calculated for unsampled storms using the 5- or 15-minute continuous-stage data for the sites. This mean discharge was applied to the predicted concentration to estimate constituent loads for unsampled storms. Increases in stage caused by snowmelt events were analyzed separately by subsetting the storm events sampled during snowmelts and using these regression relations to estimate loads for non-sampled snowmelt events. The percentage of the storms sampled at each site was somewhat dependent on the location of the surface-water site. For sites at the outlets (T-1 and C-1), the criteria necessary to be defined as a storm did not occur as frequently because these outlet sites were not as flashy as the upstream sites. The percentage of storms sampled (of the total) was 50 to 60 percent at the outlet sites. For the upstream sites, the flashiness of the system generated numerous small storm events that did not occur at the outlet sites. Typically, these storms were not sampled because of the necessity to pair storm samples with the control site; therefore, at the upstream sites, 35 to 45 percent of the total number of storms were sampled.

Constituent loads for each continuous surface-water site were estimated by summing the low-flow and stormflow loads. The annual load data for the constituents were divided by the basin drainage areas to determine constituent yields. The percentage of the total yield in stormflow was determined by summing the sampled and unsampled storm yields and dividing by the total yield. The remaining yield was attributed to low-flow periods. Data also were separated into pre- and post-treatment periods.

The distribution of nutrients, suspended sediment, field measurements (pH, water temperature, SC, and DO), and fecal streptococcus for the different sites was shown using box plots. Low-flow and stormflow data were separated into pre-treatment and post-treatment periods prior to generating plots. Grab samples collected at flows above the 97th percentile for discharge were deleted prior to analyses.

A number of significance tests were conducted on low-flow and stormflow data. For all tests, an alpha level equal to 0.05 was considered to be an acceptable level of significance. Data were first tested to determine if data for any one site significantly changed from pre- to the post-treatment. This was performed using the rank-sum test, which is a nonparametric test for two groups of varying size (Helsel and Hirsch, 1992). A second test conducted on the data was to determine if, when the data are grouped, there were significant differences between any of the sites. This was conducted for pre-treatment and post-treatment data separately using Kruskal-Wallis and Tukey multiple-comparison tests. The Kruskal-Wallis test is a nonparametric procedure that tests for significant differences between more than two groups with varying sample sizes (Helsel and Hirsch, 1992). If the Kruskal-Wallis test indicated significant differences were evident in the group, the Tukey multiple-comparison test was used to identify which sites were significantly different. The third significance test conducted on the data was the signed-rank test, which is a nonparametric test used to test for significant differences between paired observations. The test determined if the median difference between two sets of data was significantly different from zero (Helsel and Hirsch, 1992). For the signed-rank test, data were paired. For example, pairing data from T-1 and C-1 would mean that the chemistry for samples collected on the same day would be matched, so for T-1, if total N = 10 mg/L, and for C-1, total N = 9 mg/L, the difference between sites would be 1 mg/L. The differences were determined for pre- and post-treatment data, and these differences were tested to determine if there was a significant change from pre- to post-treatment.

The significance tests also were conducted on flow-weighted data. Concentrations were flow adjusted using a **LO**cally **WE**ighted **S**catterplot **S**moothing (LOWESS) procedure (Helsel and Hirsch, 1992). This procedure fits linear or quadratic equations of the predictor variable (in this case, flow) in order to "smooth" constituent concentrations. The smoothing is controlled by a smoothing factor that defines how many observations around the point are incorporated into the fitting equation (SAS Institute Inc., 1999). For this analysis, data were plotted for different smoothing factors and only one value was used to smooth all the data. Once the constituent concentrations were smoothed using this technique, significance tests were rerun in order to determine if factoring out the variability caused by flow affected the results.

The final and most important statistical method applied to the data was analysis of covariance, or ANCOVA, in order to detect effects of streambank fencing relative to pre-treatment and control conditions (Grabow and others, 1999). ANCOVA is a form of regression analysis that fits equations to the pre-treatment and post-treatment relation between paired sites (either paired basins or upstream/downstream pairs). This method tests for discrete (not gradual) changes in the relation between basins for a particular constituent. Regression lines relating the treatment data to the control (control basin or upstream site) data were generated for the pre-treatment and post-treatment data. The general form of the equation was:

$$Y_{trt} = X_{ctl} + TRT + TRT * X_{ctl}, \tag{1}$$

where Y_{trt} is the treatment data;
X_{ctl} is the control data; and
TRT is 0 if period is pre-treatment and TRT equals 1 if period is post-treatment.

Data were first tested for normality, and if necessary (and most models did require transformations), log or inverse transformations (of treatment and control data) were performed prior to analysis. Also, if the simple relation presented in equation 1 did not yield a significant model, discharge for the control data (and the associated interaction terms) were incorporated into the model. A stepwise procedure was then used to identify the model with the best fit.

A difference in the regression lines from pre- to post-treatment indicated a change in water quality due to land treatment (Grabow and others, 1999). For this analysis, an alpha level equal to 0.10 was set as the criteria for model acceptance. If a significant model was identified for the relation, then fencing did change water quality. The SAS statistical package (SAS Institute Inc., 1999) was the programming language for this analysis. An option in the program allows for output of predicted (least-square) mean values for the treatment basin that are based on the regression relation to the control data. The least-square means for pre- and post-treatment data were then used to determine a percentage change in the treatment basin for that constituent from the pre- to post-treatment period according to a procedure developed by Grabow and others (1999).

The ANCOVA procedure was conducted on transformed (if necessary) constituent concentrations and yields for both low-flow and stormflow samples. Initial analysis on constituent yields was conducted on non-weighted yields. Yields were also weighted to account for changes in flow regimes over the study period. To account for changes in flow over time, the annual mean discharge for each water year was divided by the mean discharge for the entire period for each surface-water site. The ANCOVA procedure was repeated on the weighted constituent yields.

Benthic Macroinvertebrates and Habitat Assessments

At the start of the project and within the two study basins, four sites were chosen for assessment of benthic macroinvertebrates, algae, and aquatic habitat. One site was in the control basin and the other three were in the treated basin. During the study, it became apparent that sites with smaller drainage areas had differing communities compared to the sites at the outlets (C-1 and T-1) of the study basins. To make a better assessment among the control and treated sites, a control site with a smaller drainage area (C1-2) was chosen and sampling began in May 1996. This allowed for the two larger sites (C-1 and T-1) and the three smaller sites (C1-2, T1-3, and T2-3) to be compared. Sites T1-3 and T2-3 were downstream and upstream, respectively, of a pond in the treated basin (fig. 1). All benthic-macroinvertebrate sampling locations had perennial flow over the entire study period.

Benthic Macroinvertebrates

The USEPA Rapid Bioassessment Protocol (RBP III) for benthic-macroinvertebrate sampling was used in this study (Plafkin and others, 1989). Samples were collected in May and September of each year starting September 1993 and ending May 2001. A 500-micron, mesh kick net was used. An area 1 m^2 in size was kicked in front of the net for 1 minute. This procedure was done in one riffle area and one pooled area. After kicking 1 minute, all materials captured on the net were washed into a 500-micron screen-bottom bucket. Large debris was removed, and the remaining sample was placed into one or two 5-L jars and preserved with reagent alcohol. After 24 hours, the samples were rinsed and preserved again with an 80-percent reagent-alcohol solution. The samples were shipped to Lotic Incorporated Environmental Consultants of Maine (Lotic) for analysis. A 200-organism subsample was identified to the lowest possible taxonomic level.

In addition to benthic-macroinvertebrate identification, Lotic also calculated seven community metrics for each 200-organism subsample. The metrics (calculated at the generic level) provided by Lotic included:

1. Taxa richness, which is a measurement of the total number of taxa present. Higher numbers generally reflect a more healthy macroinvertebrate community;

2. Hilsenhoff biotic index (HBI), which summarizes the overall pollution tolerance of the benthic arthropod community. The range of values is from 0-10; an increase in numbers indicates a decrease in water quality;

3. Ratio of scrapers to filter feeders, which provides an indication of the periphyton-community composition. Scrapers increase with increased abundance of diatoms and decrease with increased abundance of filamentous algae and aquatic mosses, whereas these two plant groups are utilized by filter feeders. Because filamentous algae and aquatic mosses usually are indicators of organic enrichment, a decrease in the ratio of scrapers to filter feeders is usually indicative of decreased water quality;

4. Ratio of Ephemeroptera, Plecoptera, and Trichoptera (EPT) to Chironomidae abundance. EPT are groups of invertebrates that, in general, are sensitive to pollution. Chironomidae are tolerant of pollution. A ratio near or above 1.0 indicates a healthy community;

5. Percent dominant taxa, which is a measure of the percent of dominant taxa relative to the total number of organisms. As this number increases, community health typically decreases;

6. EPT index. An increase in this index generally is indicative of increased water quality; and

7. Ratio of shredders to the total number of organisms. Shredders are sensitive to changes in the riparian zone, and, in general, the ratio should decrease as the amount of leaf material in the channel decreases. Shredders are also good indicators of toxic compounds if the compound can be adsorbed by organic matter (Plafkin and others, 1989).

Other metrics were also calculated to better understand the data. The benthic-macroinvertebrate data were reduced to family-level identification to clear some of the ambiguity from the data set and because some benthic macroinvertebrates were only reported to the family level. Three metrics were recalculated based on family-level taxonomic data: a family-level HBI (FHBI), family-level taxa richness, and family-level percent dominant taxa. Other metrics calculated were the percent EPT, percent chironomidae, and percent oligochaetes. Chironomids are known as the true midges (Pennak, 1953), and they are typically tolerant of pollution, so as their numbers increase, conditions are generally thought to be degrading. Oligochaetes are segmented worms common in mud and stagnant pools (Pennak, 1953). Generally, as the percentage of oligochaetes increases, water quality tends to decrease.

Algae

A modified National Water-Quality Assessment Program (NAWQA) protocol was used to sample algae (Porter and others, 1993). Samples were collected in May and September beginning May 1995 and ending May 2001. A sample of five rocks from the riffle area and five from the pooled area were removed from the stream and placed in a plastic tub. Water was placed in the tub to cover the surface of the rocks. A scum-getter-92 (SG-92) periphyton sampler (Porter and others, 1993) was placed against the rock giving a known sample area. A periphyton brush was used to scrape the area within the SG-92 opening. The loosened algae was sucked out of the SG-92 by use of a syringe. The algae were placed in a jar on ice.

The algal samples were shaken to ensure a homogeneous sample. A 25-mL aliquot was filtered onto a 0.7-mm pore size, 47-mm diameter filter. The filter was then rolled up with the algae on the inside of the roll, wrapped in aluminum foil, and placed onto a petri dish. The samples were sent overnight on ice to the NWQL for chlorophyll *a* analysis. Chlorophyll *a* concentrations were measured using high-performance liquid chromatography (HPLC) analysis until 1998; after 1998, a fluorometric method was used for the analysis.

Habitat

A qualitative survey of the stream habitat was completed at each site every May and September when the benthic-macroinvertebrate communities were sampled starting May 1993 and ending May 2001. The bioassessment technique used to assess stream conditions was RBP III for high-gradient streams. The biosurvey component of this protocol supplements the benthic-macroinvertebrate survey with cursory (qualitative) field observations in regards to periphyton, macrophytes, slime, and fish (Plafkin and others, 1989).

The habitat assessment using RBP III methodology involves qualitatively characterizing nine physical attributes of the habitat. These nine habitat attributes are broken down into three categories: primary (microscale), secondary (macroscale), and tertiary (bank and riparian zone) (Plafkin and others, 1989). The primary attributes generally are thought to have the greatest effect on the structure of the benthic-macroinvertebrate community and include the available cover (rocks and gravel) in the bottom substrate, the degree of embeddedness (to what extent rocks are embedded in finer sediments), and streamflow. Secondary habitat attributes are related to channel morphology and include channel alteration (evidence of channel bars or deposition zones), bottom scouring and deposition, and stream sinuosity (related to the extent of riffles and bends). Tertiary attributes include bank stability, vegetative bank stability, and streamside cover (Plafkin and others, 1989). Qualitative values were given to each of these nine attributes and total scores were calculated. The range of possible values is highest for the primary attributes (ranges from 0 to 20) and lowest for the tertiary attributes (range from 0 to 5). Thus, more weight is given to primary attributes when computing a total score. These total scores were used to give an overall idea of habitat suitability for benthic-macroinvertebrate communities. See appendix 1 for the habitat-assessment sheet used to conduct the surveys.

Water Quality

Grab samples were collected during benthic-macroinvertebrate sampling at each site. These water samples were always collected at the downstream side of the weir at the outlets of the treatment (T-1) and control basins (C-1). For the upstream sites (T1-3, T2-3, and C1-2), grab samples were either collected immediately upstream of the benthic-macroinvertebrate sample locations or downstream of the benthic-macroinvertebrate sample locations immediately prior to the benthic-macroinvertebrate sampling. Grab samples were analyzed for dissolved (ammonia, nitrite, nitrite plus nitrate, DKN, P, and orthophosphate) and total (TKN and P) constituents and suspended sediment. Chilled samples were shipped to the NWQL for nutrient analysis; samples for suspended-sediment analyses were sent to either the USGS Sediment Laboratory in Pennsylvania or Kentucky. At the time of sample collection, pH, water temperature, SC, DO, and turbidity were recorded by use of a multi-parameter probe.

Stream discharge was measured at the upstream sites during benthic-macroinvertebrate sampling with a pygmy current meter. The stage-discharge relation was used to determine stream discharge at T-1 and C-1 at the time of benthic-macroinvertebrate samping.

Data Analysis

At the family level, a benthic-macroinvertebrate aggregated index for streams was used to determine the condition of the benthic-macroinvertebrate communities at all sites for all sampling periods. This method was used by USEPA in a study conducted between 1996 and 1998 in streams within the Mill Creek Basin but outside of our study area (J.H. Green and Margaret Passmore, U.S. Environmental Protection Agency, written commun., 2004). This index consists of several benthic-macroinvertebrate community metrics being summed. Higher scores indicate a more diverse and healthy community. The scores range from 0 to 18. The community metrics used in this index are the number of EPT families, percentage of five most dominant taxa, FHBI, number and percentage of mayflies, family taxa richness, number of intolerant taxa, percentage of scrapers, and the Simpson Diversity Index.

Box plots of the generic- and family-level community metrics, habitat scores, chlorophyll *a* concentrations, and water-quality data were used to determine if there were any differences among the sites during the post-treatment period. Comparisons were conducted between the sites at the outlet (T-1 vs. C-1) and the upstream sites (T1-3 and T2-3 vs. C1-2). All box plots are presented as pre- and post-treatment with May and September samples plotted separately.

Habitat scores, chlorophyll *a* concentrations, and water-quality data were combined to form an environmental variable data set used in the multivariate canonical correspondence analysis (CCA). CCA is a method that combines the water-quality, habitat, and other environmental variables with the relative abundance of the benthic macroinvertebrates at multiple sites to aide the biologist in determining what variables are influencing the benthic-macroinvertebrate community. CCA was conducted using the CANOCO program (ter Braak and Smilauer, 1998). Prior to any CCA, all environmental data are standardized by subtracting the mean for each observation, then dividing this value by the standard deviation (Gere and Weaver, 1965). Weighted scores for benthic-macroinvertebrate species data must be calculated prior to CCA. In general, these weighted scores are calculated by dividing the abundance of a species at one site (and sample event) by the sum of the abundance for all other sites and sampling events (ter Braak, 1987). A covariance matrix is then generated between the weighted benthic-macroinvertebrate species scores and the standardized environmental variables. If the matrix is, for example, 4 columns by 4 columns, the solution for the determinant of the matrix is four eigenvalues (and eigenvectors) (Gere and Weaver, 1965). The CCA chooses the optimum weights for the environmental variables so that the dispersion of species scores is maximized along the first CCA (ordination) axis. The first eigenvalue calculated by CCA (technically, CCA is called an eigenvalue-ordination technique) is equal to the maximized dispersion of species scores along the first ordination axis (ter Braak, 1987). The scores and weights calculated for the benthic-macroinvertebrate species and environmental variables for each ordination axis represent the relation of that particular species or variable to the axis. The first ordination axis explains the majority of the variation in species and environmental variables, followed by the second axis, and so on. The species and environmental variables with the highest scores for that axis are those species and variables that show the best relation to the axis. One of the main benefits of CCA is that not all measured environmental variables are equally important and some can be combined to show a synthetic environmental gradient that better explains the species dispersion better than one variable alone (ter Braak and Verdonschot, 1995).

Any multi-collinearity issues between environmental variables were dismissed by using a Spearman-Rank correlation and normality testing of the colinear variables. When variables are strongly correlated with each other, their effect on the benthic-macroinvertebrate community cannot be separated and the canonical coefficients become unstable (ter Braak, 1986). Therefore, only one of a group of correlated variables was used in the CCA analysis. A criteria was developed to determine which correlated variables to remove from the CCA. First, variables with incomplete data sets were removed. Second, a Shapiro-Wilk normality test was used to discern which of the correlated variables would show better results. If an environmental variable was not normally distributed, the more skewness and bias there could be in the CCA results, which could possibly lead to erroneous interpretations.

Different data sets of environmental variables were analyzed using CCA. CCA was first run (for May and September separately) using water-quality and habitat variables for all five surface-water sites where benthic-macroinvertebrate samples were collected. This included data from all five sites between the years of 1993 to 2001. Following this, it was decided that instantaneous concentrations of nutrients and suspended sediment may not be related to benthic-macroinvertebrate communities as much as cumulative nutrient and suspended-sediment loads. Therefore, 1-month and 3-month nutrient and suspended-sediment loads were estimated for C-1 and T-1 (these data were not available for the upstream sites). For example, for the May 1996 benthic-macroinvertebrate sample, the loads for T-1 were estimated for April 1996 (1-month load) and February through April 1996 (3-month load). These loads were then used as environmental variables in the CCA. The methodology to estimate nutrient and suspended-sediment loads is presented in the surface-water section of this report. CCA was conducted separately using the data sets with 1-month and 3-month loads. For both data sets, May and September data were analyzed separately.

The number of samples available for CCA for the 1-month and 3-month load data sets was reduced because the upstream sites could not be included in the analysis. A general rule for CCA is that there can not be more environmental variables than samples (in this case, a sample would be the sample collected at T-1 for May 1996). Therefore, with the reduced data sets (i.e., the data sets with the 1-month and 3-month load variables), only a subset of environmental variables could be used. CCA conducted on the data sets (for May and September) that contained all five sites did not have to be subset. The final variables for the CCA that included the 1-month and 3-month loads variable were determined using Principal Components Analysis (PCA). PCA is a type of indirect gradient analysis that reduces the dimensionality of the data set so that a large number of variables can be compressed into a few components (Stiteler, 1979). As with CCA analysis, variables used in PCA must first be standardized around a mean of zero. PCA then takes these standardized variables and proceeds to reduce the variability into components. For this analysis, only the first two PCA axes were pertinent. The first PCA axis projects through the data such that the distance of any point to the axis is minimized. That is, the first axis explains most of the variation in the data set. Correlations (or factor loadings) are calculated by PCA between the principal component axes and the variables. The higher the factor loading, the more correlated that variable is to that axis (Stiteler, 1979). Those variables that showed the highest correlation to the first two principal component axes were retained for the CCA. This process was repeated for all four CCA runs.

Down-weighting of rare taxa was used as part of the CCA analysis. The theory behind down-weighting is that rare taxa are usually at a site by chance rather than being an indicator of ecological conditions; ordination techniques are sensitive to these rare taxa and can unduly influence or distort the results of the final analysis (Gauch Jr., 1982; Jongman and others, 1995; ter Braak and Smilauer, 1998). The down-weighting procedure in CANOCO was used to conduct these analyses.

Ground Water

The ground-water system in the study area was characterized on the basis of water levels, flow directions, age dating, and chemical quality. The three-dimensional configuration of the ground-water system was interpreted from well logs and the results of previous studies (Meisler and Becher, 1971; Schlosser and others, 1988; Plummer and others, 1993; Low and others, 2002). Water-level data from 8 wells and 17 piezometers were used to characterize ground-water flow directions and surface-water gain or loss in local reaches of unnamed tributaries to Big Spring Run. Results of water-quality analyses were used to relate agricultural activities (manure application and animal grazing) and recharge events to ground-water quality during the pre- and post-treatment periods.

Water samples were collected each month (from October 1993 through June 2001) from six shallow wells and two deep wells in the treatment basin (table 3). Samples were typically collected during stable stream conditions, but stable stream conditions were not a requirement for sampling. The piezometer network (table 4), which was installed after fence installation, was sampled during 1998 and 1999, with 1 to 2 samples collected from each piezometer.

Structural Framework

As a result of the regional structural complexity (Wise, 1970; Berg and Dodge, 1981; Shultz, 1999), near-infrared and black-and-white photographs of the study area were examined for linear-tonal changes to determine if large-scale structural features, such as fold-axes and faults, and small-scale features, such as formation contacts, joints, location of springs, and stream alignments, could be discerned. Intensive farming and development in the basin made it difficult to separate man-caused from naturally occurring tonal changes.

Fractured Bedrock

Although it is not possible to establish an exact thickness or maximum depth of the ground-water system, several methods can provide useful information. These methods include depth distributions of water-bearing zones reported by well drillers and borehole geophysics. The depth distributions of water-bearing zones for wells completed in the Conestoga Formation in Lancaster County, Pa., are listed in table 5. The depths of water-bearing zones in 15 wells drilled as deep as 400 ft ranged from 19 to 253 ft below land surface; the median was 128 ft below land surface (Low and others, 2002). Drillers comments regarding the results of a suite of borehole geophysical logs and video logging of the two deep wells (LN-2039 and LN-2043) are listed in tables 6 and 7, respectively. Both wells have water-producing zones, although under ambient conditions neither well exhibited vertical borehole flow. Well LN-2039 is the best yielding of the two wells.

Ground-Water Levels

Water-level recorders were installed in all eight of the wells during the first year of the study. Pressure transducers were placed in each of the wells and data loggers recorded data at 1-hour intervals throughout the study. Periodically, failure of the pressure transducers caused some loss of data, but given that four wells were in each nest, loss of data from one well was not a significant problem. Water-level data from the well network was used to characterize ground-water-level elevations near the stream so that ground-water flow directions could be determined.

The piezometer network was along the streambanks and within the stream channel. Water elevation in piezometers along the stream banks was measured using a downhole electric tape. The hydraulic potential of the piezometers in the stream channel was measured using a manometer. A manometer measures differences in hydraulic head between two water columns (Lee and Cherry, 1978; Fokkens and Weijenberg, 1968). In this case, piezometric hydraulic-head elevations were estimated relative to the surface-water gage pool on the upstream side of the weir at the surface-water sites.

Water-table maps were generated using the water-level elevations in the shallow wells, the piezometric head elevation, and the water-level elevation of the surface-water gage, in order to determine if the stream near the surface-water sites was gaining or losing water to the shallow ground-water system.

Water Quality

The frequency of ground-water sampling depended on the well and the constituent of interest or concern. For water years 1994 through 1998, all eight wells were sampled at the same frequency. All monthly samples collected from water year 1994 through water year 1998 were analyzed for dissolved nitrate plus nitrite and fecal streptococcus. In water year 1994, nine samples were collected from each well and also analyzed for analyses of DKN, dissolved ammonia, dissolved nitrite, dissolved P, and dissolved orthophosphate. From water years 1995 through 1998, quarterly samples for DKN, dissolved ammonia, dissolved nitrate plus nitrite, dissolved nitrite, dissolved P, and dissolved orthophosphate were collected from each well. From October 1998 through June 2001, the two deep wells (LN-2039 and LN-2043) were

Table 5. Number and density of water-bearing zones per 50 feet of well depth in the Conestoga Formation, Lancaster County, Pa.

[Data from Low and others, 2002]

Depth interval (feet below land surface)	Number of water-bearing zones	Mean density
0-50	7	2.33
51-100	3	28
101-150	6	55
151-200	7	89
201-250	1	25
251-300	1	25

Table 6. Comments regarding borehole geophysical (conducted Mar. 9, 1994) and video (conducted Mar. 20, 1997) logs for well LN-2039, Big Spring Run Basin, Lancaster County, Pa.

[Negative depth interval indicates that water level was in steel casing above land surface]

Depth interval (feet below land surface)	Comment
-1 to 57	Water cloudy
19	Bottom of 6-inch-diameter steel casing
19.5	6.5-inch fracture
28	Possible producing zone
57	Water clears
60	Possible producing zone
60.5	7-inch fracture
61.5	Bottom of borehole[1]

[1] Well was drilled to 63 ft, but deposition at bottom decreased well depth by 1.5 ft.

Table 7. Comments regarding borehole geophysical (conducted Mar. 9, 1994) and video (conducted Mar. 20, 1997) logs for well LN-2043, Big Spring Run Basin, Lancaster County, Pa.

Depth interval (feet below land surface)	Comment
4 to 99.5	Water cloudy
17.5	Bottom of 6-inch-diameter steel casing
18-29	Small fractures that are producing most of the water for the well
30-45	Possible producing zone
31	Possible lithology change
70-90	Possible producing zone
72-73	Possible producing zone
99.5	Bottom of borehole

sampled only quarterly for all constituents. From October 1998 through June 2001, the six shallow wells were sampled on a monthly basis for fecal streptococcus and dissolved nitrate plus nitrite and on a quarterly basis for DKN, dissolved ammonia, dissolved nitrite, and dissolved P, except for times when sample volume was insufficient because of dry conditions. Such conditions were most evident for wells LN-2037 and LN-2038 for water year 2000. At the time of sample collection, pH, water temperature, DO, and SC were recorded by use of a multi-parameter probe.

The two well nests were always sampled on the same day. Each well was either purged until dry or until at least three borehole volumes of the well were purged. Prior to well purging, an electric water sensor was used to measure the depth of the water below land surface. Each well (except for well LN-2043) was purged using a small centripetal pump with a capacity to pump about 2-4 gal/min. These pumps were placed near the screened or open interval of the well attached to 1-in. diameter polyvinyl chloride (PVC) pipe that discharged the water out of the well. A separate submersible pump with a capacity of about 10-15 gal/min was used to purge well LN-2043, the deepest (100 ft) of the eight monitor wells. After water recovery in the wells, samples for chemical and bacterial analysis were collected. For the six shallow wells, prerinsed plastic tubing (separate tubing for each well) was placed down the well and water was evacuated from the well using a peristaltic pump. The base of the tubing was fitted with a polyethylene cylinder (12-in. length) that had an inflow hole drilled near the top. This design kept the tubing from clogging with materials at the base of the well hole. Water was evacuated from the wells through the prerinsed tubing for 3 to 10 minutes prior to sample collection. Well LN-2039 was sampled from the PVC pipe used to discharge water from the well with the centripetal pump. A prerinsed tube was attached to the PVC for sample collection. Well LN-2043 was sampled using the submersible pump that had purged the well. Typically, well LN-2043 was purged until dry, so samples were collected after well recovery. The first sample collected from the wells was for fecal streptococcus and enumeration analysis was performed according to techniques described by Ehlke and others (1987) for the membrane-filter method and immediate incubation test. Three replicates were processed for each fecal streptococcus sample collected in the field. The nutrient and alkalinity samples were collected using a prerinsed 0.45-micron pore-size filter. Nutrient samples were placed on ice after sample collection. Chilled samples were shipped to the NWQL for nutrient analysis. Analyses were performed according to techniques described in Fishman and Friedman (1989). Alkalinities were measured on the same day or the following day using a Hach digital titrator and 1.6 normality sulfuric acid. The water-quality probe to measure pH, water temperature, SC, and DO was placed downhole (except for well LN-2043) after the other samples were collected. Field characteristics for well LN-2043 were measured in a bucket at land surface. Water from the submersible pump was discharged into the bucket with the water-quality probe.

Selected piezometers in the study basin were sampled from November 1998 into early May 1999. Only one or two samples were collected from any one piezometer over this entire period. Prior to sampling, the water level or potential head was measured. Piezometers were purged prior to sampling, and samples were collected using prerinsed plastic tubing and a peristaltic pump. Samples for analysis of nutrients were collected using a prerinsed 0.45-micron pore-size filter. Water was also pumped to a bucket at land surface to measure field characteristics. Nutrient samples were processed and shipped similarly to the shallow-well water samples.

As part of a broader study, some of the wells and piezometers for this study were sampled to determine age dates and nitrogen-cycling processes (Lindsey and others, 2003). Chlorofluorocarbons (CFCs) and tritium samples were analyzed in samples collected for ground-water age dating, and N isotopes were analyzed in samples collected to characterize N cycling. The methodology is described in Lindsey and others (2003).

Data Analysis

The areal and temporal distribution of nutrients, field measurements (pH, water temperature, SC, and DO), and fecal streptococcus for the eight wells was shown using box plots. Data were first separated into pre- and post-treatment periods.

A number of significance tests were conducted on the well data. For all tests, an alpha level equal to 0.05 was considered to be an acceptable level of significance. Data were first tested to determine if data for any one well significantly changed from the pre- to the post-treatment period. This was performed using the rank-sum test (Helsel and Hirsch, 1992). A second test was conducted on data groupings to determine whether significant differences existed between any of the wells. Statistical tests were conducted on pre- and post-treatment data separately using Kruskal-Wallis and Tukey multiple-comparison tests (Helsel and Hirsch, 1992). The third significance test conducted on the data was the signed-rank test, which was used to determine if the median difference between two sets of paired data was significantly different from zero (Helsel and Hirsch, 1992). Data pairs for the wells were the shallow wells inside the fence paired with the well outside the fence at the two well nests. The differences would be determined for pre- and post-treatment data, and these differences would then be tested to determine if there was a significant change from the pre-treatment to the post-treatment.

Similar to surface water, streambank-fencing impacts on shallow ground-water quality were identified using ANCOVA. Regression lines relating the treatment (inside fence) to the control (outside fence) wells were generated for the pre- and post-treatment data. For this analysis, an alpha level equal to 0.10 was set as the criteria for model acceptance. If a significant model was identified for the relation, then fencing did change shallow ground-water quality. The least-square means for pre- and post-treatment data were then used to determine a percentage change in the treatment well for that constituent from the pre- to post-treatment period.

If the simple relation expressed in equation 1 did not produce a significant model, water-level data from the control well (and the associated interaction terms) was incorporated into the model. Additional analysis included the development of surrogates for recharge based on precipitation amounts. The amount of time to the last precipitation event (0.5-in., 1-in., and 2-in. storm) prior to ground-water sampling was calculated. These variables were then placed into equation 1 with interaction terms. A stepwise procedure was used to identify the model with the best fit.

Quality Control

Quality-control (QC) samples were collected to measure bias and variability of the data. Quality-control samples were collected in concert with surface-water grab and storm samples, benthic-macroinvertebrate samples, and ground-water samples.

Surface Water

A variety of surface-water QC samples were collected for water-quality and benthic-macroinvertebrate samples. Replicate samples were used to evaluate sample and laboratory variability. Blank (de-ionized water) samples were collected to test for bias from the introduction of contamination from the sampling equipment. Standard-reference samples were collected to measure laboratory accuracy. QC samples also were collected during storm events to determine if the automatic storm samplers at each site were collecting a representative sample of the water passing the weir, which was within 5 ft of the intake tubing for the automatic samplers. The samples collected to check automatic sampler accuracy are hereinafter referred to as AUTO/DI (for comparison of automatic and depth integrated) samples.

A total of 1,670 surface-water samples (not counting QC samples) were submitted for chemical analyses during the study; 1,074 grab samples and 596 composited storm samples were collected. The number of replicate samples collected for the grab and storm samples was 59 and 34, respectively (tables 8 and 9). Grab-sample replicates were collected in another bottle at the downstream side of the weir at the same time the regular sample was collected. As stated earlier, the storm samples from the automatic sampler were composited into one bottle. For the replicate sample, the volume composited was doubled so that the replicate and regular sample were taken from the same bottle (split replicate).

Results from the replicate analyses indicated the median percentage difference for most constituents was less than 10 percent. Median percentage differences of 10 percent or less are acceptable for nutrient analysis (Witt and others, 1992). The median percentage difference for suspended sediment (29 percent) replicates in grab samples was much higher than any other percent difference. Conversely, the percentage difference for replicates of suspended-sediment samples for storms was only 3.6 percent. Median percentage differences typically increase as the detection limit is approached. The average suspended sediment concentrations for grab and storm samples were about 15 and 300 mg/L, respectively. Therefore, the higher median percentage difference in suspended sediment for grab samples was attributed to relatively low concentrations near the detection limit. The detection limit for suspended sediment was 1 mg/L. This high median percentage difference was not considered a problem because higher percentage differences are expected near the detection limit and because stormflow is the predominant transport mechanism of suspended sediment in the basin. The percentage difference for total ammonia plus organic nitrogen (TKN) (13 percent) replicates in storm samples was close to the 10-percent acceptance limit for nutrient samples; therefore, no adjustments were made to these data.

Table 8. Summary of percentage differences for replicate grab samples collected from the surface-water sites in Big Spring Run Basin, Lancaster County, Pa.

[TKN, total ammonia plus organic nitrogen; DKN, dissolved ammonia plus organic nitrogen]

Constituent	Number of replicate samples	Percentage difference		
		Minimum	Median	Maximum
Dissolved nitrate plus nitrite	59	0	1.0	12
Dissolved nitrite	59	0	0	160
Dissolved ammonia	59	0	3.9	91
TKN	58	0	5.3	67
DKN	59	0	5.7	130
Total phosphorus	58	0	9.8	140
Dissolved phosphorus	59	0	3.8	120
Dissolved orthophosphate	47	0	0	100
Suspended sediment	48	0	29	170

Table 9. Summary of percentage differences for replicate storm samples collected from the surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.

[TKN, total ammonia plus organic nitrogen; DKN, dissolved ammonia plus organic nitrogen]

Constituent	Number of replicate samples	Percentage difference		
		Minimum	Median	Maximum
Dissolved nitrate plus nitrite	34	0	3.1	31
Dissolved nitrite	34	0	2.2	22
Dissolved ammonia	34	0	6.9	130
TKN	34	0	13	89
DKN	34	0	3.5	26
Total phosphorus	34	.46	8.3	87
Dissolved phosphorus	34	0	5.0	47
Dissolved orthophosphate	34	0	3.3	110
Suspended sediment	34	0	3.6	70

AUTO/DI samples were collected during storm events; 82 and 84 samples were collected for comparison of nutrients and suspended sediment, respectively. Theses samples were paired; that is, of the 82 samples, 41 were collected at the weir at the same time as the automatic sample. To determine if the differences between the automatic and depth-integrated (DI) samples at each site were significantly different from zero, a signed-rank test was conducted on the paired data (table 10). The test indicated a significant difference between paired samples for suspended-sediment concentrations at C-1 and T-1. Suspended-sediment concentrations were higher in the automatic samples than in the DI samples for both sites. Samples collected by the automatic sampler were collected from a perforated intake tube oriented in the stream channel at an angle generally perpendicular to the streamflow direction. A bottle was placed at the end of the intake tube in order to keep the tubing above the channel bottom. However, the intake tubing could have pulled some sediment from the bottom of the channel during storm events, and this could account for the higher concentration of suspended sediment. It is likely that if more AUTO/DI samples were collected at T-4, there also would have been a significant difference between automatic and DI samples at T-4. Regression relations between automatic and DI samples were generated for C-1, T-1, and T-4 (table 11). These relations were used to correct surface-water load estimations for these three sites. It was assumed that the relation between the automatic and DI samples did not change over time.

Analyses of AUTO/DI samples collected for nutrients did not indicate any significant differences between the automatic and DI samples (table 10). Except for the samples collected at T-1, the percentage difference in AUTO/DI for nutrient samples indicated DI samples tended to have higher concentrations than automatic samples; however, the differences did not warrant regression equations to correct for the bias.

The other QC samples collected were blanks and standard-reference samples (table 12). Thirty-two (equipment and bottle) blanks were submitted to test for contamination of water samples with constituents of concern. The blank samples were designated as either surface-water or ground-water QC samples. Most of the blanks were processed in the laboratory; only some were processed in the field. The blank samples did not show any contamination problems. Dissolved ammonia was detected in the first few blank samples submitted, but this problem was not evident after 1995. Five standard-reference samples were submitted to the laboratory to check for sample bias and precision (U.S. Environmental Protection Agency,

Table 10. Summary of paired surface-water samples collected by the automatic storm samplers and at the downstream side of the weir (referred to as a depth-integrated (DI) sample) in the Big Spring Run Basin, Lancaster County, Pa.

[TKN, total ammonia plus organic nitrogen; P, total phosphorus; SED, suspended sediment]

Site	Constituent	Number of paired samples	Median percentage difference[1]	Signed-rank probability[2]
C-1	TKN	11	-7.8	0.23
T-1	TKN	16	1.7	.17
T-2	TKN	7	-8.5	.09
T-4	TKN	7	-.077	.31
C-1	P	11	-18	.21
T-1	P	16	.48	.70
T-2	P	7	-1.6	.69
T-4	P	7	-5.5	.30
C-1	SED	12	9.6	.0093
T-1	SED	20	4.9	.017
T-2	SED	6	-1.6	.56
T-4	SED	4	26	.25

[1]Median percentage difference is equal to the concentration for the automatic sample subtracted by the concentration for the DI sample divided by the mean difference between the two samples multiplied by 100.

[2]The value for signed-rank probability indicates whether the difference between the paired samples was significantly different from zero. Probabilities equal to or less than 0.05 indicate a significant difference between pairs with 95-percent confidence.

Table 11. Linear regressions relating suspended-sediment concentrations in samples collected by the automatic sampler to grab samples collected at the weir in the Big Spring Run Basin, Lancaster County, Pa.

[AUTO(SED), automatic sampler suspended-sediment concentration in milligrams per liter; GRAB(SED), grab sample suspended-sediment concentration in milligrams per liter]

Site	Regression
C-1	AUTO(SED) = -3.610 + (1.152 x GRAB(SED))
T-1	AUTO(SED) = -0.872 + (1.013 x GRAB(SED))
T-4	AUTO(SED) = 10.929 + (1.273 x GRAB(SED))

2002a). These samples also showed no analytical problems. All results from the standard-reference samples were within the acceptable limits for the samples submitted. The standard-reference samples submitted during 1995 required mixing of reagents with inorganic-free water. Thus, these samples showed more differences relative to the mean recovery value as opposed to the standard-reference sample submitted in 2000. The standard-reference sample in 2000 was a pre-prepared liquid solution that simply had to be poured into a sample bottle.

Benthic Macroinvertebrates

Quality control for the benthic macroinvertebrates consisted of Lotic, Inc., taking a second 200-animal subsample from a specified site and identifying samples to the lowest possible taxonomic level, for a total of 14 replicate samples. The Index of Similarity is used as a quality-assurance tool when evaluating variance between two communities, a control and a reference, or along a gradient such as above and below a pollution impact area (Klemm and others, 1990). The Index of Similarity scores between the “normal” and replicate samples ranged from 59 to 80 percent (table 13). These results indicated the replicates split from the samples in the laboratory were within the acceptable range and the samples were valid. Data for replicate samples was published in USGS annual data reports (Durlin and Schaffstall, 1996, 1997a, 1997b, 1998, 1999, 2000, 2001, 2002). The streams were not large enough nor was there suitable habitat within the 100-m reach to take a field replicate sample. Results from the replicate samples did not indicate any significant problem with replicating the “normal” sample; therefore, it was determined that benthic sample processing methods did adequately represent the benthic sample.

The QC sample for chlorophyll *a* consisted of a split replicate sample analyzed at the laboratory (table 14). The streams generally were not large enough to take a field replicate. The percentage difference for the chlorophyll *a* sample (28 percent) indicated that replication of the “normal” sample was not very precise.

Ground Water

A variety of ground-water QC samples were collected; replicate samples were the most common type. Blank samples also were collected to identify any sources of nutrients in the sampling equipment (table 12).

A total of 580 regular samples were collected for analysis of dissolved nitrate plus nitrite. Additional nutrients were sampled in 265 of the 580 samples collected for nitrate plus nitrite. Twenty-two replicate samples were submitted for analysis of nitrate plus nitrite and 10-14 replicate samples were submitted for analysis of the other nutrients (table 15). Replicate data showed little, if any, variation between environmental and replicate samples.

Blank and standard-reference samples submitted for evaluation of ground-water sampling procedures also did not indicate any problems with sampling or laboratory procedures (table 12). Only one value (concentration of analyte) was above detection limit for all ground-water blank samples submitted, and, similar to surface water, this was for dissolved ammonia and occurred early in the project (1994). An analysis of DKN for one blank ground-water sample reported a concentration, but this was actually below the laboratory reported detection limit.

Effects of Streambank Fencing

The effects of streambank fencing on surface- and ground-water quality were determined using a paired-basin and upstream/downstream comparison approach. Given the amount of climatic variability during the study and the subsequent effect on water quality, it was apparent by the end of the study that studying only the treatment basin would have dramatically changed conclusions drawn from the data. Data from controls (either surface or ground water) were pertinent in quantifying the effects of fencing and factoring out effects caused by climatic variability.

Ancillary Data

Precipitation amounts and agricultural practices in the study basins were two important factors that affected water quality to some extent. Precipitation measured at T-1 was assumed to be representative of both basins, which was reasonable considering the total area for both basins was 3.1 mi^2. Agricultural activity, on the other hand, did vary between basins, and any changes in nutrient applications or number of pastured cows in either basin had to be identified. However, no attempt was made to directly relate any changes in nutrient applications to a change in water quality because there can be a considerable lag period between the time when a nutrient is applied to the landscape and when that nutrient reaches a surface- or ground-water system (assuming some other process does not remove the nutrient from the water column). Focazio

Table 12. Summary of quality-control samples from the Big Spring Run Basin, Lancaster County, Pa.

[N, nitrogen; DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen; P, phosphorus; SW, surface-water quality-control sample; GW, ground-water quality-control sample; SW/GW, quality-control sample processed from equipment or bottles used for either surface- or ground-water samples; <, less than; –, no data; values in ( ) are known concentrations for reference samples; all units in milligrams per liter]

Date	Time	Sample type	Dissolved nitrate plus nitrite N, as N	Dissolved nitrite, as N	Dissolved ammonia, as N	DKN, as N	TKN, as N	Total P	Dissolved P	Dissolved orthophosphate, as P	Suspended Sediment	Quality-control type[1]
03/14/1994	1510	SW	0.11	0.01	0.04	<0.2	–	–	<0.01	<0.01	–	Equipment blank
03/14/1994	1512	SW	<.05	<.01	.03	<.2	–	–	<.01	<.01	–	Equipment blank
04/13/1994	1116	SW	–	–	–	–	–	–	–	–	<1	Bottle blank
06/10/1994	1016	SW	–	–	–	–	–	–	–	–	<1	Bottle blank
07/01/1994	1000	SW	<.05	<.01	.03	<.2	–	–	.03	<.01	–	Equipment blank
07/12/1994	1200	GW	<.05	<.01	.04	–	–	–	–	.01	–	Equipment blank
07/12/1994	1210	SW/GW	<.05	<.01	.03	–	–	–	–	.01	–	Equipment blank
08/30/1994	1530	SW/GW	<.05	<.01	.03	<.2	–	–	<.01	<.01	–	Equipment blank
08/30/1994	1531	SW	<.05	<.01	.04	<.2	–	–	<.01	<.01	–	Equipment blank (replicate)
02/13/1995	1000	SW	<.05	<.01	.02	<.2	–	–	<.01	.01	–	Equipment blank
02/13/1995	1100	SW	.05	<.01	<.015	<.2	–	–	<.01	<.01	–	Equipment blank
02/13/1995	1430	SW	3.5 (3.22)	<.01	15 (15.1)	16	–	–	6.9	6.2 (6.31)	–	Reference sample
02/13/1995	1431	SW	3.5 (3.22)	<.01	14 (15.1)	15	–	–	7.0	5.9 (6.31)	–	Reference sample (replicate)
05/10/1995	0600	SW	.07	<.01	.02	<.2	–	–	<.01	<.01	–	Equipment blank
07/05/1995	0800	SW	<.05	<.01	<.015	<.2	<0.2	0.01	<.01	.02	–	Equipment blank
09/07/1995	1005	SW	<.05	<.01	<.015	<.2	<.2	.01	<.01	<.01	–	Equipment blank
10/11/1995	1050	SW	–	–	–	–	–	<.01	–	–	–	Bottle blank
12/12/1995	0800	SW/GW	2.6	<.01	14	9.5 (10.5)	9.7 (10.5)	6.0 (6.5)	5.1 (6.5)	5.0	–	Reference sample
12/12/1995	0900	SW/GW	3.7 (3.22)	<.01	15 (15.1)	13	14	6.2	5.9	6.1 (6.31)	–	Reference sample (replicate)
04/22/1997	1130	SW	.063	<.01	<.015	<.2	<.2	<.01	<.01	<.01	–	Equipment blank
09/09/1997	0600	SW	<.05	<.01	<.015	<.2	<.2	<.01	<.01	<.01	–	Bottle blank
09/09/1997	0630	GW	<.05	<.01	<.015	<.2	–	–	<.01	<.01	–	Bottle blank
09/09/1997	0700	GW	<.05	<.01	<.015	<.2	–	–	<.01	<.01	–	Equipment blank
09/09/1997	0730	SW	<.05	<.01	<.015	<.2	<.2	<.01	<.01	<.01	–	Bottle blank

Table 12. Summary of quality-control samples from the Big Spring Run Basin, Lancaster County, Pa.—Continued

[N, nitrogen; DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen; P, phosphorus; SW, surface-water quality-control sample; GW, ground-water quality-control sample; SW/GW, quality-control sample processed from equipment or bottles used for either surface- or ground-water samples; <, less than; –, no data; values in ( ) are known concentrations for reference samples; all units in milligrams per liter]

Date	Time	Sample type	Dissolved nitrate plus nitrite N, as N	Dissolved nitrite, as N	Dissolved ammonia, as N	DKN, as N	TKN, as N	Total P	Dissolved P	Dissolved orthophosphate, as P	Suspended Sediment	Quality-control type[1]
09/09/1997	0800	SW	<0.05	<0.01	<0.015	<0.2	<0.2	<0.01	<0.01	<0.01	–	Bottle blank
09/09/1997	0830	SW	<.05	<.01	<.015	<.2	<.2	<.01	<.01	<.01	–	Equipment blank
09/09/1997	0900	GW	<.05	<.01	<.015	<.2	–	–	<.01	<.01	–	Equipment blank
06/28/1999	1330	SW	<.05	<.01	<.02	<.1	.103	<.004	<.004	<.01	0.3	Equipment blank
06/28/1999	1420	SW	<.05	<.01	<.02	<.1	–	–	<.004	<.01	–	Equipment blank
06/28/1999	1430	SW	<.05	<.01	<.02	<.1	–	–	<.004	<.01	–	Equipment blank
10/19/2000	0900	SW/GW	16.0 (15.8)	.006	1.29 (1.32)	1.35	–	–	3.33	3.066 (3.33)	–	Reference sample
12/20/2001	0800	SW	<.05	<.008	<.04	<.1	<.01	<.004	<.004	<.02	–	Bottle blank
12/20/2001	0800	GW	<.05	<.008	<.04	.077	–	–	<.004	.009	–	Equipment blank
12/20/2001	0900	SW	<.05	<.008	<.04	<.1	<.1	<.004	<.004	<.02	–	Equipment blank
12/20/2001	0900	GW	<.05	<.008	<.04	<.1	–	–	<.004	.023	–	Equipment blank
12/20/2001	1000	SW	<.05	<.008	<.04	<.1	<.1	.0019	<.004	<.02	–	Equipment blank
12/20/2001	1100	SW	<.05	<.008	<.04	<.1	<.1	<.004	<.004	<.02	–	Equipment blank

[1]Definitions for quality-control type were referenced from U.S. Environmental Protection Agency (2002a).

Table 13. Index of Similarity for replicate 200-organism subsamples identified by Lotic, Inc., for the benthic macroinvertebrates collected in the Big Spring Run Basin, Lancaster County, Pa.

Date	Site	Index of Similarity between two samples (percent)
September 1993	T-1	59
May 1994	T-1	59
September 1994	C-1	80
May 1995	T1-3	65
September 1995	T2-3	78
May 1997	T-1	59
September 1997	T-1	68
May 1998	C-1	71
September 1998	T-1	64
May 1999	C1-2	71
September 1999	T2-3	60
May 2000	C-1	74
September 2000	T-1	79
May 2001	C-1	72

Table 14. Summary of quality-control replicate sample collected at surface-water site T-1 for chlorophyll *a*, Big Spring Run Basin, Lancaster County, Pa.

[units are in milligrams per square meter]

Site	Chlorophyll *a*	Chlorophyll *a* split	Percent difference
T-1	33.2	42.5	28

and others (1998) collected spring samples in the Chesapeake Bay Basin and found that residence times (the time the nutrient first reaches the water table until it is discharged to a surface-water system) can vary from a few years to more than 40 years. The residence time was between 4 and 20 years for over 50 percent of the samples collected (Focazio and others, 1998).

Precipitation

Precipitation was quite variable during the study period (fig. 6). The highest and lowest annual precipitation totals measured for an entire water year were for water years 1996 (54.6 in.) and 1995 (34.2 in.), respectively. The highest monthly total was recorded in September 1999 (12.0 in.); 6.7 in. fell as the remnants of Hurricane Floyd moved over the basins. The average monthly amount of precipitation during the pre- and post-treatment periods was 3.7 and 3.3 in., respectively. This equates to about 44.9 and 39.8 in. of annual precipitation during the pre- and post-treatment period, respectively.

Agricultural Activity

Nutrient applications in the form of inorganic and organic fertilizers in the treatment and control basins showed differences between the pre- and post-treatment periods (table 16). The average annual estimated amount of N applied during the pre- and post-treatment periods in the treatment basin was 49,800 and 36,100 lb/mi^2, respectively. The average annual estimated amount of P applied during the pre- and post-treatment periods in the treatment basin was 8,800 and 5,900 lb/mi^2, respectively. The percent decrease from the pre- to post-treatment periods in the annual amount of N and P applied in the treatment basin was 27 and 33 percent, respectively. The average annual estimated amount of N applied during the pre- and post-treatment periods in the control basin was 65,100 and 63,400 lb/mi^2, respectively. The average annual estimated amount of P applied during the pre- and

Table 15. Summary of percentage differences for replicate samples collected from the ground-water wells at surface-water sites T-1 and T-2 in the Big Spring Run Basin, Lancaster County, Pa.

[DKN, dissolved ammonia plus organic nitrogen]

Constituent	Number of replicate samples	Percentage difference		
		Minimum	Median	Maximum
Dissolved nitrate plus nitrite	22	0	0.76	8.7
Dissolved nitrite	12	0	0	120
Dissolved ammonia	14	0	0	14
DKN	12	0	1.8	89
Dissolved phosphorus	12	0	0	67
Dissolved orthophosphate	10	0	0	67

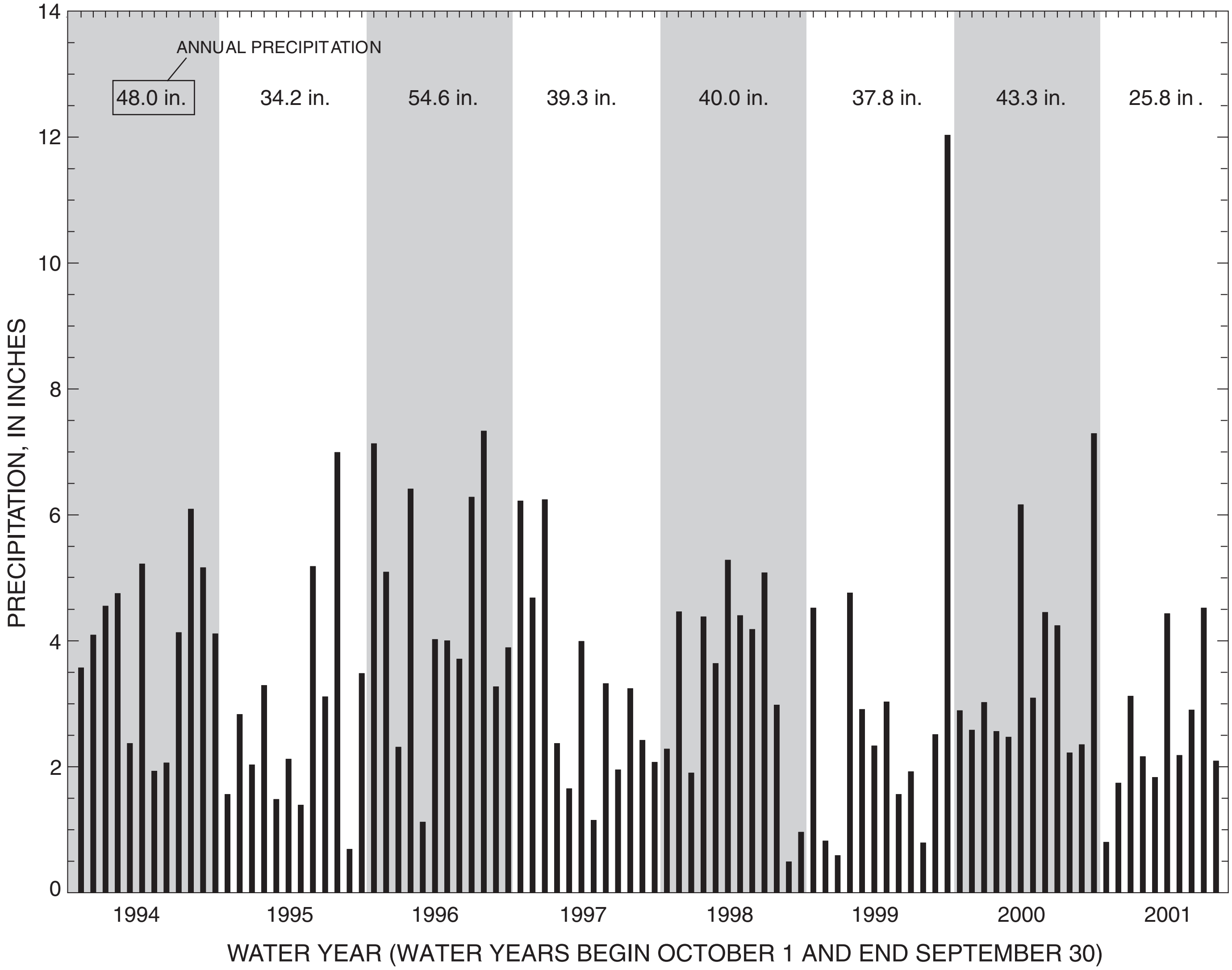


Figure 6. Bargraph showing measured monthly and annual precipitation at the outlet of the treatment basin in the Big Spring Run Basin, Lancaster County, Pa., from October 1993 through July 2001.

post-treatment periods in the control basin was 12,200 and 13,100 lb/mi^2, respectively. The percentage change from the pre- to post-treatment periods in the annual amount of N and P applied in the control basin was a 3 percent decrease and 7 percent increase, respectively. Thus, nutrient-application rates in the control basin remained relatively unchanged from the pre- to post-treatment period; however, nutrient-application rates in the treatment basin decreased from the pre- to the post-treatment period.

Table 16. Estimated annual applications of nitrogen and phosphorus from inorganic and organic fertilizers to the treatment and control basins of the Big Spring Run Basin, Lancaster County, Pa., by water year.

[All units are in pounds per square mile]

Water year[1]	Treatment basin		Control basin	
	Nitrogen	Phosphorus	Nitrogen	Phosphorus
1994	20,600	3,300	56,700	9,700
1995	61,200	10,900	71,900	13,800
1996	61,700	11,200	61,100	11,300
1997	57,400	10,000	72,200	14,300
1998	38,400	6,400	67,900	14,200
1999	40,300	6,700	68,400	14,100
2000	35,000	5,300	58,200	11,600
[2]2001	23,600	3,900	55,700	11,800

[1]Water years begin October 1 and end September 30.

[2]Data for water year 2001 were collected from October 2000 through June 2001.

Table 17. Summary of cow-pasture data for the treatment and control basins of the Big Spring Run Basin, Lancaster County, Pa., from October 1, 1993 (beginning of water year 1994), through June 30, 2001.

[Units are the average daily number of cows in pasture for 24 hours divided by the total acreage of pasture along streambanks in each basin]

Water year[1]	Treatment basin	Control basin
1994	0.89	2.5
1995	1.3	3.3
1996	1.4	3.4
1997	1.6	3.3
1998	1.4	3.4
1999	1.5	3.3
2000	1.5	2.4
[2]2001	.79	1.8

[1]Water years begin October 1 and end September 30.

[2]Cow-pasture data for water year 2001 were collected from October 1, 2000, through June 30, 2001.

The number of cows pastured remained relatively consistent from WY 1995 through WY 2000 (table 17). Approximately 54 and 52 acres of pasture along stream channels existed in the treatment and control basins, respectively. The amount of pasture did not change during the study period. The number of cows in pasture per unit area in the treatment basin averaged about 44 percent of the total number in the control basin. During the pre- and post-treatment periods, the number of cows in pasture in the treatment basin averaged about 40 and 47 percent, respectively, of the total number in the control basin. The number of cows in both basins decreased noticeably from WY 2000 to WY 2001. The control basin showed about a 25-percent decrease in the number of cows from WY 1999 to WY 2000, and another 25-percent decrease from WY 2000 to WY 2001. The treatment basin showed about a 50-percent decrease in the number of cows from WY 2000 to WY 2001.

Surface Water

The paired-basin and upstream/downstream monitoring approaches to detect changes in water quality caused by land-use changes are critical for all studies designed to quantify effects on a watershed scale. This is especially important for study periods that show a large degree of climatic variability, which was the case for this study.

Streamflow

The quantity of water discharging from the study area showed a marked difference from the pre- to the post-treatment period (fig. 7). Mean discharge for the post-treatment period was about 56-63 percent of the mean discharge for the pre-treatment period (table 18). A lower mean during the post-treatment period was expected given that the precipitation record, on average, showed 5 in. more annual precipitation

Table 18. Summary of pre- and post-treatment mean discharge for the four continuous surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.

[All units are in cubic feet per second]

Site	Overall mean discharge[1]	Pre-treatment mean discharge[2]	Post-treatment mean discharge
T-1	1.65	2.13	1.20
C-1	2.87	3.56	2.23
T-2	.49	.63	.38
T-4	.36	.50	.28

[1]The overall mean discharge for T-1, C-1, and T-2 included data from October 1, 1993 through July 31, 2001. T-4 data included the period from May 1, 1995 through July 31, 2001.

[2]The pre-treatment mean was determined using data through July 31, 1997.

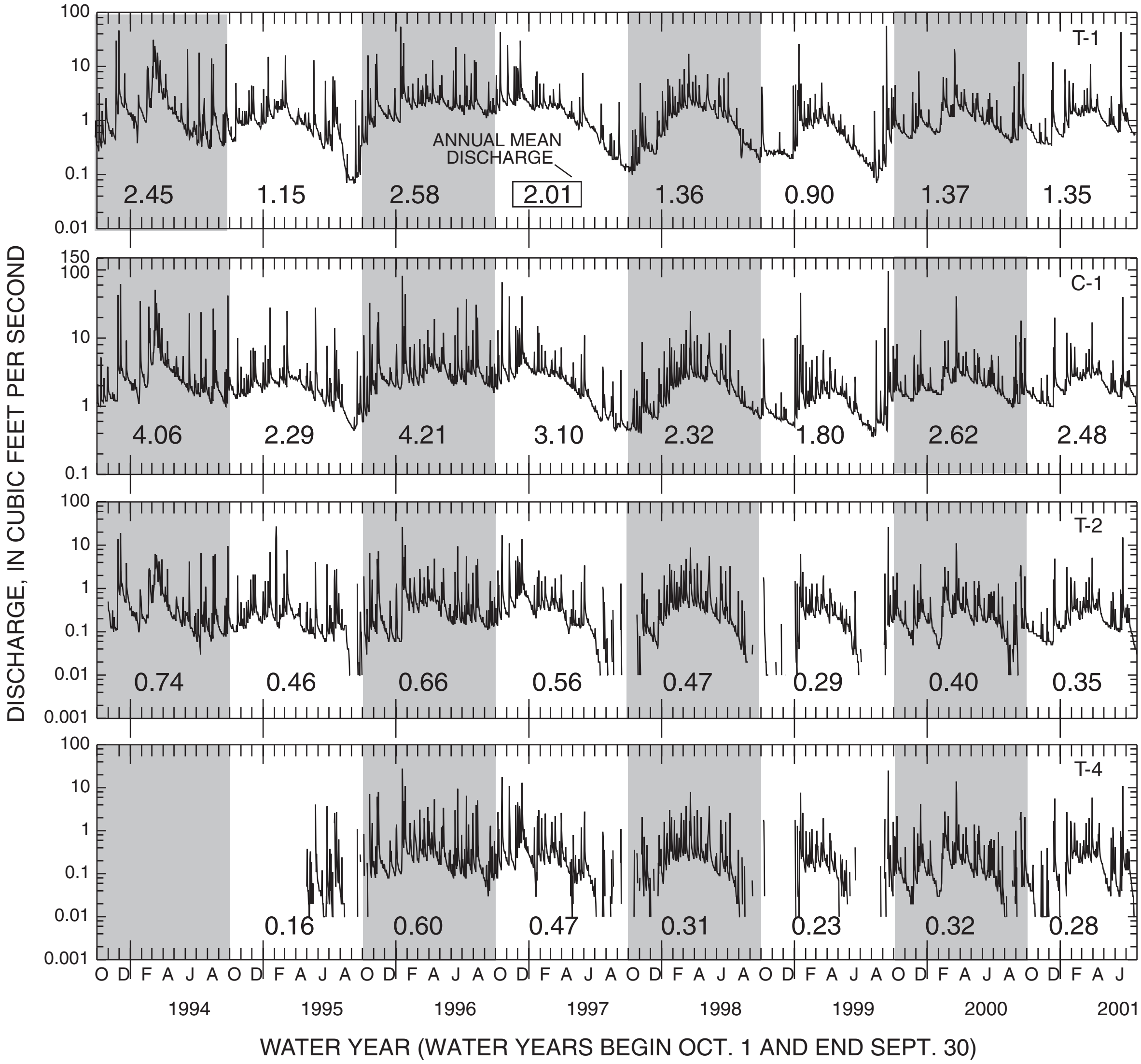


Figure 7. Graphs of daily and annual mean discharge for the four continuous surface-water sites in the Big Spring Run Basin, Lancaster County, Pa., from October 1993 through July 2001.

during the pre-treatment period relative to the post-treatment period. The lowest mean annual discharge (for an entire year of record) for each site was in WY 1999. Drought conditions existed for much of summer 1999. WY 1999 had the second lowest (37.8 in.) recorded annual amount of precipitation (fig. 6). WY 1995 had less precipitation than WY 1999, but much (12 in.) of the precipitation during WY 1999 was in September, and it is possible that much of that precipitation did not increase stream base flow until after the end of the water year. The highest mean discharge for three of the four sites was in WY 1996 when 54.6 in. of precipitation was recorded.

The amount of discharge per unit area was different between the treatment and control basins. Stream discharge for station C-1 indicated an average of 1.62 $(ft^3/s)/mi^2$ of drainage. The stations in the treatment basin averaged from 1.12 (T-4) to 1.36 $(ft^3/s)/mi^2$ (T-2), with the average for T-1 of 1.19 $(ft^3/s)/mi^2$.

Water Quality

The quality of surface water indicated by analyses of samples collected during periods of low flow and stormflow were analyzed separately and then compared to determine the effects of streambank fencing. It was expected that effects of fencing would vary with changes in flow. Data were flow-adjusted using LOWESS procedures, but the flow-adjusted data did not affect results from significance tests or ANCOVA; therefore, only unadjusted flow data are discussed in the results section.

Low Flow

Nutrient and suspended-sediment concentrations for the five surface-water sites sampled during low flow indicated agricultural activities did influence water quality, which would be expected considering that land use in the study basins was 80-90 percent agricultural.

Changes in Pre- and Post-Treatment Constituent Concentrations and Field Water-Quality Characteristics

The median concentration for nitrate-N for all the base-flow samples collected in both basins was 10.4 mg/L. The median value for nitrate-N was indicative of a basin with a high percentage of agricultural land use. A study conducted in 1995 in another agricultural basin in Lancaster County found concentrations of nitrate N from intensively (greater than 80 percent agricultural land use) farmed basins ranged from 5.6 to 22 mg/L; 44 percent of the samples were above 10 mg/L (Langland and Reed, 1996). Another study showed that nitrate-N concentrations increased with an increase in agricultural land use (Ott and others, 1991). This study showed concentrations of nitrate N for basins with about 20-30 percent agricultural land use averaged about 0.5-1.5 mg/L, whereas a basin with about 60 percent agricultural land use had nitrate-N concentrations of about 7-9 mg/L. Median values for total-P (0.04 mg/L) and suspended-sediment (14 mg/L) concentrations for all the low-flow samples collected in the Ott and others (1991) study did not show elevated concentrations relative to sites with less agricultural land use.

Field characteristics (pH, DO, water temperature, and SC) and fecal-streptococcus samples were only collected during low-flow sampling. Median values for DO (10.2 mg/L) and water temperature (13.8 °C) for all low-flow data indicated that, in general, the surface-water system was saturated or slightly above saturation for DO (Drever, 1982). DO concentrations at or slightly above saturation are indicative of systems where the oxygen is not depleted significantly by fauna consuming organic matter and photosynthesizing plants that are releasing oxygen are present in the channel (Hem, 1985). The pH and SC data were reflective of a carbonate system. Median pH and SC values for all the low-flow samples were 7.7 standard units and 672 μS/cm. These data corresponded well with ground-water samples summarized by Poth (1977), who reported median values for pH and SC for ground-water samples collected from the Conestoga Formation of 7.25 standard units and 690 μS/cm, respectively. The median value for fecal streptococcus for all the low-flow samples was 1,200 colonies per 100 milliliter (col/100 mL). Fecal-streptococcus samples collected from 15 predominantly agricultural sites in the Tulpehocken Creek Basin in Pennsylvania had median values that ranged from 1,200 to 38,000 col/100 mL (Barker, 1978). Another study in Lancaster County reported a median range for fecal streptococcus of 1,420 to 61,000 col/100 mL (Unangst, 1992). Thus, fecal-streptococcus data for all the sites did not indicate high concentrations relative to other agricultural sites in Pennsylvania.

The rank-sum test indicated significant differences at all sites in dissolved-nitrate concentrations from the pre-treatment to the post-treatment period. Each of the five sites where low-flow samples were collected showed a significant decrease (at an alpha value equal to 0.05) in nitrate. Percent reductions in nitrate from the pre- to the post-treatment period at T-1 and T-2 were 26 and 22 percent, respectively; the upstream sites (T-3 and T-4) showed 10- and 21-percent reductions, respectively. C-1 showed a 10-percent reduction in nitrate from pre- to the post-treatment period (fig. 8b). Approximately 96 percent of the dissolved N in all low-flow samples collected was in the form of nitrate; thus, significant reductions in nitrate equated to significant reductions in dissolved N. Dissolved N accounted for about 99 percent of the concentration of total N for all the low-flow samples collected.

The other forms of N at low flow included organic, ammonia, and nitrite (fig. 8b and 8c). Organic N accounted for about 3 percent of the dissolved-N fraction. Ammonia N and nitrite N both make up about 0.5 percent of the dissolved-N fraction. Concentrations of organic N were calculated from analysis of Kjeldahl N (organic plus ammonia N) and ammonia N; the difference between the two equaled organic N. The median dissolved organic-N concentration for all low-flow samples was 0.20 mg/L; median concentrations of dissolved

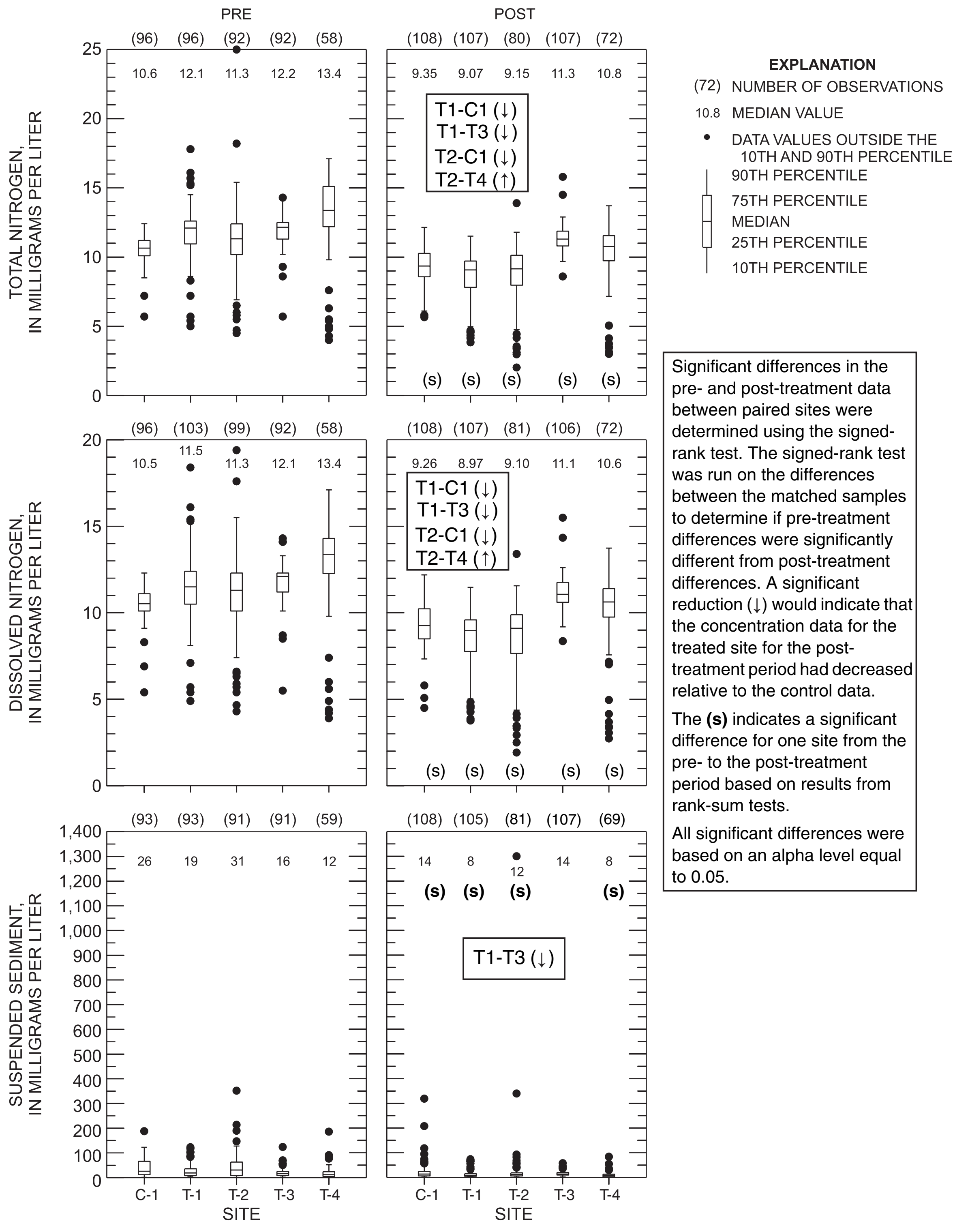
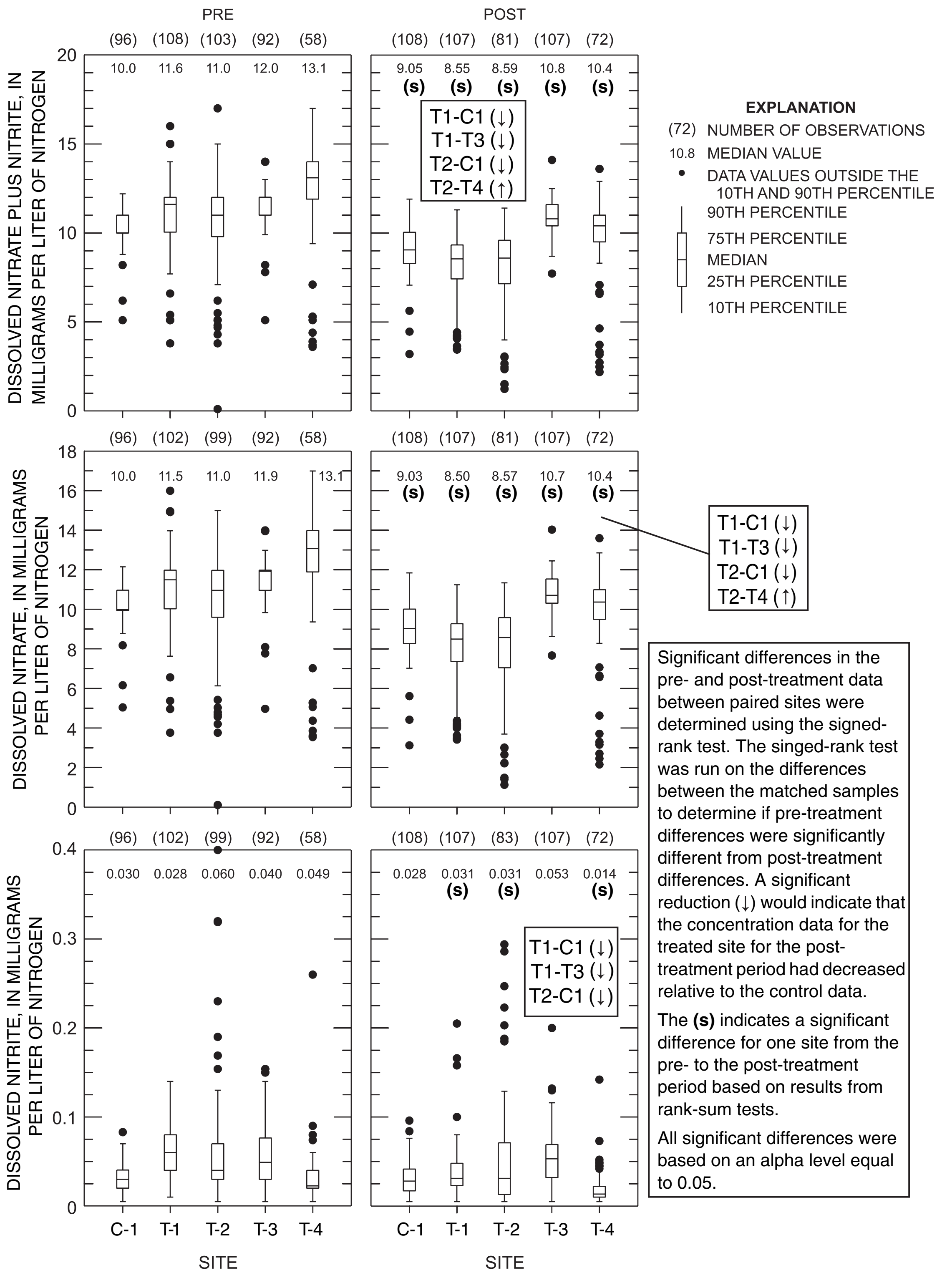


Figure 8a. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.



Significant differences in the pre- and post-treatment data between paired sites were determined using the signed-rank test. The singed-rank test was run on the differences between the matched samples to determine if pre-treatment differences were significantly different from post-treatment differences. A significant reduction (↓) would indicate that the concentration data for the treated site for the post-treatment period had decreased relative to the control data.

The **(s)** indicates a significant difference for one site from the pre- to the post-treatment period based on results from rank-sum tests.

All significant differences were based on an alpha level equal to 0.05.

Figure 8b. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

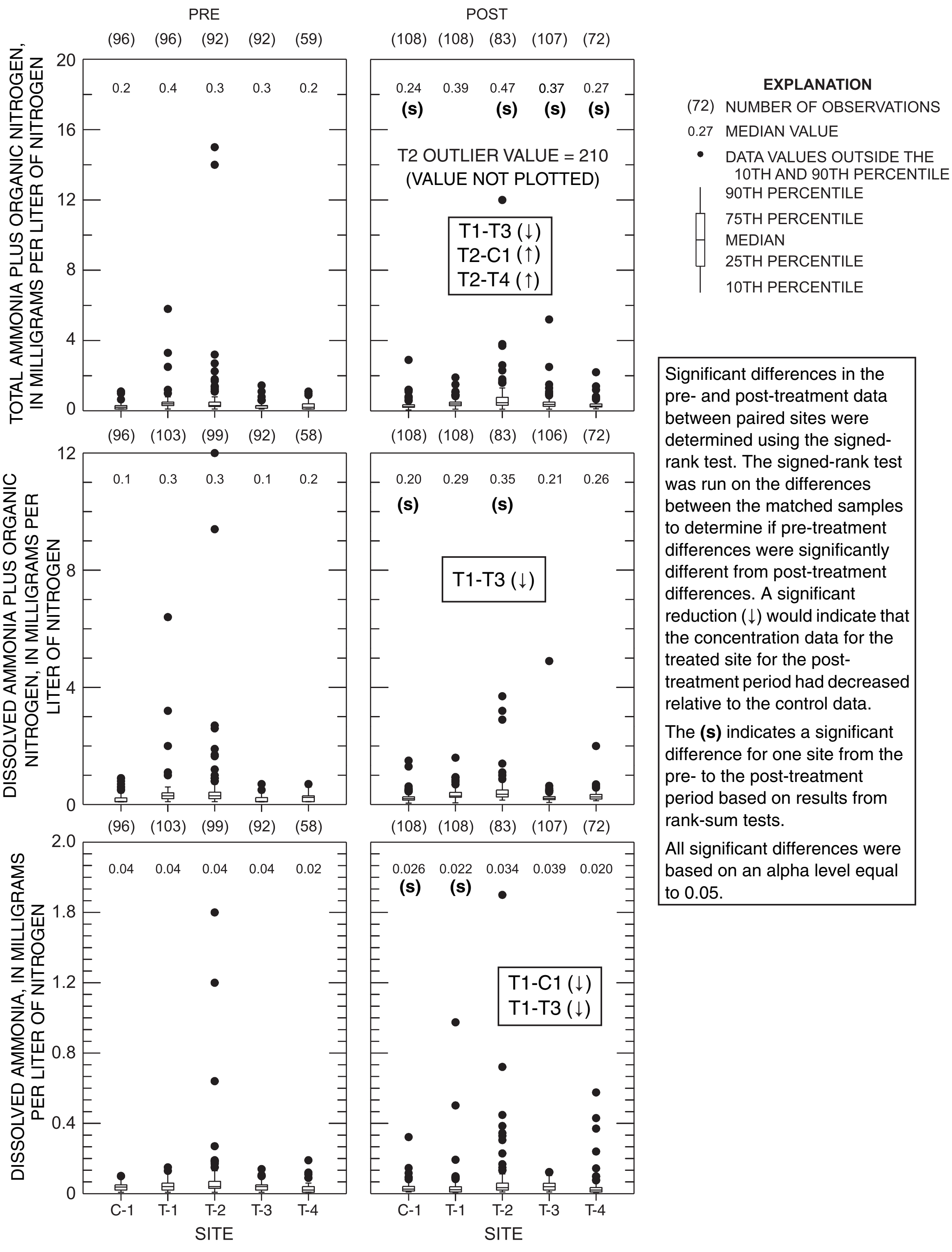


Figure 8c. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

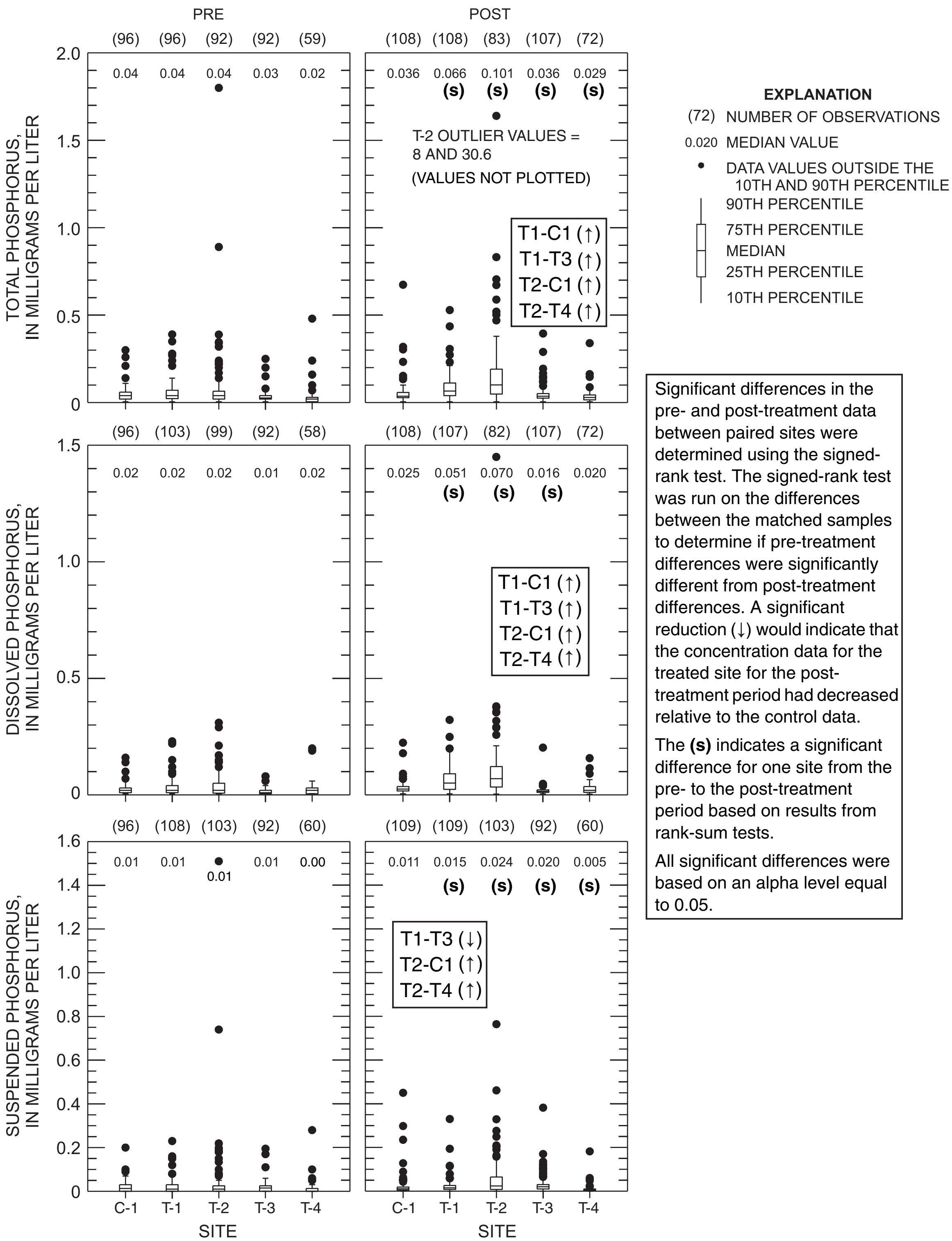


Figure 8d. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

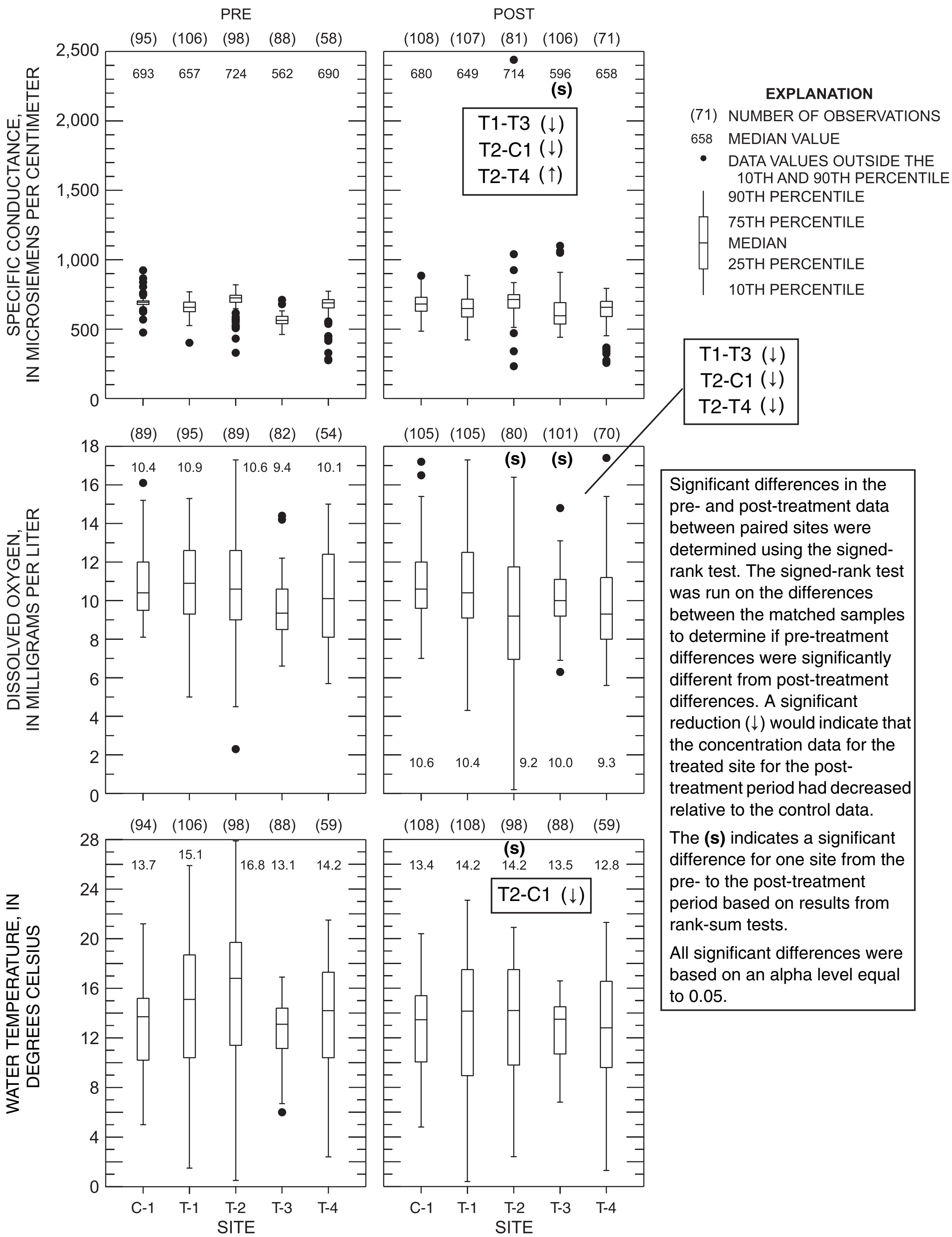


Figure 8e. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

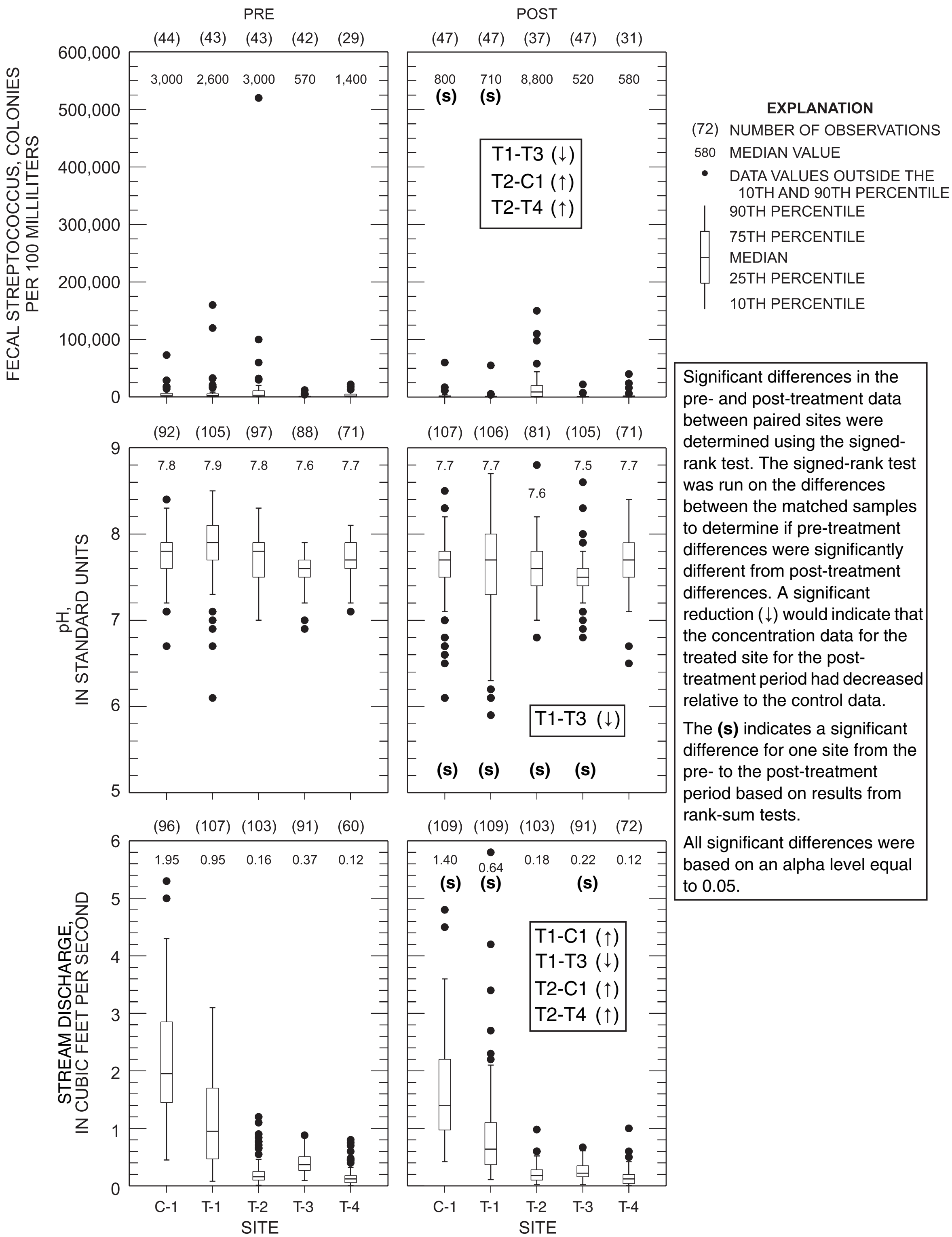


Figure 8f. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

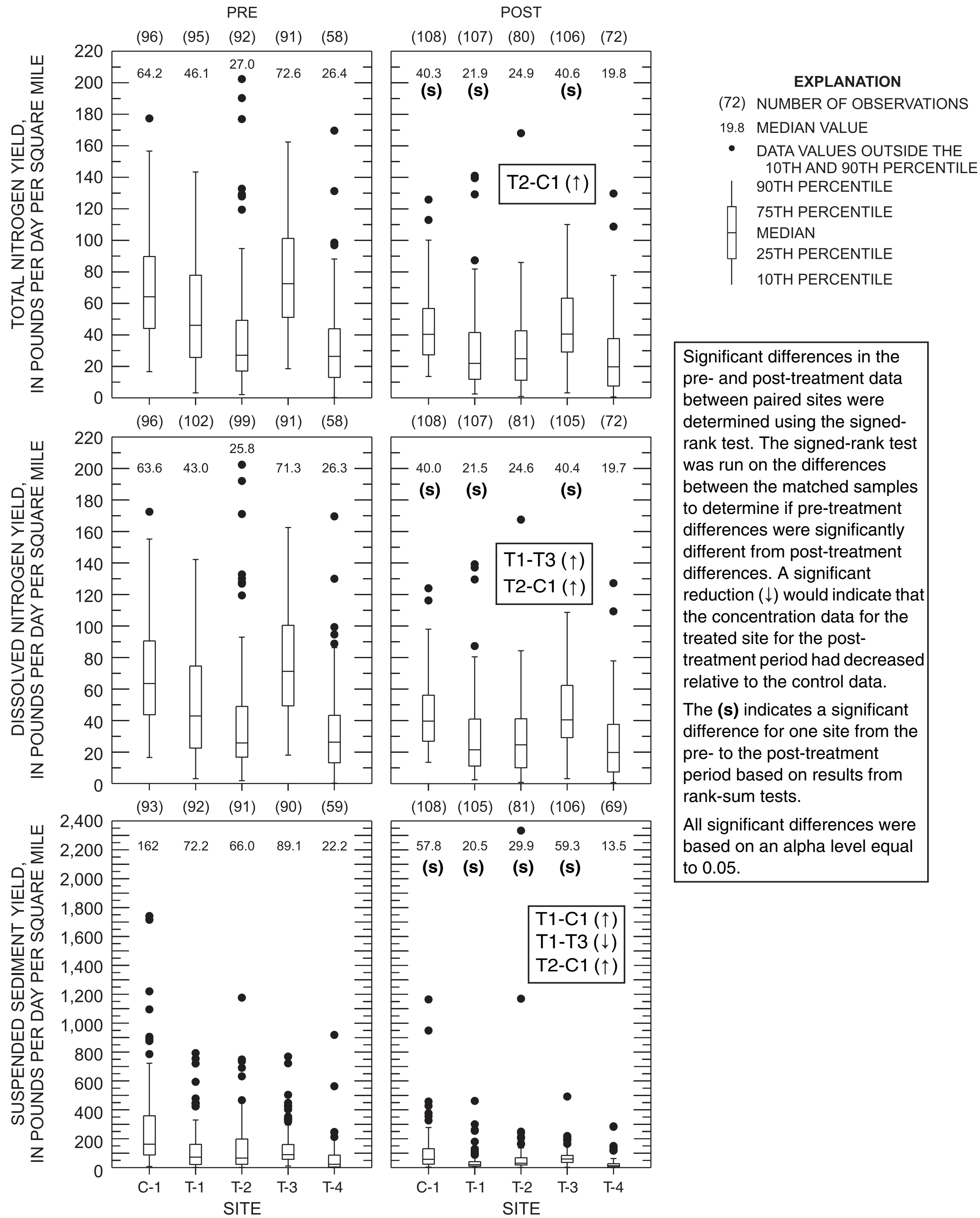


Figure 8g. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

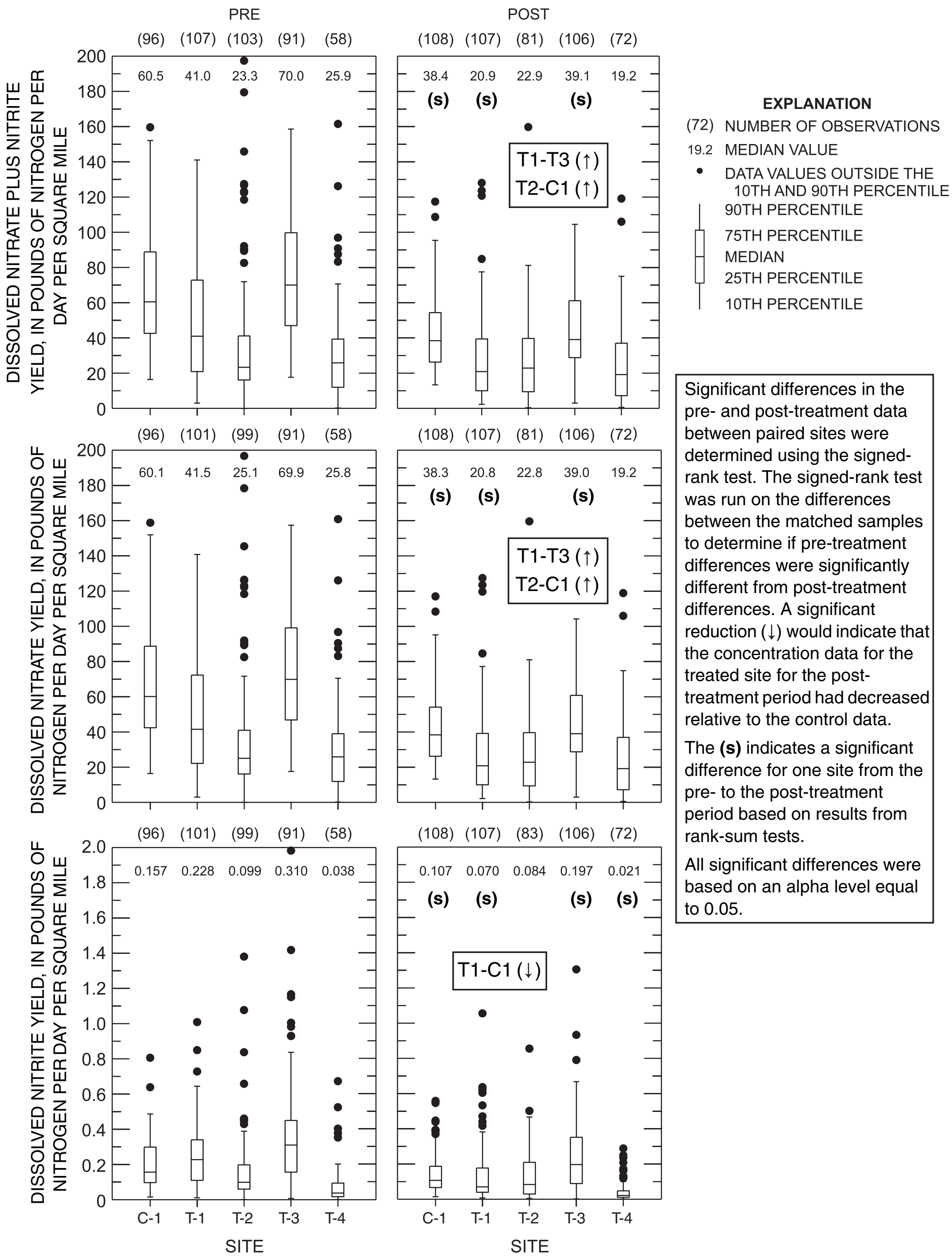


Figure 8h. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

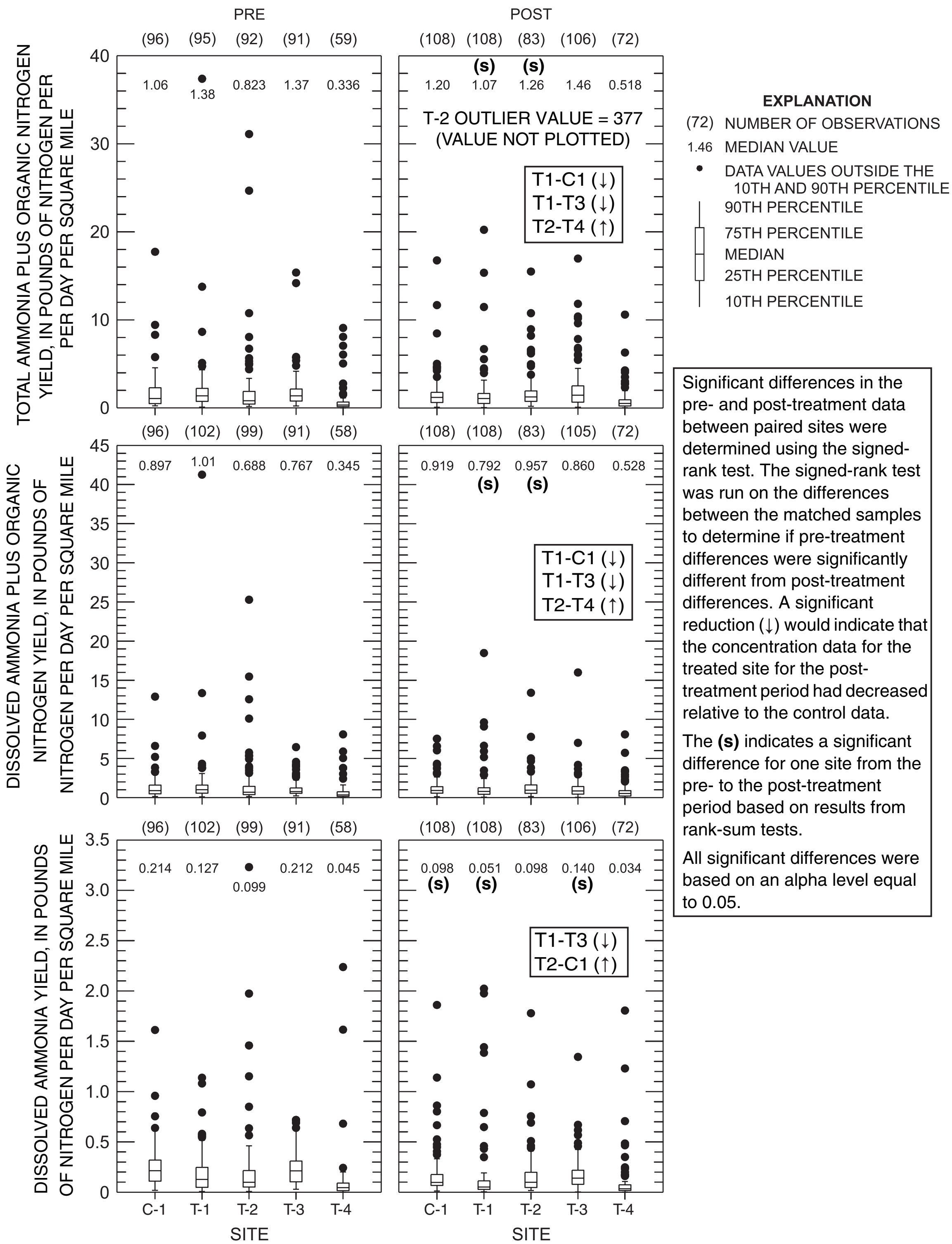


Figure 8i. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

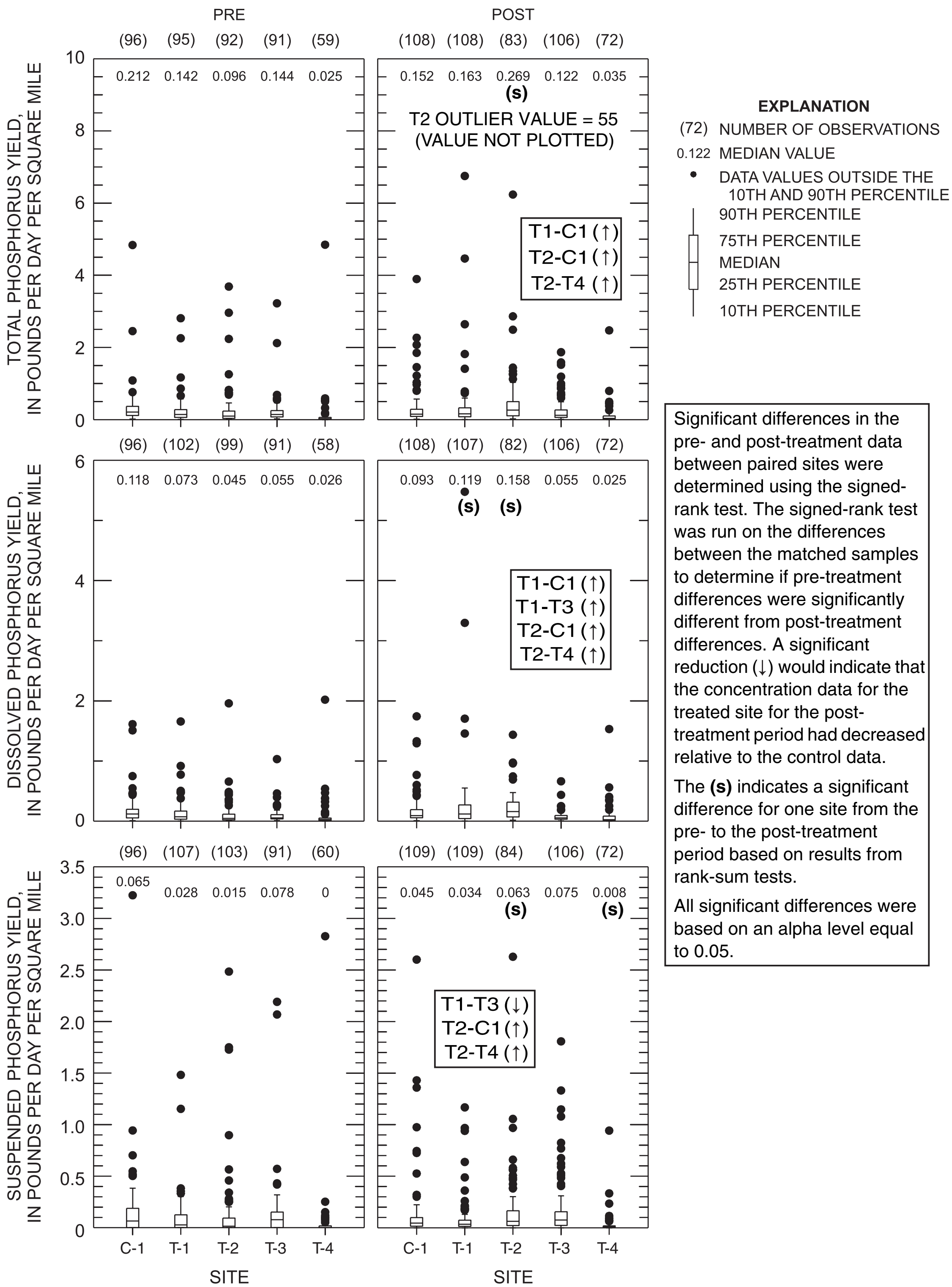


Figure 8j. Ranges of constituents, constituent yields, and discharge for low-flow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at five surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

nitrite N and dissolved ammonia N were 0.04 and 0.03 mg/L, respectively. Data for C-1 indicated a significant increase in dissolved organic N and a significant decrease in dissolved ammonia N from pre- to the post-treatment period, but no significant change in nitrite. Concentrations of dissolved organic N also showed significant increases from pre- to post-treatment for T-2 and T-3. Median concentrations for dissolved ammonia N and dissolved nitrite N for T-1 showed significant decreases of about 50 percent from the pre- to the post-treatment period. T-2 and T-4 also showed significant decreases in concentrations of dissolved nitrite N.

P concentrations for low-flow samples showed major differences between periods (pre- and post-treatment) in the treatment basin (fig. 8d). The percentage of dissolved P to total P for C-1, T-1, and T-2 for low-flow samples was about 70-73 percent. The percentage of dissolved P to total P for T-3 and T-4 was 57 and 96 percent, respectively. This indicated that almost all the total P measured at T-4 during low-flow sampling was in the dissolved form. Thus, activities upstream of T-4 were not contributing much suspended material that could act as a transport mechanism for P. Each site in the treatment basin showed a significant increase in concentrations of total and suspended P from the pre- to the post-treatment period. All sites in the treatment basin except for T-4 showed a significant increase in concentrations of dissolved P from the pre- to the post-treatment period. Conversely, C-1 did not show any significant difference from the pre- to post-treatment period in concentrations of total, dissolved, and suspended P. The greatest increase in concentration of total P from the pre- to the post-treatment period was at T-2, where the median concentration increased by 150 percent. Most of the increase at T-2 was caused by higher concentrations of dissolved P during the post-treatment period. The median concentrations of dissolved P at T-2 for the pre- and post-treatment periods were 0.02 and 0.07 mg/L, respectively. Median concentrations of dissolved P for T-4 did not change (remained at 0.02 mg/L) from pre- to the post-treatment period; therefore, elevated concentrations of dissolved P for T-2 were caused by processes downstream of the T-4 site. T-2 was only about 1,300 ft downstream from T-4, with 0.04 mi^2 more drainage at T-2 (fig. 1). The increased P in the upper part of the treatment basin was also evident at T-1. Median concentrations of total and dissolved P at T-1 from pre- to the post-treatment period increased about 0.03 mg/L (fig. 8d).

Suspended-sediment and fecal-streptococcus data showed similarities between basins over the study period. Suspended-sediment concentrations for low-flow samples showed significant decreases from the pre- to post-treatment period for all sites except for T-3 (fig. 8a). Median suspended-sediment concentrations for T-3 and T-4 decreased by 12 and 33 percent, respectively, from pre- to the post-treatment period; for T-1 and T-2, the percent decreases were 50 and 61 percent, respectively. The decrease in suspended-sediment concentration of 46 percent for C-1 from the pre- to the post-treatment period indicated streambank fencing was not the only factor in reducing suspended-sediment concentrations in the treatment basin; decreases in stream discharge also contributed to suspended-sediment reductions. Fecal-streptococcus data also showed significant decreases for T-1 and C-1 from the pre- to the post-treatment period (fig. 8f). The median values for fecal streptococcus for T-1 and C-1 decreased by 73 percent from the pre- to the post-treatment period. Sites (T-2, T-3, and T-4) in the upper part of the treatment basin did not show any significant change in fecal-streptococcus data before and after treatment.

Significant changes in field data from the pre- to the post-treatment period were most evident in the upper part of the treatment basin. T-3 showed a significant increase in DO concentration and SC, and a significant decrease in pH. T-2 showed significant decreases in DO, pH, and water temperature (fig 8e and 8f). At the outlets of the study basins (T-1 and C-1), there was only a significant decrease in pH. The decrease in water temperature at T-2 after fence installation was likely caused by vegetation reducing solar inputs to the stream channel. Water temperatures did decrease at T-1 (median decreased from 15.1 to 14.2 °C) from the pre- to the post-treatment period, but it was not significant, likely indicating that the effects of fencing on water temperature were being somewhat overwhelmed by solar inputs in stream stretches where fencing was not installed.

Changes in Instantaneous Yields of Nutrients and Suspended Sediment

Instantaneous yields for N species for low-flow samples showed significant decreases from the pre- to the post-treatment period for sites along the mainstem (fig. 8g, 8h, and 8i). This was related to less stream discharge during the post-treatment period. Sites along the main channel in both basins were C-1, T-1, and T-3. During the post-treatment period, 107-108 samples were collected at each of these sites during low flow. Only 81 and 72 samples, respectively, were collected at T-2 and T-4 during the post-treatment period. The lower sample numbers at T-2 and T-4 were because of no flow at the time the sample was to be collected. The days of no flow at T-2 and T-4 were likely the days with the least amount of flow at the mainstem sites. Instantaneous yields are determined by multiplying concentration by discharge, so these days with the least amount of flow had the lowest yields, and this helped to produce a significant decrease in N yields during the post-treatment period. Stream discharge at the time of sample collection showed a significant decrease from the pre- to post-treatment period for C-1, T-1, and T-3. T-2 and T-4 did not show significant changes in the distribution of stream discharge from the pre- to the post-treatment period (fig. 8f). C-1, T-1, and T-3 showed significant decreases in yield for nitrate-N (38, 49, and 44 percent, respectively), nitrite-N (32, 69, and 29 percent, respectively), and ammonia-N (54, 60, and 33 percent, respectively).

Instantaneous yields for P showed major differences between the control and treatment basins (fig. 8j). C-1 data showed no significant change in P yields from the pre- to post-treatment period, but median yields did decrease. Upstream sites in the treatment basin also did not show significant changes in P yields (except for T-4, which showed a significant increase in yields of suspended P from the pre- to post-treatment period). T-2 showed significant increases in the yield of dissolved (250-percent increase) and suspended P (319-percent increase); T-1 only showed an increase in yield of dissolved P (63-percent increase). The significant increase in instantaneous P yields at T-2 indicated some process(es) were contributing P to the stream channel.

Instantaneous suspended-sediment yields for low-flow samples showed significant decreases from the pre- to post-treatment period for all sites except T-4 (fig. 8g). T-4 did show a 39-percent decrease in the median suspended-sediment yield from the pre- to post-treatment period; for T-3, the median yield was reduced only 33 percent, but the decrease was significant. Percent decreases in suspended-sediment yield for the other sites ranged from 55 (T-2) to 72 percent (T-1).

Post-Treatment Changes

Post-treatment changes in water-quality constituents for low-flow samples were quantified using the ANCOVA approach along with equation 1. These quantified changes during the post-treatment period were determined for concentrations and yields. Changes in yield were also determined using a weighted approach that used annual variations from the overall mean discharge to adjust flow.

Table 19. Summary of percent change in water-quality constituents between treated and control/upstream surface-water sites in the Big Spring Run Basin, Lancaster County, Pa., for low-flow samples collected during the post-treatment relative to the pre-treatment period.

[N, nitrogen; TKN, total ammonia plus organic nitrogen; DKN, dissolved ammonia plus organic nitrogen; P, phosphorus; NS, no significant relation]

Constituent	Percent change[1]			
	(T-1)-(C-1)	(T-1)-(T-3)	(T-2)-(C-1)	(T-2)-(T-4)
Total-N concentration	-24	-22	-19	-4.2
Dissolved-N concentration	-24	-22	-19	-5.5
Dissolved-(nitrate+nitrite) concentration	-24	-21	-17	-5.1
Dissolved-nitrate concentration	-24	-21	-17	-5.1
Dissolved-nitrite concentration	-39	-40	-21	-2.8
TKN concentration	-7.6	-19	+40	+40
DKN concentration	-7.1	-12	-1.3	+11
Dissolved-ammonia concentration	-35	-34	-6.5	-13
Total-P concentration	+57	+42	+190	+140
Dissolved-P concentration	+57	+65	+140	+100
Suspended-P concentration	+43	+16	+140	+150
Suspended-sediment concentration	-40	-52	-24	NS
Fecal-streptococcus counts	-46	-75	+300	+190
Dissolved-oxygen concentration	NS	-6.9	-18	-6.7
pH	-1.3	-3.0	-1.2	-.86
Specific conductance	-.13	-6.5	-2.0	+.31
Water temperature	-4.7	-6.7	-8.8	-.76
Total-N yield	-20	-.35	+20	-4.6
Dissolved-N yield	-21	+1.1	+20	-5.8
Dissolved-(nitrate+nitrite) yield	-14	+2.3	+32	-6.1
Dissolved-nitrate yield	-22	+.78	+32	-6.2
Dissolved-nitrite yield	-46	-47	NS	+3.8
TKN yield	-27	-31	+42	+62
DKN yield	-25	-22	NS	+15
Dissolved-ammonia yield	NS	-43	NS	+3.8
Total-P yield	+31	+53	+210	+180
Dissolved-P yield	+36	+46	+160	+130
Suspended-P yield	+46	+9.2	+140	+160
Suspended-sediment yield	-43	-53	-6.7	+3.6

[1] The percent change is negative if there was a decrease for the treated site relative to the control site during the post-treatment period. For example, the -24 value for total N for (T-1)-(C-1) indicates a 24-percent reduction in total-N concentration for T-1 during the post-treatment period when the relation to C-1 is taken into account.

Concentrations for N species generally showed a decrease for the treated sites during the post-treatment period relative to control sites (table 19). The paired relations showed a reduction of 22-24 percent in concentrations of total and dissolved N and nitrate N at the outlet of the treatment basin (T-1); for T-2, the reduction ranged from 4 to 19 percent. Ammonia-N concentrations also showed a larger percent reduction at T-1 (34-35 percent) than at T-2 (average reduction of about 10 percent). Total (TKN) and dissolved (DKN) forms of Kjeldahl N showed a decrease (range of 7 to 19 percent) for T-1, but data for T-2 showed a significant increase in TKN (40 percent). DKN data for T-2 showed either a very small reduction (1.3 percent) or a slight increase (11 percent). Kjeldahl N is the sum of organic and ammonia N. Given that ammonia N showed a reduction at T-2, the increase in TKN was caused by increased organic N.

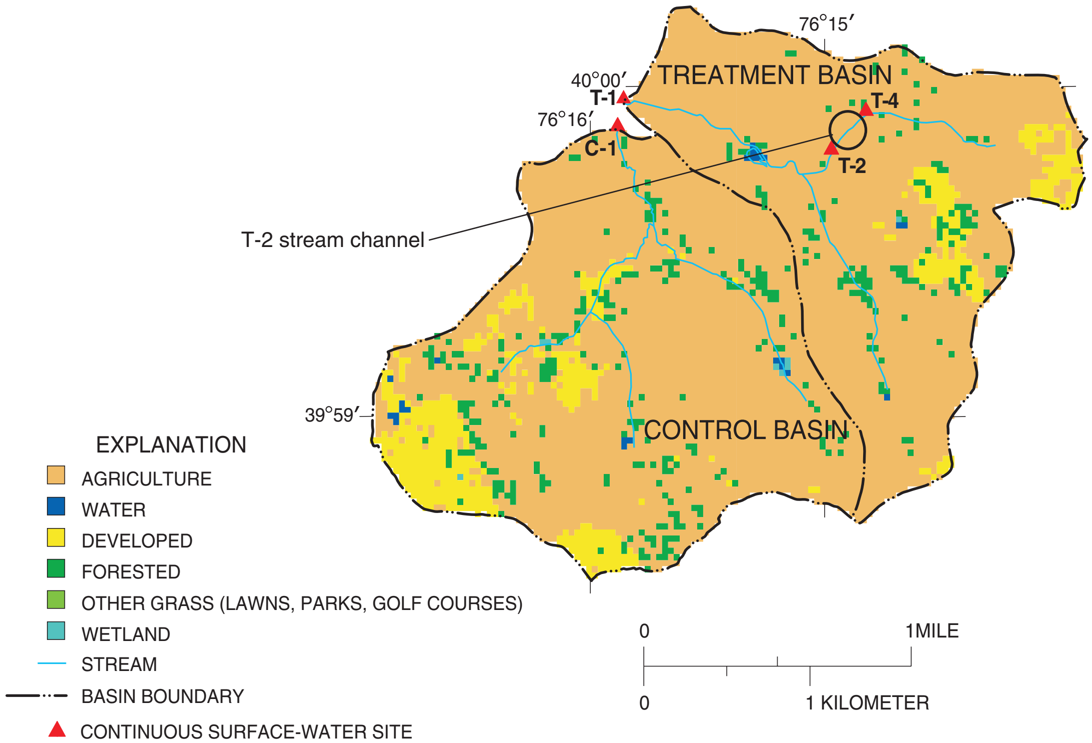
Concentrations of P for low-flow samples increased after fence installation in the treatment basin relative to control sites. Data from the study basin indicated the source of the elevated P concentrations was the drainage area and possibly the stream channel between sites T-4 and T-2. During August 2000, seven grab samples were collected in the stream channel between T-4 and T-2 (fig. 9). These data showed increased concentrations of dissolved P, DKN, and ammonia immediately downstream of cattle crossings. Concentrations of dissolved P increased from 0.029 mg/L 800 ft upstream of the T-2 weir to 0.291 mg/L for a sample collected immediately downstream of the weir. Concentrations of dissolved P for shallow ground-water wells at T-2 showed a significant increase (for two of three wells) from the pre- to the post-treatment period. Median concentrations of dissolved P for wells LN2037, LN2038, and LN2040 were 0.005, 0.005, and 0.0075 mg/L, respectively, during the pre-treatment period; medians increased to 0.008, 0.0155, and 0.0350 mg/L, respectively, during the post-treatment period.

Suspended-sediment and fecal-streptococcus data for low-flow samples at T-1 showed significant reductions relative to control sites during the post-treatment period, but data for T-2 were variable (fig. 8a, 8f, and 8g). Suspended-sediment concentrations showed significant reductions for T-1 and T-2. Similar to N, the reductions at T-1 (about 45-percent reduction) were greater than at T-2. Fecal-streptococcus data indicated major differences between T-1 and T-2. Percent reductions in fecal-streptococcus colonies for T-1 ranged from 46 to 75 percent. Conversely, fecal-streptococcus data for T-2 indicated a 200-300 percent increase during the post-treatment period relative to control sites. The percentage increase in fecal-streptococcus colonies was probably related to the significant increase in TKN for T-2 during the post-treatment period. Increased organic-N concentrations for T-2 during the post-treatment period caused the significant increase in TKN. Thus, for T-2 during the post-treatment, significant increases occurred in organic-N concentration and fecal-streptococcus colonies. After fence installation the authors observed that dairy cows in the pasture at T-2 would tend to excrete waste products when using the cattle crossings. Increased concentrations of some nutrients downstream of the crossings verified this personal observation (fig. 9), as did the data for fecal streptococcus.

Field characteristics showed similar responses at T-1 and T-2 relative to control sites during the post-treatment period. Values for pH, DO, SC, and water temperature at T-1 and T-2 decreased relative to control sites during the post-treatment period, except for a 0.31 percent increase in SC at T-2 relative to T-4. The largest percent reduction was for DO (average decrease of 10.5 percent). This decrease may be because of decreased flow during the post-treatment period. Personal observation of the stream channel between T-2 and T-4 during the post-treatment period showed more stagnant pools or slow-moving water compared to the pre-treatment period. Reductions in water temperature were likely caused by increased vegetative cover (which would reduce solar inputs to water in channel) along the stream channel in the treatment basin.

Post-treatment changes in the instantaneous yield of N species for low-flow samples showed differences between T-1 and T-2. Percent changes in yield for low-flow samples from the pre- to the post-treatment period were determined for unweighted (table 19) and weighted-discharge approach (table 20) ANCOVA models. Data for T-1 indicated, in general, reductions in yields for all N species during the post-treatment period relative to control sites. Unweighted yields for dissolved nitrate N for T-1 did show a slight percent increase (0.78 percent) relative to T-3, but weighted yield for this same comparison showed a 34-percent reduction. The average weighted yield reduction for T-1 for total and dissolved N and nitrate N was about 25 percent; the unweighted average reduction was about 10 percent. The average weighted and unweighted yield increase for T-2 for total and dissolved N and nitrate N for low-flow samples was about 8-10 percent (tables 19 and 20). Instantaneous yields of DKN, TKN, and ammonia N for T-1 also showed percent reductions (average reduction of 21, 25, and 35 percent, respectively) during the post-treatment period. Instantaneous yields of DKN, TKN, and ammonia N for T-2 also showed percent increases (average increases of 31, 74, and 15 percent, respectively) at T-2 during the post-treatment period.

Instantaneous P yields for low-flow samples showed significant increases for both treatment sites (T-1 and T-2) during the post-treatment period relative to control sites (tables 19 and 20). Both dissolved and suspended forms of P showed significant increases. The average increase during the post-treatment period at T-1 in dissolved and suspended forms of P was about 42-43 percent, which equated to a 58 percent increase in the instantaneous yield of total P. The average increases during the post-treatment period at T-2 in dissolved and suspended forms of P were about 170 and 180 percent, respectively, which equated to about a 210-percent increase in the instantaneous yield of total P. Stream discharge during the post-treatment period (table 18) showed the ratio of stream discharge of T-2 to T-1 was 32 percent. The ratio of the percent increase in instantaneous P yields between T-1 and T-2 was 20-30 percent. Thus, it appears the source of the increased instantaneous P



Distance above T2 weir (feet)	Discharge (cfs)	Dissolved phosphorus (mg/L)	DKN-N (mg/L)	Dissolved Nitrate-N (mg/L)
0	0.021	0.291	0.58	5.1
50		CATTLE CROSSING		
53	-	0.078	-	6.40
200	0.015	0.071	0.31	6.71
370	-	0.058	0.31	7.04
520	-	0.129	0.60	6.51
610		CATTLE CROSSING		
620	0.0005	0.089	0.30	3.63
660	-	0.031	0.20	4.06
770	-	0.029	0.43	3.79

Figure 9. Study area map with stream channel above T-2 identified, and table of water-quality data collected in channel on August 17, 2000 in the Big Spring Run Basin, Lancaster County, Pa. [cfs, cubic feet per second; mg/L, milligrams per liter; DKN, dissolved ammonia plus organic nitrogen; N, nitrogen; -, no data].

Table 20. Summary of percent change (determined using a weighted-discharge approach) in constituent yields between treated and control/upstream surface-water sites in the Big Spring Run Basin, Lancaster County, Pa., for low-flow samples collected during the post-treatment relative to the pre-treatment period.

[N, nitrogen; TKN, total ammonia plus organic nitrogen; DKN, dissolved ammonia plus organic nitrogen; P, phosphorus; NS, no significant relation]

Constituent	Percent change[1]			
	(T-1)-(C-1)	(T-1)-(T-3)	(T-2)-(C-1)	(T-2)-(T-4)
Total-N yield	-15	-34	+14	+5.4
Dissolved-N yield	-15	-34	+13	+3.6
Dissolved-(nitrate+nitrite) yield	-14	-33	+17	+2.0
Dissolved-nitrate yield	-14	-34	+17	+1.9
Dissolved-nitrite yield	-39	-45	-6.2	+5.7
TKN yield	-11	-31	+80	+110
DKN yield	-10	-27	+33	+45
Dissolved-ammonia yield	-20	-41	+39	+2.6
Total-P yield	+51	+98	+270	+200
Dissolved-P yield	+48	+43	+210	+170
Suspended-P yield	+43	+71	+210	+220
Suspended-sediment yield	-32	-55	+2.6	+18

[1] The percent change is negative of there was a decrease for the treated site relative to the control site during the post-treatment period. For example, the -15 value for total-N yield for (T-1)-(C-1) indicates a 15-percent reduction in total-N concentration for T-1 during the post-treatment period when the relation to C-1 is taken into account.

yield for T-1 during the post-treatment period was the drainage area and (or) stream channel between T-2 and T-4.

Changes in the instantaneous suspended-sediment yields for low-flow samples during the post-treatment period were different at T-1 and T-2. The average percent reduction in the instantaneous suspended-sediment yield during the post-treatment period for T-1 was 46 percent; for T-2, the average percent increase was 4 percent (tables 19 and 20).

Stormflow

Nutrients and suspended-sediment concentrations for stormflow indicated water chemistry was much different than for low-flow samples. The median concentrations of total and dissolved N for all stormflow samples collected in the study basin were 5.24 and 3.99 mg/L, respectively; for all low-flow samples collected, the median concentrations for total and dissolved N were 10.87 and 10.70 mg/L, respectively. Nitrate N was the only nutrient species with a higher median concentration for low flow (10.35 mg/L) compared to stormflow (2.88 mg/L). The median ammonia-N concentration for all stormflow samples (0.18 mg/L) was 0.15 mg/L higher than the median for low-flow samples. The increased ammonia N in stormflow samples helped to increase TKN and DKN concentrations for stormflow relative to low flow, but organic-N concentrations were also higher in stormflow. P concentrations in stormflow were at least an order of magnitude higher than in low-flow samples. The median concentrations of total P for all low-flow and stormflow samples were 0.04 and 0.57 mg/L, respectively. The median concentrations of dissolved P for stormflow (0.22 mg/L) was a magnitude higher than the median for low-flow samples. Median suspended-sediment concentrations were much higher for stormflow samples (227 mg/L) than for low-flow samples (14 mg/L); this was expected given that the mean discharge at which low-flow samples were collected was 0.39 ft^3/s, and the mean discharge for all the storm samples was 6.3 ft^3/s.

Changes in Pre- and Post-Treatment Constituent Concentrations

The rank-sum tests indicated concentrations of N species for stormflow samples collected in the treatment basin did not change significantly during the post-treatment period relative to pre-treatment conditions. Data for T-1 and T-2 did not show any significant changes in N species during the post-treatment period. Significant decreases from the pre- to the post-treatment period were evident for nitrate plus nitrite and nitrate for stormflow samples collected at C-1 (fig. 10a). For all stormflow samples, nitrate N made up 73 percent of dissolved N or 58 percent of the total amount of N. Organic N in stormflow accounted for 36 percent of the total N; the remaining N fraction was made up of ammonia N (4.5 percent) and nitrite N (1 percent). Data for C-1 for low-flow samples also indicated a significant decrease in nitrate-N concentrations during the post-treatment period (fig. 8b). For all sites except T-2, there was a significant decrease during the post-treatment in the

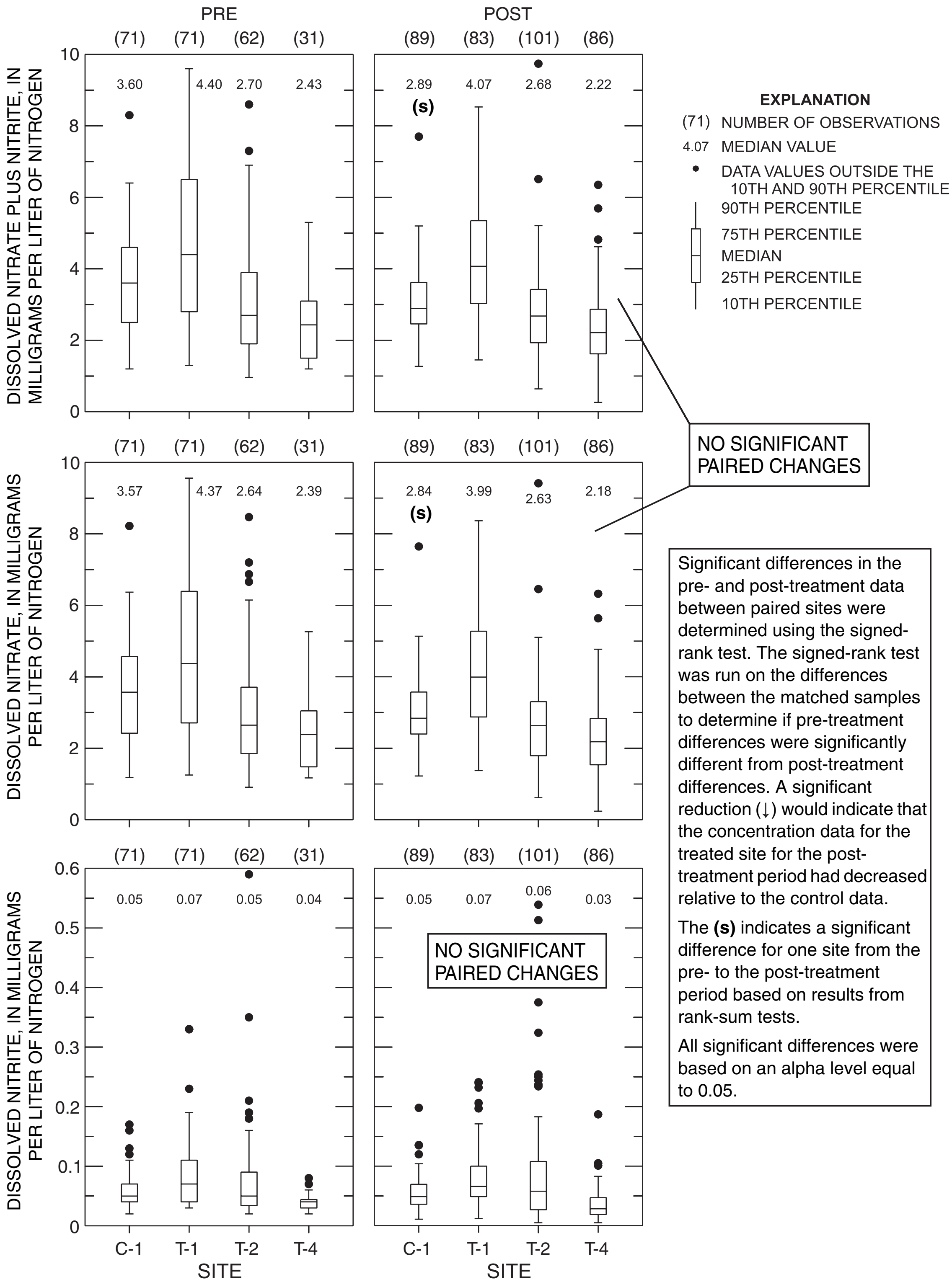


Figure 10a. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.

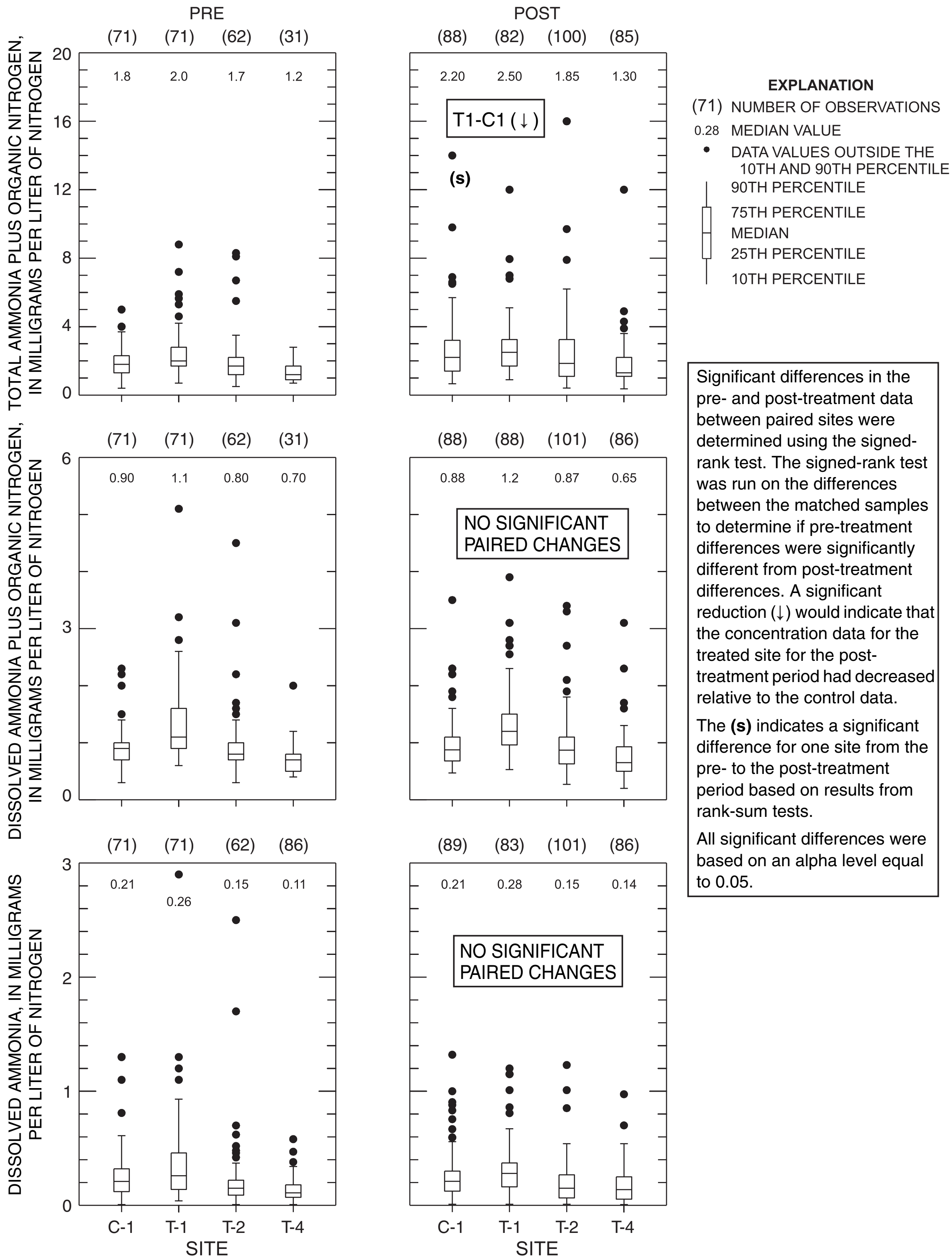


Figure 10b. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

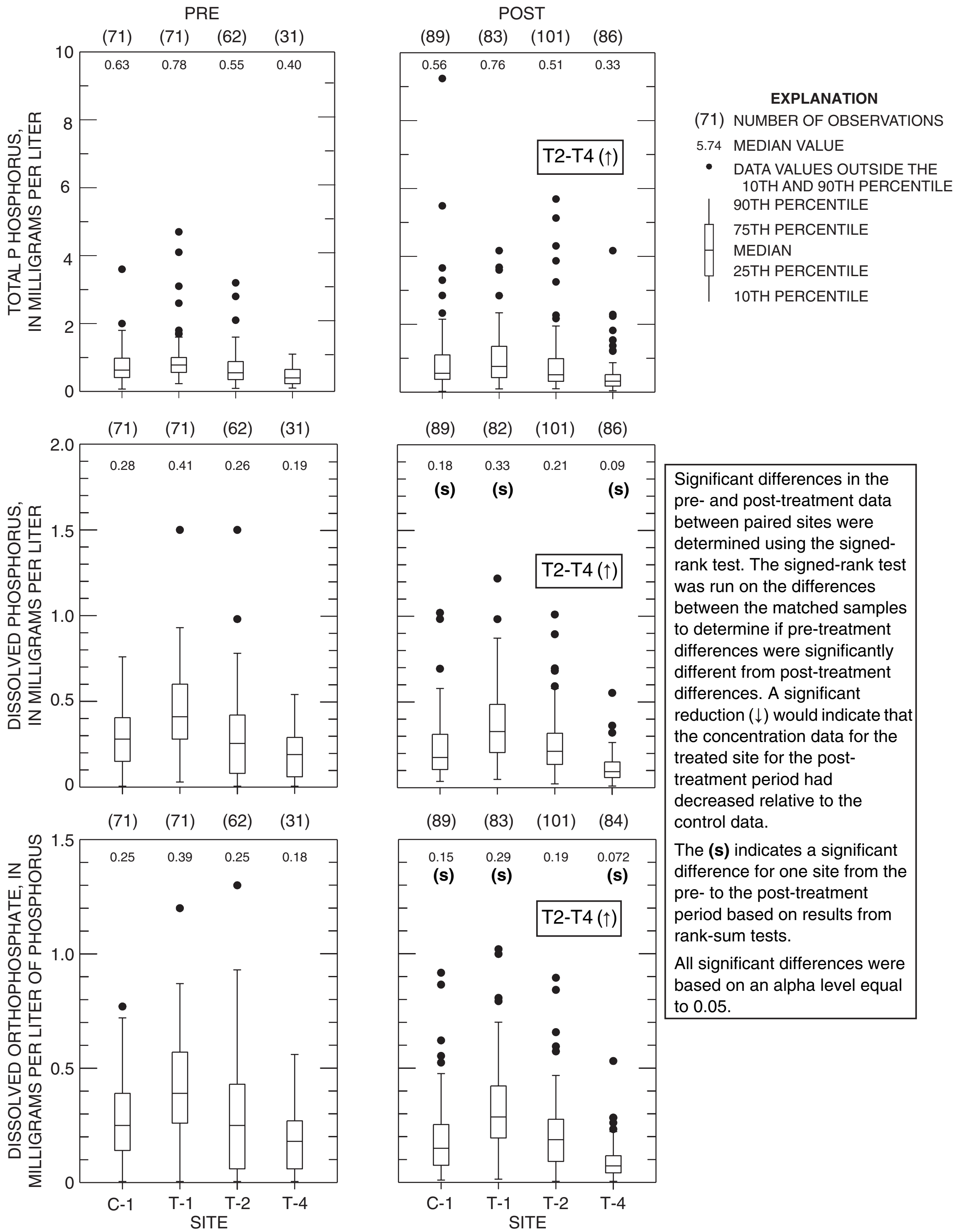


Figure 10c. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

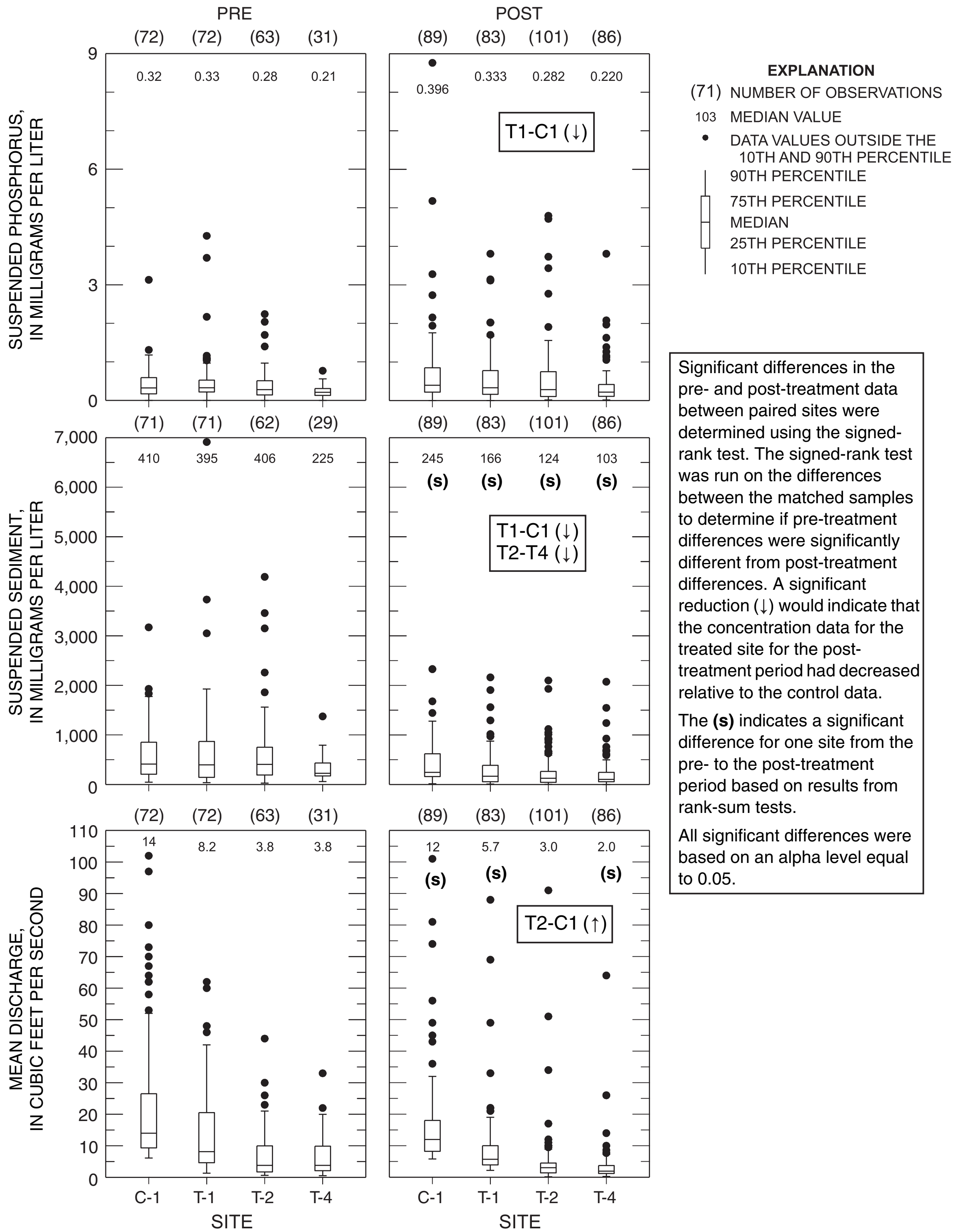


Figure 10d. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

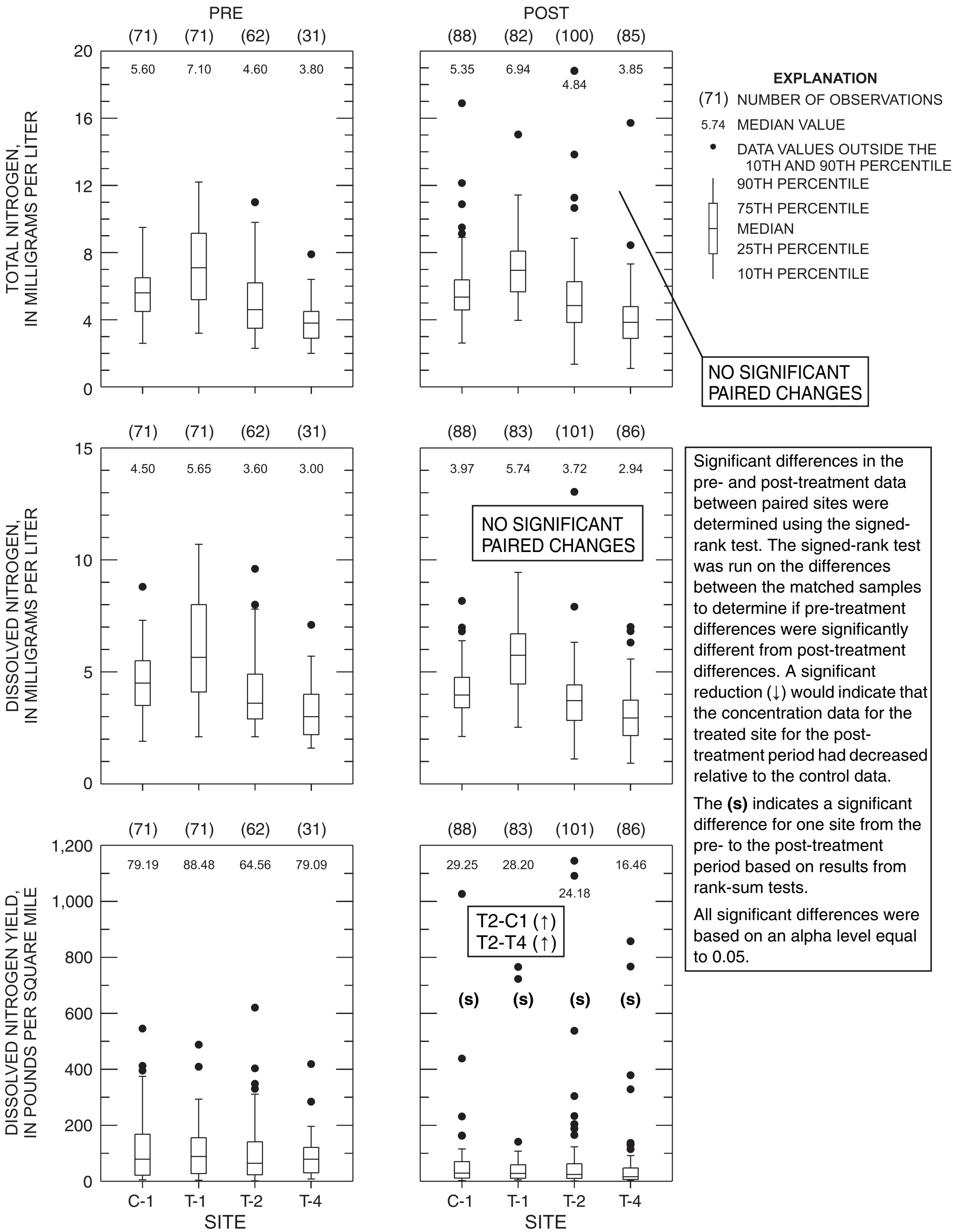
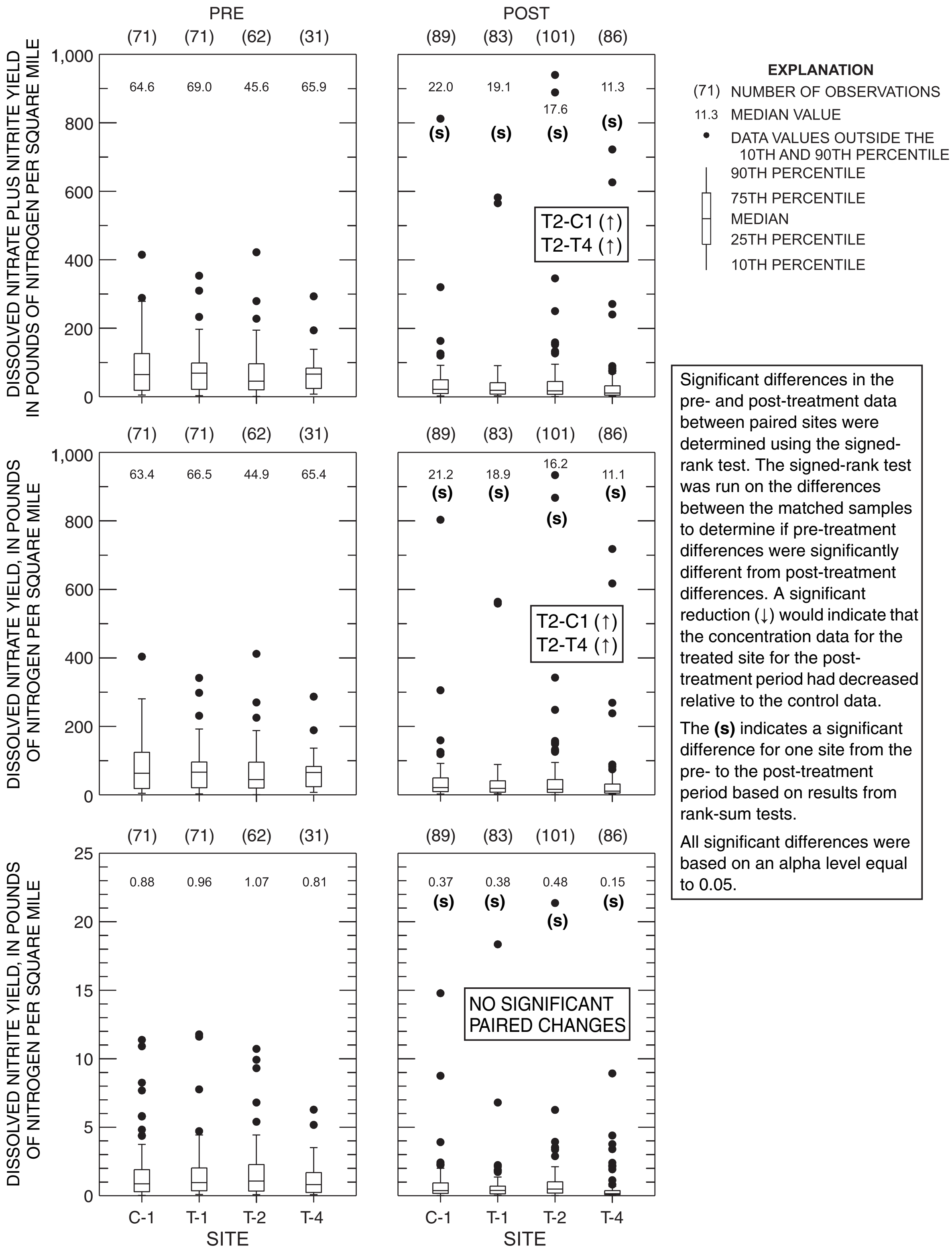


Figure 10e. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued



Significant differences in the pre- and post-treatment data between paired sites were determined using the signed-rank test. The signed-rank test was run on the differences between the matched samples to determine if pre-treatment differences were significantly different from post-treatment differences. A significant reduction (↓) would indicate that the concentration data for the treated site for the post-treatment period had decreased relative to the control data.

The **(s)** indicates a significant difference for one site from the pre- to the post-treatment period based on results from rank-sum tests.

All significant differences were based on an alpha level equal to 0.05.

Figure 10f. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

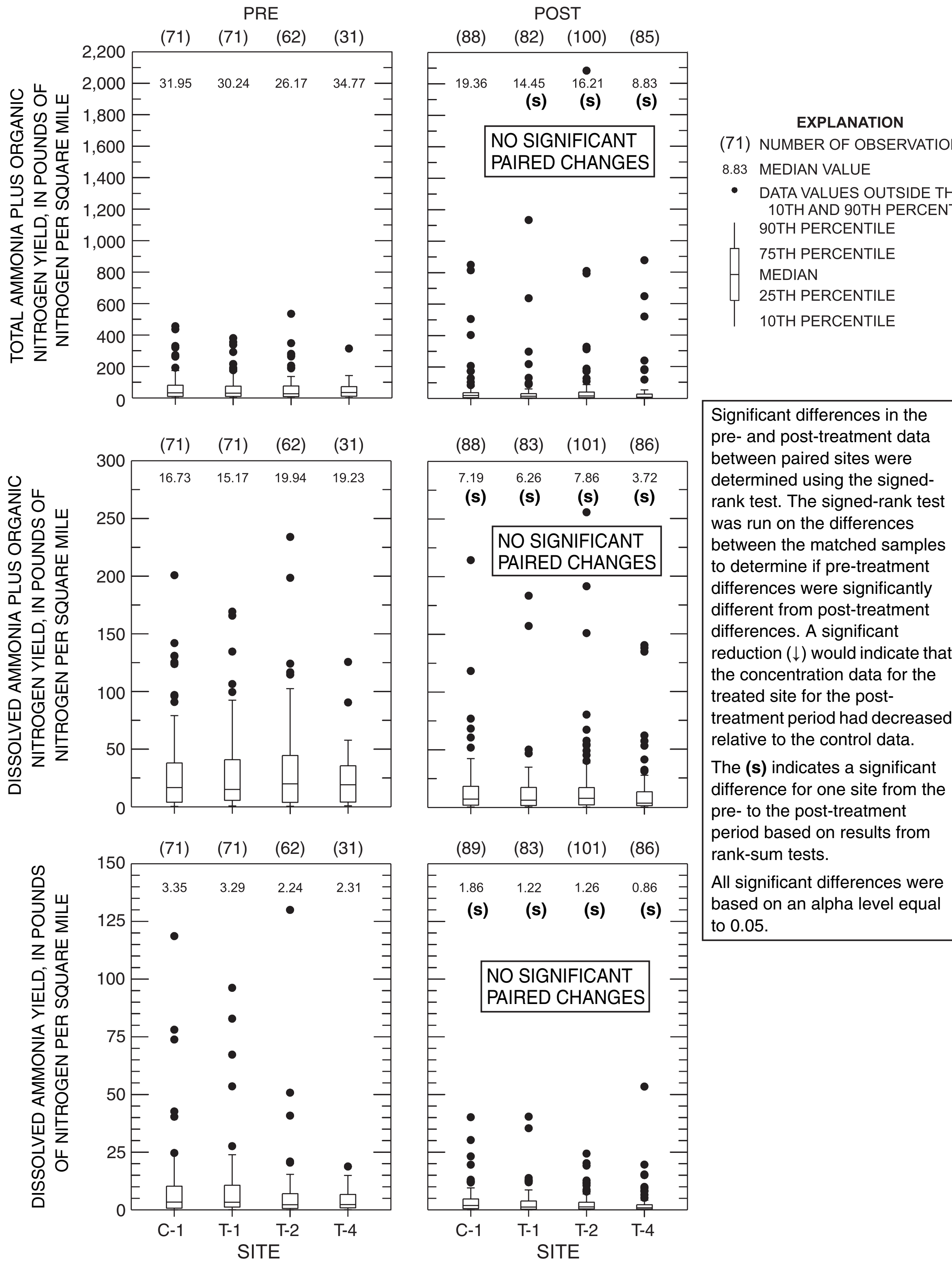


Figure 10g. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

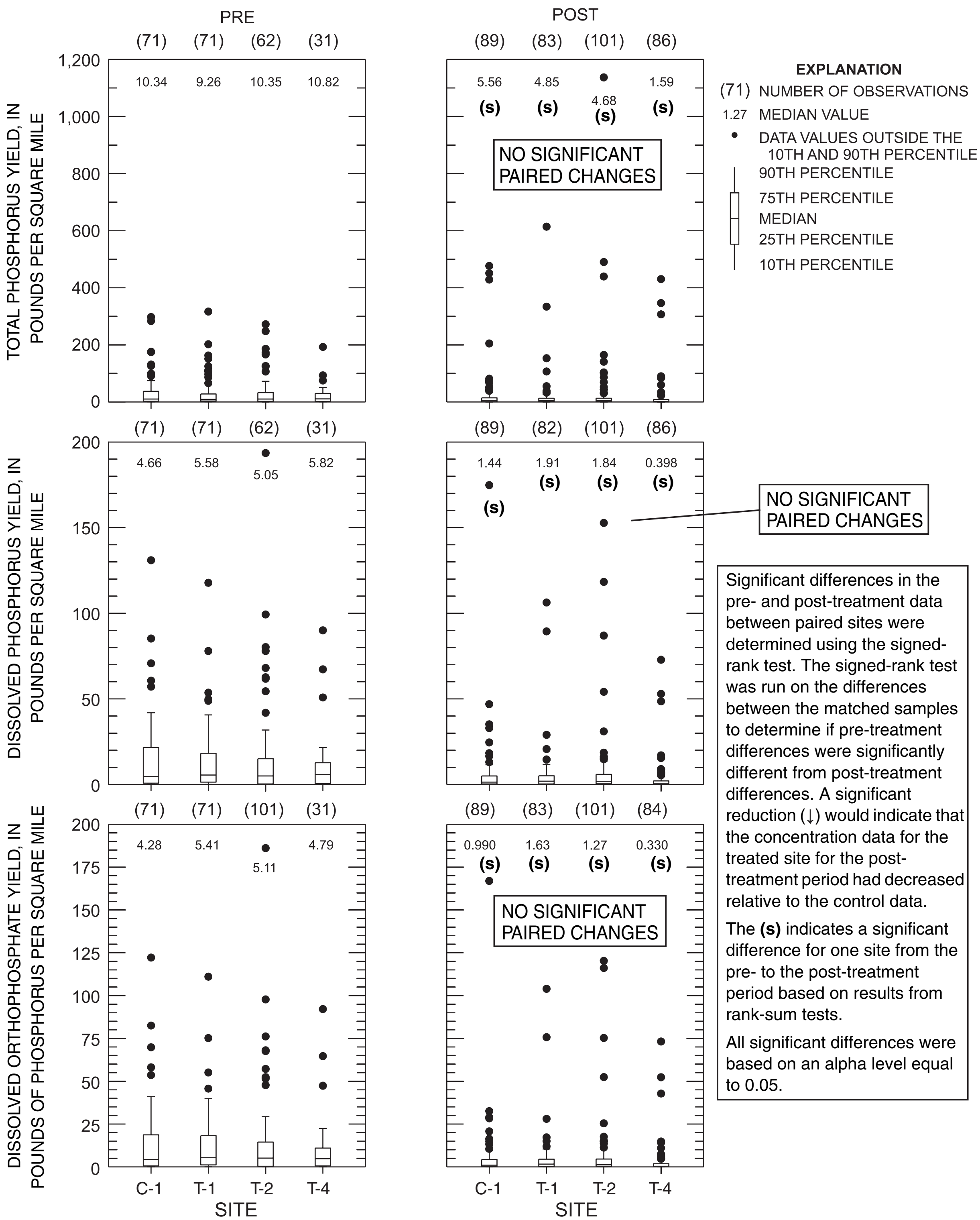
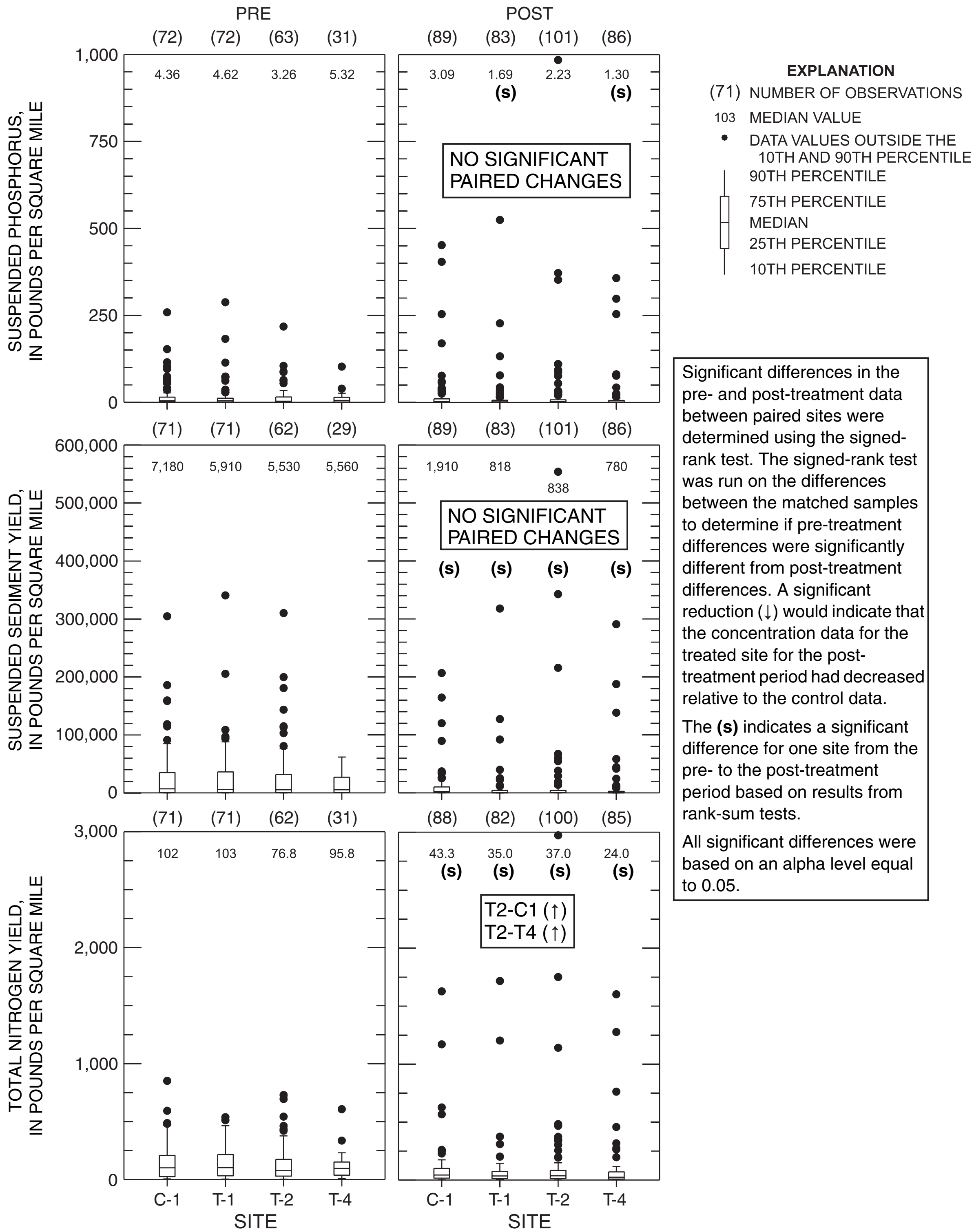


Figure 10h. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued



Significant differences in the pre- and post-treatment data between paired sites were determined using the signed-rank test. The signed-rank test was run on the differences between the matched samples to determine if pre-treatment differences were significantly different from post-treatment differences. A significant reduction (↓) would indicate that the concentration data for the treated site for the post-treatment period had decreased relative to the control data.

The **(s)** indicates a significant difference for one site from the pre- to the post-treatment period based on results from rank-sum tests.

All significant differences were based on an alpha level equal to 0.05.

Figure 10i. Ranges of constituents, constituent yields, and discharge for stormflow samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at four surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued mean discharge sampled for storm events. The significant decrease during the post-treatment period in concentrations of nitrate N in stormflow samples was partially caused by the lower concentration of nitrate N in the base-flow component of the stormflow.

P concentrations for stormflow samples showed similar changes from the pre- to post-treatment period for the all sites except for T-2. Concentrations of dissolved P and orthophosphate P decreased significantly during the post-treatment relative to the pre-treatment period for T-1, C-1, and T-4. C-1 and T-4 showed larger percentage reductions than T-1. Decreases in the median concentrations of dissolved P and orthophosphate P ranged from 37 to 50 percent and 40 to 60 percent, respectively, for C-1 and T-4. Decreases in the median concentrations of dissolved P and orthophosphate P for T-1 were 20 and 27 percent, respectively (fig. 10c). The lack of significant reductions in P for T-2 was expected given that low-flow data indicated a significant increase in dissolved P for T-2 (fig. 8d). This increased dissolved P evident at T-2 during low flow likely contributed to the lower percentage decreases (relative to the control sites) in dissolved P in stormflow for T-1.

Suspended-sediment concentrations for stormflow samples showed significant decreases at all sites from the pre- to post-treatment period. The largest percent decreases were at T-1 and T-2 (56 and 69 percent, respectively) rather than C-1 and T-4 (40 and 54 percent, respectively). The primary reason suspended-sediment concentrations decreased at all sites from the pre- to the post-treatment period was the decreased mean discharge for storms sampled during the post-treatment period (fig. 10d). The decrease in the median discharge sampled at all sites during the post-treatment period relative to pre-treatment period ranged from 14 (C-1) to 49 percent (T-4). The larger percentage decreases in suspended-sediment concentrations for the treated sites were likely because of the fence installation and the establishment of a vegetative buffer strip along the stream channel.

Changes in Stormflow Yields of Nutrients and Suspended Sediment

The yield of N species in stormflow samples showed significant decreases from pre- to the post-treatment period for all sites. The largest percentage decreases in N species during the post-treatment period were at T-4 (figs. 10e, 10f, and 10g). This was likely related to differences in the sampling periods between T-4 and the other sites. The first storm sample at T-4 was collected in June 1995; the first samples at the other continuous sites were collected in winter 1993 - spring 1994. WY 1996 had the highest annual mean discharge at each site of any water year (fig. 7). Most storm samples collected at T-4 during the pre-treatment period were collected during WY 1996. The higher flows for WY 1996 probably caused the yield distribution for T-4 to be relatively higher during the pre-treatment period than the other sites, hence the higher percentage reductions in N species for T-4 during the post-treatment period. Decreases in median yields of total N from the pre- to post-treatment period for stormflow samples ranged from 52 percent for T-2 to 75 percent for T-4. The overall decreases in N yields were caused by the lower mean discharges (table 17) during the post-treatment period. In general, the N species that showed the greatest reduction from pre- to post-treatment period was nitrate; reductions ranged from 61 (T-2) to 83 percent (T-4).

The yield of total and dissolved P and suspended sediment in stormflow also showed significant reductions for all sites from the pre- to the post-treatment period (figs. 10h and 10i). Again, the greatest percentage reductions were evident for T-4 and this can also be attributed to differences in the flow distribution for T-4 relative to the other sites. The decrease in the median yield of total P from the pre- to the post-treatment period was 85 percent for T-4; the range for the other sites was 46 (C-1) to 55 percent (T-2). The decrease in the median suspended-sediment yield from the pre- to the post-treatment period was 86 percent for T-4, while the range for the other sites was 73 (C-1) to 86 percent (T-1). Decreased P and suspended-sediment yields for all the sites were again partially caused by decreased mean discharge during the post-treatment period relative to pre-treatment period.

Post-Treatment Changes

Post-treatment changes in water-quality constituents for stormflow samples were quantified using the ANCOVA approach along with equation 1. These quantified changes during the post-treatment period were determined for concentrations and yields. Changes in yield were also determined using a weighted approach that used annual variations from the overall mean discharge to adjust flow.

ANCOVA results indicated post-treatment changes (at treated sites relative to control sites) in concentrations of N species for stormflow samples were variable. The only N species that showed a reduction during the post-treatment period at the treated sites relative to control sites was ammonia (table 21). Reductions in concentrations of ammonia N at the treated sites (relative to the relation to control sites) ranged from 14 to 30 percent. Stormflow data for N species showed N concentrations in the treatment basin increased with an increase in drainage area (fig. 10a, 10b, and 10e). T-2 showed greater reductions in N species (relative to C-1) than T-1. However, relative to T-4, T-2 showed increases in all N species during the post-treatment period. The percentage change of total-N concentrations during the post-treatment period for the treated sites ranged from a 13-percent reduction to about a 9-percent increase.

ANCOVA results showed a decrease in concentrations of total P for T-1 and an increase for T-2 during the post-treatment period relative to control sites. Both T-1 and T-2 showed percentage increases (ranging from 8 to 56 percent) in concentrations of dissolved P during the post-treatment period (table 21). The reasons for increased concentrations in the treatment basin were discussed earlier. Given that T-2 also showed higher percentage increases relative to C-1 and

T-4 in concentrations of total P for low flow (table 19), it was expected that T-2 would also show higher percentage increases for stormflow. The percentage increase in concentrations of total P for T-2 relative to control sites ranged from 8 to 30 percent. Percentage increases in concentrations of dissolved P for T-2 were greater than percentage increases in concentrations of suspended P. T-1 showed a 13-percent decrease in total-P concentration during the post-treatment period relative to C-1 data, mainly because of a 20-percent reduction in suspended-P concentrations.

ANCOVA results showed a decrease in suspended-sediment concentrations in stormflow samples for T-1 and T-2 during the post-treatment period relative to control sites (table 21). Percentage reductions in suspended-sediment concentrations were greater at T-2 than T-1. The range in the percentage reductions for suspended-sediment concentrations was from 31 to 58 percent.

Post-treatment changes in the instantaneous yield of N species for stormflow samples showed differences between T-1 and T-2. Percentage changes in yield for stormflow samples from the pre- to the post-treatment period were determined for unweighted (table 21) and weighted-discharge approach (table 22) ANCOVA models. T-1 showed percentage reductions in all N species for unweighted and

Table 21. Summary of percentage change in water-quality constituents between treated and control/upstream surface-water sites in the Big Spring Run Basin, Lancaster County, Pa., for stormflow samples collected during the post-treatment relative to the pre-treatment period.

[N, nitrogen; TKN, total ammonia plus organic nitrogen; DKN, dissolved ammonia plus organic nitrogen; P, phosphorus; NS, no significant relation]

Constituent	Percent change[1]		
	(T-1)-(C-1)	(T-2)-(C-1)	(T-2)-(T-4)
Total-N concentration	-6.2	-13	+8.9
Dissolved-N concentration	+3.5	-11	+1.8
Dissolved-(nitrate+nitrite) concentration	+6.2	-15	+1.8
Dissolved-nitrate concentration	+6.3	-15	+2.4
Dissolved-nitrite concentration	-18	-21	+32
TKN concentration	-6.5	-14	+15
DKN concentration	NS	-15	+11
Dissolved-ammonia concentration	-23	-30	-14
Total-P concentration	-13	+8.4	+30
Dissolved-P concentration	+8.1	+18	+56
Orthophosphate concentration	+22	+23	+60
Suspended-P concentration	-20	+2.5	+20
Suspended-sediment concentration	-31	-58	-35
Total-N yield	-23	+18	+50
Dissolved-N yield	-20	+19	+53
Dissolved-(nitrate+nitrite) yield	-33	+23	+54
Dissolved-nitrate yield	-33	+24	+53
Dissolved-nitrite yield	-28	+6.7	+41
TKN yield	-30	+13	+42
DKN yield	-23	+1.9	+55
Dissolved-ammonia yield	-41	-15	+25
Total-P yield	-26	+27	+69
Dissolved-P yield	NS	+62	+130
Orthophosphate yield	+6.0	+64	+150
Suspended-P yield	-40	+36	+35
Suspended-sediment yield	-38	-47	-50

[1] The percent change is negative of there was a decrease for the treated site relative to the control site during the post-treatment period. For example, the -6.2 value for total-N yield for (T-1)-(C-1) indicates a 6.2-percent reduction in total-N concentration for T-1 during the post-treatment period when the relation to C-1 is taken into account.

weighted-discharge approach ANCOVA results. Averaging unweighted and weighted results, reductions in yields of total and dissolved N for T-1 were 19 and 15 percent, respectively. Conversely, for T-2, average increases in the yields of total and dissolved N were 25 and 30 percent, respectively. Ammonia, which showed the greatest reductions in concentrations using ANCOVA results, also showed the greatest reductions in yield for all N species for T-1, and, for T-2, ammonia was the only N species to show a reduction for one of the paired relations for T-2 (tables 21 and 22).

ANCOVA results for P yields for stormflow samples showed major differences between T-1 and T-2. Similar to ANCOVA results for concentrations of total P, yields of total P during the post-treatment period showed a reduction for T-1 (average of 22 percent), and for T-2, the average increase in yields of total P was 46 percent (tables 21 and 22). The reduction in total-P yield for T-1 was mostly attributed to reductions in the yield of suspended P, which showed an average reduction of 36 percent. The percent increase in yield of total P for T-2 during the post-treatment period was mainly attributable to increased yields of dissolved P. The average increase from ANCOVA results for T-2 was 88 percent.

Table 22. Summary of percent change (determined using a weighted-discharge approach) in constituent yields between treated and control/upstream surface-water sites in the Big Spring Run Basin, Lancaster County, Pa., for stormflow samples collected during the post-treatment relative to the pre-treatment period.

[N, nitrogen; TKN, total ammonia plus organic nitrogen; DKN, dissolved ammonia plus organic nitrogen; P, phosphorus]

Constituent	Percent change[1]		
	(T-1)-(C-1)	(T-2)-(C-1)	(T-2)-(T-4)
Total-N yield	-15	+16	+30
Dissolved-N yield	-10	+19	+31
Dissolved-(nitrate+nitrite) yield	-8.7	+22	+32
Dissolved-nitrate yield	-8.6	+23	+31
Dissolved-nitrite yield	-17	+17	+42
TKN yield	-24	+30	+26
DKN yield	-14	+23	+34
Dissolved-ammonia yield	-33	-6.4	+32
Total-P yield	-18	+37	+52
Dissolved-P yield	+14	+62	+99
Orthophosphate yield	+22	+74	+120
Suspended-P yield	-33	+55	-68
Suspended-sediment yield	-35	-47	-39

[1] The percent change is negative of there was a decrease for the treated site relative to the control site during the post-treatment period. For example, the -15 value for total-N yield for (T-1)-(C-1) indicates a 15-percent reduction in total-N concentration for T-1 during the post-treatment period when the relation to C-1 is taken into account.

ANCOVA results for suspended-sediment yields showed percent reductions for T-1 and T-2 during the post-treatment period. Post-treatment reductions at the treated sites ranged from 35 to 50 percent. The average percent reductions in the suspended-sediment yield at T-1 and T-2 were 36 and 46 percent, respectively. These results show that streambank fencing, even with a buffer width of 5-12 ft, helped to reduce suspended-sediment yields during storm events.

Annual Yields

Annual yields of nutrients and suspended sediment for the four continuous sites were estimated for low-flow and stormflow data separately using multiple-regression procedures. The percentage of low flow was based on the storm-sampling procedures for each site. That is, the criteria used to initiate and end storm sampling was used to isolate stormflow periods for which storm samples were not collected. Multiple-regression procedures were used to estimate loads for low-flow periods and non-sampled storm events. The loads for sampled storms were known. The estimated and known loads were summed in order to determine the percentage of low flow and stormflow for each water year. Any periods not considered stormflow were considered low-flow periods (table 23).

The overall yields and the breakdown of yields into base-flow and stormflow components were somewhat similar between T-1 and C-1 and between T-2 and T-4; however, sites at the outlet (T-1 and C-1) did show variations relative to upstream sites (T-2 and T-4). Mean annual yields for nitrate N, nitrite N, and ammonia N generally were higher for T-1 and C-1 than for T-2 and T-4 (table 23). Conversely, mean annual yields for DKN, TKN, dissolved P, and total P generally were lower for T-1 and C-1 than T-2 and T-4.

Nitrate N accounted for most of the yield of total N at each site. Ninety percent of the yield of total N for C-1 was nitrate N. Nitrate N accounted for 82, 78, and 74 percent of the yield of total N for T-1, T-2, and T-4, respectively. TKN accounted for only 9 percent of the yield of total N for C-1; for T-1 and T-2, TKN accounted for 17 percent of the yield of total N.

The dissolved to total P ratios for annual yields were similar for outlet sites but dissimilar for T-2 and T-4. The dissolved to total P ratio for C-1 and T-1 was 34 percent. Only 17 percent of the yield of total P for T-2 was in dissolved form; the ratio for T-4 was 55 percent. The dissimilarity in the ratio for T-2 and T-4 was partially because of the large difference in suspended-sediment yield between the sties. The average annual yield of suspended sediment for T-2 and T-4 was 774,000 and 497,000 lb/mi^2, respectively. Dissolved P is defined as any P that passes through a 0.45-micron filter during sample filtration; thus, suspended P is attached to particles with a diameter greater than 0.45 microns. Suspended sediment, for the methodology used by the USGS sediment laboratories, is defined as any particle with a diameter size equal to or greater than 0.5 microns (Guy, 1969).

Table 23. Annual yields (in pounds per square mile) by water year for nutrients and suspended sediment, total flow, and the percentage of base flow at the four continuous surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.

[N, nitrogen; P, phosphorus; DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen; %Base, percent base flow; Mft^3, millions of cubic feet]

Constituent		Water year								Annual mean
		1994	1995	1996	1997	1998	1999	2000	[1]2001	
					C-1					
Dissolved- nitrate N	Yield	31,600	22,100	36,100	60,900	19,600	13,100	20,400	15,700	28,300
	%Base	88	92	88	94	90	87	90	92	91
Dissolved-nitrite N	Yield	146	82.6	172	103	87.0	63.1	96.9	64.4	105
	%Base	51	63	55	53	67	51	60	58	52
Dissolved- ammonia N	Yield	790	166	638	179	205	265	312	156	350
	%Base	16	42	20	47	33	15	23	31	23
(DKN)-N	Yield	2,520	923	2,510	1,410	996	1,060	1,390	725	1,490
	%Base	23	44	36	20	48	26	41	40	33
(TKN)-N	Yield	5,180	1,360	3,850	2,710	1,530	2,430	3,200	1,950	2,870
	%Base	15	30	25	18	34	12	18	21	20
Dissolved P	Yield	826	246	725	469	167	524	331	187	448
	%Base	15	24	19	11	29	5.3	10	23	16
Total P	Yield	2,270	530	1,890	742	442	1,410	1,970	872	1,310
	%Base	6.4	17	10	14	18	3.2	5.4	8.2	8.2
Suspended sediment	Yield	1,850,000	808,000	1,460,000	782,000	293,000	466,000	487,000	357,000	839,000
	%Base	14	11	9.3	14	10	4.0	4.8	6.4	11
Total flow, Mft^3	Flow	130	73.2	132	97.2	72.8	55.9	82.0	60.4	88.8
	%Base	67	79	69	71	77	62	76	80	72

Table 23. Annual yields (in pounds per square mile) by water year for nutrients and suspended sediment, total flow, and the percentage of base flow at the four continuous surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

[N, nitrogen; P, phosphorus; DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen; %Base, percent base flow; Mft^3, millions of cubic feet]

Constituent		Water year								Annual mean
		1994	1995	1996	1997	1998	1999	2000	[1]2001	
					T-1					
Dissolved- nitrate N	Yield	32,300	16,500	34,100	27,800	15,400	7,890	15,600	10,800	20,700
	%Base	91	91	90	90	95	86	94	92	91
Dissolved- nitrite N	Yield	120	96.6	185	170	611	362	352	453	303
	%Base	56	72	71	71	12	6.4	12	5.6	23
Dissolved- ammonia N	Yield	717	169	670	321	424	179	311	218	392
	%Base	22	29	19	28	73	16	63	49	35
(DKN)-N	Yield	1,890	771	2,100	1,230	795	674	801	561	1,140
	%Base	28	42	33	40	64	33	50	40	38
(TKN)-N	Yield	4,660	1,410	3,740	2,050	6,750	3,470	5,640	4,960	4,220
	%Base	13	29	22	28	90	62	76	70	56
Dissolved P	Yield	834	247	623	494	143	243	241	143	383
	%Base	17	21	15	5.2	30	20	44	14	18
Total P	Yield	2,550	599	1,760	933	400	783	774	1,050	1,140
	%Base	11	16	6.9	6.6	18	8.4	15	4.2	9.6
Suspended sediment	Yield	2,240,000	685,000	1,450,000	347,000	108,000	194,000	230,000	341,000	722,000
	%Base	5.2	6.0	4.7	23	32	4.8	4.6	2.3	6.6
Total flow, Mft^3	Flow	79.3	36.4	82.0	63.1	42.4	28.3	43.1	32.8	51.5
	%Base	65	77	70	71	81	62	80	75	72

Table 23. Annual yields (in pounds per square mile) by water year for nutrients and suspended sediment, total flow, and the percentage of base flow at the four continuous surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

[N, nitrogen; P, phosphorus; DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen; %Base, percent base flow; Mft³, millions of cubic feet]

Constituent		Water year								Annual mean
		1994	1995	1996	1997	1998	1999	2000	[1]2001	
					T-2					
Dissolved- nitrate N	Yield	29,900	16,300	25,200	25,800	18,700	10,000	13,300	11,200	19,600
	%Base	87	78	82	92	88	78	80	86	85
Dissolved- nitrite N	Yield	192	86.0	120	101	69.6	42.7	94.5	61.0	95.6
	%Base	53	49	53	61	52	42	37	41	50
Dissolved- ammonia N	Yield	516	118	465	266	140	120	306	112	256
	%Base	11	30	12	25	30	28	13	21	17
(DKN)-N	Yield	4,830	802	2,540	4,540	1,140	961	1,710	916	2,180
	%Base	66	46	35	83	46	35	23	27	58
(TKN)-N	Yield	10,100	1,220	3,550	5,600	1,470	2,100	4,880	3,290	3,980
	%Base	70	33	28	77	36	17	14	8.6	45
Dissolved P	Yield	691	1,020	827	575	237	296	422	281	567
	%Base	8.8	2.6	5.0	7.7	24	13	17	15	8.8
Total P	Yield	9,280	7,850	2,630	1,330	485	1,020	1,760	1,520	3,380
	%Base	.66	.51	3.4	7.9	21	10	9.3	3.2	2.8
Suspended sediment	Yield	1,160,000	928,000	1,410,000	475,000	126,000	446,000	675,000	712,000	774,000
	%Base	11	2.9	2.3	9.8	17	1.9	3.2	1.8	5.1
Total flow, Mft³	Flow	21.2	14.7	21.3	17.7	14.9	9.40	13.5	9.44	15.3
	%Base	53	40	49	66	65	51	52	62	55

Table 23. Annual yields (in pounds per square mile) by water year for nutrients and suspended sediment, total flow, and the percentage of base flow at the four continuous surface-water sites in the Big Spring Run Basin, Lancaster County, Pa.—Continued

[N, nitrogen; P, phosphorus; DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen; %Base, percent base flow; Mft³, millions of cubic feet]

Constituent		Water year								Annual mean
		1994	1995	1996	1997	1998	1999	2000	[1]2001	
T-4										
Dissolved- nitrate N	Yield		1,970	18,700	18,900	12,300	8,120	9,210	9,970	12,800
	%Base		54	82	89	87	81	81	84	84
Dissolved- nitrite N	Yield		16.8	178	73.5	35.4	42.3	40.8	41.7	69.4
	%Base		21	21	61	44	63	20	45	36
Dissolved- ammonia N	Yield		63.0	463	127	95.8	84.1	226	191	203
	%Base		4.6	8.2	31	34	15	5.5	17	14
(DKN)-N	Yield		313	2,800	1,520	1,050	2,510	1,260	879	1,680
	%Base		7.2	17	54	33	78	22	37	41
(TKN)-N	Yield		500	6,580	8,440	1,830	4,410	3,200	2,050	4,380
	%Base		5.5	39	77	20	57	8.3	23	47
Dissolved P	Yield		71.9	783	2,170	148	2,850	210	154	1,030
	%Base		4.6	25	84	28	94	17	35	75
Total P	Yield		163	1,540	2,700	296	4,920	1,110	745	1,860
	%Base		1.7	17	77	17	87	4.2	10	59
Suspended sediment	Yield		198,000	1,060,000	351,000	138,000	450,000	535,000	335,000	497,000
	%Base		0.20	2.3	12	11	4.3	2.0	4.4	4.2
Total flow, Mft³	Flow		2.08	19.1	14.0	9.83	7.29	9.79	7.38	10.7
	%Base		22	39	59	55	43	42	58	48

[1] Yields for water year 2001 were estimated from October 2000 through June 2001.

The percentage of base flow that contributed to overall yields for nitrate N, DKN, total P, dissolved P, and suspended sediment were very similar between T-1 and C-1 (table 23). The average percentage of total flow that was base flow for T-1 and C-1 was 72 percent. Ninety-one percent of the yield of nitrate N for both sites was transported during base-flow periods. Conversely, most of the P and suspended-sediment yields was transported during stormflow. Only about 9 and 17 percent of the yields of total and dissolved P for both sites was transported during base flow. Base flow contributed only 6.6 to 11 percent of the suspended-sediment yields for T-1 and C-1.

The percentage of base flow that contributed to overall yields for N and suspended-sediment yields were somewhat similar between T-2 and T-4; however, the base-flow component for P was different. For T-2 and T-4, base flow made up less of the total amount of flow than for T-1 and C-1. This was partially caused by the fact that during dry periods in the summer, flow was either zero, or close to zero at T-2 and T-4; during these same periods at T-1 and C-1, flow never approached zero. Similar to the outlet sites, the yield of nitrate N for T-2 and T-4 mostly was transported during base flow (average about 85 percent), and most of the suspended-sediment yield (about 95 percent) was transported during stormflow. The most dramatic difference between T-2 and T-4 for base-flow contributions to constituent yields was for P. The yield of total P for T-4 had a much higher contribution from base flow (59 percent) than the yield for T-2 (2.8 percent). One reason for the large difference could be the installation of a sediment-retention pond immediately upgradient of T-4. Construction of the retention pond began in winter 1995 and was completed by March 1996. The outflow from the retention pond fed into the main channel about 20-30 ft upstream of the weir at T-4. The discharge of fine sediments and possibly dissolved P being released from the sediments in the retention pond could have supplied P at T-4 even after stormflow had subsided. Another possible cause of the difference in base-flow contribution of P between T-2 and T-4 could have been because of agricultural land use between the T-4 and T-2 gages. The drainage area (0.04 mi^2) downgradient of T-4 was strictly agricultural. During storm events, the transport of P from subsurface zones and overland flow in this 0.04-mi^2 area could have supplied T-2 with additional P that reached the stream channel downgradient of the T-4 gage.

Annual yields for the basin were somewhat similar to values reported by Ott and others (1991) for the Conestoga-River Basin, which is an agricultural drainage basin (reported to have land use of 62.7 percent agriculture in1989) in Lancaster County. In fact, the streams from the study area eventually feed into the Conestoga River. Average annual yields of total N and total P for the Conestoga River from 1985 to 1989 were 22,275 and 1,515, lb/mi^2, respectively (Ott and others, 1991). The average annual yield of total N for the four sites for this study ranged from 17,180 (T-4) to 31,170 lb/mi^2 (C-1). The average annual yield of total P for the four sites for this study ranged from 1,140 (T-1) to 3,380 lb/mi^2 (T-2).

Summary

Discharge of water from the study basin reflected the difference in precipitation from the pre- to post-treatment period. Mean discharge for surface-water sites during the post-treatment period was about 60 percent of the mean discharge for the pre-treatment period. In general, this caused a decrease in the mean discharge sampled for low flow and stormflow during the post-treatment period. The overall mean discharge for the four continuous surface-water sites ranged from 0.36 ft^3/s for T-4 to 2.87 ft^3/s for C-1.

Low-flow data showed that 96 percent of the total-N concentration for all low-flow samples was in the form of nitrate N, and 99 percent of the total N was dissolved. The median concentration of nitrate N for all low-flow samples was 10.4 mg/L. T-1 and T-2 showed 22-26 percent reductions in concentrations of nitrate N from the pre- to post-treatment period; whereas C-1 showed a 10-percent reduction.

Instantaneous N yields based on low-flow samples showed significant decreases from the pre- to post-treatment period at mainstem sites. Decreases in instantaneous yields of nitrate N from the pre- to post-treatment period at the mainstem sites (C-1, T-1, and T-3) ranged from 38 to 49 percent. Decreases in instantaneous yields of ammonia N from the pre- to post-treatment period at the mainstem sites ranged from 33 to 60 percent. Tributary sites (T-2 and T-4) did not show significant changes in instantaneous nitrate and ammonia yields from the pre- to post-treatment period.

P concentrations for low-flow data showed significant increases from the pre- to post-treatment period at T-1 and T-2. The concentration of total P at T-2 increased by 150 percent from the pre- to post-treatment period. Median concentrations of total P for T-1 and T-2 increased by 0.03 and 0.06 mg/L, respectively. Most of the increase in concentration of total P was caused by increased concentrations of dissolved P.

Instantaneous P yields based on low-flow samples indicated significant differences between treated and control sites. The yield of dissolved P for T-1 and T-2 showed significant increases from the pre- to post-treatment period of 63 and 250 percent, respectively; C-1 showed no significant change. The instantaneous yield of suspended P for T-2 also showed a 319-percent increase from the pre- to post-treatment period.

Post-treatment changes in concentrations for low-flow samples at treated sites relative to control sites showed significant reductions in N, significant increases in P, and varied results for fecal-streptococcus colonies. Total and dissolved N and nitrate N showed 22-23 percent relative reductions at T-1; for T-2, reductions for the same constituents were about 12 percent (table 24). Average relative reductions in concentrations of ammonia N during the post-treatment period at T-1 and T-2 ranged from 10 (T-2) to 34 percent (T-1). Average percent increases during the post-treatment in total P at treated relative to control sites ranged from 50 (T-1) to 160 percent (T-2); average increases in dissolved P ranged from 61 (T-1) to 120 percent (T-2). Fecal-streptococcus data for T-1 showed an

average relative reduction of 60 percent during the post-treatment period; T-2 showed a 240 percent relative increase.

Post-treatment changes in N and P yields for low-flow samples at treated sites relative to control sites showed varied results. T-1 showed relative reductions in yields for all N species during the post-treatment period. Average relative reductions in instantaneous yields of total N and nitrate N were 17 percent for T-1 during the post-treatment period (table 24). Average relative increases in instantaneous yields of total N and nitrate N were 9 to 11 percent for T-2 during the post-treatment period. Instantaneous yields of ammonia N showed an average relative reduction of 35 percent at T-1 and an increase of 15 percent at T-2. The instantaneous yield of total P showed average relative percent increases during the post-treatment period at T-1 and T-2 of 51 and 220 percent, respectively.

Table 24. Summary of percent changes in water-quality constituents at surface-water sites (T-1 and T-2) in the Big Spring Run Basin, Lancaster County, Pa., based on analysis of covariance results for paired and upstream-downstream comparisons.

[N, nitrogen; TKN, total ammonia plus organic nitrogen; DKN, dissolved ammonia plus organic nitrogen; P, phosphorus; NS, no significant relation; NA, not available]

Constituent	Low flow (percent change)		Stormflow (percent change)	
	T-1[1]	T-2[2]	T-1[3]	T-2
Total-N concentration	-23	-12	-6.2	-2.0
Dissolved-N concentration	-23	-12	3.5	-4.6
Dissolved-nitrate concentration	-22	-11	6.3	-6.3
Dissolved-nitrite concentration	-40	-12	-18	5.5
TKN concentration	-13	40	-6.5	.50
DKN concentration	-9.6	4.8	NS	-2.0
Dissolved-ammonia concentration	-34	-9.8	-23	-22
Total-P concentration	50	160	-13	19
Dissolved-P concentration	61	120	8.1	37
Suspended-P concentration	30	140	-20	11
Suspended-sediment concentration	-46	-24	-31	-46
Fecal-streptococcus counts	-60	240	NA	NA
Dissolved-oxygen concentration	-6.9	-12	NA	NA
pH	-2.2	-1.0	NA	NA
Specific conductance	-3.3	-.84	NA	NA
Water temperature	-5.7	-4.8	NA	NA
Total-N yield	-17	8.7	-19	28
Dissolved-N yield	-17	7.7	-15	30
Dissolved-nitrate yield	-17	11	-21	33
Dissolved-nitrite yield	-44	1.1	-22	27
TKN yield	-25	74	-27	28
DKN yield	-21	31	-18	28
Dissolved-ammonia yield	-35	15	-37	8.9
Total-P yield	51	220	-22	46
Dissolved-P yield	43	170	14	88
Suspended-P yield	42	180	-36	14
Suspended-sediment yield	-46	4.4	-36	-46

[1] The percentage change at T-1 during the post-treatment period for low-flow samples is based on comparisons to C-1 and T-3. Average percentage changes in yield are based on weighted and unweighted analysis of covariance results.

[2] The percentage change at T-2 during the post-treatment period for low-flow and stormflow samples is based on comparisons to C-1 and T-4. Average percentage changes in yield are based on weighted and unweighted analysis of covariance results.

[3] The percentage change at T-1 during the post-treatment period for stormflow samples is based on comparisons to C-1. Average percentage changes in yield are based on weighted and unweighted analysis of covariance results.

Stormflow data, as expected, showed much different concentrations for nutrients and suspended sediment than for low-flow samples. The median concentrations of total N for all stormflow and low-flow samples were 5.24 and 10.87 mg/L, respectively. The median concentration of nitrate N for all stormflow samples was 2.88 mg/L, which was 7.52 mg/L lower than the median for low-flow samples. The median concentrations of total P for all stormflow and low-flow samples were 0.57 and 0.04 mg/L, respectively. The median suspended-sediment concentration for all stormflow samples (227 mg/L) was much greater than the median for low-flow samples (14 mg/L).

N and P concentrations for stormflow from the pre- to post-treatment period showed some differences. N-species concentration data for T-1 and T-2 did not show any significant differences from the pre- to post-treatment period; data for C-1 showed significant decreases in concentrations of nitrate N. Concentrations of total P did not change significantly from the pre- to post-treatment period at any sites; however, concentrations of dissolved P showed significant decreases at the untreated sites (C-1 and T-4) of 37-50 percent; T-1 showed a significant percentage decrease of 20 percent and T-2 showed no significant change.

Suspended-sediment concentrations for stormflow samples showed significant decreases at all sites from the pre- to post-treatment period. Largest percentage decreases were at T-1 and T-2 (56-69 percent) rather than C-1 and T-4 (40-54 percent). Percentage reductions for all sites were partially attributable to a decrease in the mean discharge for sampled storms from the pre- to post-treatment period.

Stormflow yields for N, P, and suspended sediment showed significant reductions at all sites from the pre- to post-treatment period. Decreases in median yields of total N from the pre- to post-treatment period for stormflow samples ranged from 52 percent for T-2 to 75 percent for T-4. The decrease in the median yield of total P from pre- to the post-treatment period was 85 percent for T-4; the range for the other sites was 46 (C-1) to 55 percent (T-2). The decrease in the median suspended-sediment yield from pre- to the post-treatment period ranged from 73 (C-1) to 86 percent (T-1 and T-4). Decreased yields for all the sites were caused by decreased mean discharge during the post-relative to pre-treatment period.

Post-treatment changes in concentrations for stormflow samples at treated sites relative to control sites showed varied results for nutrients and a relative decrease in suspended sediment. Concentrations of total N and dissolved ammonia were the only N species for stormflow samples that showed relative reductions during the post-treatment period for T-1 and T-2; average relative reductions in total N and dissolved ammonia were 2 to 6 percent and 22 to 23 percent, respectively (table 24). Concentrations of total P showed an average relative decrease at T-1 (13 percent) and a relative increase of 19 percent for T-2 during the post-treatment period. Both sites showed relative increases (8 to 37 percent) in concentrations of dissolved P during the post-treatment period. Average percent reductions in suspended-sediment concentrations were greater at T-2 (46 percent) than T-1 (36 percent).

Post-treatment changes in N and P yields for stormflow samples at treated sites relative to control sites showed different results by site; however, both treated sites showed significant reductions in suspended-sediment yield. The average reduction in yield of total N for T-1 was 19 percent during the post-treatment period; the average relative increase in yield of total N for T-2 was 28 percent. Similarly, the average reduction in yield of total P for T-1 was 22 percent during the post-treatment period, and the average relative increase in yield of total P for T-2 was 46 percent. The average percent relative reductions in the suspended-sediment yield during the post-treatment period at T-1 and T-2 were 36 and 46 percent, respectively.

Overall changes in constituent yields were determined by summing low-flow and stormflow ANCOVA results. The percentage changes determined from the ANCOVA procedure for low-flow and stormflow data were combined with the annual yield data presented in table 23. The percentage of the constituent yield attributed to low-flow and stormflow periods as presented in table 23 was applied to the ANCOVA results to determine an overall change for each constituent.

Overall changes to constituent yields at T-1 relative to C-1 showed that all constituents except dissolved P showed an overall reduction during the post-treatment period (table 25). The greatest overall reduction at T-1 occurred for suspended sediment (37 percent) and dissolved ammonia (36 percent). T-1 showed an 18 percent overall reduction in nitrate-N, and, when combining all the changes in N species for T-1 during the post-treatment period, the overall total-N reduction for T-1 was 19 percent. The total-P yield for T-1 showed an

Table 25. Overall water-quality changes in constituent yields for the treated sites (T-1 and T-2) of the Big Spring Run Basin, Lancaster County, Pa., for the post-treatment period based on analysis of covariance (ANCOVA) results and the separation of constituent yields into base-flow and stormflow components.

[DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen]

Constituent	T-1 (percent change)	T-2 (percent change)
Dissolved nitrate	-18	+15
Dissolved nitrite	-28	+15
Dissolved ammonia	-36	+10
DKN	-20	+30
TKN	-26	+43
Dissolved phosphorus	+19	+94
Total phosphorus	-14	+51
Suspended sediment	-37	-44

overall reduction of 14 percent even though dissolved-P yield increased by 19 percent.

Conversely, data for T-2 showed increases in all constituent yields during the post-treatment period (relative to T-4 and C-1) except for suspended sediment. The overall post-treatment changes at T-2 showed that the range in the increase of N species was 10 percent for ammonia N to 43 percent for TKN as N, with the change in all N species for T-2 giving a total-N yield increase of 21 percent. The dissolved-P yield showed the largest increase (94 percent) of all constituents listed in table 25, and this large increase for dissolved P was the primary cause for the 51 percent increase in total-P yield for T-2 during the post-treatment period. Suspended-sediment yield showed a 44 percent reduction at T-2 during the post-treatment period, which is similar to the reduction evident for T-1.

The summed results for T-1 and T-2 can basically be viewed as the end result of streambank fencing at the end of the study period. The results indicate that the effects of streambank fencing on surface-water quality had a cumulative impact as one moved downstream within the study basin. These data suggest that streambank fencing may not indicate an improvement in water quality as the stream drainage area decreases because of microscale processes such as what was seen at T-2 for changes in P.

Benthic Macroinvertebrates and Habitat

The streams in the study area are first- and second-order streams that can benefit from a good riparian buffer. For example, riparian buffers can benefit these streams by reducing diurnal stream temperature fluctuations, providing habitat for fauna, and trapping sediment in overland flow. First- and second-order streams are numerous, sensitive to sedimentation and other impacts, and discharge into larger streams and rivers. One of the best ways to protect larger streams and rivers is to protect the smaller streams that discharge into them (University of Georgia, 2003). This section will discuss the habitat, water quality, and benthic-macroinvertebrate community at the treated and untreated sites over time by use of box plots and environmental variables that influence the macroinvertebrate communities among the sampling sites as determined by CCA.

Habitat

The first change in habitat observed at the treated sites after the fence was installed was the amount of grassy vegetation on the banks. One stream reach at site T2-3 became overgrown with vegetation, thus blocking the stream from view. The herbaceous plants that grew along the stream channel after fence installation are shown in figure 4.

The bottom substrate available cover (BSAC) at all sites ranged from fair to excellent during the pre- and post-treatment periods (fig. 11). At the outlets, C-1 and T-1 showed improvement in BSAC from the pre- to post-treatment period for May samples. T-1 showed little change (mean value for pre-treatment was 14 and mean for post-treatment was 13) in BSAC for September samples, and C-1 showed a mean decrease of four units. For the upstream sites, T2-3 showed a slight improvement in BSAC from the pre- to post-treatment period (mean increased from 11 to 13); T1-3 and C1-2 basically showed no change. Overall, the mean BSAC scores for the control sites showed no change from the pre- to post-treatment period; the treatment sites showed an increase in the mean BSAC scores of one unit. Scores for BSAC increase as the percentage of rubble, gravel, submerged logs, undercut banks, or other stable habitat increases (Plafkin and others,

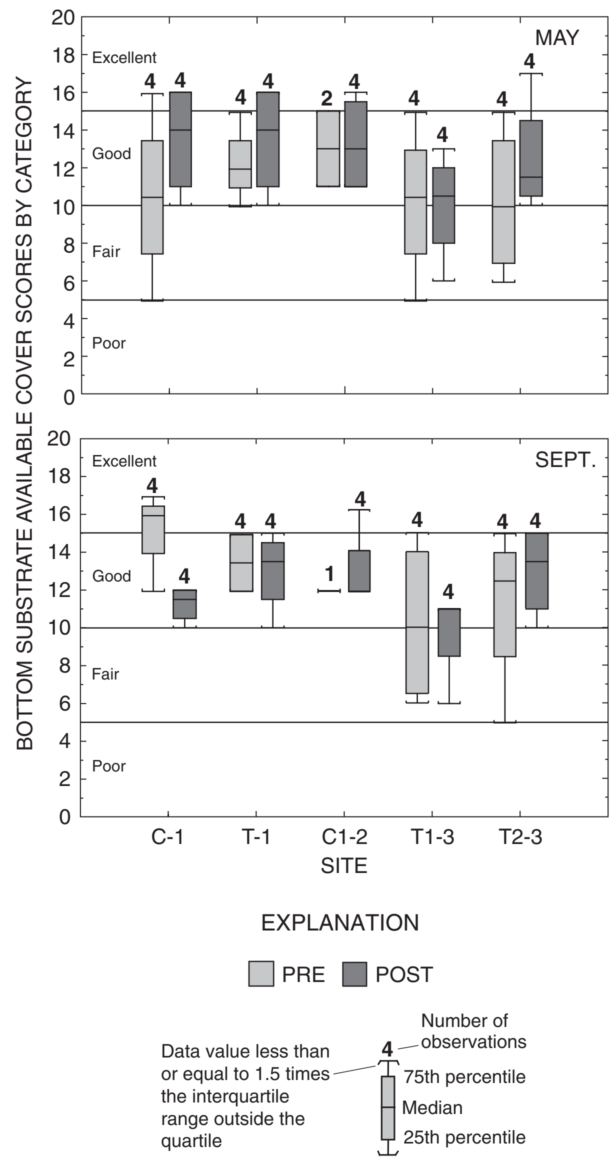


Figure 11. Distribution of bottom substrate available cover using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

1989). Thus, it appears that streambank fencing had a small positive influence on BSAC, probably because of less fine sediment in the creek and more exposure of the gravel and rock stream bottom.

The embeddedness scores for all sites generally ranged from fair to good (fig. 12). At the outlets, the changes in embeddedness were somewhat similar to changes in BSAC during the post-treatment period. That is, for BSAC and embeddedness, T-1 showed an increase in scores relative to C-1 for post-treatment September samples, but there was virtually no change between the outlets for post-treatment May samples (figs. 11 and 12). Similar trends in BSAC and embeddedness were somewhat expected given they are a measure of similar physical characteristics. At the upstream sites, T2-3 and C1-2 showed an improvement in embeddedness scores from the pre- to post-treatment period (mean scores increased from 10 to 12 and 12 to 13, respectively), and T1-3 had lower scores (mean scores went from 10 to 8) (fig. 12). Thus, T2-3 showed improvement in BSAC and embeddedness during the post-treatment period; T1-3 showed no improvement, and scores indicated some degradation. The major difference between T1-3 and T2-3 was the fact that T1-3 was below a pond and T2-3 was immediately upstream of the pond. Water flowed from T2-3 into the pond and the outlet of the pond was just upstream of T1-3. It is possible that BSAC and embeddedness did not show improvement at T1-3 during the post-treatment period because of sediment discharge from the pond. This would tend to decrease scores for BSAC and embeddedness. An increase in the embeddedness score is good because it shows that embeddedness decreased, which translates to more rock and gravel niches for invertebrates to inhabit (Plafkin and others, 1989).

The velocity to depth ratio (VDR) scores showed markedly different trends from the pre- to post-treatment periods (fig. 13). Mean VDR scores for May samples increased from the pre- to post-treatment period at each site; whereas for September samples, each site showed a decrease in mean VDR scores. VDR is a qualitative measure of the ability of the stream to provide a stable aquatic environment (Plafkin and others, 1989). VDR is qualitatively assessed at the time of sample collection and values are somewhat dependent on the amount of flow. That is, if VDR is assessed during relatively low conditions, pools might not be evident and scores would subsequently be lower. Streams with deep and shallow pools and riffle areas would indicate a better VDR than a stream with only deep, stagnant pools or only riffles. Stream discharge at the time of benthic-macroinvertebrate sample collection also showed differences between the pre- and post-treatment periods (fig. 14). Mean stream discharge for May samples increased at each site (mean increases ranged from 21 percent for C1-2 to 200 percent for T2-3) from the pre- to post-treatment period; conversely, mean stream discharge for September samples decreased at each site (mean decreases ranged from 15 percent for T-1 to 63 percent for T1-3 and T2-3) from the pre- to post-treatment period. Overall, the mean stream discharge measured for all sites except C1-2 increased from the pre- to post-treatment period by 16 (C-1) to 109 percent (T2-3). This increase in measured stream discharge for the benthic-macroinvertebrate sampling was unexpected because, during the post-treatment period, stream discharge was 56-63 percent of the flow measured during the pre-treatment period at the four continuous stream discharge sites in the study basin (table 18). Nevertheless, the increased stream discharge measured during the post-treatment period for May samples produced VDR scores that indicated better habitat; the converse was true for the September samples. Therefore, it appears that changes in VDR scores were more

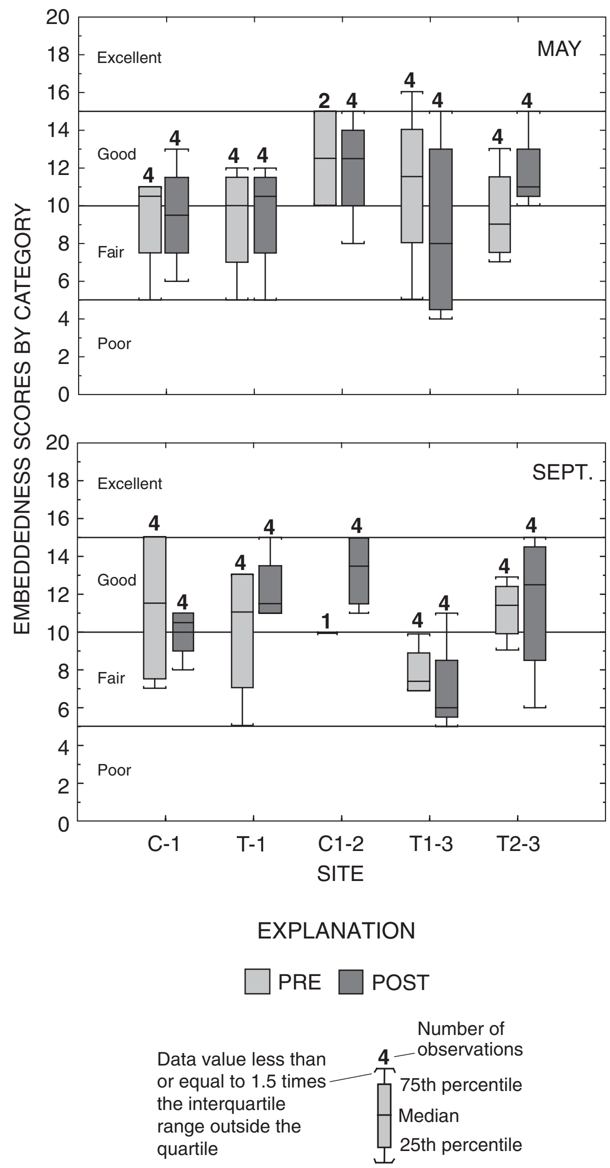


Figure 12. Distribution of embeddedness using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

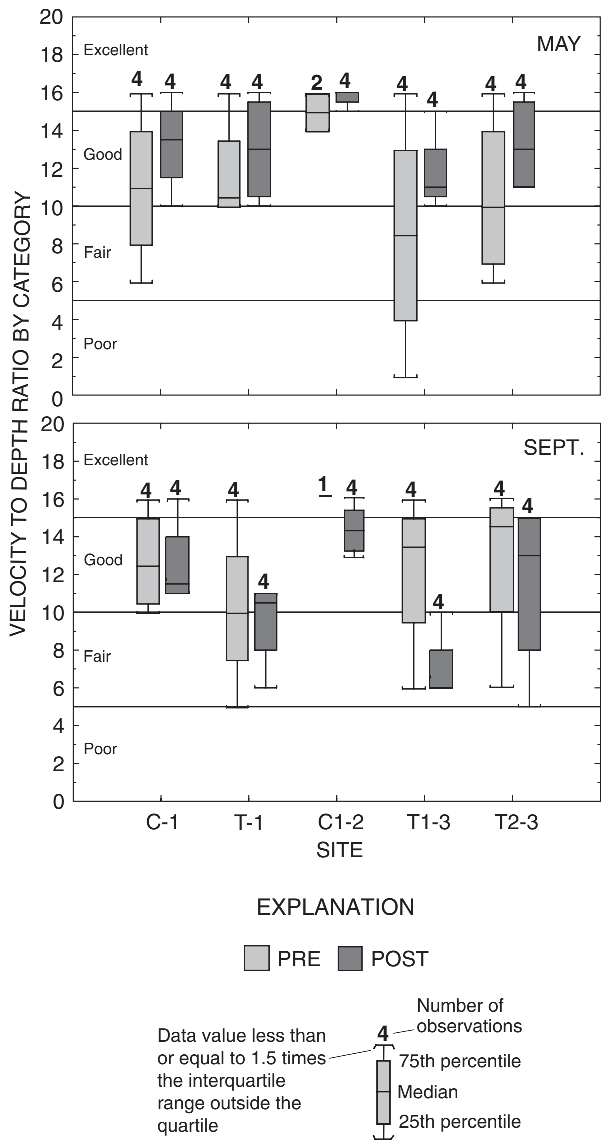


Figure 13. Distribution of velocity to depth ratio using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

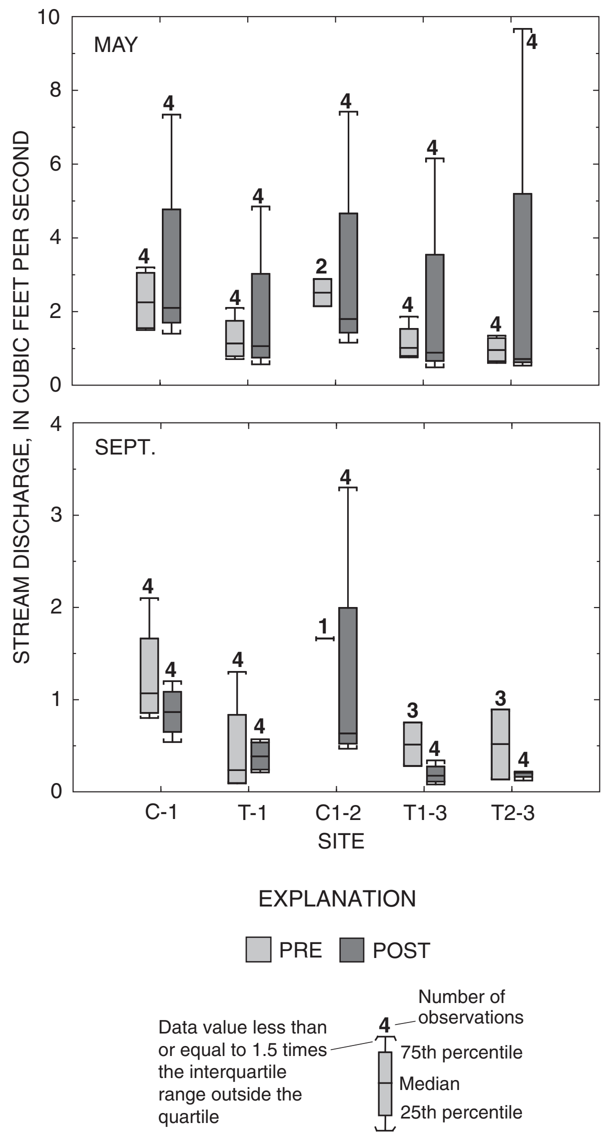


Figure 14. Distribution of stream discharge for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

related to differences in stream discharge as opposed to changes caused by fence installation.

Channel-alteration scores showed different trends from pre- to post-treatment periods for upstream and outlet sites (fig. 15). The mean channel-alteration score for T-1 was basically the same (mean=11) for all samples collected during the pre- and post-treatment period; whereas for C-1, the mean score increased from 9 to 13 from the pre- to post-treatment period. Upstream sites (T1-3 and T2-3) in the treatment basin during the post-treatment period showed improvement in channel-alteration scores relative to the upstream site in the control basin (C1-2). The overall mean score for channel alteration for the upstream sites in the treatment basin increased from 9 to 12 from the pre- to post-treatment period; the mean for C1-2 decreased from 13 to 12. Channel-alteration scores at the control sites during the post-treatment period were rated as excellent for each sample event; however, C1-2 was also rated as excellent during the pre-treatment period for all sample events. Thus, the upstream sites in the treatment basin showed improvement relative to the upstream site in the control basin, but the upstream site was already rated as excellent, so a substantial increase in channel-alteration scores was not possible. Overall, it was difficult to determine the effect of fence installation on channel alteration given that the upstream site in the control basin (C1-2) was rated as excellent throughout the study and C-1 improved relative to T-1 during the post-treatment period. Scores for channel alteration are influenced by stream velocities. Channel alteration is rated as excellent if there is no indication of point-bar development in the channel and the sinuosity of the channel is maintained; poor channel alteration would have heavy deposits of fine materials, bar development, and increased channelization (Plafkin and others, 1989).

Scores for bottom scouring and deposition (BSD) were also difficult to interpret given relations between the treated and control sites. The outlet site (T-1) of the treatment basin during the post-treatment period showed improvement relative to C-1 for May and September sample events (fig. 16). The upstream sites in both basins showed an improvement in BSD scores from the pre- to post-treatment period. The overall mean score for T1-3 and T2-3 increased from 9 to 11 from the pre- to post-treatment period. C1-2 showed a similar increase; however, as with all data for C1-2, the pre-treatment data were limited. BSD, like channel alteration, is dependent

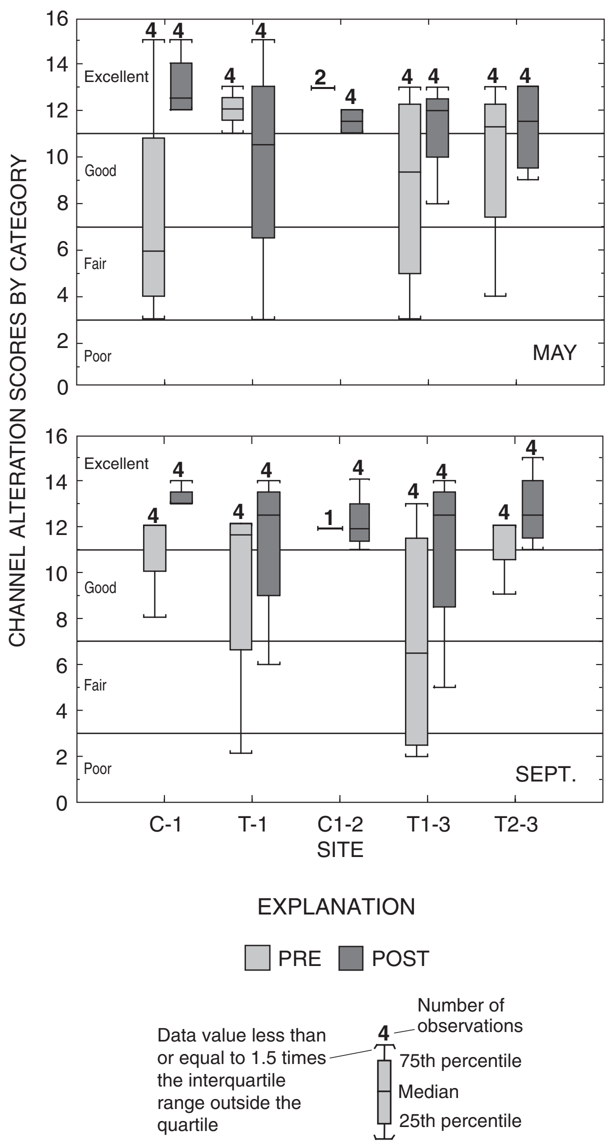


Figure 15. Distribution of channel alteration using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

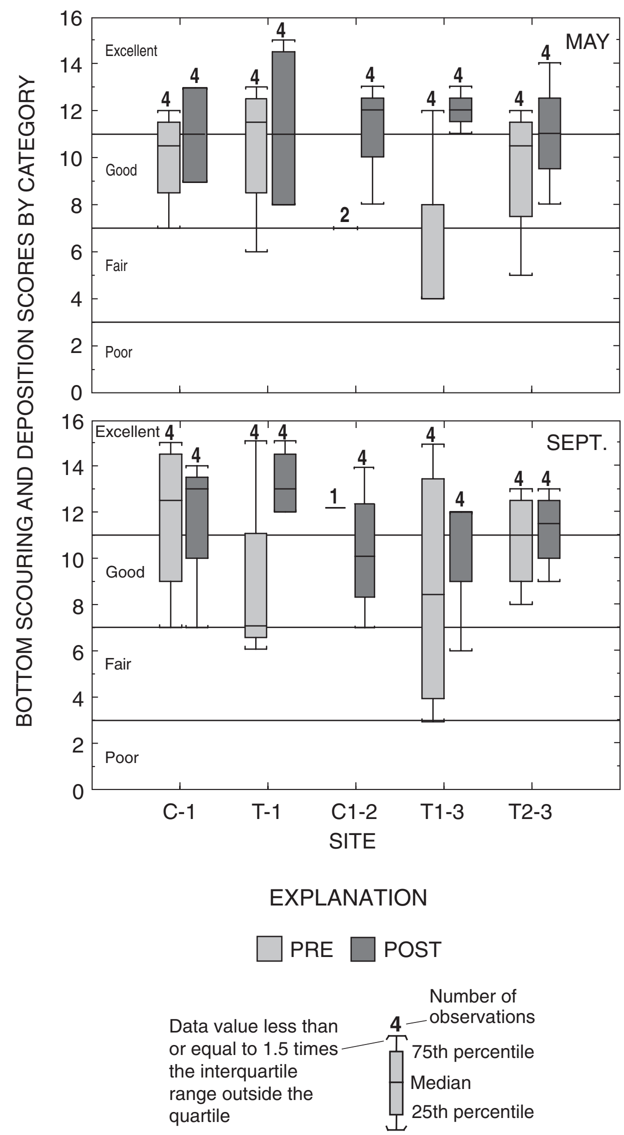


Figure 16. Distribution of bottom scouring and deposition using May and September site scores at benthic-macroinvertebrate sites for the pre- and post- treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

on velocity. BSD is rated by determining the percentage of the stream channel affected by scour and deposition, with higher percentages indicating an unstable habitat (Plafkin and others, 1989). However, BSD scores are inversely related to the percentage of bottom scour and deposition, so the higher the BSD score, the better the habitat for benthic macroinvertebrates (see appendix 1). Overall, it appeared that fence installation and subsequent establishment of a vegetative filter along the stream channel helped to reduce bottom scour and deposition at the outlet of the treatment basin. It was likely that fencing helped to reduce bottom scouring at upstream sites, but the limited amount of data for C1-2 reduced the robustness of this finding.

The pool/riffle, run/bend ratio (PRRBR) is a qualitative assessment of the ratio of pools to riffles and bends to straight stream segments. As the percentage of riffles and bends increases, habitat is improved (Plafkin and others, 1989). C-1 and C1-2 had lower mean PRRBR scores during the post-treatment relative to the pre-treatment period; however, pre-treatment data for C1-2 indicated it was rated as excellent for PRRBR scores (fig. 17). No treatment sites had any scores in the excellent PRRBR category throughout the study. Mean PRRBR scores for all the treated sites were higher during the post-treatment than the pre-treatment period for May samples, whereas for September, mean PRRBR scores were lower during the post-treatment than the pre-treatment period for all sites. Overall, the mean PRRBR scores for C-1 and C1-2 decreased by 2 and 4 units, respectively, from the pre- to post-treatment period, whereas for the treatment sites, overall mean scores did not change. This indicated that relative to PRRBR, the treated sites did show some improvement relative to control sites during the post-treatment period.

Bank stability is a qualitative measure based on whether the site shows evidence of bank erosion and bank slopes (appendix 1). At these sites, bank slopes greater than 30 percent were not evident. Banks with a slope greater than 30 percent are given lower bank-stability scores. No treatment site during the pre-treatment period was rated as excellent for this habitat characteristic, whereas during the post-treatment, T-1 and T2-3 showed at least some scores in the excellent category for bank stability (fig. 18). T-1 showed the most improvement of the five sites in bank-stability scores from the pre- to post-treatment period; overall mean scores increased from 6 to 8. Over the same period, the overall mean score for C-1 did not change. For the upstream sites, the low number of observations (3) during the pre-treatment period for C1-2 made it difficult for comparisons, but given the data available, overall mean scores indicated T2-3 improved relative to C1-2 from the pre- to post-treatment period. At T2-3, lush grassy vegetation established itself after fence installation, whereas for T1-3, increased vegetative growth after fence installation was not as noticeable. Overall, it appeared that fence installation and subsequent vegetative development along the banks, as expected, had a positive influence on bank-stability scores. As the banks stabilized, there was less bank erosion and subsequently less deposition of fine materials in the stream bottom.

Scores for bank vegetative stability (BVS) during the post-treatment period showed a similar response as bank-stability scores, which was expected given the similarities in these habitat characteristics. T-1 showed the most improvement in BVS scores of the five sites from the pre- to post-treatment period (fig. 19). There was virtually no change in BVS scores for either of the control sites. T2-3 showed an increase in overall mean BVS scores from the pre- to post-treatment period; T1-3 actually showed a slight decrease. This response generally followed the results for bank-stability scores. Overall, it appeared fence installation and subsequent

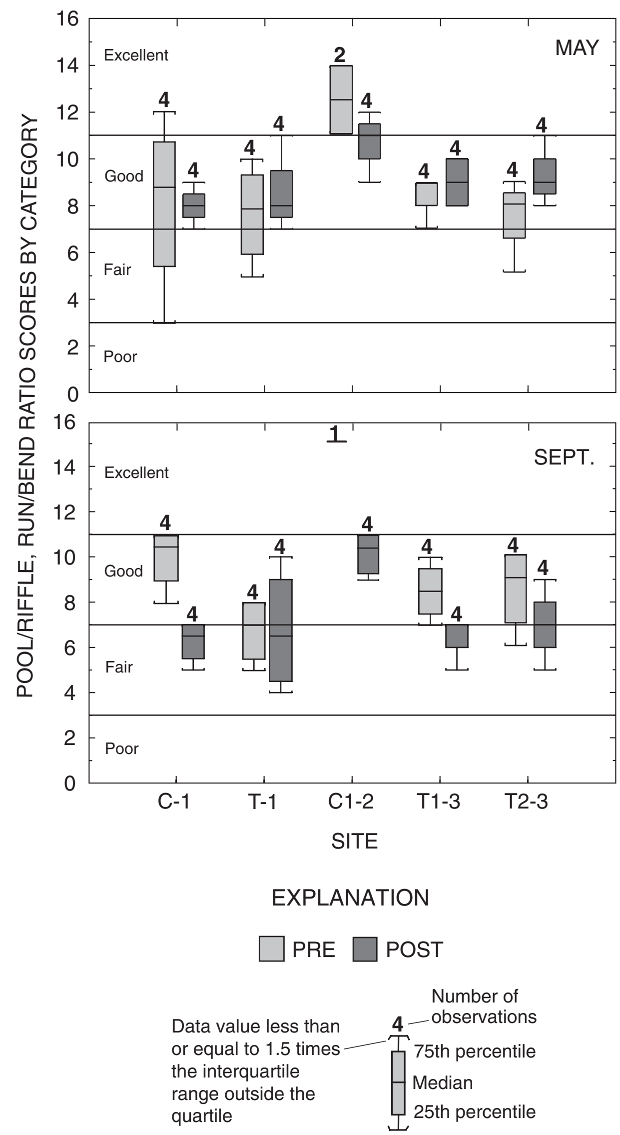


Figure 17. Distribution of pool/riffle, run/bend ratio using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

vegetative development along the banks, as expected, had a positive influence on BVS scores. BVS scores are based on the percentage of the streambank covered by vegetation or boulders or cobbles. The more vegetation, the more stable the banks and the less bank erosion (Plafkin and others, 1989).

The streamside-cover scores showed basically no change for any of the sites from the pre- to post-treatment period (fig. 20). All sites stayed within the fair category throughout the study. Streamside cover is based on vegetative type-shrubs score as the best cover, then trees, then grass and forbes. The lowest scores are for sites with no vegetative materials present (Plafkin and others, 1989). These sites were vegetated in grasses and other herbaceous vegetation, which were rated as fair for streamside-cover scores.

The total-habitat score is a sum of qualitative scores (see appendix 2) derived from the RBP III assessment of each site. The sum of the scores typically are not rated into categories (such as poor, fair, etc.); instead, the scores are compared to reference sites. No reference (undisturbed) sites were located for our five study sites because an undisturbed site would dictate that no agricultural activity was present in the reference basin, and no such site was identified in Lancaster County

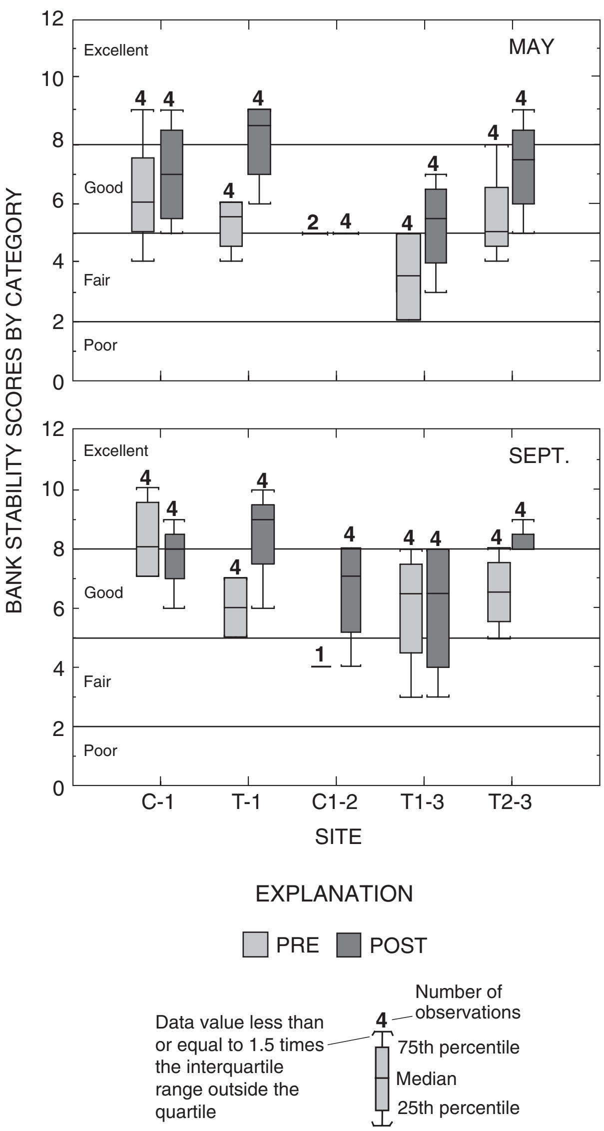


Figure 18. Distribution of bank stability using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

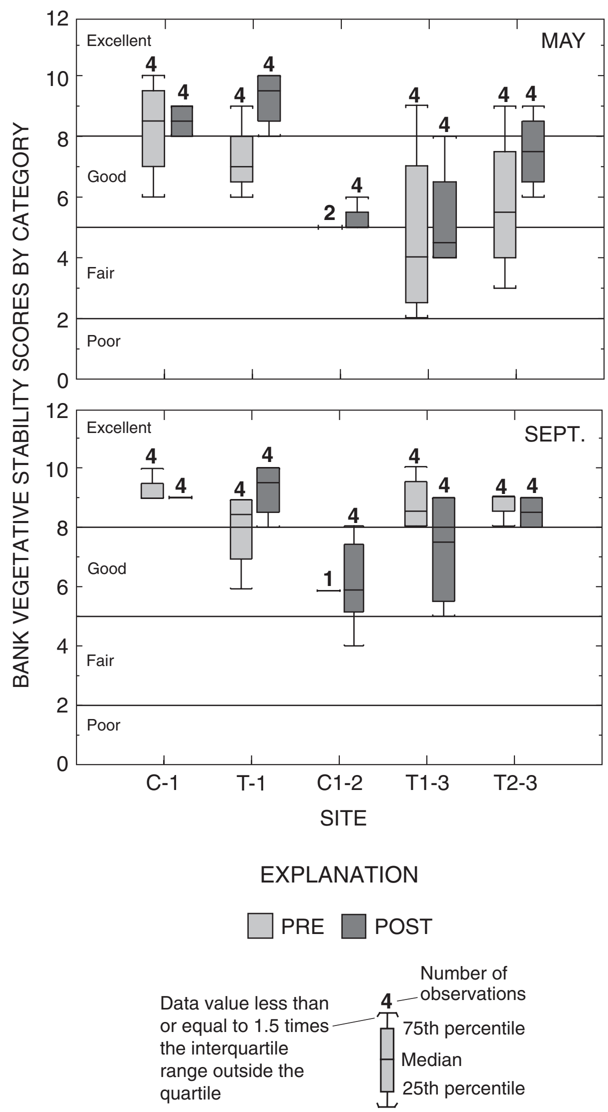


Figure 19. Distribution of bank vegetative stability using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

where karst conditions exist. Nevertheless, it was possible to see how fence installation affected total habitat scores relative to control sites. For the upstream sites, C1-2 basically showed no change in the overall mean total habitat score from the pre- to post-treatment period (fig. 21); T1-3 and T2-3 showed overall mean increases from the pre- to post-treatment period of 3 and 10 units, respectively. Total-habitat scores for T-1 showed an overall mean increase of 9 units from the pre- to post-treatment period, and the overall mean C-1 scores increased by 3 units. Overall, fence installation helped to improve habitat conditions at the treatment-basin sites. T-1 and T2-3 showed the most improvement; improvement caused by fencing at T1-3 may have been somewhat muted because of the presence of an outflowing pond directly upstream of the site. This may have acted as a source of sediment during the post-treatment period that could have negatively affected some of the habitat characteristics.

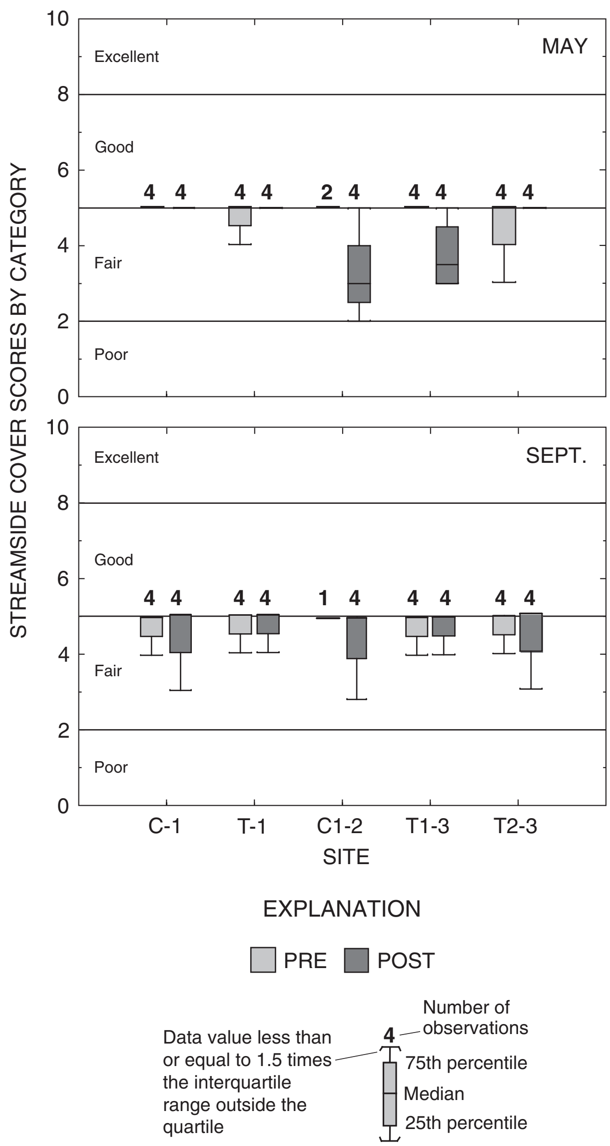


Figure 20. Distribution of streamside cover using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

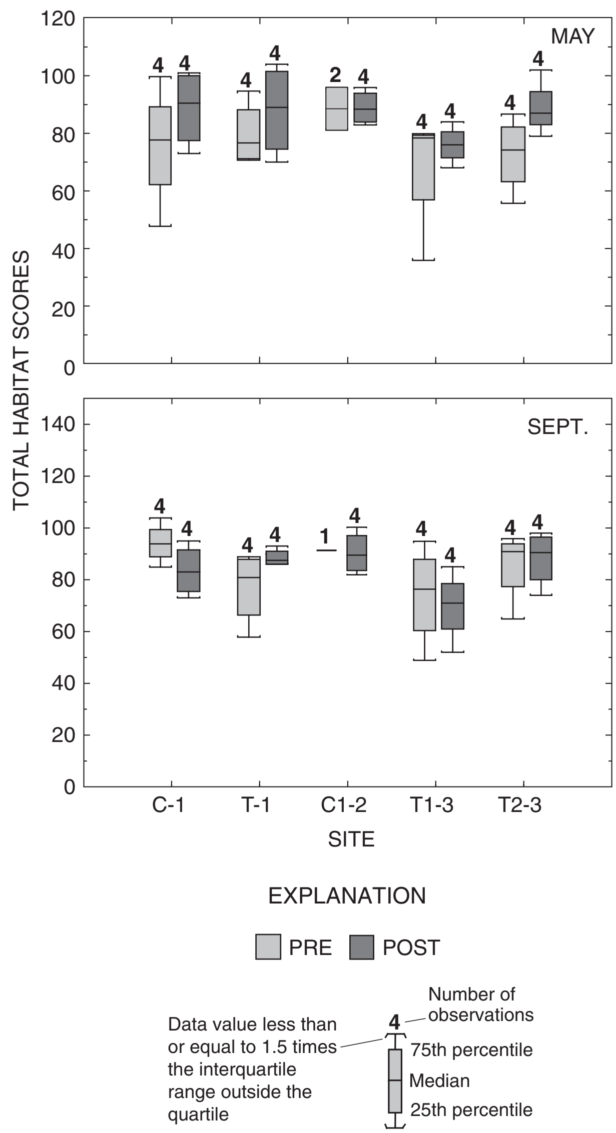


Figure 21. Distribution of total habitat using May and September site scores at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

Water Quality

The water-quality data collected during the benthic-macroinvertebrate sampling was a snapshot of conditions and was certainly not reflective of conditions during storm events. The total yield of nutrients and suspended sediment to the five benthic-macroinvertebrate sampling sites was very dependent on storm events, and this was especially true for water-quality constituents that have a suspended component, such as suspended sediment, total P, and TKN (see table 23 for constituent yields separated into low-flow and stormflow components for the four continuous surface-water sites). A discussion of the effects of fencing on water-quality constituents was given in a previous section. Here, the data collected during benthic-macroinvertebrate sampling events will be presented with limited discussion.

Water-quality samples collected indicated a decrease in concentrations of dissolved nitrate at all sites after the fencing was installed (fig. 22). This was similar to results for the five surface-water sites where low-flow samples were collected. Median concentrations of dissolved nitrate as N in low-flow samples decreased at C-1 and T-1 by 10 and 26 percent, respectively, from the pre- to post-treatment period (fig. 8b).

Concentrations of dissolved nitrite were measured at the time of benthic-macroinvertebrate sampling. These data were generally consistent with the concentrations of dissolved nitrite for low-flow samples. Median concentrations of dissolved nitrite for low-flow samples collected at C-1 and T-1 showed virtually no change from the pre- to the post-treatment period (fig. 8b); concentrations of dissolved nitrite averaged about 0.03 mg/L as N. For the upstream benthic-macroinvertebrate sites, the dissolved nitrite concentrations at the time of benthic-macroinvertebrate sampling were higher than for the outlet sites. Overall mean concentrations of dissolved nitrite for C1-2, T1-3, and T2-3 were 0.04, 0.09, and 0.06 mg/L as N, respectively. Again, it appeared the pond was acting as a source, because the upstream concentrations of nitrite at T2-3 were about 0.03 mg/L less than at T1-3.

Concentrations of dissolved ammonia were basically unchanged from the pre- to post-treatment period at the upstream sites, but slight changes were evident at the outlet sites (fig. 23). Concentrations of dissolved ammonia at the upstream sites in the treatment basin appeared to be affected by the pond, because the site upstream of the pond (T2-3) had a mean dissolved-ammonia concentration of 0.05 mg/L as N and T1-3 (downstream of the pond) had an overall mean concentration of 0.11 mg/L as N. The overall mean concentration of dissolved ammonia for T-1 (when benthic-macroinvertebrate samples were collected) decreased by 0.01 mg/L from the pre- to post-treatment period; C-1 showed a 0.01 mg/L increase over the same period. For the low-flow samples collected at T-1 and C-1, median concentrations of dissolved ammonia decreased by 45 and 35 percent, respectively, from the pre- to post-treatment period.

During benthic-macroinvertebrate sampling, concentrations of DKN and TKN increased (figs. 24 and 25) at all sites except C1-2. However, these results did not necessarily coincide with the results from the low-flow sampling. Median concentrations of DKN and TKN increased by 0.1 and 0.04 mg/L, respectively, from the pre- to post-treatment period for C-1 for low-flow samples; conversely, T-1 showed no change in DKN and TKN concentrations for low-flow samples over the same period (fig. 8c). As with concentrations of dissolved ammonia, T1-3 had higher generally higher concentrations of DKN and TKN than T2-3, which would again indicate the pond was acting as a source for these constituents.

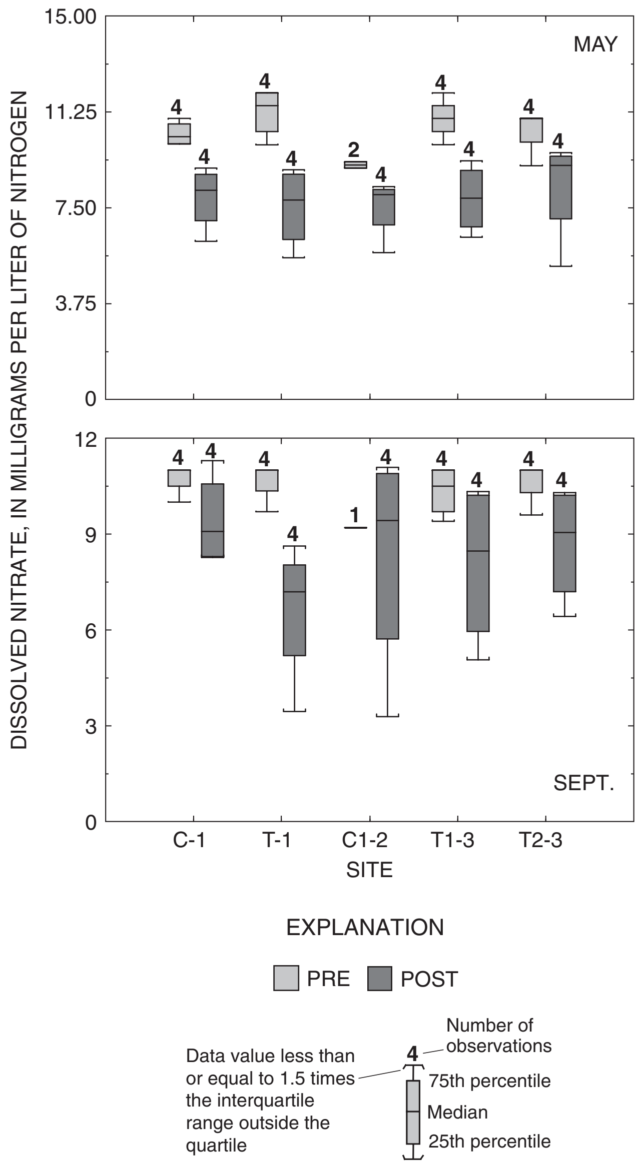


Figure 22. Distribution of concentrations of dissolved nitrate as nitrogen for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

All benthic-macroinvertebrate sites showed increased concentrations of total and dissolved P (figs. 26 and 27). This was consistent with P results for low-flow samples collected in the treatment basin (fig. 8d). In the treatment basin, the source of the elevated P concentrations appeared to be the drainage area between continuous surface-water sites T-4 and T-2 (fig. 9). T-2 is on a tributary branch that feeds into the mainstem in the treatment basin above T2-3 (fig. 1). Median concentrations of dissolved and total P at T-2 from the pre- to post-treatment period for low-flow samples increased by 0.05 (250-percent increase) and 0.06 mg/L (150-percent increase), respectively. The upstream sites in the treatment basin showed average percent increases from the pre- to post-treatment periods in concentrations of dissolved and total P of 350 and 310 percent, respectively. The elevated P concentrations evident at tributary site T-2 were transported downstream to the outlet (T-1), where post-treatment increases in dissolved and total P were 155 and 65 percent, respectively. Unlike the treatment basin, changes in concentrations of dissolved and total P at C-1 from the pre- to post-treatment period were not significant (fig. 8d); however, water-quality samples collected during benthic-macroinvertebrate sampling showed that the

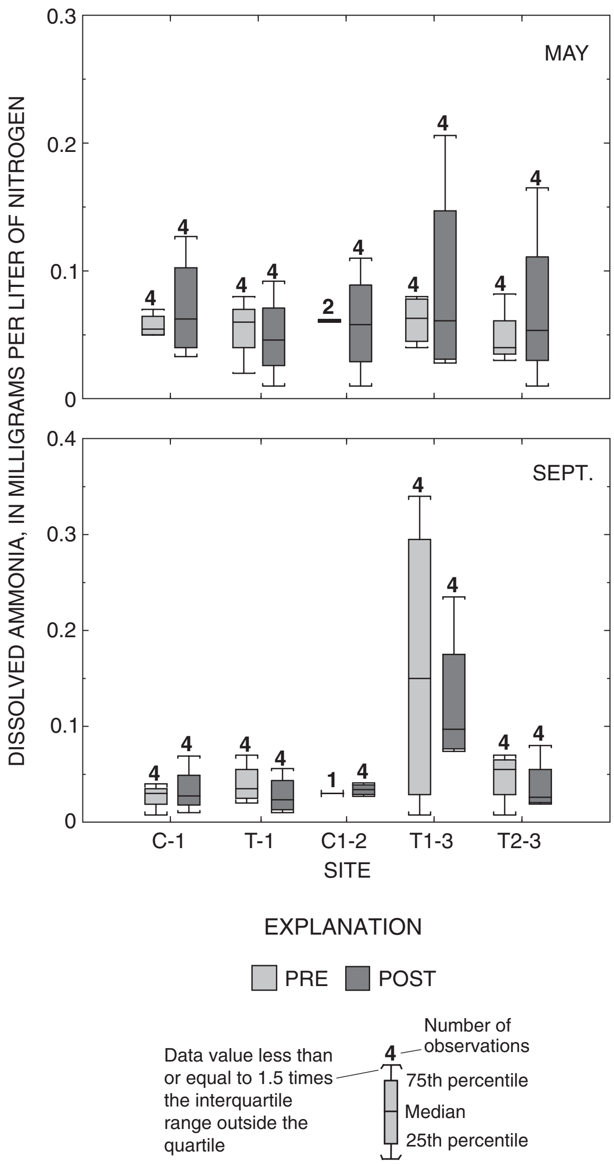


Figure 23. Distribution of concentrations of dissolved ammonia as nitrogen for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

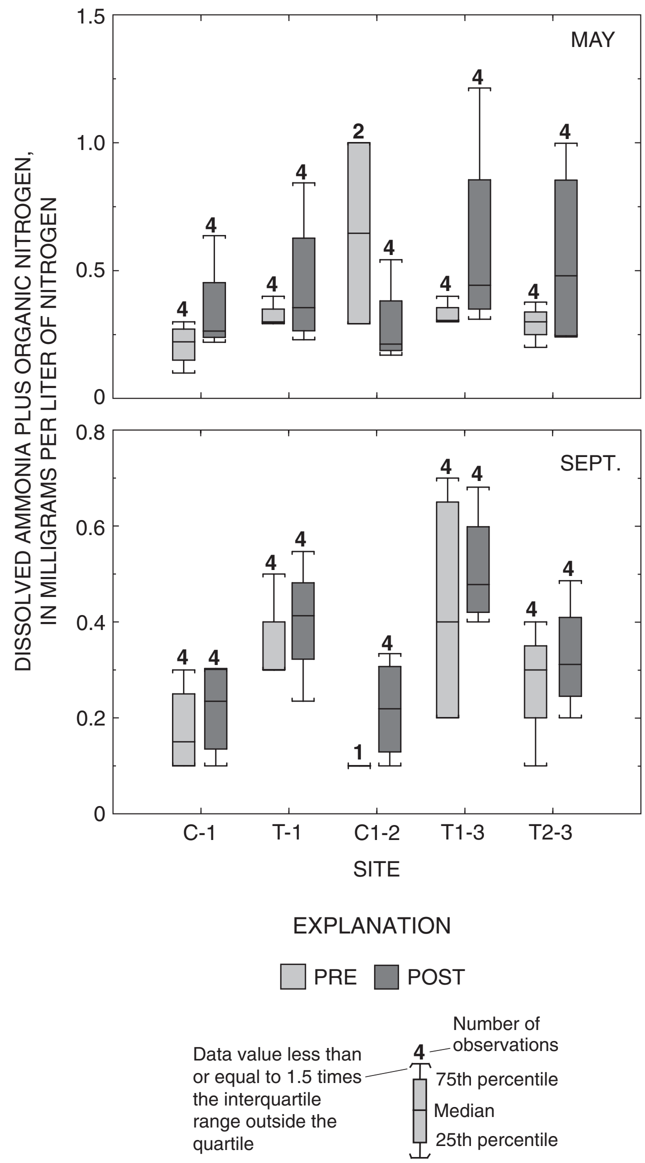


Figure 24. Distribution of concentrations of dissolved ammonia plus organic nitrogen as nitrogen for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

control sites also showed an increase in dissolved- and total-P concentrations from the pre- to post-treatment period (figs. 26 and 27).

Suspended-sediment concentrations and turbidity are two different measures of the amount of materials in the water. Suspended sediment is a measure of the actual concentration of suspended materials, but prior to measurement, organic materials are removed. Turbidity is a measure of the cloudiness in the water. Turbidity was measured instream, and organic materials were present. So, there may not be a high amount of correlation between suspended-sediment concentrations and turbidity, because the correlation is somewhat dependent on the amount of organic material. For this study, virtually no correlation existed between these two water-quality constituents. It should also be noted that for the continuous surface-water sites, the amount of the suspended-sediment yield occurring during low flow was about 5-10 percent (table 23); thus, samples collected during low flow were not

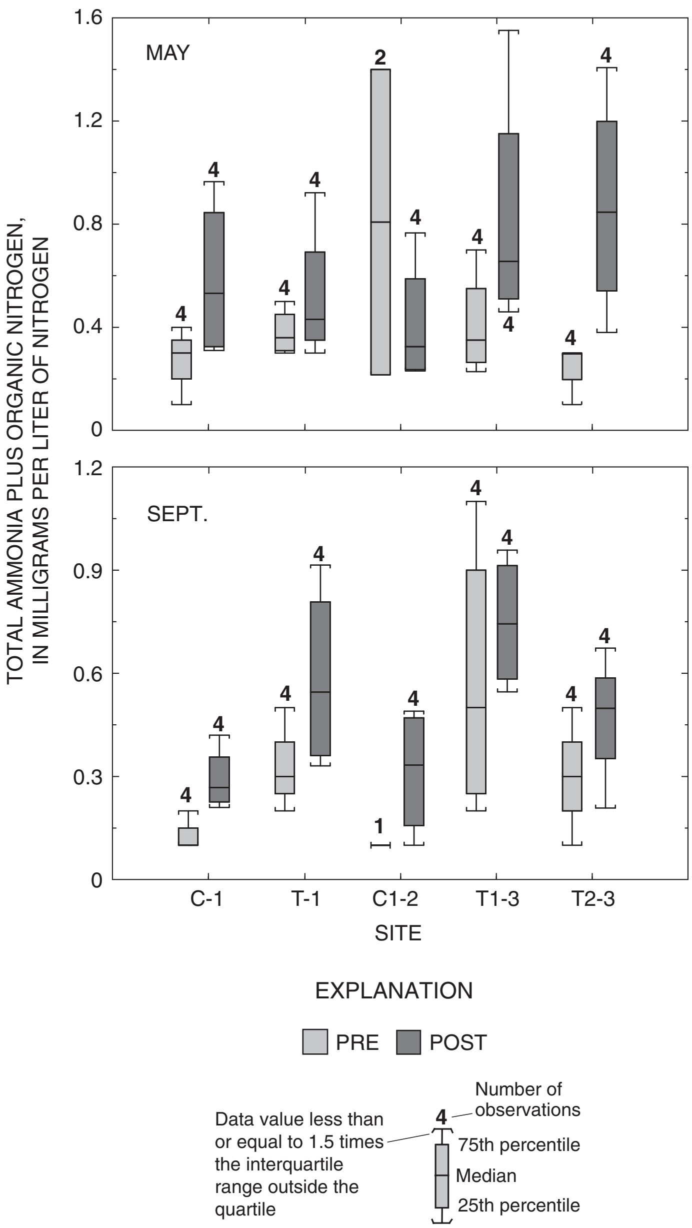


Figure 25. Distribution of concentrations of total ammonia plus organic nitrogen as nitrogen for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

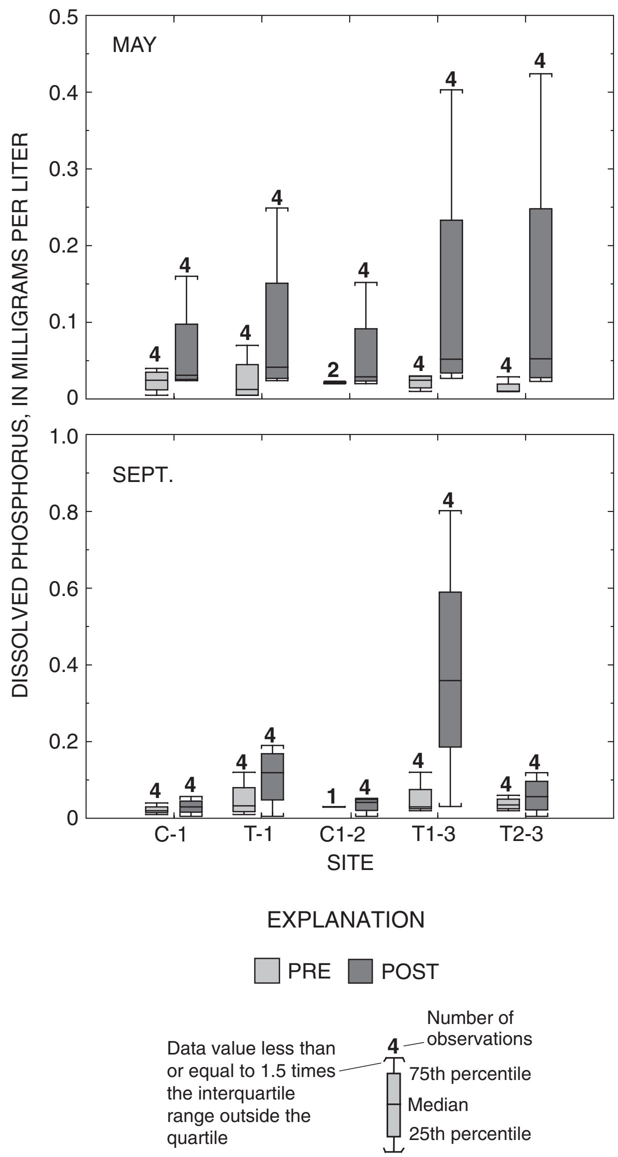


Figure 26. Distribution of concentrations of dissolved phosphorus for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

representative of suspended-sediment concentrations that occurred during stormflow events (see figure 10d). Suspended-sediment samples collected during benthic-macroinvertebrate sampling showed overall mean increases at the control sites from the pre- to post-treatment period; however, treated sites all showed an overall mean decrease (fig. 28). The increase for the control sites does not follow from the results for low-flow samples collected at C-1. C-1 showed a 46-percent decrease in suspended-sediment concentrations from the pre- to post-treatment period for low-flow samples (fig. 8a). Suspended-sediment concentrations during benthic-macroinvertebrate sampling for the treated sites showed an overall decrease from the pre- to post-treatment period of about 36 percent, which is similar to the percent decrease evident for T-1 for low-flow samples (58 percent). Overall (combining May and September samples), turbidity increased at all benthic-macroinvertebrate sites from the pre- to post-treatment period, except for T-1 (fig. 29). The post-treatment turbidity

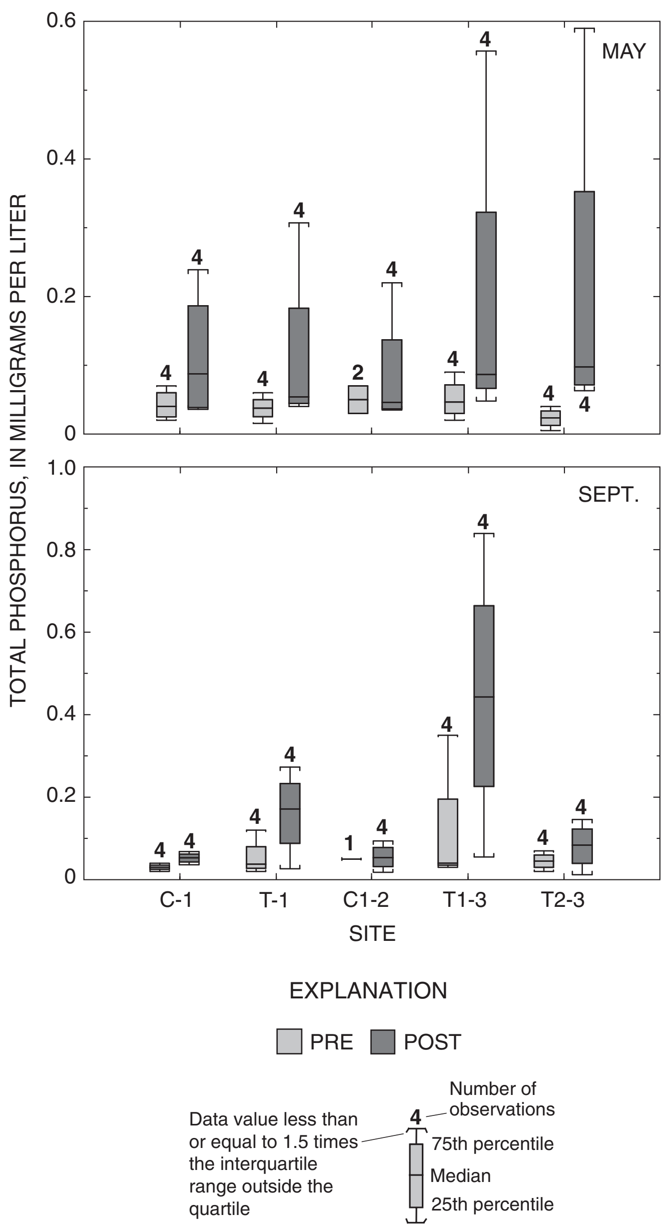


Figure 27. Distribution of concentrations of total phosphorus for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

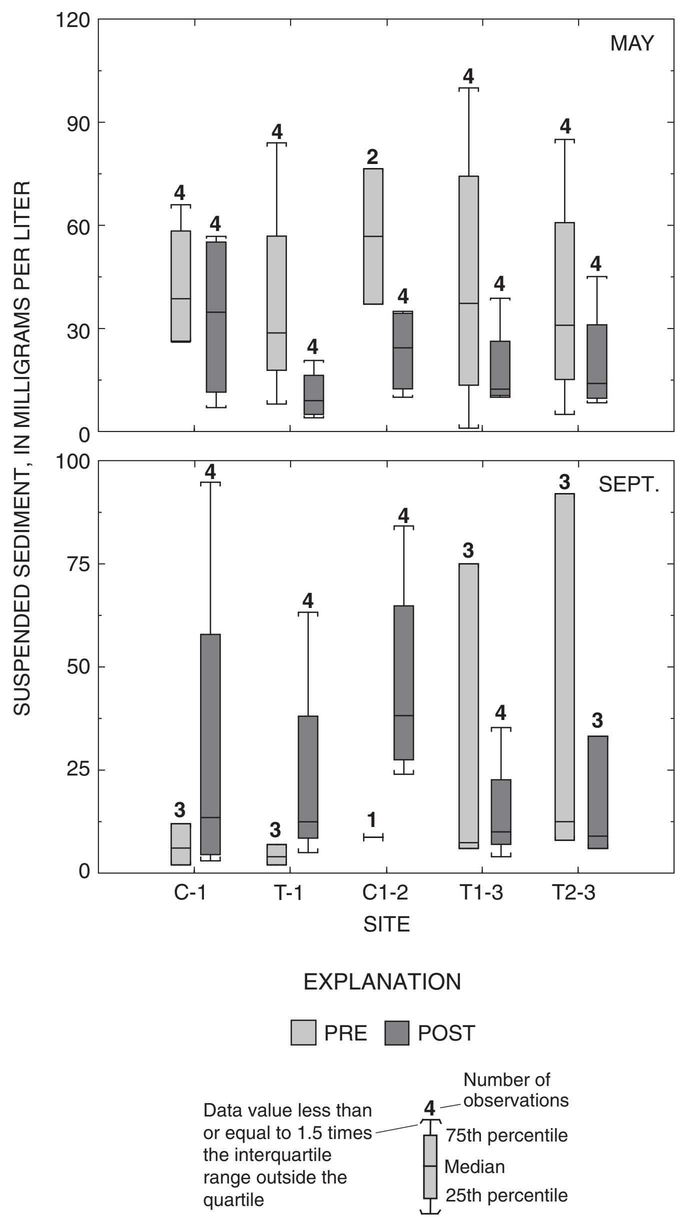


Figure 28. Distribution of concentrations of suspended sediment for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

values for May showed a much wider range than the pre-treatment samples. This was likely because of the larger range in stream discharge during the time of benthic-macroinvertebrate collection for post-treatment May samples (fig. 14).

Water temperature, pH, SC, and DO were also measured during benthic-macroinvertebrate sampling. May median temperatures for all sites decreased from the pre- to post-treatment period, and September median water temperatures increased slightly at all sites except T2-3 (fig. 30). Overall, mean water temperatures measured at each benthic-macroinvertebrate site from the pre- to post-treatment period decreased by 0.2 to 1.3 °C. This followed results for the low-flow samples collected at the four continuous surface-water sites, because these samples showed that sites C-1, T-1, and T-2 all showed lower median water temperatures from the pre- to post-treatment period (fig. 8e). Streambank fencing has been found to help moderate temperature fluctuations (Osmond and Gilliam, 2002). It has also been demonstrated that trees are better at controlling stream temperature than grasses (Connecticut River Joint Commissions, 1998; University of Georgia, 2003),

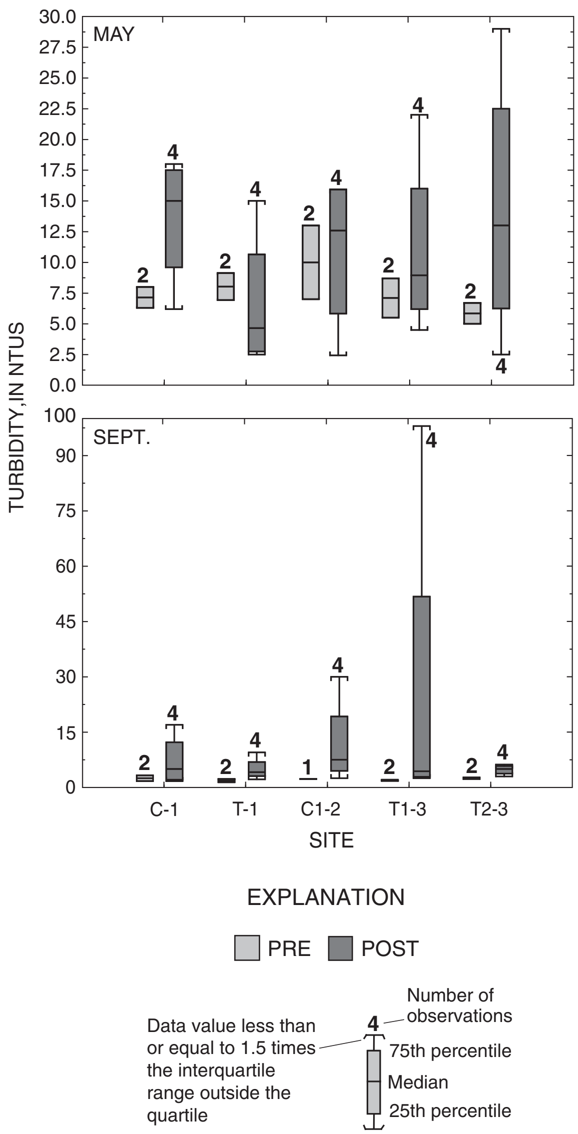


Figure 29. Distribution of turbidity values for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa..

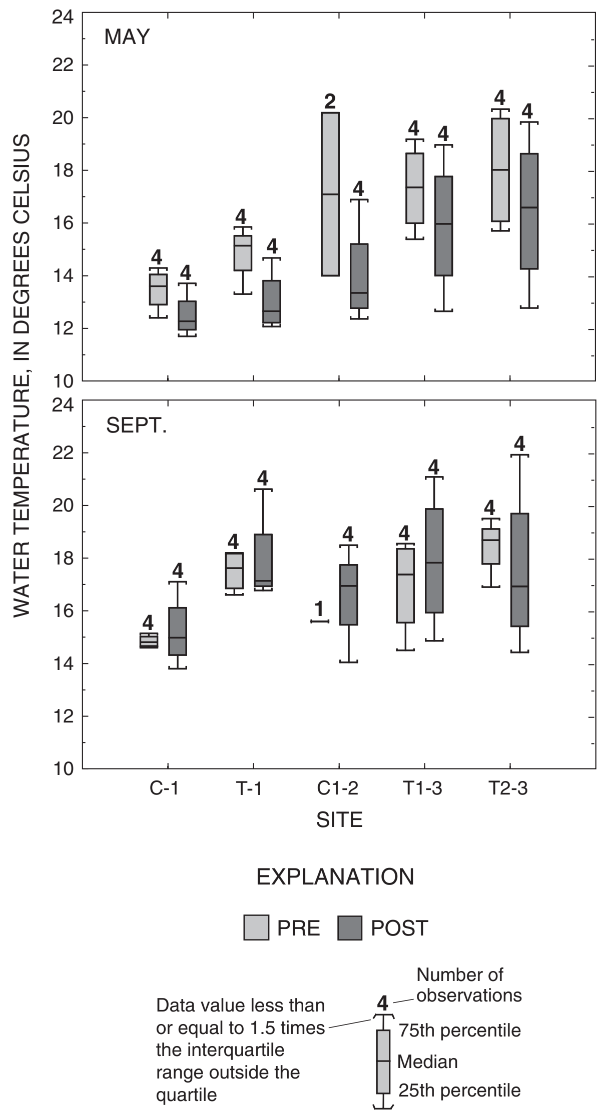


Figure 30. Distribution of water temperature for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

and, as stated earlier, grasses were the dominant vegetative type in the buffer strip established after fence installation. The pH measured during benthic-macroinvertebrate sampling showed an overall decrease from the pre- to post-treatment period at each site except C1-2 (and this could have been because of the low sample number for C1-2 for the pre-treatment period) (fig. 31). The mean pH decrease for the benthic-macroinvertebrate sites from the pre- to post-treatment period ranged from 0.1 to 0.4 units. Again, this followed data collected for low-flow samples at the four continuous surface-water sites because pH for these samples collected at C-1, T-1, and T-2 showed a 0.1-0.2 pH unit decrease from the pre- to post-treatment period (fig. 8f). SC measured during benthic-macroinvertebrate sampling also showed a trend similar to that measured for low-flow samples. SC at the five benthic-macroinvertebrate sites showed a decrease from the pre- to post-treatment period. The overall mean decrease ranged from 18 (C-1) to 47 μS/cm (T-1) (fig. 32). Similarly, for the low-flow samples, median SC values at C-1, T-1, and T-2 showed a decrease from the pre- to post-treatment period of

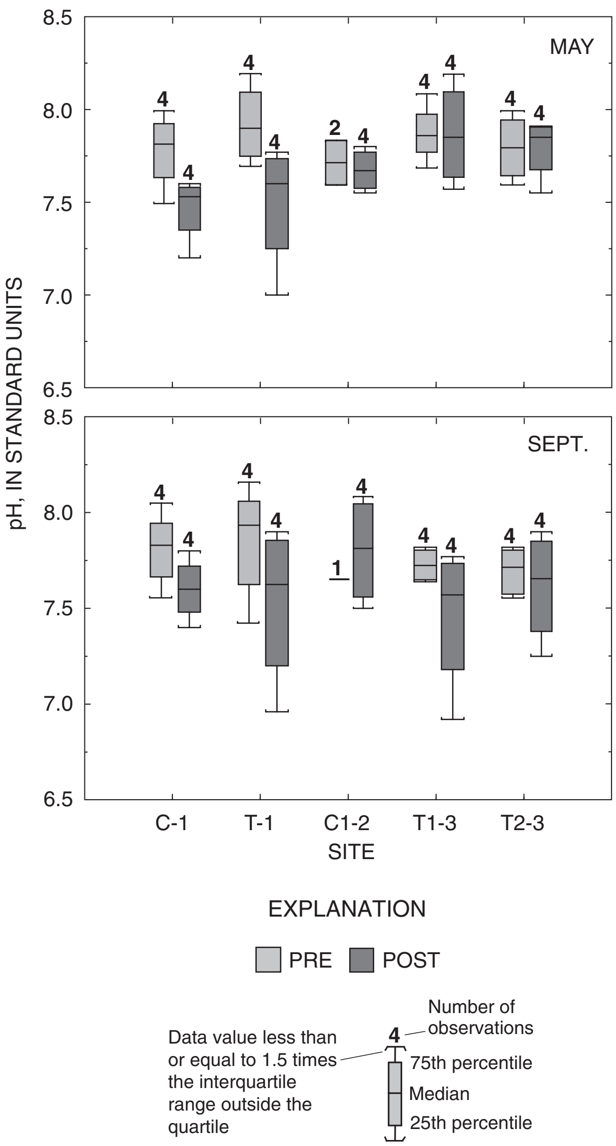


Figure 31. Distribution of pH for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

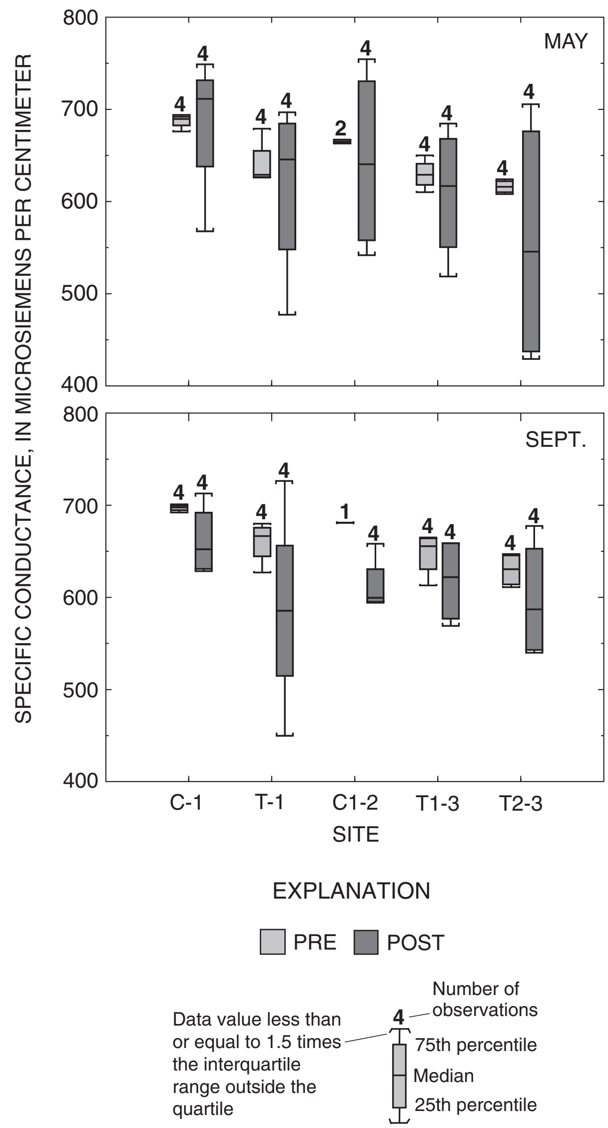


Figure 32. Distribution of specific conductance for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post- treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

8 to 13 μS/cm (fig. 8e). Measured DO concentrations during benthic-macroinvertebrate sampling decreased at all sites from the pre- to post-treatment period (fig. 33). The overall mean decrease in DO from the pre- to post-treatment period ranged from 0.1 (C1-2) to 1.1 mg/L (T1-3). These data did not necessarily follow the trends evident for low-flow samples because data for C-1 showed a 0.2 mg/L increase in median DO concentrations from the pre- to post-treatment period; however, data for T-1 and T-2 over the same period showed a 0.5 and 1.4 mg/L decrease, respectively (fig. 8e).

Chlorophyll *a* concentrations, which are a measure of algal growth (on substrates in this case), were highly variable (fig. 34). Given the high concentrations of nutrients available at these benthic-macroinvertebrate sites, it is unlikely that either N or P was limiting; thus, changes in chlorophyll *a* concentrations were more likely related to the physical habitat. Variability could be related to the availability of substrate and (or) the most recent flow regime. That is, if a high flow event occurred prior to benthic-macroinvertebrate sampling, it is possible that algal growth on substrates was scoured, and this could dramatically affect chlorophyll *a* concentrations measured. From the pre- to post-treatment period, the overall mean concentrations of chlorophyll *a* showed basically no change at C-1 and a decrease of 44 $\mu g/m^2$ at T2-3 (fig. 34); conversely, over the same period, C1-2 and T1-3 showed an increase of 10-16 $\mu g/m^2$ and T-1 showed an increase of 36 $\mu g/m^2$. The decreased concentration of chlorophyll *a* for T2-3 during the post-treatment period may be because of decreased available substrate because vegetation tended to overgrow the channel, and possibly the vegetation reduced solar inputs to substrates available to the algae. The increased concentration of chlorophyll *a* for T-1 during the post-treatment period may be because of more available substrate because this outlet site was not overgrown with vegetation. The habitat characteristics for T-1 did show the habitat was improved for benthic macroinvertebrates, and more beneficial stream habitat for benthic macroinvertebrates typically also indicates better habitat for algae.

Benthic Macroinvertebrates

Several benthic-macroinvertebrate metrics were calculated at the generic and family level. The generic data gives more definition at each site. Family-level analysis was used for most of the data analysis because of the number of individual animals that were not identified to the generic level and for comparability of results to a previous USEPA study. The list of metrics includes percent dominant taxa (generic and family), EPT index, generic EPT/Chironomidae ratio, EPT/total number, percent Chironomidae, shredders/total taxa ratio, scrapers/filterers ratio, Hilsenhoff Biotic Index (HBI) (generic and family), taxa richness (generic and family), and percent Oligochaeta.

The percent dominant taxa (generic level) (PDTG) showed differences between season and sites (fig. 35). The outlet sites and T1-3 had lower PDTG values for September relative to May samples, indicating that benthic-macroinvertebrate communities were more diverse in the fall at these sites. The upstream sites had higher PDTG values than outlet sites, indicating the benthic-macroinvertebrate communities were more diverse at the outlets, which would be expected because the outlets had more physical habitat (stream channels were wider). Overall, the mean PDTG decreased by 2-4 percent at the outlets sites from the pre- to post-treatment period. At the upstream sites, C1-2 showed that only a few taxa were dominant throughout the study. The upstream sites in the

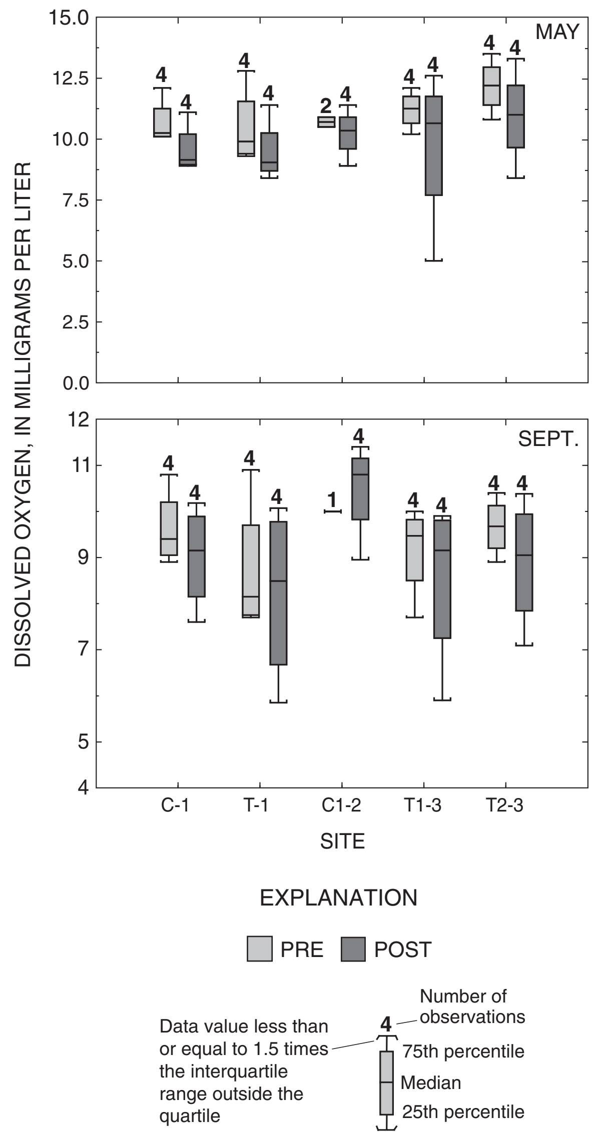


Figure 33. Distribution of concentrations of dissolved oxygen for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

treatment basin consistently showed more diversity than C1-2, but the overall PDTG means for T1-3 and T2-3 increased slightly (less than 1 percent) and by 13 percent, respectively, from the pre- to post-treatment period. When a community is dominated by one or a few taxa, this is usually indicative of some environmental stress affecting the community (Klemm and others, 1990), and as the percentage of the dominant taxa decreases, balance in the benthic-macroinvertebrate community increases, which is generally indicative of a healthy community (Barbour and others, 1999). However, these first- to second-order, limestone streams typically are not very diverse (Klemm and others, 1990) and dominance by one or two taxa is expected. Whatever the case, streambank fencing did not seem to improve benthic-macroinvertebrate community structure based on PDTG.

The percent dominant taxa (family level) (PDTF) also showed differences between season and sites (fig. 36). The outlet sites and T1-3 had lower PDTF values for September relative to May samples. The upstream sites also had higher PDTF values than outlet sites. Overall, the mean PDTF values

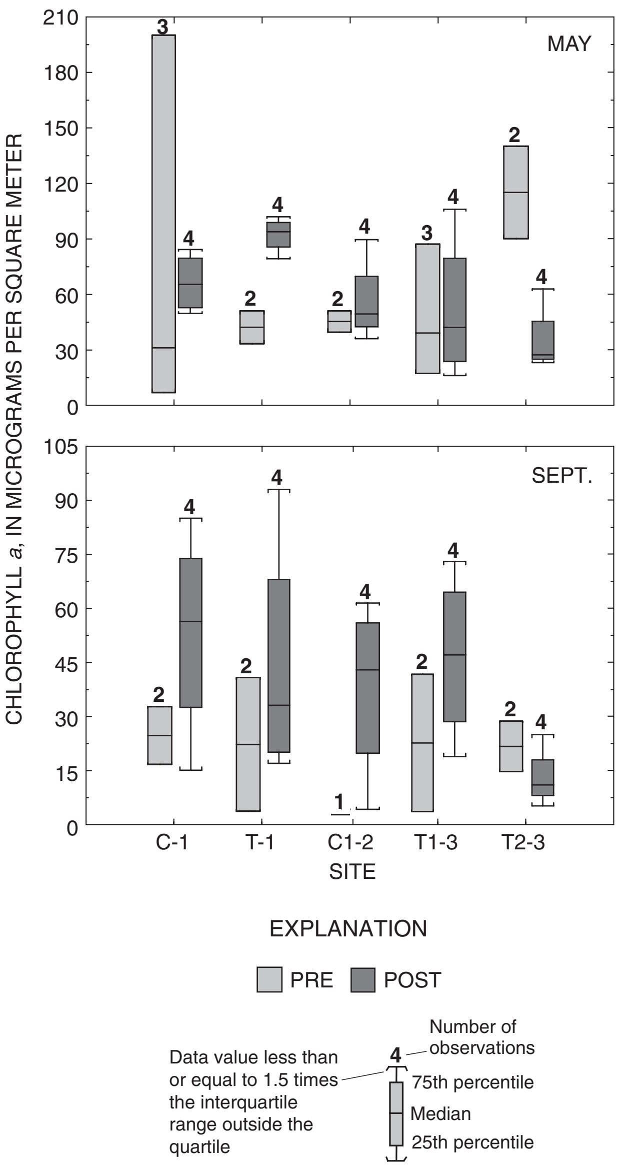


Figure 34. Distribution of concentrations of chlorophyll *a* for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

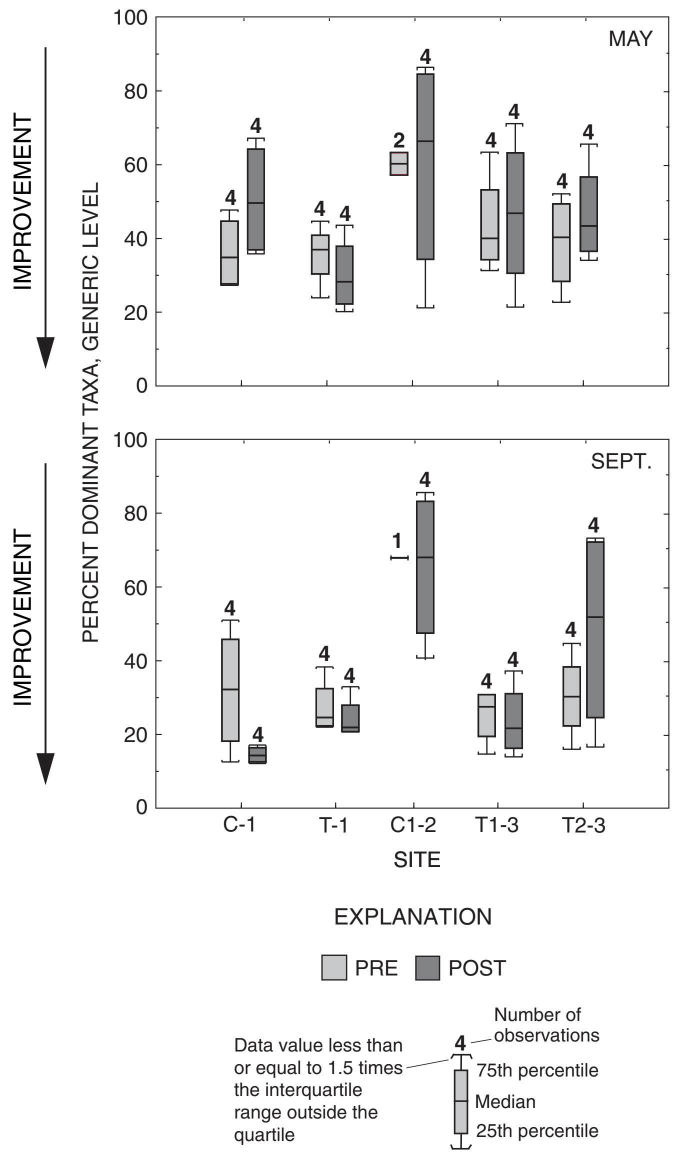


Figure 35. Distribution of percent dominant taxa (generic level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

at C-1 and T-1 increased by 3 and decreased by 6 percent, respectively, from the pre- to post-treatment period. Thus, T-1 did show some improvement relative to C-1. For the upstream sites, C1-2 again showed higher (less diverse) values for PDTF than T1-3 and T2-3. Unlike PDTG data, PDTF data for T1-3 showed a decrease of about 9 percent from the pre- to post-treatment period; however, data for T2-3 showed an increase in PDTF of about 10 percent over the same period. The improvement evident for T1-3 PDTF data (and not PDTG data) was likely because of the incorporation of family-level data into the metric (for families that were not identified to the generic level).

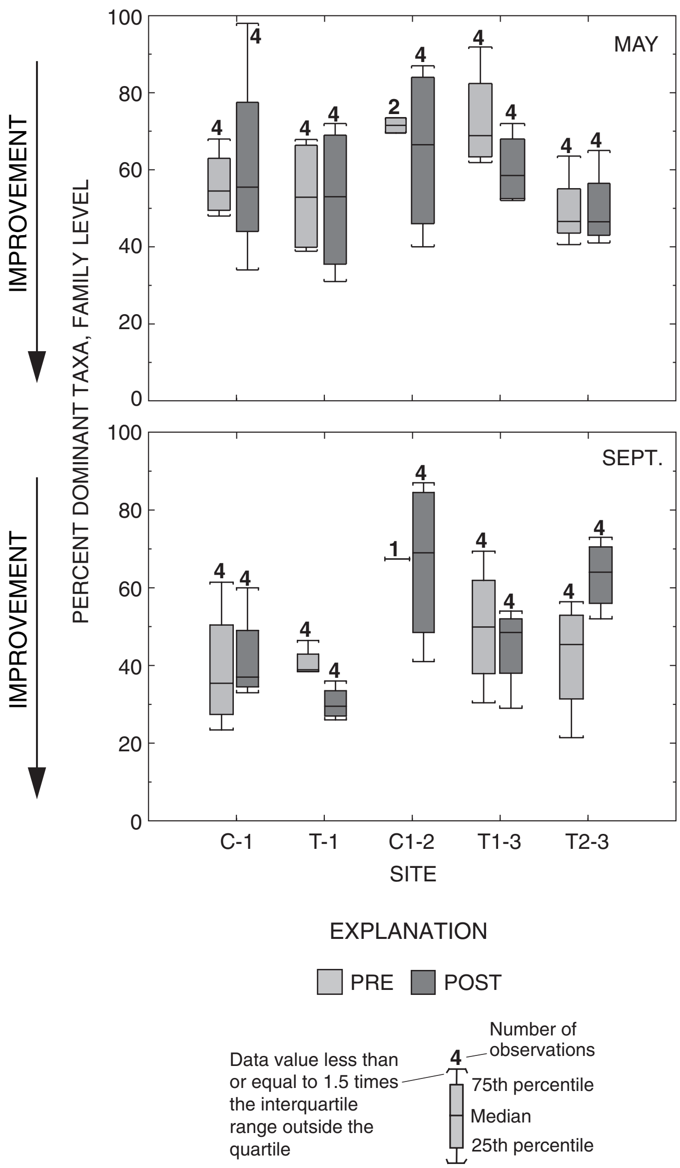


Figure 36. Distribution of percent dominant taxa (family level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

Considering all sites together for all years of observation, the dominant taxa in May were Chironomidae, Gammaridae, Naididae, and Tubificidae. All these families are semi-tolerant to organic enrichment (Voshell, 2002). The dominant taxa in September were Gammaridae, Tubificidae, Elmidae, Physidae, Baetidae, Chironomidae, and Simuliidae. All these families are moderately to very tolerant of organic enrichment (Voshell, 2002). This indicates the more sensitive taxa were not able to become dominant members of the benthic-macroinvertebrate community after the fences were installed in the treatment basin. Sensitive taxa may not have been present or only a few individuals were present during the post-treatment period because of 1) not enough time for the system to equilibrate to the new conditions, or 2) because these are spring-fed, first- to second-order limestone streams. Limestone streams typically consist of communities including Ephemeroptera (mayflies), Diptera (Chironomidae - midges), Amphipods (Gammaridae - scuds), and Isopods (Asellidae -pillbugs and sowbugs) (Shaffer, 1991) and all were present in the treated and untreated streams. The sensitive taxa that were present in few numbers included Promoresia (Elmidae), Oxyethira (Hydroptilidae), Antocha (Tipulidae), and two species of Chironomidae (*Pagastia* and *Prodiamesa*).

The EPT index showed differences between upstream and outlet sites and season. The EPT index for outlet sites was consistently higher than for upstream sites (fig. 37). The overall mean EPT value for C-1 and T-1 was 3; the overall mean for the upstream sites was 1 (C1-2) or 2 (T1-3 and T2-3). The upstream sites all showed decreased median EPT values from the pre- to post-treatment period for May and September samples; however, the overall mean decrease during this period was either 1 (C1-2) or zero (T1-3 and T2-3). Generally, EPT values for September samples were about one unit higher than for May samples, which would indicate EPT taxa were more prevalent in the fall. A decrease in the EPT index is generally indicative of a decrease in water quality (Plafkin and others, 1989). EPT taxa are considered to be pollution sensitive, but in first- to second-order streams it has been noted that benthic-macroinvertebrate communities are typically less diverse and some organic enrichment may actually increase the EPT taxa present (Klemm and others, 1990). In these streams, because the water chemistry was fairly consistent between upstream and outlet sites, it may be that there was more available habitat for EPT taxa at the outlet sites. It can not be stated that streambank fencing had a positive effect on EPT taxa.

The EPT/Chironomidae ratio showed an overall mean decrease for all sites from the pre- to post-treatment period (fig. 38). The overall percent mean decrease from the pre- to post-treatment period was greater for the control sites. The mean percent decreases for C-1 and C1-2 from the pre- to post-treatment period were 56 and 77 percent, respectively; mean percent decreases for T-1, T1-3, and T2-3 were 24, 52, and 52 percent, respectively. Thus, treatment sites showed some improvement in this metric relative to control sites during the post-treatment period. A higher ratio is indicative of improvement (Klemm and others, 1990). Chironomidae typically are found where there is siltation and detritus for them to feed on (Merritt and Cummins, 1984), and EPT typically are found in rocky-bottomed streams (Merritt and Cummins, 1984; Voshell, 2002). It would be expected to see a rise in the EPT to Chironomidae index after fencing because of less sediment flowing into the stream. Generally, an increase in the EPT to chironomidae ratio indicates less stressful water chemistry (in terms of heavy metals, nutrients, sediment, or other compounds such as low-level concentrations of pesticides) (Plafkin and others, 1989).

The percentage of EPT (PEPT) taxa to the total number of individuals showed similar results to EPT-index data. As with EPT data, PEPT values for the outlet sites were consistently higher than for the upstream sites, and September data were consistently higher than May (fig. 39). The overall mean PEPT values decreased for all sites from the pre- to post-treatment period. PEPT values for the outlet sites decreased by 30 (C-1) and 56 percent (T-1); the upstream sites decreased by

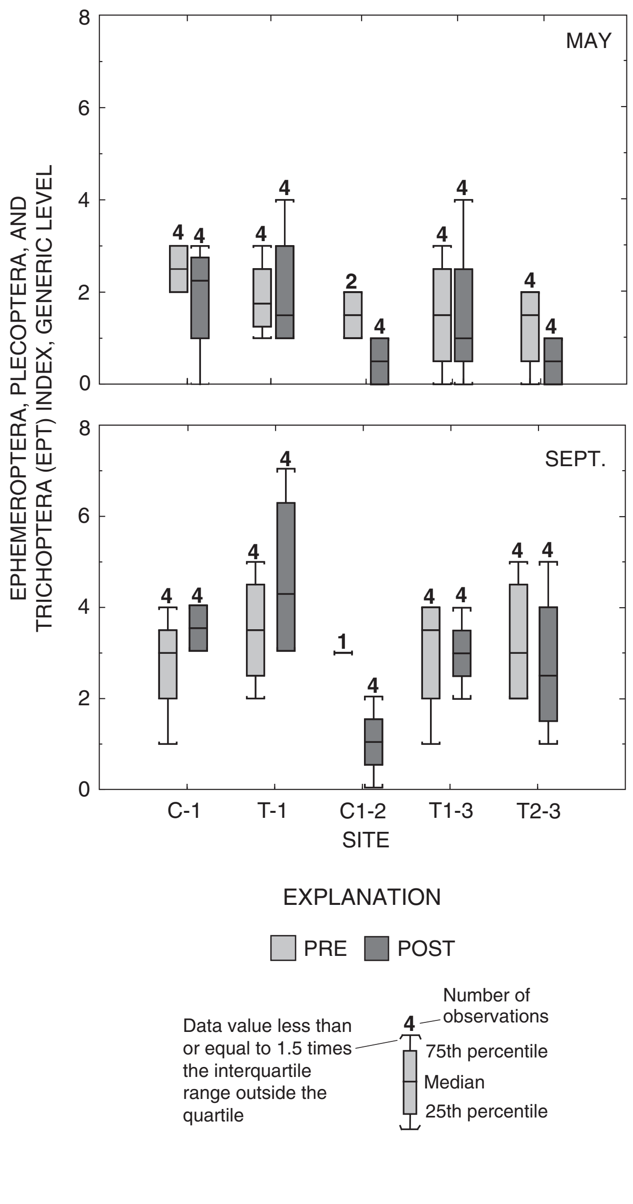


Figure 37. Distribution of Ephemeroptera, Plecoptera, and Trichoptera (EPT) index (generic level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

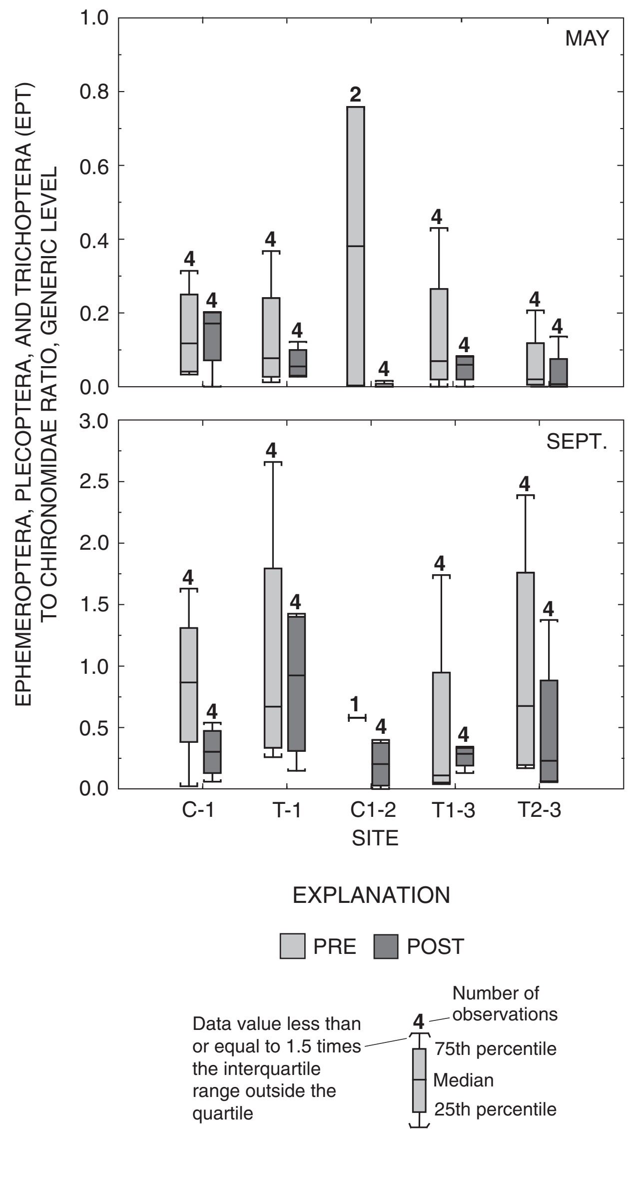


Figure 38. Distribution of Ephemeroptera, Plecoptera, and Trichoptera (EPT) to Chironomidae ratio (generic level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

78-82 percent. Based on these data, it again can not be stated that trends in EPT taxa indicated an improvement in benthic-macroinvertebrate community structure in the treatment basin after streambank fencing.

The percentage of Chironomidae also showed decreasing trends from the pre- to post-treatment period for all benthic-macroinvertebrate sites (fig. 40). The overall mean for the percentage of Chironomidae showed a percentage decrease from the pre- to post-treatment period of 17 (C-1) and 25 percent (T-1) at the outlets and 38 (T2-3) to 57 percent (C1-2) at the upstream sites. A decrease in the percentage of Chironomidae tends to indicate water quality is improving (Plafkin and others, 1989). Therefore, based on this metric, post-treatment data indicated an improvement at the outlet of the treatment basin relative to the control site; upstream data indicated a degradation at the treatment sites relative to the control site.

A review of the Chironomidae genera indicated that the dominant at C-1 and T-1 were moderately tolerant of organic pollution (Mandaville, 2002). For May samples collected at C-1, *Orthocladius* was the dominant genera in 1994, 1997,

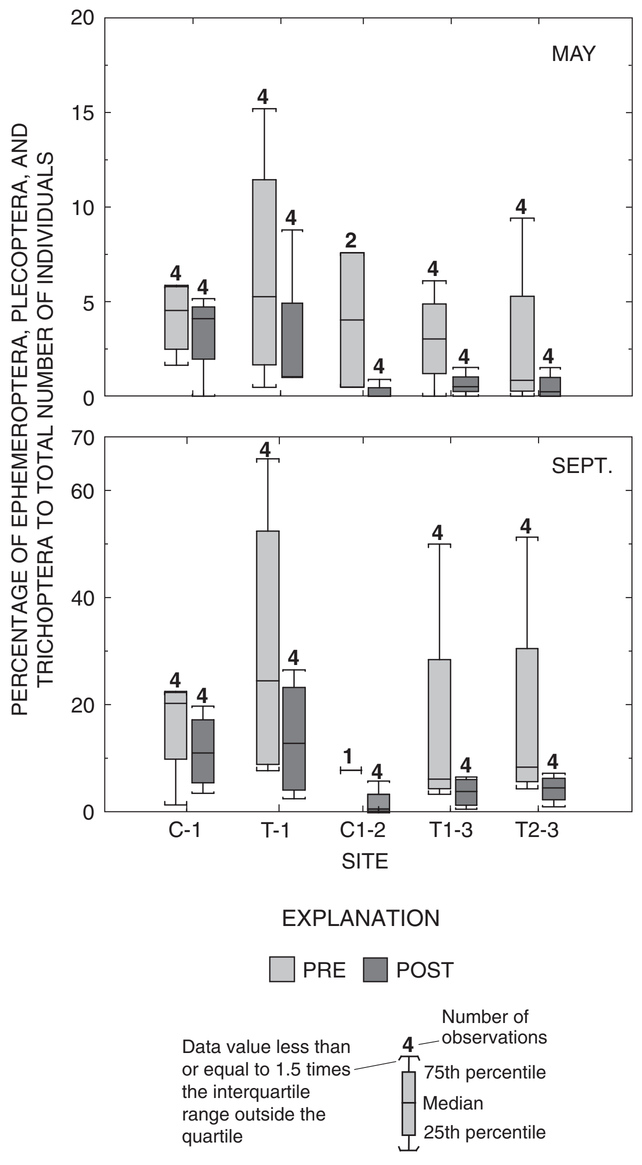


Figure 39. Distribution of the percentage of Ephemeroptera, Plecoptera, and Trichoptera (EPT) to total number of individuals for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

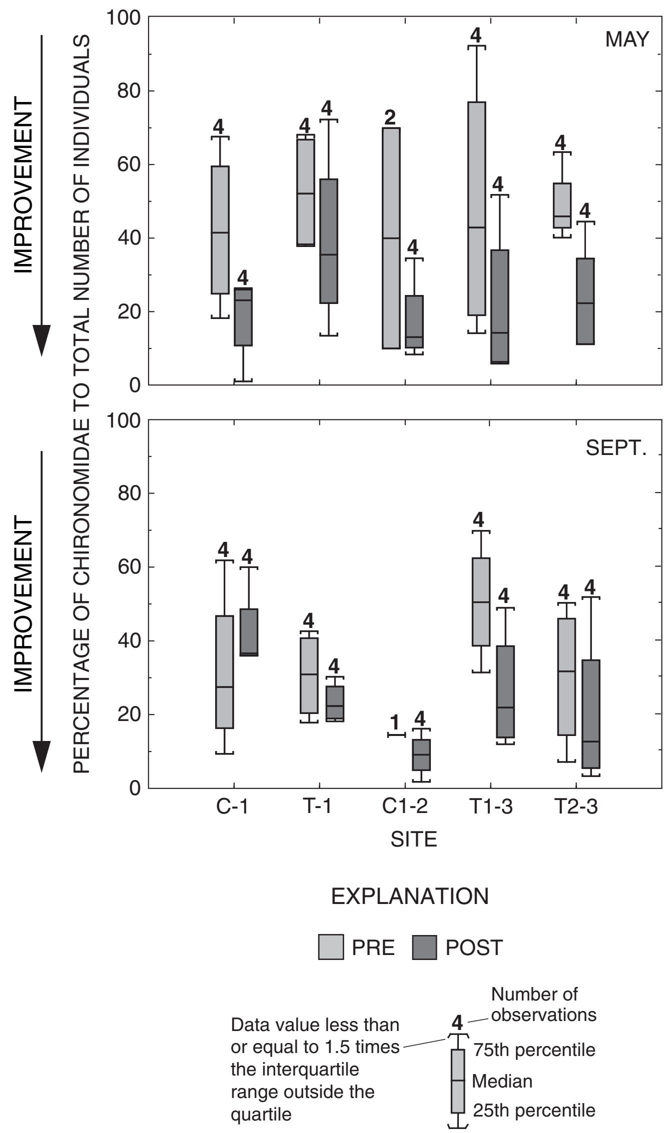


Figure 40. Distribution of the percentage of Chironomidae to total number of individuals for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

1998, and 2001; *Micropsectra* in 1995; *Dicrotendipes* in 1996; and *Cricotopus* in 1999 and 2000. These genus are primarily collector/gatherers. The dominant species at T-1 for samples collected in May were similar to those identified at C-1. For May samples collected at T-1, *Orthocladius* was dominant in 1994, 1997, 1999, and 2001; *Cricotopus* in 1995, 1996, and 2000; and *Micropsectra* in 1998. At T-1 for May samples, the shredders were dominant in 1995, 1996, and 2000, which is a little different than C-1, but most taxa groups were the same before and after fencing (appendix 3). For September samples, the Chironomidae genera dominant at C-1 and T-1 are moderately tolerant of organic pollution (Mandaville, 2002). The dominant genera at C-1 for September samples were *Rheotanytarsus* in 1993, *Orthocladius* in 1994 and 1998, *Cricotopus* in 1995, *Polypedilum* in 1996, *Tvetenia* in 1997, *Dicrotendipes* and *Tanytarsus* in 1999, and *Dicrotendipes* in 2000. For 1995 and 1996 September samples, the shredders dominated the Chironomidae, and the other years were dominated by taxa that are collector/gatherers and collector/filterers (Voshell, 2002). For September samples, T-1 was dominated by *Rheotanytarsus* in 1993, 1995, 1998, and 2000, *Cricotopus* in 1994 and 1997, *Polypedilum* in 1996, and *Clinotanypus* in 1999. For September samples collected at T-1, the shredders dominated in 1994, 1996, and 1997 similar to C-1. The only predator to dominate at any site for any year was *Clinotanypus* for September samples collected during 1999 at T-1 (appendix 3).

A similar pattern in Chironomidae genera was observed at the upstream sites. For samples collected in May, C1-2 was dominated by *Orthocladius* in 1996, 1997, 1998, and 1999 and *Cricotopus* in 2000 and 2001. For May samples collected at T1-3, the dominant genera varied from year to year; *Orthocladius* was dominant in 1994 and 1997, *Dicrotendipes* in 1995 and 2000, *Cricotopus* in 1996 and 2000, *Micropsectra* in 1998, 1999, and 2000, and *Stictochironomus* in 2001. May samples collected at T2-3 also showed varied dominant genera from year to year; *Orthocladius* was dominant in 1994, 1996, 1997, 1999, and 2001, *Dicrotendipes* in 1995, *Micropsectra* in 1998, *Cricotopus* in 2000, and *Stictochironomus* was co-dominant with *Orthocladius* in 2001 (appendix 3). For September samples collected at the upstream sites, C1-2 was dominated by *Cricotopus* and *Polypedilum* in 1996, *Dicrotendipes* in 1997, 1999, and 2000, and *Orthocladius* in 1998. The shredders were dominant for the September 1996 sample at C1-2 (similar to C-1 and T-1), and the collector/gatherers were dominant for the September samples collected in the other years at C1-2. T1-3 had fairly consistent dominant taxa over the years for September samples. For September samples collected at T1-3, *Rheotanytarsus* was dominant in 1993, 1994, 1995, 1998, and 2000 as a collector/filterer; *Phaenopsectra*, a scraper, was dominant in 1996; and *Chironomus*, a very tolerant genera of collector/gatherers, was dominant in 1997. T1-3 was the only site that the shredders did not dominate the Chironomidae taxa in 1996 for September samples. For September samples collected at T2-3, the dominant taxa were *Cricotopus*, a collector/gatherer, in 1993; *Dicrotendipes*, a collector/gatherer, in 1994 and 1995, *Polypedilum*, a shredder, in 1996; *Chironomus*, a collector/gatherer, in 1997, and *Paratendipes*, a collector/gatherer, in 1998 and 1999. Only five genera were identified in the September 2000 sample at T2-3 and each genera had only one individual identified for a total of five Chironomidae individuals (appendix 3).

The percentage of shredders to the total taxa (SHRED) showed a decrease from the pre- to post-treatment period at each site for May and September samples, except for May samples collected at C1-2 (fig. 41). Overall, each site showed a decrease in mean SHRED values from the pre- to post-treatment period. The overall mean decrease in SHRED at C-1 and C1-2 from the pre- to post-treatment period was 32 percent, whereas the overall mean decrease for T-1 and T2-3 was about 50 percent and T1-3 showed a 90-percent decrease. Thus, the treatment sites did not show improvement in the number of shredders relative to the total number of individuals relative to control sites during the post-treatment period. Given that streamside cover, a physical habitat variable, did not show any change in tree density along the riparian zone after fence installation (fig. 20), it is not unexpected that there was not a detected change in the number of shredders to the total number of individuals. Shredders are sensitive to changes (such as increased tree density) in the riparian zone and usually are good indicators of toxic effects when toxicants are absorbed into riparian-zone plant leaves (Plafkin and others, 1989). The shredders eat the leaves that fall into the stream and their numbers and taxa numbers can decrease with increased pollution in their food source.

The scrapers to filterers ratio (SFR) increased at all sites except C-1 from the pre- to post-treatment period (fig. 42). There was much less variability in SFR for outlet sites as opposed to upstream sites; data collected during the post-treatment period showed large fluctuations at the upstream sites. At the outlets from the pre- to post-treatment period, the overall mean SFR values decreased by 47 percent at C-1 and increased by 67 percent at T-1. At the upstream sites, overall mean increases from the pre- to post-treatment period exceeded 200 percent; C1-2 showed the largest percentage increase. Changes in SFR from the pre- to post-treatment period did not follow trends evident for chlorophyll *a* data, which showed increased concentrations of chlorophyll *a* at T-1, C1-2, and T1-3, and decreased concentrations at T2-3 (and no change at C-1) from the pre- to post-treatment period (fig. 34). Typically, algae attached as periphyton are food sources for scrapers. Our measure of chlorophyll *a* for this study was sampling of algal from substrate surfaces, so there should be some relation between our chlorophyll *a* concentration data and SFR values; however, the relation was not evident. It could be that the community for filterers, which feed on filamentous algae and aquatic mosses, changed from the pre- to post-treatment period. Changes in concentrations of free-flowing algae (such as filamentous) were not sampled for in this study. Whatever the case, during the post-treatment period, SFR values increased dramatically at the upstream sites.

The generic-level HBI data showed the control sites had consistently lower HBI scores than the treatment sites (when comparing upstream and outlet sites separately) (fig. 43). For May and September sampling events, T2-3 was the only site that had lower median HBI values; C-1 was the only site that had higher median HBI values. Overall, based on the HBI index, the sites were rated as poor to good. The overall changes in the mean HBI values from the pre- to post-treatment period showed a 1 and 13 percent decrease at T-1 and C-1, respectively; conversely, upstream sites showed increases of 2 (C1-2), 3 (T1-3), and 11 percent (T2-3). As HBI values decrease, this generally indicates water quality, as defined by benthic-macroinvertebrate community structure, improves. That is, a decrease in HBI indicates the benthic-macroinvertebrate community is reverting to a community structure less tolerant of organic pollution. Therefore, it appeared that, based on HBI trends relative to control sites, there was some improvement in benthic-macroinvertebrate community structure in the treatment basin.

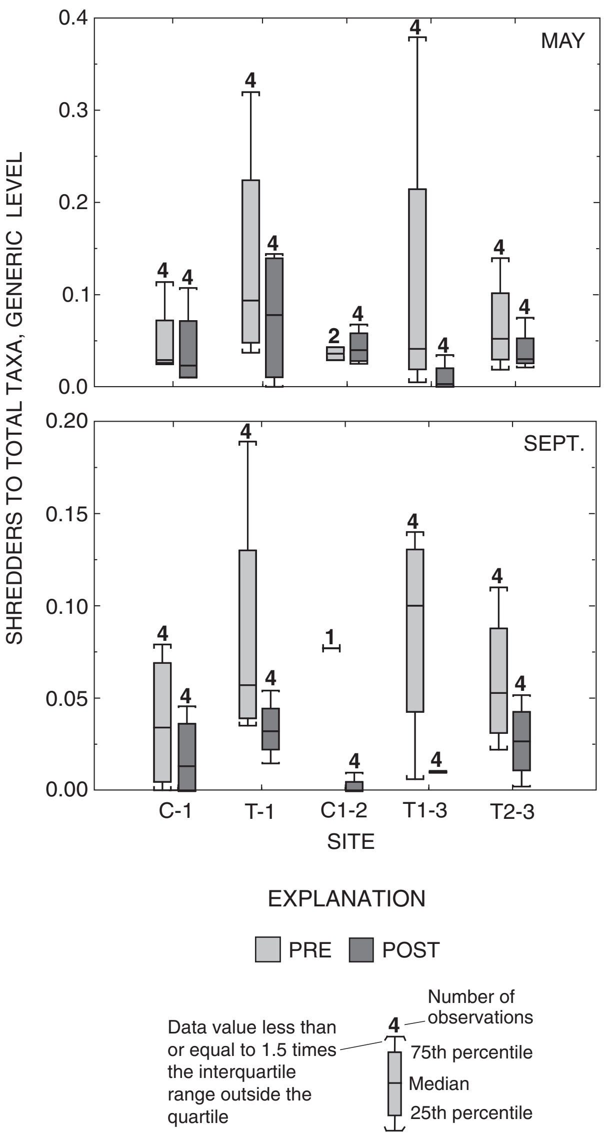


Figure 41. Distribution of shredders to total taxa (generic level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

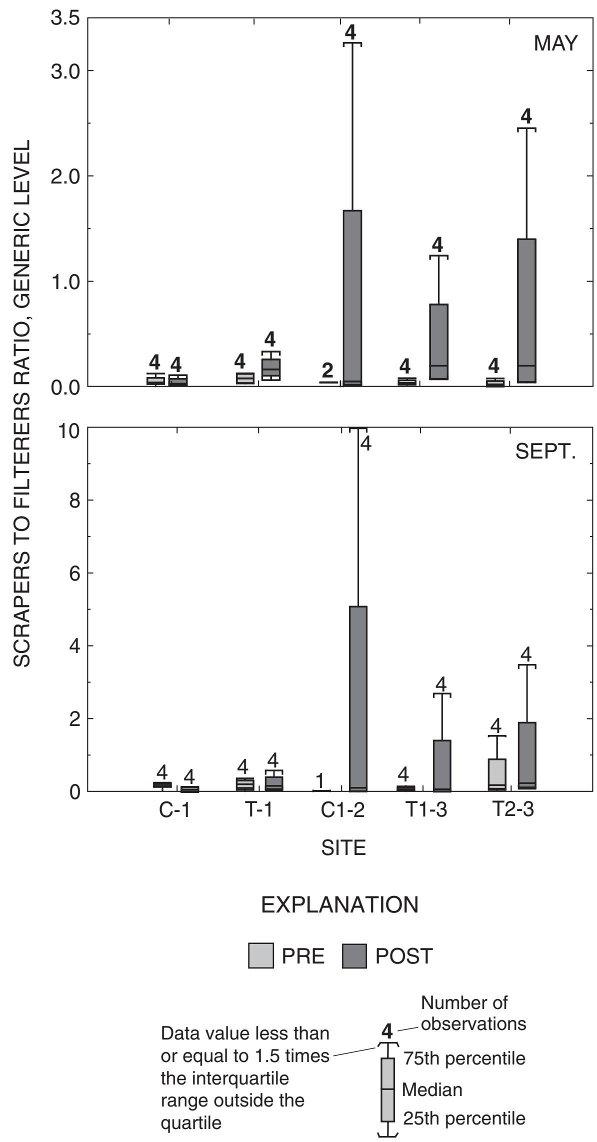


Figure 42. Distribution of scrapers to filterers (generic level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

Family-level HBI (FHBI) data showed different results than generic-level HBI, specifically for the upstream sites. FHBI for the outlet sites basically showed similar results to generic-level HBI because the overall mean percentage increase in FHBI from the pre- to post-treatment period at C-1 and T-1 was 11 and 2 percent, respectively (fig. 44). Unlike generic-level HBI, the overall mean FHBI for the upstream sites increased from the pre- to post-treatment period by 1 (T2-3), 4 (C1-2), and 6 percent (T1-3). Again, the reason for this is probably the incorporation of benthic macroinvertebrates identified to the family level into calculation of the index. On the basis of results for FHBI, it can not be concluded that fence installation had a positive influence on benthic-macroinvertebrate community structure (as defined by HBI) for the upstream sites; however, FHBI trends for the outlets showed improvement after fence installation at T-1 relative to C-1.

Taxa richness between the generic and family level showed similar results. In general, generic- and family-level taxa richness showed improvement (more species) at the

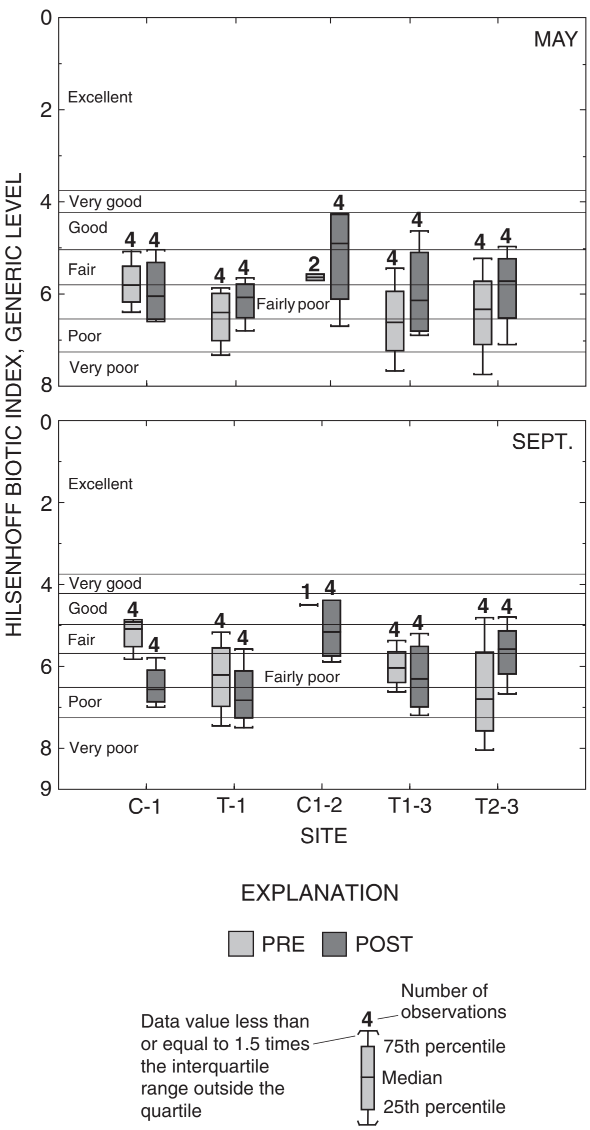


Figure 43. Distribution of Hilsenhoff Biotic Index (generic level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

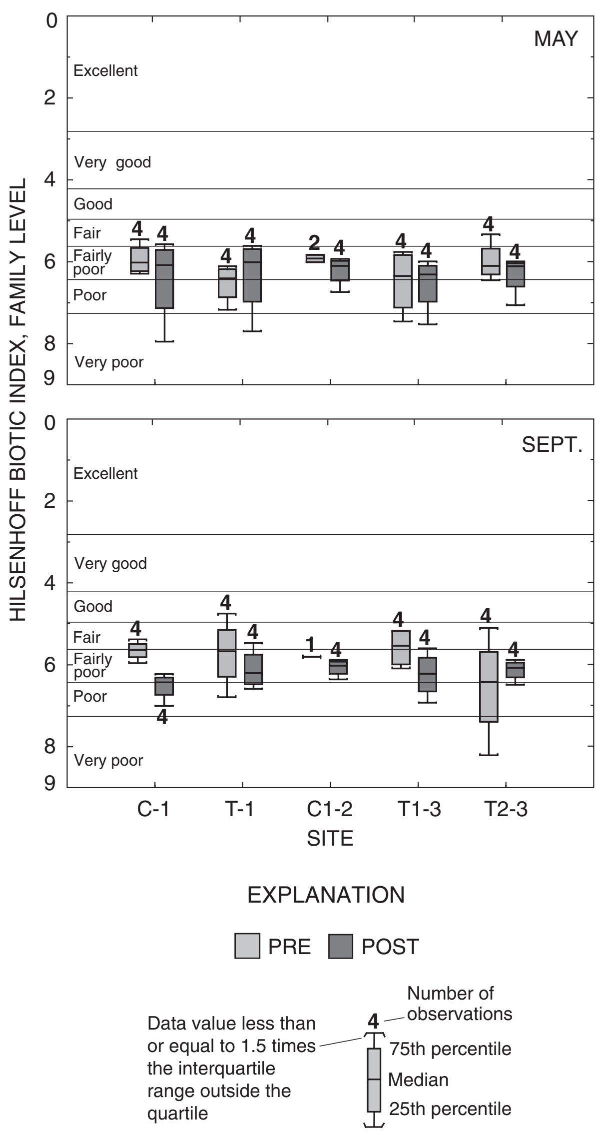


Figure 44. Distribution of Hilsenhoff Biotic Index (family level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

treated relative to control sites from the pre- to post-treatment period. The generic-level data showed more definitive improvements with the overall mean number of species increasing by 3 and 6 at T-1 and T1-3, respectively, from the pre- to post-treatment period (T2-3 showed no change): data for C-1 and C1-2 showed a overall mean increase of 1 species and decrease of 4 species, respectively (fig. 45). The family-level taxa richness data showed overall mean increases from the pre- to post-treatment period at C-1 and T-1 of 3 and 4 species, respectively. Over the same time period for the upstream sites, C1-2 and T2-3 both showed an overall mean decrease in family-level taxa richness of 1 species; T1-3 showed an overall mean increase of 4 species (fig. 46). Therefore, overall, it appeared streambank fencing had a positive influence on the number of species in the benthic-macroinvertebrate community. Stabilization of the riparian zone improved instream qualities, thus allowing better habitat for more taxa.

The mean percentage of Oligochaeta increased at the control sites and decreased at the treated sites from the pre- to post-treatment period (fig. 47). However, it should be noted

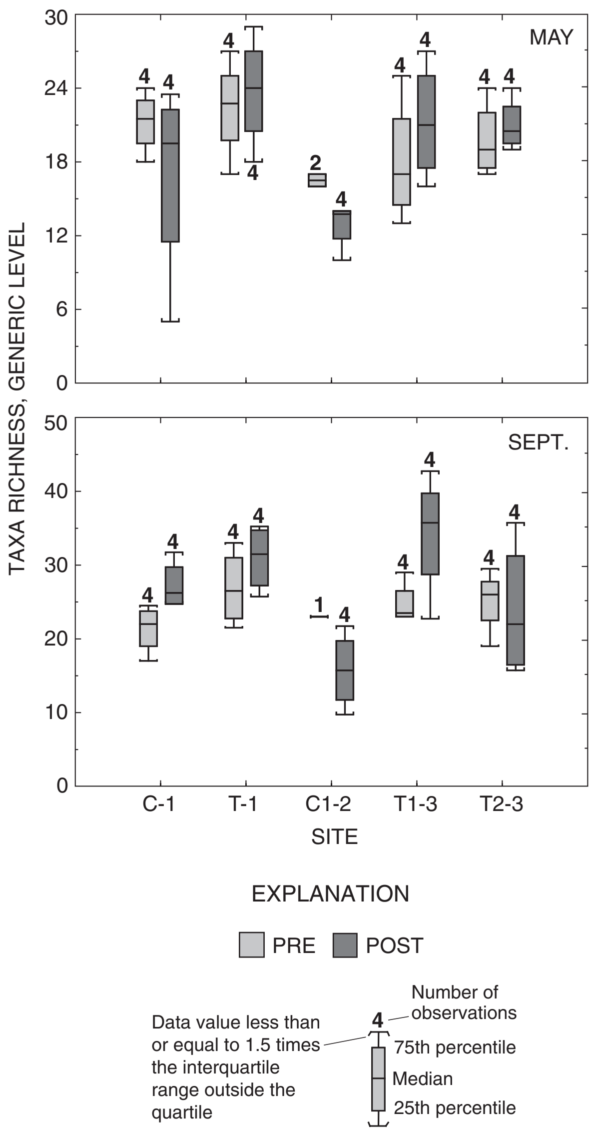


Figure 45. Distribution of taxa richness (generic level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

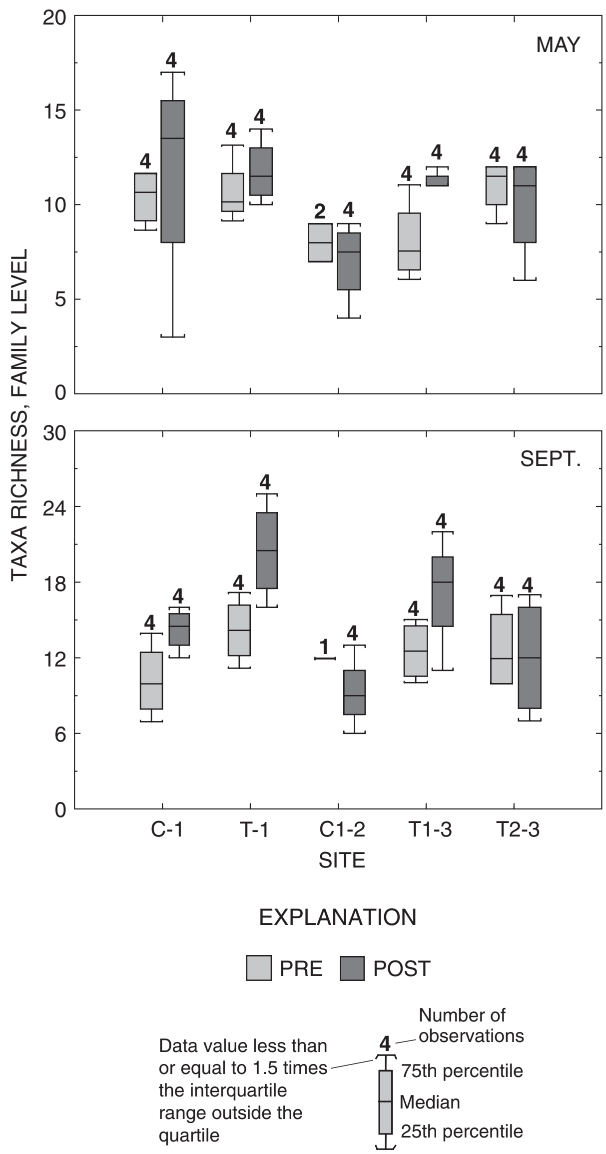


Figure 46. Distribution of taxa richness (family level) for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

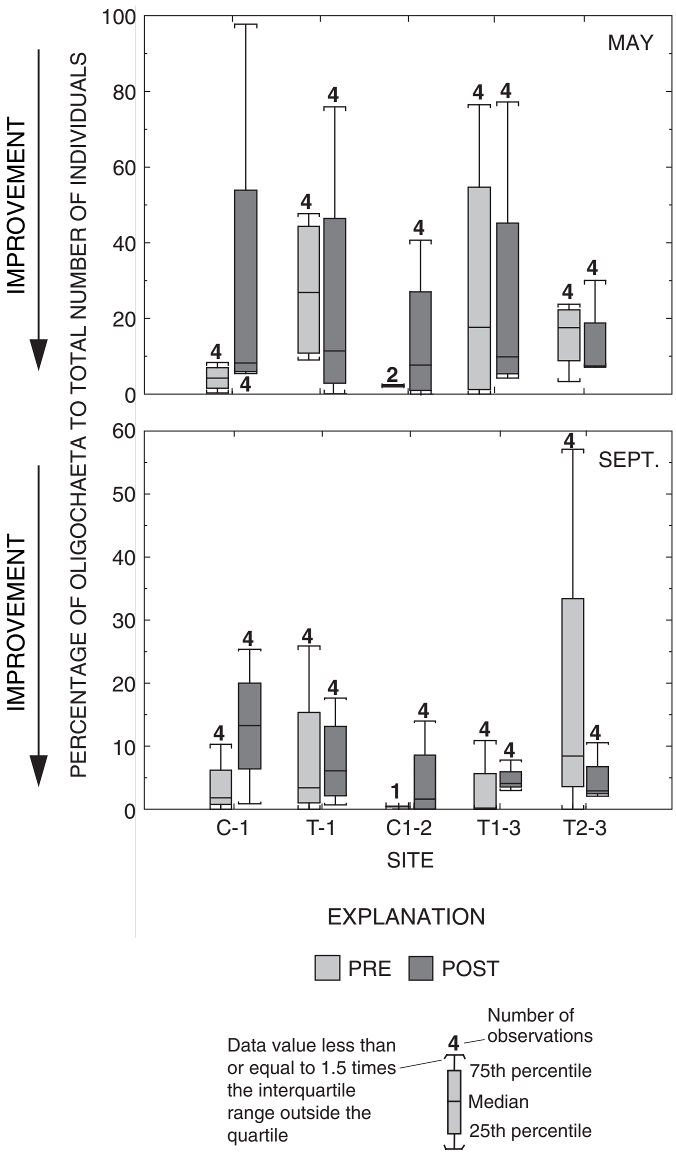


Figure 47. Distribution of the percentage of Oligochaeta to total number of individuals for May and September sampling events at benthic-macroinvertebrate sites for the pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa.

pre-treatment mean values for the percentage of Oligochaetes were lower at the control sites relative to treatment sites by 13 to 16 percent; the pre-treatment means at both control sites were less than 5 percent. Nonetheless, results from this metric indicated the benthic-macroinvertebrate community sampled in the treatment basin had less oligochaetes during the post-treatment period. This was expected because streambank fencing reduced sediment inputs to the stream channel (table 25) and oligochaetes tend to live in stagnant pools or slow-moving water with muddy (or sediment-laden) substrates. Oligochaetes usually become more dominant with increasing organic pollution (Klemm and others, 1990).

A Macroinvertebrate Aggregated Index (MAI) for streams was used to analyze benthic-macroinvertebrate data in a USEPA study in the Pequea and Mill Creek Basins between 1996 and 1998 (J.H. Green and Margaret Passmore, U.S. Environmental Protection Agency, written commun., 2004). All USEPA sites were downstream (that is, in higher order streams) than the sites sampled as part of this project. The USEPA approach was adopted for this study. The USEPA study showed an increase in the quality of the benthic-macroinvertebrate communities in all streams in the Pequea and Mill Creek Basins for the year 1997 relative to 1996, and it was concluded the increase in quality was because of the lower amounts of rainfall over the area in the spring of 1997. Less rainfall would contribute less nutrients and sediments reaching the streams in 1997, and cause less degradation of the streams for that year (J.H. Greene and Margaret Passmore, written commun., 2004). Also, less rainfall also reduced total flow and the lack of significant storm events (relative to previous years) may have helped to maintain benthic-macroinvertebrate communities (as opposed to causing washouts after which time the community would have to reestablish itself). The improvement was still evident in 1998 when the USEPA study was concluded.

In comparison to the USEPA study, data from this study for the May samples showed a somewhat similar trend in the MAI with higher scores in 1997 and 1998 relative to 1996. The mean MAI scores for May data for all five sites for 1996, 1997, and 1998 were 5, 7, and 6, respectively (table 26). The mean MAI score for May decreased at C-1 and C1-2 by 1 and 2 units, respectively, from the pre- to post-treatment

Table 26. Macroinvertebrate Aggregated Index for benthic-macroinvertebrate surface-water sites in the Big Spring Run Basin, Lancaster County, Pa., by station and year for May and September samples.

[–, no data]

Station identifier	1993		1994		1995		1996		1997		1998		1999		2000		2001	
	May	Sep	May	Sep	May	Sep	May	Sep	May	Sep	May	Sep	May	Sep	May	Sep	May	Sep
C-1	–	7	5	11	6	8	7	8	7	9	8	10	0	7	6	11	7	–
T-1	–	10	6	12	3	11	6	9	8	13	7	15	5	10	6	12	7	–
C1-2	–	–	–	–	–	–	3	6	6	3	3	8	4	8	2	2	1	–
T1-3	–	8	5	7	3	6	6	9	6	10	7	9	4	12	3	9	3	–
T2-3	–	6	5	8	3	12	4	10	7	10	4	8	4	5	5	7	3	–

period; however, for the treated sites over the same period, T-1 showed a 1 unit increase, and T1-3 and T2-3 showed no change. Thus, according to May MAI scores, the benthic-macroinvertebrate community structure at the treated sites showed a slight improvement relative to control sites after fence installation.

The September MAI scores also indicated an overall improvement during 1997 and 1998 relative to 1996 (table 26). The mean MAI scores for September data for all five sites for 1996, 1997, and 1998 were 8, 9, and 10, respectively. From the pre- to post-treatment period, the mean MAI score for September increased at C-1 by 1 unit and decreased at C1-2 by 1 unit. The treated sites also showed different trends in MAI scores for September. From the pre- to post-treatment period, mean MAI scores for September increased by 2 units for T-1 and T1-3, but decreased by 1 unit at T2-3 (table 26). Thus, according to September MAI scores, the benthic-macroinvertebrate community structure at the treated sites showed a slight improvement relative to control sites (except for T2-3) after fence installation.

Canonical Correspondence Analysis

The Canonical Correspondence Analysis (CCA) results showed little distinction among the control and treated sites. Variables used in the CCA are listed in table 27. There was not enough variation among the different variables to show a gradient and to make a strong statement about which variables had the most influence on the benthic-macroinvertebrate community.

The first CCA axis for May samples relating habitat and water-quality variables to species data shows water quality as the main driving force influencing benthic-macroinvertebrate communities for these streams. The environmental variables that most influenced the benthic-macroinvertebrate community for the May samples were determined using the CCA downweighted method. The major variables delineating the sites along the first axis are DO (correlation to first axis equal to -0.3785), suspended-sediment concentration (0.2421), and dissolved nitrite (-0.2266) (fig. 48). The eigenvalue of the first axis is 0.2609 and this axis explains 42 percent of the species/environmental relation. Suspended-sediment concentration has an inverse relation to DO and nitrite along the first CCA axis. According to this CCA, sites (and sample dates) that had water-quality data with higher suspended-sediment concentrations should be on the right half of figure 48 (paralleling the vector for suspended-sediment concentration); conversely, sites and sample dates with higher DO and concentrations of dissolved nitrite are on the left.

The first CCA axis for May samples relating habitat and water-quality variables to species data shows that suspended-sediment concentration was negatively correlated to bank stability, bottom substrate available cover (BSAC), bottom

Table 27. May and September detrended correspondence analysis regression p-values for the first axis scores and environmental variables, and variables retained for canonical correspondence analysis for benthic-macroinvertebrate sites located in the Big Spring Run Basin, Lancaster County, Pa.

[CCA, canonical correspondence analysis; Bottom substrate, bottom substrate available cover; (c), concentration]

May			September		
Environmental variable	p-value	Keep for CCA	Environmental variable	p-value	Keep for CCA
Bottom substrate	0.5973	yes	Channel alteration	0.0679	yes
Channel alteration	.5811	yes	Chlorophyll *a* (c)	.3396	yes
Chlorophyll *a* (c)	.2732	yes	Specific conductance	.1135	yes
Specific conductance	.6216	yes	Discharge	.7026	yes
Discharge	.0303	no	Dissolved oxygen (c)	.4479	yes
Dissolved oxygen (c)	.0522	yes	Dissolved phosphorus (c)	.1711	yes
Dissolved phosphorus (c)	.3469	yes	Embeddedness	.9706	yes
Fencing	.7780	yes	Fencing	.2116	yes
Nitrite (c)	.2852	yes	Nitrite (c)	.0945	yes
pH	.3914	yes	pH	.8133	yes
Pool/riffle, run/bend ratio	.0528	yes	Suspended sediment (c)	.5908	yes
Suspended sediment (c)	.5101	yes	Water temperature	.8327	yes
Stream cover	.0337	no	Bank vegetative stability	.0087	no
Water temperature	.7129	yes			
Bank vegetative stability	.7485	yes			

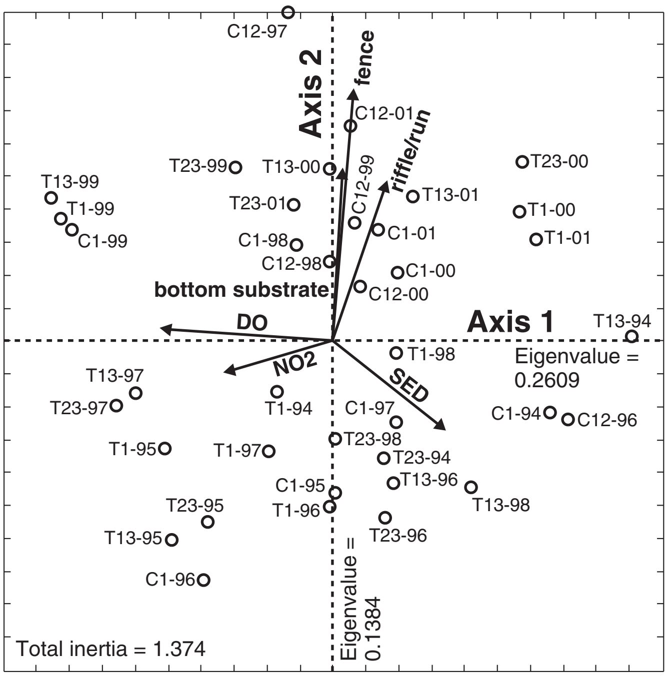


Figure 48. Canonical Correspondence Analysis of habitat and water-quality variables in relation to sites by year for May benthic-macroinvertebrate samples collected in the Big Spring Run Basin, Lancaster County, Pa. [DO, concentration of dissolved oxygen; NO2, concentration of dissolved nitrite as nitrogen; riffle/run, pool/riffle, run/bend ratio; SED, concentration of suspended sediment; points are labelled indicating site and year, so T23-96 was collected at T2-3 in May 1996].

scouring and deposition (BSD), the velocity to depth ratio (VDR), and fencing (table 28). The negative correlation does not necessarily indicate a relation between suspended sediment and the other variables, but it does indicate that, for example, if suspended-sediment concentrations were higher, bank stability scores were lower. Thus, lower suspended-sediment concentrations (at the time of benthic-macroinvertebrate sampling) were correlated with higher values for habitat variables that would tend to improve with streambank fencing. The correlation of suspended-sediment concentration to VDR was likely produced by the relation of VDR to discharge because suspended-sediment concentrations were negatively correlated to stream discharge, and VDR showed relations to stream discharge.

The first CCA axis for May samples relating habitat and water-quality variables to species data shows that DO and the concentration of dissolved nitrite were also important water-quality constituents (fig. 48). Unlike the concentration of suspended sediment, DO and the concentration of dissolved nitrite were negatively correlated with stream discharge (table 28). The vector directions for DO and the concentration of dissolved nitrite on figure 48 are generally opposite of the direction for the concentration of suspended sediment. According to the CCA, concentrations of dissolved nitrite at T-1 had less of an effect on the benthic-macroinvertebrate community over time after the fence installation. T1-3 and T2-3 showed no definite change in concentrations of dissolved nitrite after fence installation. The control sites were dispersed (showed no definitive relation to explanatory variables) before and after fence installation for the CCA plot relating May species data to habitat and water-quality variables.

The second CCA axis for May samples relating habitat and water-quality variables to species data was correlated to status of the fencing (0.5382), the availability of habitable bottom substrate (0.3562) for the invertebrates, and the pool riffle/run bend ratio (0.3348) (fig. 48). The eigenvalue of this axis was 0.1384. Another 22 percent of the environmental/species relation can be explained by this axis for a total of 64 percent with both axes. This axis pertained more to the morphology of the streams compared to the first axis being more water quality oriented.

The CCA plot for May samples relating habitat and water-quality variables to species data shows that the majority of the sites (and sample events) below the center horizontal line (axis 2) were sites sampled prior to fence installation (but in both the treatment and control basins) (fig. 48), indicating that fence installation had a positive effect on the benthic-macroinvertebrate community. Those sites above this line were post-treatment sites (in both the control and treated basins), except for one (the sample collected at T1-3 in May 1994). Inputs of suspended sediment during the post-treatment period were reduced relative to pre-treatment period (see table 23 for suspended-sediment yields for T-1 and C-1). As suspended-sediment inputs decreased, BSAC scores rose as well as the pool/riffle, run/bend ratio. With less sediment covering the rocks, the bottom substrate was exposed and created more riffle-type environment rather than a pooled environment. The most desirable habitat type generally is rock and gravel (Plafkin and others, 1989). The pool/riffle, run/bend ratio is used as a key channel morphology feature because it is assumed a stream with riffles provides more diverse habitat for the benthic macroinvertebrates than a straight run or pooled environment (Plafkin and others, 1989).

The environmental variables that most influenced the benthic-macroinvertebrate community for the September samples were also determined using the CCA downweighted method. The variables used in the September analysis can be found in table 27. The first axis of the CCA plot for the September samples correlated with several variables: dissolved nitrite (0.6028), dissolved phosphorus (-0.5115), channel alteration (-0.3513), SC (0.2035), and fencing (-0.1731) (fig. 49). This axis explained 32 percent of the site and species relation to the habitat variables. Channel alteration was the only habitat variable that influenced the benthic macroinvertebrates; the other significant variables were water-quality constituents. This is different from the CCA plot for May samples, which

Table 28. Canonical Correspondence Analysis variables for May and September samples and their correlated variables with sign of correlation for benthic-macroinvertebrate sites located in the Big Spring Run Basin, Lancaster County, Pa.

[CCA, Canonical Correspondence Analysis; +, positive correlation; -, negative correlation; Bottom substrate, bottom substrate available cover; DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen; (c), concentration]

May		September	
CCA environmental variable	**Correlated variable/sign**	**CCA environmental variable**	**Correlated variable/sign**
Bottom substrate	Specific conductance + Embeddedness + Bottom scouring and deposition + Suspended sediment (c) - Velocity/depth ratio +	Channel alteration	Dissolved nitrate (c) - Bottom scouring and deposition + Stream cover +
Channel alteration	Bank stability + Pool/riffle, run/bend ratio + Velocity/depth ratio +	Embeddedness	Bank stability + Bottom substrate + Dissolved nitrite (c) -
Pool/riffle, bend/run ratio	Channel alteration + Embeddedness + Velocity/depth ratio +	Fencing	Specific conductance - Stream discharge - DKN (c) + Dissolved phosphorus (c) + Dissolved nitrate (c) + Pool/riffle, run/bend ratio - TKN (c) + Total phosphorus (c) + Velocity/depth ratio -
Vegetative bank stability	Bank stability + Dissolved oxygen (c) - Stream cover + Water temperature -	Chlorophyll *a* (c)	Bottom substrate - Specific conductance -
Fencing	Bank stability + DKN (c) + Dissolved phosphorus (c) + Dissolved nitrate (c) - Suspended sediment (c) - TKN (c) + Total phosphorus (c) +	Stream discharge	Specific conductance + DKN (c) - Fencing - Pool/riffle, run/bend ratio + Water temperature - TKN (c) -
Chlorophyll *a* (c)	Embeddedness -	Dissolved oxygen (c)	Dissolved phosphorus (c) - Dissolved nitrate (c) + Pool/riffle, run/bend ratio +
Dissolved oxygen (c)	Stream discharge - Dissolved nitrate (c) + pH + Water temperature + TKN (c) - Total phosphorus (c) - Bank vegetative stability -	pH	TKN (c) - Total phosphorus (c) -
pH	Bank stability - Stream discharge - Dissolved oxygen (c) + Dissolved nitrate (c) + Bottom scouring and deposition - Water temperature +	Specific conductance	Bottom substrate + Chlorophyll *a* (c) - Stream dischage + Fencing - TKN (c) -
Specific conductance	Bottom substrate + DKN (c) -	Water temperature	Stream discharge -

Table 28. Canonical Correspondence Analysis variables for May and September samples and their correlated variables with sign of correlation for benthic-macroinvertebrate sites located in the Big Spring Run Basin, Lancaster County, Pa.—Continued

[CCA, Canonical Correspondence Analysis; +, positive correlation; -, negative correlation; Bottom substrate, bottom substrate available cover; DKN, dissolved ammonia plus organic nitrogen; TKN, total ammonia plus organic nitrogen; (c), concentration]

May		September	
CCA environmental variable	**Correlated variable/sign**	**CCA environmental variable**	**Correlated variable/sign**
Water temperature	Bank stability - Stream discharge - Dissolved oxygen (c) + Dissolved phosphorus (c) - Dissolved nitrate (c) + pH + Bank vegetative stability -	Dissolved nitrite (c)	DKN (c) + Embeddedness - Pool/riffle, run/bend ratio - TKN (c) + Total phosphorus (c) + Velocity/depth ratio -
Dissolved nitrite (c)	Stream discharge - Dissolved nitrate (c) + TKN (c) - Total phosphorus (c) -	Dissolved phosphorus (c)	Dissolved oxygen (c) - DKN (c) + Dissolved nitrate (c) - Fencing + TKN (c) + Total phosphorus (c) +
Dissolved phosphorus (c)	Bank stability + DKN (c) + Fencing + Dissolved nitrate (c) - Bottom scouring and deposition + Water temperature - TKN (c) + Total phosphorus (c) +	Suspended sediment (c)	Bank vegetative stability - Stream cover - Turbidity + Chlorophyll *a* (c) +
Suspended sediment (c)	Bank stability - Bottom substrate - Stream discharge + Fencing - Bottom scouring and deposition - Velocity/depth ratio -		

showed only water-quality constituents influencing the first CCA axis (fig. 48).

For September samples, sites and sampling events with higher concentrations of dissolved nitrite and SC fell to the right side and sites and sampling events with higher concentrations of dissolved P fell to the left of the CCA graph (fig. 49). September samples collected in the treatment basin after fence installation were generally on the left side of the graph, indicating concentrations of dissolved P increased after fencing was installed. Concentrations of dissolved P were higher for benthic-macroinvertebrate samples collected during the post-treatment period in the treatment basin (fig. 26).

The second axis of the CCA plot for September samples relating sites and sampling events to environmental variables was correlated with DO (0.5069) and discharge (0.4798) (fig. 49). The first and second axes explained 53 percent of the environmental/species relation for the September CCA plot. Less variation was explained for the first two axes of the September CCA than for the May CCA (64 percent). The correlation of stream discharge and DO to the second CCA axis for September data indicate these water-quality constituents were somewhat correlated. The box plots for benthic-macroinvertebrates samples for stream discharge (fig. 14) and DO (fig. 33) showed that pre-treatment stream discharge and DO at all five sites were generally higher during the pre- relative to the post-treatment period, and review of the site and sample point locations on figure 49 indicate the majority of points above axis 1 are post-treatment samples. This would correspond with the location of the vectors along the second axis for stream discharge and DO.

The benthic macroinvertebrates collected at the five sites in May were typical of mid- to pollution-tolerant families (Barbour and others, 2002), with 22 and 14 families, respectively, for the two different tolerant levels. Only one intolerant family, Heptageniidae, was identified at T-1 in May 1997, with only one individual in the sample. More of the families that require higher DO, less sedimentation, and are tolerant of organic pollution, such as Hygrobatidae and Ceratopogonidae, fell to the left side of the CCA plot (fig. 50). Those families moderately tolerant of some sedimentation, organic pollution, and moderate levels of DO, such as Dytiscidae and Elmidae, fell more at the center of the graph. Those families that live in the sediments, can tolerate low DO levels, and are moderately tolerant of organic pollution, such as Naididae and Caenidae, were on the right side of the graph. The mid-pollution families are well distributed among all sites whether fenced or not. The more tolerant families were fewer in number at the top of the graph where fencing is a strong variable (fig. 50). As the banks became more vegetated, shredders (such as Tipulidae and Haliplidae) moved into the fenced sites; otherwise, most of the families present on the CCA plot are collector/gatherers (such as Asellidae and Baetidae), scrapers (such as Heptageniidae and Psephenidae), and predators (such as Hygrobatidae and Lebertiidae).

The species present in streams for September samples were mostly semi-tolerant to very tolerant of organic enrichment. Looking at the benthic macroinvertebrates, there really was no discernible difference between pre- and post-treatment samples (fig. 51). The lower half (below the first axis) of the CCA plot contained those benthic macroinvertebrates that can tolerate low DO or are facultative. The changes for September samples seen from the pre- to post-treatment period in benthic-macroinvertebrate community structure were more reflective of the habitat at each site.

The incorporation of 1-month load data for nutrients and suspended sediment along with instantaneous data into the CCA (for this CCA, only C-1 and T-1 were analyzed because the load data were not available for upstream sites) conducted on May samples to relate habitat and water-quality variables to species data showed the first axis of the CCA plot was

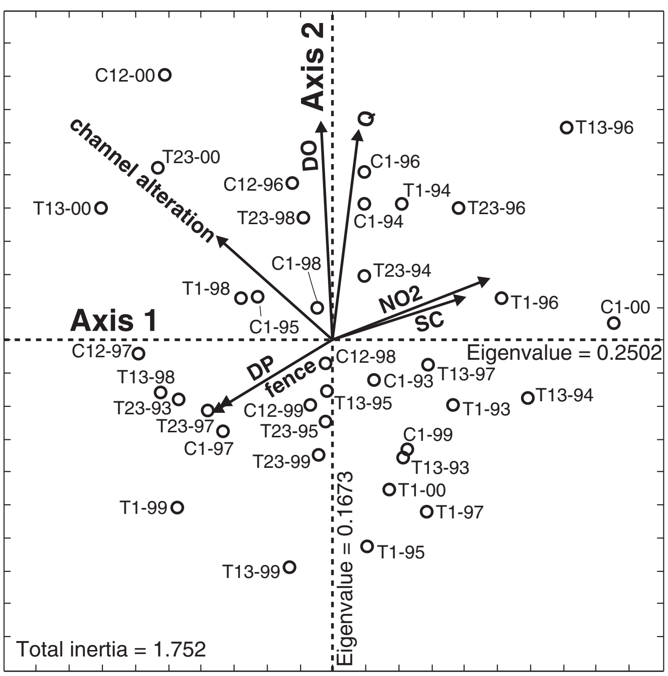
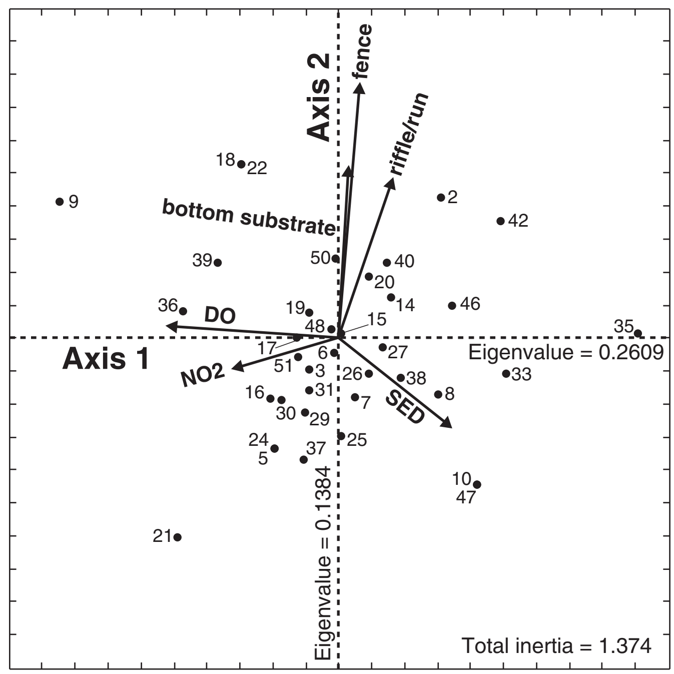


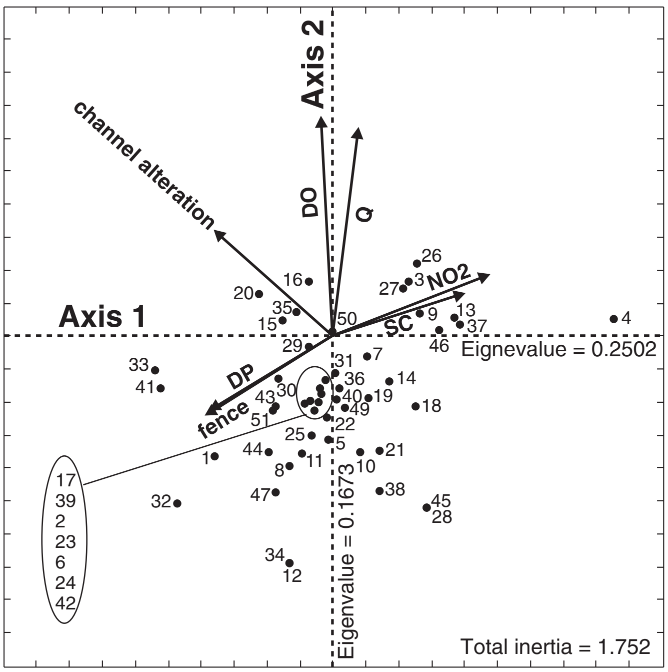
Figure 49. Canonical Correspondence Analysis of habitat and water-quality variables in relation to sites by year for September benthic-macroinvertebrate samples collected in the Big Spring Run Basin, Lancaster County, Pa. [DO, concentration of dissolved oxygen; DP, concentration of dissolved phosphorus; NO_2, concentration of dissolved nitrite as nitrogen; Q, stream discharge; SC, specific conductance; points are labelled indicating site and year, so T23-96 was collected at T2-3 in May 1996].



1, Aeshnidae	18, Ephydridae	35, Muscidae
2, Asellidae	19, Erpobdellidae	36, Naididae
3, Baetidae	20, Gammaridae	37, Ostracoda
4, Bivalvia	21, Glossiphoniidae	38, Physidae
5, Caenidae	22, Haliplidae	39, Pisidiidae
6, Ceratopogonidae	23, Haplotaxidae	40, Planariidae
7, Chironomidae	24, Heptageniidae	41, Planorbidae
8, Coenagrionidae	25, Hydrophilidae	42, Psephenidae
9, Copepoda	26, Hydropsychidae	43, Psychodidae
10, Corixidae	27, Hydroptilidae	44, Scirtidae
11, Culicidae	28, Hydrozetidae	45, Sialidae
12, Dryopidae	29, Hygrobatidae	46, Simuliidae
13, Dugesiidae	30, Isotomidae	47, Siphlonuridae
14, Dytiscidae	31, Lebertiidae	48, Sperchonidae
15, Elmidae	32, Libellulidae	49, Tabanidae
16, Empididae	33, Lumbriculidae	50, Tipulidae
17, Enchytraeidae	34, Lymnaeidae	51, Tubificidae

Figure 50. Canonical Correspondence Analysis of habitat and water-quality variables in relation to family level taxa for May benthic-macroinvertebrate samples collected in the Big Spring Run Basin, Lancaster County, Pa. [DO, concentration of dissolved oxygen; NO_2, concentration of dissolved nitrite as nitrogen; riffle/run, pool/riffle, run/bend ratio; SED, concentration of suspended sediment].

most highly correlated with instantaneous stream discharge (-0.4798) and embeddedness (-0.3881). The discharge at T-1 for the May samples was consistently less than C-1 (see fig. 14), which is noted by most of the T-1 sites falling further right on the graph than the C-1 sites. The two sites and samples (T1-99 and C1-99) farther to the right on the CCA plot were caused by the drought in 1999, when stream discharges at both sites were lower than normal (see fig. 7). The first axis of this May CCA plot explained 49 percent of the environmental/species relation.



1, Aeshnidae
2, Asellidae
3, Baetidae
4, Bivalvia
5, Caenidae
6, Ceratopogonidae
7, Chironomidae
8, Coenagrionidae
9, Copepoda
10, Corixidae
11, Culicidae
12, Dryopidae
13, Dugesiidae
14, Dytiscidae
15, Elmidae
16, Empididae
17, Enchytraeidae
18, Ephydridae
19, Erpobdellidae
20, Gammaridae
21, Glossiphoniidae
22, Haliplidae
23, Haplotaxidae
24, Heptageniidae
25, Hydrophilidae
26, Hydropsychidae
27, Hydroptilidae
28, Hydrozetidae
29, Hygrobatidae
30, Isotomidae
31, Lebertiidae
32, Libellulidae
33, Lumbriculidae
34, Lymnaeidae
35, Muscidae
36, Naididae
37, Ostracoda
38, Physidae
39, Pisidiidae
40, Planariidae
41, Planorbidae
42, Psephenidae
43, Psychodidae
44, Scirtidae
45, Sialidae
46, Simuliidae
47, Siphlonuridae
48, Sperchonidae
49, Tabanidae
50, Tipulidae
51, Tubificidae

Figure 51. Canonical Correspondence Analysis of habitat and water-quality variables in relation to family level taxa for September benthic-macroinvertebrate samples collected in the Big Spring Run Basin, Lancaster County, Pa. [DO, concentration of dissolved oxygen; DP, concentration of dissolved phosphorus; NO_2, concentration of dissolved nitrite as nitrogen; Q, stream discharge; SC, specific conductance].

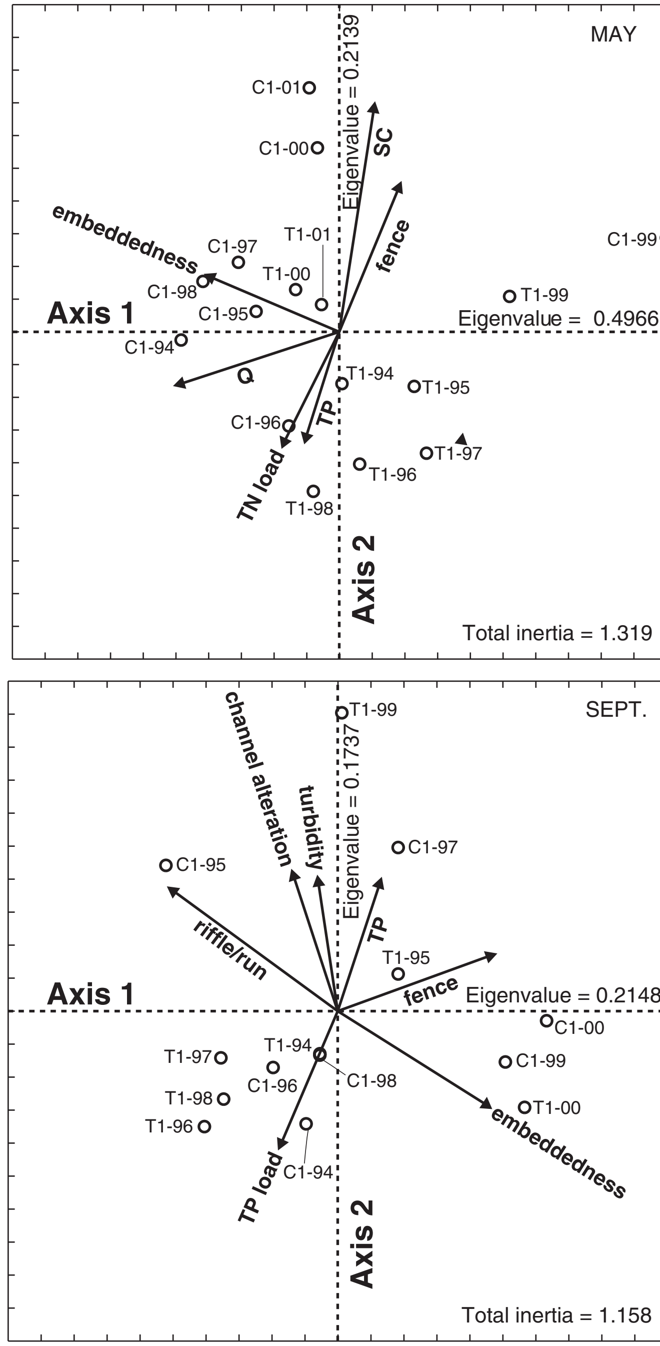


Figure 52. A Canonical Correspondence Analysis graph showing the relation of sites C-1 and T-1 to habitat and water quality using monthly mean nutrient, sediment, and discharge data for the one month prior to the May and September benthic-macroinvertebrate sampling events in the Big Spring Run Basin, Lancaster County, Pa. [Q, stream discharge; riffle/run, pool/riffle, run/bend ratio; SC, specific conductance; TN, concentration of total nitrogen; TP, concentration of total phosphorus; points are labelled indicating site and year, so T1-96 was collected at T-1 in 1996].

The second axis of the plot for CCA that included May 1-month load data for nutrients and suspended sediment along with instantaneous data separated sites on the basis of SC (0.6749), the presence of fencing (0.4360), the 1-month load of total N (-0.3302), and concentration of total P (-0.3145) (fig. 52). The second axis indicates nutrient levels are higher at T-1 than C-1, and this was likely because of increased concentrations of total P at T-1 during the post-treatment period (fig. 26). However, for T-1 and C-1, the CCA plot indicates post-treatment samples appeared to be less concentrated with nutrients than pre-treatment samples, and this could be because of the significant decrease at T-1 and C-1 in the instantaneous yield of total N for low-flow samples (fig. 8g). The percentage of the environmental/species relation explained by the second axis is 21 percent. The first and second axes together explain 70 percent of the environmental/species relations at C-1 and T-1.

The incorporation of 1-month load data for nutrients and suspended sediment along with instantaneous data into the CCA conducted on September samples to relate habitat and water-quality variables to species data showed that the physical habitat of the streams was more important than nutrient concentrations, which is similar to the May CCA that also used the 1-month load and instantaneous data (fig. 52). The first axis of the September plot for CCA (with loads incorporated) explained 23 percent of the site and species relations to the habitat variables. Along the first axis, the sites separated based on the presence of the pool/riffle, run/bend ratio (-0.4952), fencing (0.4598), and embeddedness (0.4528).

The second axis of the plot for CCA that included September 1-month load data for nutrients and suspended sediment along with instantaneous data separated sites on the basis of channel alteration, turbidity, and total P (both concentrations and loads) (fig. 52). The second axis explained 19 percent of the site and species relations to the habitat and water-quality variables. Together, the first and second axes explained 42 percent of the environmental/species relations seen at C-1 and T-1. Interestingly, the vectors for instantaneous concentration of total P and monthly load of total P separated the sites differently. Instantaneous concentration of total P (0.3817), channel alteration (0.4099), and turbidity (0.3900) all pull the sites to the top of the CCA plot and monthly load of total P (-0.3964) pulls sites to the bottom of the CCA. The reason for this was that the highest 1-month loads of total P at T-1 and C-1 for the September samples were during 1994 and 1996. Most of load for total P during the months prior to September 1994 and 1996 were caused by storm events. The concentrations of total P for the 1994 and 1996 September samples were the lowest concentrations for all the September samples; thus, there was basically an inverse relation between 1-month loads of total P and instantaneous concentrations of total P. This would explain the vectors for concentration of total P and 1-month load of total P going in opposite directions.

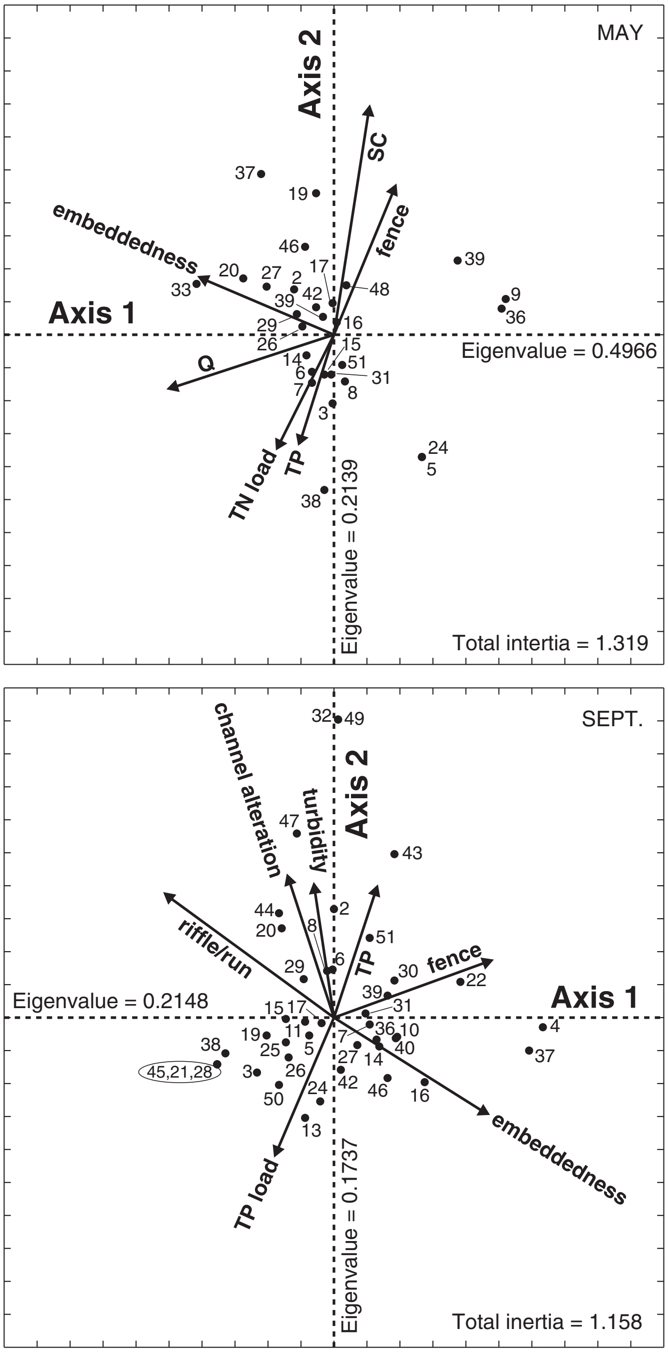
The May CCA plot that relates family-level taxa to habitat and water-quality variables (with 1-month load data included) has benthic macroinvertebrates needing a less embedded bottom substrate on the right half of the plot and those that can tolerate a little more embeddedness and higher discharge on the left of the plot (fig. 53). When looking at the benthic macroinvertebrates, the more pollution-sensitive invertebrates were near the middle of the CCA plot and away from the highest nutrient concentrations and loads. Although none of the benthic macroinvertebrates on the plot are pollution sensitive, a slight gradient from those that are tolerant (such as Caenidae) to moderately tolerant (such as Hygrobatidae) can be seen.

The September CCA plot that relates family-level taxa to habitat and water-quality variables (with 1-month load data included) showed benthic macroinvertebrates tolerant of higher embeddedness and less riffle turbulence to the right of the CCA (fig. 53), and benthic macroinvertebrates requiring less turbid water towards the bottom (below the first axis) of the CCA plot. This CCA for September samples showed that habitat was more influential than nutrient concentrations and loads on the presence of a species at a particular site.

The incorporation of 3-month load data for nutrients and suspended sediment along with instantaneous data into the CCA conducted on May data showed results similar to CCA for May that used the 1-month load data. Instantaneous discharge (-0.4342) was most important variable affecting the first axis along with the 3-month load for TKN (-0.2958), pool/riffle, run/bend ratio (-0.2851), and the velocity to depth ratio (-0.2295) (fig. 54). As with the CCA using 1-month load data for May samples, discharge was still the most important variable for separating these sites. Again, the drought of 1999 is very visible because the samples for 1999 are to the far right of the plot, which would indicate these samples were associated with relatively low flows. The first axis explained 42.5 percent of the environmental/species relation.

The second axis for the CCA for May with tri-monthly load data incorporated was dominated by the 3-month load for TKN (-0.4946) (fig. 54). Bank stability (-0.1937) was a minor influence compared to the 3-month TKN load (and this was evident on the basis of the length of the vector for bank stability compared to the vector length for the 3-month TKN load). Fencing was not a major separating variable on the basis of this CCA. T-1 samples showed higher 3-month loads of TKN than C-1 samples, which was verified from the 3-month TKN load data that showed four of the six highest TKN loads for May samples (if T-1 and C-1 data are combined) were at T-1.

The CCA plot for September samples using the 3-month load data showed the first axis was correlated to stream temperature (0.3886), velocity to depth ratio (-0.3874), concentration of chlorophyll *a* (0.3846), suspended-sediment concentration (0.3277), and 3-month load of TKN (-0.2417) (fig. 54). The first axis explained 22 percent of the site and species relations to the habitat variables. The CCA for September samples using the 3-month load data showed that C-1 samples were generally on the left side and T-1 to the right side of the plot. Clustering of C-1 along the first axis to the left of the second axis and T-1 to the right of the second axis was likely



1, Aeshnidae
2, Asellidae
3, Baetidae
4, Bivalvia
5, Caenidae
6, Ceratopogonidae
7, Chironomidae
8, Coenagrionidae
9, Copepoda
10, Corixidae
11, Culicidae
12, Dryopidae
13, Dugesiidae
14, Dytiscidae
15, Elmidae
16, Empididae
17, Enchytraeidae
18, Ephydridae
19, Erpobdellidae
20, Gammaridae
21, Glossiphoniidae
22, Haliplidae
23, Haplotaxidae
24, Heptageniidae
25, Hydrophilidae
26, Hydropsychidae
27, Hydroptilidae
28, Hydrozetidae
29, Hygrobatidae
30, Isotomidae
31, Lebertiidae
32, Libellulidae
33, Lumbriculidae
34, Lymnaeidae
35, Muscidae
36, Naididae
37, Ostracoda
38, Physidae
39, Pisidiidae
40, Planariidae
41, Planorbidae
42, Psephenidae
43, Psychodidae
44, Scirtidae
45, Sialidae
46, Simuliidae
47, Siphlonuridae
48, Sperchonidae
49, Tabanidae
50, Tipulidae
51, Tubificidae

Figure 53. A Canonical Correspondence Analysis graph showing the relation of macroinvertebrates at C-1 and T-1 to habitat and water quality using monthly mean nutrient, sediment, and discharge data for the one month prior to the May and September benthic-macroinvertebrate sampling events in the Big Spring Run Basin, Lancaster County, Pa. [Q, stream discharge; riffle/run, pool/riffle, run/bend ratio; SC, specific conductance; TN, concentration of total nitrogen; TP, concentration of total phosphorus].

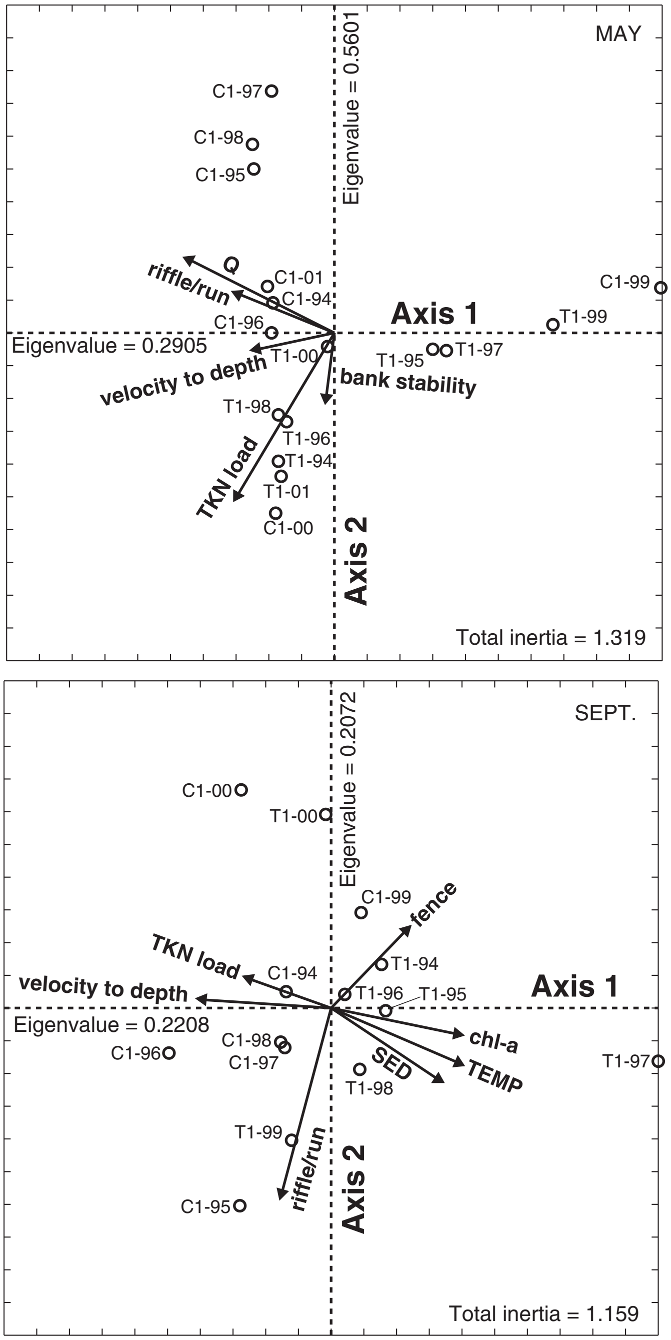


Figure 54. A Canonical Correspondence Analysis graph showing the relation of sites C-1 and T-1 to habitat and water quality using tri-monthly mean nutrient, sediment, and discharge data for the three months prior to the May and September benthic-macroinvertebrate sampling events in the Big Spring Run Basin, Lancaster County, Pa. [chl-a, concentration of chlorophyll *a*; Q, stream discharge; riffle/run, pool/riffle, run/bend ratio; SED, concentration of suspended sediment; TEMP, water temperature; TKN, concentration of total ammonia plus organic nitrogen as nitrogen; velocity to depth, velocity to depth ratio; points are labelled indicating site and year, so T1-96 was collected at T-1 in 1996].

caused by lower stream temperatures at C-1 (fig. 30) and a better velocity to depth ratio at C-1 (fig. 13).

The second axis of the CCA plot for September samples using 3-month load data was correlated to the pool riffle/run bend ratio (-0.5611) and the presence of fencing (0.2365) (fig. 54). The second axis explained 21 percent of the environmental/species relations. Both axes together explained 43 percent of the environmental/species relations. The greater the riffle area in the stream, the lower (below the first axis) the site was on the CCA plot. Even though the presence of fencing was significant along the second axis, T-1 samples during the post-treatment period did not appear to be clustered along the fencing vector.

The May CCA plot that relates family-level taxa to habitat and water-quality variables (with 3-month load data included) showed benthic macroinvertebrates separated out as those that live in faster (higher stream discharge) areas with deep riffles and pools and those that live in slower areas with only occasional riffles (fig. 55). Along the second axis, the benthic-macroinvertebrates separated on the 3-month TKN load and stabilization of the streambanks. The 3-month TKN loads for May samples showed that the four highest values (if combining C-1 and T-1 data) were for T-1; thus, benthic-macroinvertebrate families along the top of the vector for TKN should be more prevalent at T-1. The two families near the top of the 3-month load vector for TKN are Psephenidae and Erpobdellidae, and these were only identified at T-1 for May samples.

The September CCA plot that relates family-level taxa to habitat and water-quality variables (with 3-month load data included) generally showed the more intolerant benthic macroinvertebrates were to the right along the first axis of the CCA plot (fig. 55). The location of the intolerant families on the right corresponds with figure 54 and the location of samples collected at T-1. Thus, this CCA plot indicates site T-1 had more intolerant families for September samples than C-1. The four families (Physidae, Hydrozetidae, Glossiphoniidae, and Sialidae) to the far right of the CCA plot for September were only identified at T-1. Little variance in tolerance was seen along the second axis. Along the second axis, benthic macroinvertebrates ranged from those found in good quality riffles and pools (at the bottom of the CCA plot) to those able to tolerate poorer quality of riffles and pools towards the middle and top of the CCA plot.

Environmental variables that had the most significant effect on species data according to the CCA analysis were the presence or absence of the fence, dissolved nitrite and DO concentrations, and stream discharge. These first three variables were significant for CCA with all five sites included. Stream discharge was most important for CCA when only T-1 and C-1 data were included. These environmental variables, along with others used in CCA, were correlated with other environmental variables (table 28).

The fencing in the treatment basin was correlated with a number of environmental variables for the May and September samples (table 28). Fence installation in May and September

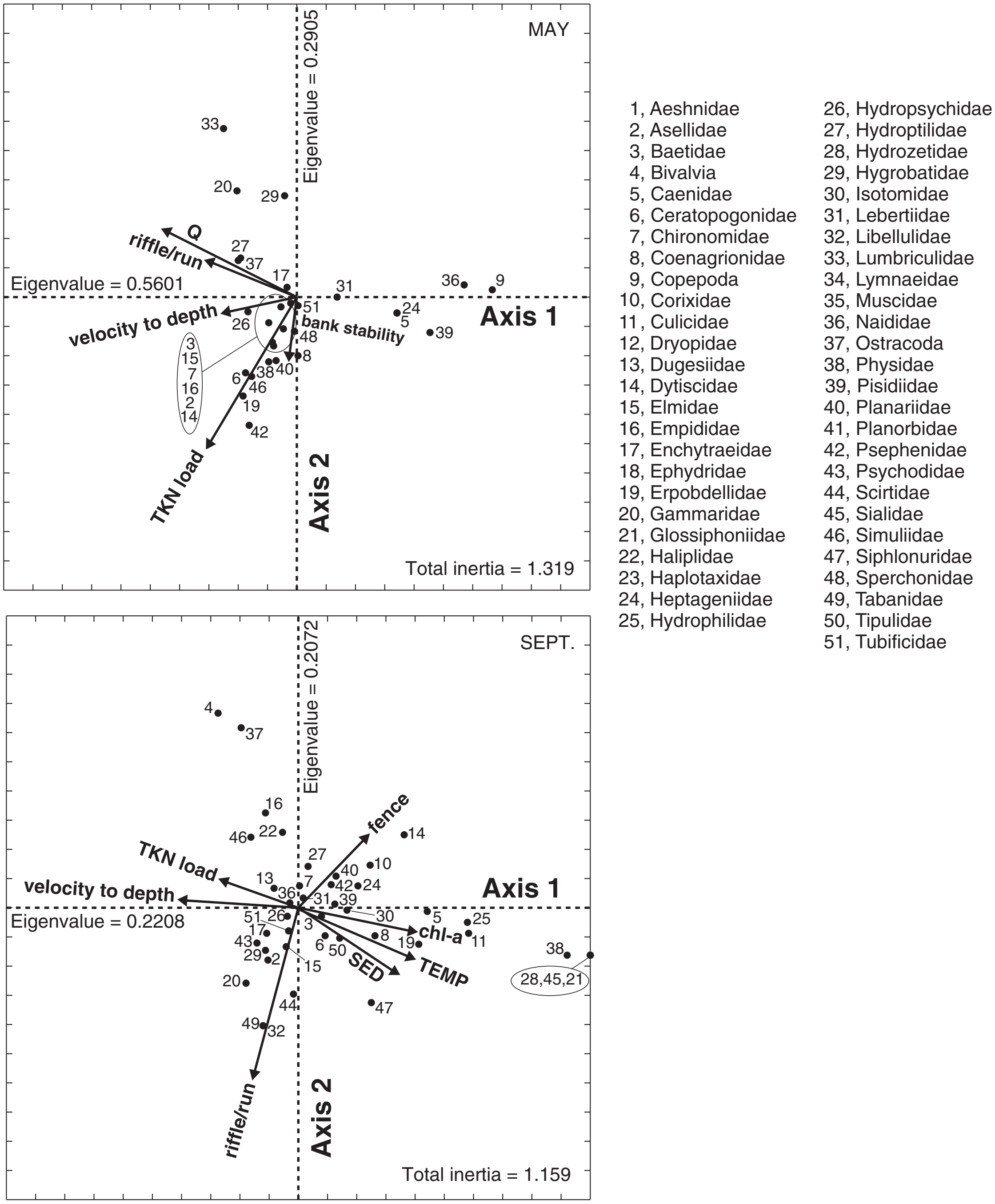


Figure 55. A Canonical Correspondence Analysis graph showing the relation of macroinvertebrates at C-1 and T-1 to habitat and water quality using tri-monthly mean nutrient, sediment, and discharge data for the three months prior to the May and September benthic-macroinvertebrate sampling events in the Big Spring Run Basin, Lancaster County, Pa. [chl-a, concentration of chlorophyll *a*; Q, stream discharge; riffle/run, pool/ riffle, run/bend ratio; SED, concentration of suspended sediment; TEMP, water temperature; TKN, concentration of total ammonia plus organic nitrogen as nitrogen; velocity to depth, velocity to depth ratio].

was positively correlated with DKN, TKN, and the concentration of dissolved and total P (table 28). All four of these nutrient constituents showed increased median values for May and September benthic-macroinvertebrate samples during the post-treatment period in the treatment basin (figs. 23-26). It should be noted that low-flow samples collected at T-1 during the post-treatment period did not show increased concentrations of TKN and DKN (fig. 8c) relative to the pre-treatment period. Fencing was also positively correlated with bank stability and negatively correlated with the concentration of suspended sediment for May samples, and these follow from the data presented. Fencing was negatively correlated with stream discharge, which was likely caused by the decrease in stream discharge from the pre- to post-treatment period at T-1 (table 18).

Dissolved-nitrite concentrations were consistently low during benthic-macroinvertebrate sampling even as peak concentrations occurred in 1998. Nitrite can be toxic to macroinvertebrates, fish, and humans at concentrations much higher than those seen at the fencing sites (Pit and others, 1999). Nitrite binds with the hemoglobin present in the blood causing suffocation because of insufficient oxygen in the blood (U.S. Environmental Protection Agency, 2002b). In fish, severe damage to the gills, liver, spleen, kidneys, and the nervous system can occur (PondGuy, 2004; Algone.com, 2004). At 0.55 mg/L of nitrite N, yearling rainbow trout suffered a 55 percent mortality rate after a 24-hour exposure period; exposure to 0.15 mg/L of nitrite N in 48 hours resulted in no deaths (Pit and others, 1999). Several grab samples collected during benthic-macroinvertebrate sampling had concentrations of nitrite N close to 0.15 mg/L, but only one sample exceeded 0.15 mg/L (0.2 mg/L of nitrite as N was measured at T1-3 during September 1996).

DO concentrations were important in CCA. DO concentrations decreased from the pre- to post-treatment period for all sites (fig. 33). The negative correlation of DO to stream discharge for May samples (table 28) indicates the lower DO concentrations during the post-treatment period were partially related to lower stream discharge over the same period. It stands to reason that respiration of benthic macroinvertebrates would be affected by lower DO concentrations. These DO changes from the pre- to post-treatment period were not related to fence installation because control and treatment sites showed similar trends.

Stream discharge, although not significant for all the CCA that included the five sites, obviously had a significant effect on benthic macroinvertebrates. Stream discharge affects most of the habitat parameters that were qualitatively assessed using RBP III protocols. The stream discharge also affects the transport of nutrients and suspended sediment through the stream system. Storm events prior to benthic-macroinvertebrate sampling could also have played an important role in benthic-macroinvertebrate community structure that was identified because higher discharge during storm events can displace macroinvertebrates and redistribute them throughout the stream network.

Summary

Overall, the nine habitat characteristics that were qualitatively assessed during benthic-macroinvertebrate sampling indicated that habitat did improve for benthic macroinvertebrates after fence installation in the treatment basin. In reviewing the primary (microscale), secondary (macroscale), and tertiary (bank and riparian zone) categories for the habitat characteristics, the most noticeable effect was for the tertiary category (bank stability, vegetative stability, and streamside cover). These three characteristics showed improvement at the treatment sites relative to control sites from the pre- to post-treatment period (table 29). Streamside-cover scores were least affected by fence installation, because of the fact that only herbaceous vegetation was established inside the fenced area, and streamside-cover scores increase as tree cover increases. The primary category of habitat variables (and believed to be the most important variables relative to benthic macroinvertebrates) showed improvement during the post-treatment period relative to control sites at the outlet (T-1) and upstream (T1-3 and T2-3) sites except for VDR at T-1; BSAC and embeddedness improved at the outlet and upstream sites. The secondary (channel alteration, BSD, and PRRBR) category of habitat variables is related to macroscale processes and these variables appeared to be least effected by fence installation; nevertheless, these variables showed an overall improvement at treated sites during the post-treatment period. These variables would be most effected by flow regime, and it may be that the lower stream discharge during the post-treatment period reduced any changes that might have taken place in the treatment basin relative to the control basin.

Overall, the installation of the fencing allowed vegetation to establish itself along the streambanks, and this helped to trap sediment prior to it reaching the stream channel. The habitat along the banks and within the stream improved in quality, allowing for less sedimentation of the stream bottom and more niches for benthic macroinvertebrates to inhabit. When looking at the overall numbers for total habitat (table 29), the results were positive and the habitat improved at the outlet and at upstream sites in the treatment basin.

Water-quality data collected during benthic-macroinvertebrate sampling showed varied changes in nutrient constituents following fence installation. Dissolved nitrate and ammonia decreased at the treated sites relative to control sites during the post-treatment period (table 29). TKN and DKN either decreased or showed no change at T-1 relative to C-1 during the post-treatment period; however, data for upstream sites indicated an increase in TKN and DKN relative to C1-2. Concentrations of total and dissolved P increased at all treatment sites relative to control sites during the post-treatment period, which was expected given the results for the low-flow sampling conducted as part of the surface-water aspect of this study.

The most consistent water-quality change in the treatment basin during the post-treatment period was the decrease in suspended-sediment concentrations. Suspended sediment

Table 29. Summary of differences between treatment and control benthic-macroinvertebrate data for habitat characteristics, benthic-macroinvertebrate indices, and water quality collected in the Big Spring Run Basin, Lancaster County, Pa.

[HBI, Hilsenhoff biotic index; EPT, Ephemeroptera, Plecoptera, and Trichoptera; μS/cm, microsiemens per centimeter; °C, degrees Celsius; mg/L, milligrams per liter; N, nitrogen; TKN, total ammonia plus organic nitrogen as nitrogen; DKN, dissolved ammonia plus organic nitrogen as nitrogen]

	May		September		Combined	
Habitat	**Outlet[1]**	**Upstream[2]**	**Outlet**	**Upstream**	**Outlet**	**Upstream**
Bottom substrate available cover	-2	3	4	-3	0.9	0.9
Embeddedness	0	2	4	-4	2	.1
Velocity to depth ratio	-1	2	.5	-4	-.2	2
Channel alteration	-8	3	-.2	-.9	-4	2
Bottom scouring & deposition	-.2	-2	4	-.1	2	-2
Pool/riffle, run/bend ratio	1	3	4	2	2	2
Bank stability	2	1	3	-1	2	.3
Bank vegetative stability	2	2	2	-1	2	1
Streamside cover	.2	1	.2	.1	.2	.6
Total habitat score	-6	18	20	-12	7	7
Indices						
Taxa richness (generic)	6	5	-2	12	2	8
Taxa richness (family)	-.8	3	2	7	.9	4
HBI (generic)	-.50	.19	-.84	.30	-.67	.25
HBI (family)	-.66	-.26	-.47	.78	-.57	.068
Scrapers to filterers	.13	-.28	.16	-1.8	.14	-1.0
Shredders to total taxa	-.053	-.11	-.032	-.022	-.043	-.072
EPT index	.8	-.2	.4	.4	.6	.2
EPT to chironomidae ratio	-.060	.25	.34	-1.3	.14	-.25
Percent EPT	-2.8	-.71	-12	-41	-7.5	-14
Percent dominant taxa (generic)	-19	8.9	14	16	-2.5	8.7
Percent dominant taxa (family)	-5.5	-.75	-13	29	-9.4	8.8
Percent chironomidae	10	-.40	-18	2.1	-3.9	1.5
Percent oligochaetes	-28	-17	-11	.53	-19	-12
Water quality						
Specific conductance (μS/cm)	-24	-16	-34	22	-29	6.1
pH (standard units)	-.11	.08	-.11	-.48	-.11	-.18
Water temperature (°C)	-.80	1.46	.03	-1.91	-.38	.16
Dissolved oxygen (mg/L)	.10	-.95	.10	-1.53	.10	-1.45
Dissolved-nitrate N (mg/L)	-1.23	-.50	-2.74	-1.71	-1.99	-.98
Dissolved-ammonia N (mg/L)	-.020	.012	-.018	-.077	-.019	-.013
TKN (mg/L)	-.17	.87	.09	-.11	-.04	.52
DKN (mg/L)	-.01	.62	.01	-.20	0	.33
Total phosphorus (mg/L)	.006	.123	.084	.218	.045	.170
Dissolved phosphorus (mg/L)	.026	.078	.052	.188	.039	.133
Suspended sediment (mg/L)	-17.7	18.6	-5.53	-32.1	-11.4	-1.76

[1] Outlet comparison is between T-1 and C-1. Data in table are mean difference between T-1 and C-1 data from the pre- to post-treatment period. The mean differences for the pre- and post-treatment periods were the means for T-1 subtracted by the means for C-1. The post-treatment period difference was subtracted by the pre-treatment period difference.

[2] Upstream comparisons are between T1-3 to C1-2 and T2-3 to C1-2. Data in table are the average difference between T1-3 and C1-2 and the difference between T2-3 and C1-2 from the pre- to post-treatment period. The mean differences for the pre- and post-treatment periods were the means for T1-3 and T2-3 subtracted by the means for C1-2. The post-treatment period difference was subtracted by the pre-treatment period difference for both pairs, with these differences averaged.

decreased at all treated sites compared to the control sites after the fences were installed. This decrease in suspended sediment corresponds well with results from the surface-water sampling. Surface-water sampling results showed 36 to 46 percent reductions in suspended-sediment yield at treated sites relative to control sites during the post-treatment period. This change was reflected in the benthic-macroinvertebrate communities where the numbers of chironomidae and oligochaeta decreased because of less siltation and habitat where they could live. This also equates to the improvement seen for some of the habitat variables affected by sediment loadings.

Field water-quality characteristics measured during benthic-macroinvertebrate sampling generally showed different trends during the post-treatment period at outlet and upstream sites in the treatment basin. SC, water temperature, and DO showed improvement at T-1 relative to C-1 during the post-treatment; upstream sites showed degradation relative to these characteristics (SC and water temperature increased and DO decreased) (table 29). For low-flow data collected at T-1 and T-2, SC, water temperature, and DO all decreased relative to control/upstream sites during the post-treatment period.

Benthic-macroinvertebrate indices that showed improvement during the post-treatment period at the outlet and upstream sites in the treatment basin relative to control sites were taxa richness, EPT index, and percent oligochaetes (table 29). Taxa richness showed relative increases at the treated sites at the generic and family level. The relative number of taxa showed a larger increase at the upstream sites. The EPT index and percent oligochaetes showed more improvement at the outlet than the upstream sites (relative to control sites) during the post-treatment period. An increase in taxa richness was likely related to habitat improvement. Positive changes in the EPT index and a decrease in the percent oligochaetes usually are correlated with improving water quality. Fewer oligochaetes is indicative of a healthier, less silt-covered stream bottom. Oligochaetes are tolerant of low DO (Mandaville, 2002) and can live in areas of high siltation; more sensitive macroinvertebrates would not be able to exist.

Two benthic-macroinvertebrate metrics showed negative responses during the post-treatment period at the outlet and upstream sites in the treatment basin relative to control sites. The total number of EPT taxa and shredders decreased at treated relative to control sites during the post-treatment period. An increase in shredders was not expected because tree cover did not increase. The relative decrease in percent EPT was more prominent at the upstream sites in the treatment basin. Given the relative increase in the EPT taxa, the decrease in total numbers of EPT taxa indicates that even though the number of families in the post-treatment samples increased in the treatment basin, the total number of EPT decreased.

All other benthic-macroinvertebrate metrics showed different responses during the post-treatment period at the treated relative to control sites. The HBI scores at generic and family level showed improvement at T-1 compared to C-1 meaning that with time more pollution sensitive benthic macroinvertebrates inhabited T-1 than before the fencing was installed; conversely, the upstream sites showed degradation in HBI scores during the post-treatment period (table 29). The scraper to filterer ratio (SFR) increased at T-1 relative to C-1 during the post-treatment period but upstream sites showed a decrease relative to C1-2. The upstream sites decreased in SFR relative to C1-2 because of a large increase in the number of scrapers identified at C1-2. Overall, SFR did not improve in the treatment relative to the control basin during the post-treatment period. The percent dominant taxa (for generic and family level) decreased at T-1 relative to C-1 during the post-treatment period; upstream sites showed the reverse trend relative to C1-2. When this percentage declines, it indicates more evenness in the community indicating a healthier community. The relatively higher numbers at the upstream sites indicated these sites were becoming more dominated by fewer taxa. This is typical of headwater limestone streams. However, it should be noted that taxa richness increased at the upstream sites. For the upstream sites, this indicated that even though the total number of individuals were being dominated by a few taxa, the total number of taxa increased. Percent chironomidae showed relatively small changes at the treated sites relative to the control sites during the post-treatment period, with a small relative decrease evident at T-1 and a small relative increase at the upstream sites (table 29). Chironomidae typically are considered pollution and siltation tolerant animals, and a decrease in number indicates an improvement in water quality and stream quality. The upstream response of more chironomids could be an artifact of the streams going back to their natural state and not an indication of site degradation.

Canonical Correspondence Analysis (CCA) was used to integrate all the results discussed above in order to determine which variables appeared to have the most significant effect on benthic-macroinvertebrate community structure. CCA was conducted on May and September data separately. Overall, the most important variables identified by CCA (those typically associated with the first CCA axis) to affect benthic-macroinvertebrate communities were the presence of a fence, concentrations of dissolved nitrite, DO, suspended-sediment concentration, and stream discharge. Fencing as a significant variable in the CCA indicated that fencing did influence the structure of the benthic-macroinvertebrate community. Following from this, water-quality data showed suspended-sediment concentrations and loads decreased in the treatment relative to the control basin during the post-treatment period. Stream discharge, which decreased from the pre- to post-treatment period, as expected, also was shown to significantly affect the benthic-macroinvertebrate community. The one variable that typically is not known to affect the benthic-macroinvertebrate community is dissolved nitrite. Nitrite is an intermediate product in different N-cycling processes, and, in this case, it would be expected that it is intermediate to denitrification processes that occur in relatively anaerobic conditions. These anaerobic conditions and evidence for denitrification were identified as part of the ground-water aspect of this study conducted in the treatment basin. With the CCA conducted, dissolved nitrite and DO were significant on some of the same CCA axis.

Therefore, it may be that dissolved nitrite as an important CCA variable may be somewhat related to DO. It would be expected that DO would have a significant effect on the structure of the benthic-macroinvertebrate community.

Habitat variables such as embeddedness, PRRBR, and VDR also had an influence on the structure of the benthic-macroinvertebrate community, but these variables tended to be more correlated with the second CCA plot axis, indicating their influence was less than those variables associated with the first CCA axis. These habitat variables are considered primary and secondary habitat characteristics, which, relative to tertiary characteristics, are supposed to have a greater effect on benthic macroinvertebrates. Generally, with the improvements in habitat structure that occurred because of fence installation, the sites tended to be related to these habitat variables in a gradient along the axis that indicated the treated sites had better habitat characteristics than control or untreated sites.

Incorporation of 1- and 3-month loads (the 1- or 3-month cumulative load for a constituent) into the CCA was conducted to determine if cumulative loads were more important to the structure of the benthic-macroinvertebrate community than instantaneous concentration data collected at the time of benthic-macroinvertebrate sampling. Some 1- and 3-month load variables significantly affected the benthic-macroinvertebrate community. Total 1-month loads of N and P were significant, as were the 3-month loads for TKN. For the CCA run incorporating 3-month load data, TKN was the only nutrient variable (whether load or concentration) that was significant, and it was along the first axis. Thus, this analysis suggests that incorporation of 1- and 3-month load data into CCA for benthic macroinvertebrates should be done if the data are available.

Overall, it can be stated that streambank fencing helped to improve the habitat for benthic macroinvertebrates. This improvement in habitat was reflected in the biological metrics that generally indicated improvement at the treated sites relative to control sites. In general, habitat and metrics indicated the outlet site improved a bit more than upstream sites in the treatment basin, but this may just be indicative of more overall habitat for the benthic macroinvertebrates. The outlet site had a larger stream width and deeper pools and riffles.

Ground Water

The study design for the effects of fencing on ground water focused on water quality in the treatment basin with treated wells inside the fenced area and control wells outside the fenced area. Some characterization of subsurface water in the basins was conducted through the installation and sampling of a piezometer network and one-time spring sampling. Knowledge of the geology of the study area was generally dependent on the findings of previous workers such as Meisler and Becher (1971) and Poth (1977). Analysis of near-infrared and black-and-white photographs supplemented the existing work.

Structural Framework

Much of the structural framework and geology of the region has been described by Meisler and Becher (1971). In general, the region has undergone several periods of deformation that have produced a complex structural pattern of folds, thrust faults, various sets of cleaved, warped, and folded axial planes and bedding. Many of the major structural features trend east-west, but local variation is common.

In the study area, near-infrared (105 linear- and 2 curvilinear-tonal features identified) and black-and-white (62 linear-tonal features identified) photographs confirmed the local, dominant structural trend is east-west, with a minor north-south trend (fig. 56). The linear-tonal features reflect the deformation history of the Cambrian sediments and may represent minor folds or possibly joints.The near-infrared photography also indicated several areas where ground water may be at or near the surface. Although the Vintage Formation is generally cut or bounded by faults (Poth, 1977, p. 23), the contact between the Vintage Formation and the Conestoga Formation was not identified on either the near-infrared or black-and-white photographs. Tonal features did, however, correlate well with the contact between the Vintage Formation and the Antietam and Harpers Formations.

Ground-Water Flow

Ground-water flow generally results in a water table that is a subdued replica of the land surface. Precipitation that infiltrates the soil, regolith, and fractured bedrock recharges the ground-water system and, through springs and streambed seepage, provides the base flow to maintain the surface-water system between precipitation events and through periods of drought.

In the study area, four springs were sampled (fig. 1). Spring SP76 is in the treatment basin at an altitude of approximately 330 ft and has a discharge sufficient to supply the domestic needs of the spring owner and associated dairy operation. Springs SP72 and SP73 are in the control basin near surface-water site C-1 at an altitude of about 300 ft. SP72 is along the streambank of Big Spring Run and has an estimated discharge of a few gallons per minute. SP73 is east of Big Spring Run and has a discharge sufficient to supply the domestic needs of the spring owner and associated dairy operation. Spring SP17 is also in the control basin, at an approximate altitude of 350 ft with an estimated discharge of about 50 gal/min.

A series of piezometers was installed at surface-water sites T-1, T-2, and C-1 to better understand the local flow system and to determine if the stream channels were gaining or losing water (table 4). Water levels measured from the piezometers at T-1 between October 22, 1998, and July 1, 1999, exhibited minimum change, varying less than 1.0 ft. Water-level altitudes were consistently greater at the T-1 stilling well than in any of piezometers or wells (fig. 57), thus indicating this tributary section of Big Spring Run was a los-

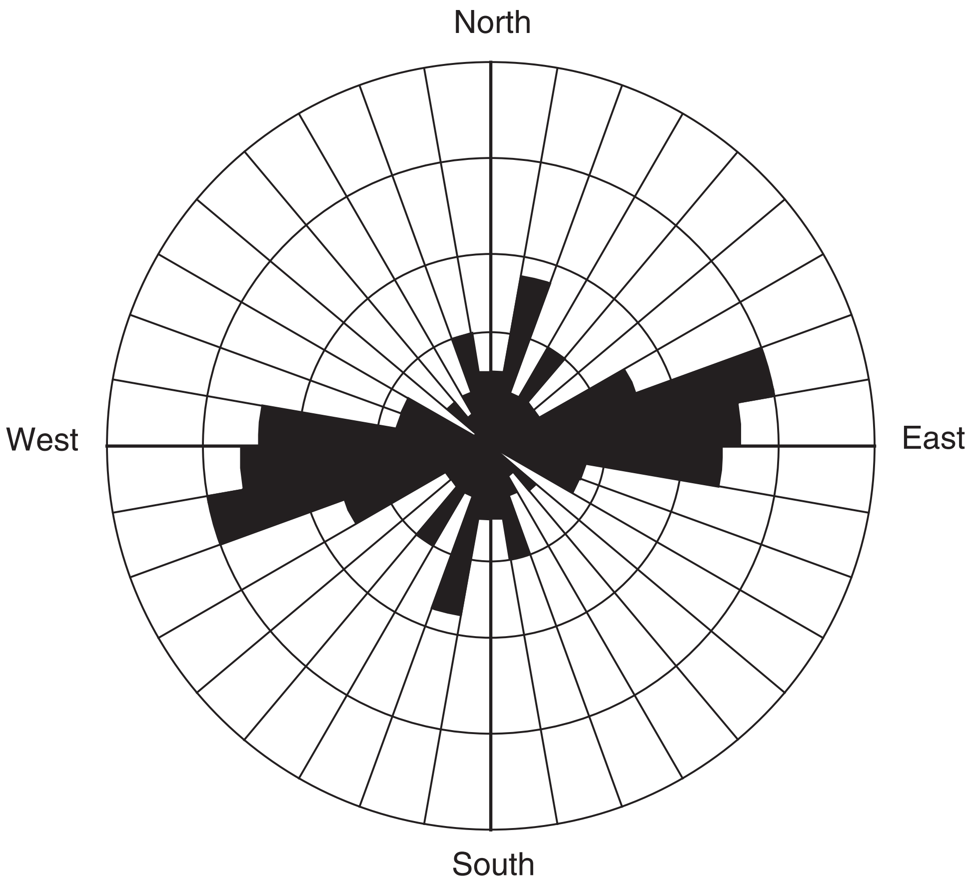


Figure 56. Rose diagram showing linear-tonal features identified on near-infrared photography for the Big Spring Run Basin, Lancaster County, Pa. The rose-diagram (plotted in 10-degree increments) shows that there is a dominant east-west trend in the linear-tonal features that reflect the structural framework of the study area.

ing stream (recharges ground water). The water-level altitudes in the shallow wells also were greater than for the deep well, indicating this reach recharged not only the shallow ground water but also the deep ground-water system.

Water levels measured from the piezometers and wells at surface-water site T-2 between October 29, 1998, and May 3, 1999, exhibited minimum change, varying less than 1.1 ft except in the deep well where the maximum variation was about 2.9 ft. Water-level altitudes at the T-2 stilling well were greater than in piezometers LN-2078 and LN-2079, but lower than the remaining piezometers or wells (fig. 58). The exception was piezometer LN-2074, which twice (water levels in the piezometers were measured five to eight times over the study period) had water-level altitudes greater than in the stilling well. These water levels indicated this tributary section of Big Spring Run gained water along the eastern bank, but lost water along the western bank. This was probably related to an input of ground-water recharge from the adjacent watershed to the east, which discharged near T-2. On occasion, the water-level altitudes in the shallow wells were greater than for the deep well. The varying water levels in the wells indicated this reach had more dynamic ground-water flow than at T-1 and was an area of ground-water discharge for the shallow and deep systems.

Water levels measured at surface-water site C-1 between April 27, 1999, and July 1, 1999, exhibited more variation (maximum change of 5.3 ft) than water levels measured in the piezometer and well networks at T-1 and T-2. Water-level altitudes at the C-1 stilling well were consistently lower at all piezometers, the exception was piezometer LN-2095, which exhibited the greatest variation in altitude (291.42 to 296.19 ft) and had an altitude greater than the stilling well on July 1, 1999. The inconsistent response of piezometer LN-2095 suggests that it (1) may not be completed properly, or (2) may be completed in a different hydrologic environment. During repeated visits to C-1, it was noted that a sanitary sewer line

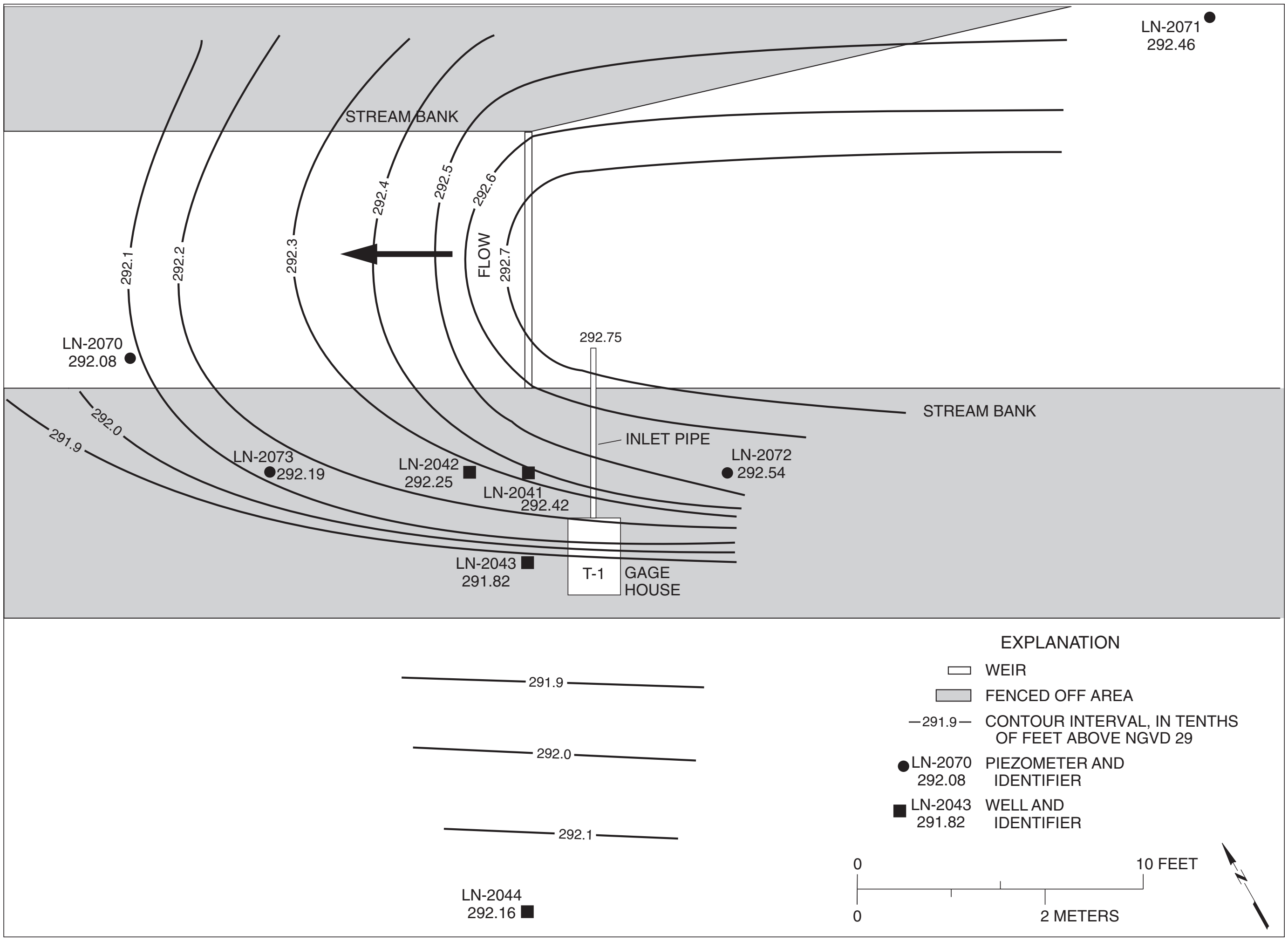


Figure 57. Potentiometric surface for piezometers, ground-water wells, and surface water at surface-water site T-1 on April 5, 1999 in the Big Spring Run Basin, Lancaster County, Pa.

was buried near the stream and it is possible that piezometer LN-2095 may be completed in a gravel-filled fracture related to construction of the sewer line. Ignoring piezometer LN-2095, the water-level altitudes indicate this reach of Big Spring Run was a gaining stream with recharge entering from streambanks and through the bottom of the stream.

Water-quality data can provide additional insight regarding ground-water flow paths. Although the water-level data collected from wells and piezometers at T-1 indicated this stream segment was a source of ground-water recharge, the chemistry in the wells and piezometers indicated otherwise. Concentrations of dissolved ammonia in the stream were significantly lower than concentrations of ammonia in the wells and piezometers. Concentrations of dissolved nitrate plus nitrite, however, were significantly greater in the stream than in the wells and piezometers. Hence, the water-quality data suggested this stream was not in good communication with the shallow or deep ground-water system.

Statistical analysis of water-quality data for wells at surface-water site T-1 indicated the water in deep well LN-2043 was significantly different from the water in wells LN-2041, LN-2042, and LN-2044 for most of the constituents sampled. Elevated values for pH, SC, and alkalinity in well LN-2043 indicated the water-bearing fractures intercept water with longer flow paths and greater residence times than in the shallow wells. DO concentrations, however, were greater in well LN-2043 than in the shallow wells. This may be the result of water cascading into the well and becoming oxygenated. Well LN-2043 was the lowest yielding of the eight wells drilled and it did not recover to pre-pumping water levels prior to monthly sampling.

Water-quality data in wells and piezometers at surface-water site T-2 were also different than in the stream. Concentrations of dissolved ammonia in the stream were somewhat higher than in the wells and piezometers; concentrations of dissolved nitrate plus nitrite showed the opposite. These differences, however, were not as great as seen at T-1 for the shallow wells relative to the stream chemistry. Deep well LN-2039 showed the greatest difference in water chemistry between a well and the stream of all wells at T-2. These data suggested the shallow wells at T-2 were in fairly good connection with the stream, but the deep well was not.

Age Dating

During the summer of 1999, water samples were collected from selected wells, piezometers, and springs in the Big Spring Run Basin (table 30) as part of a larger scale effort to age date ground water in the Chesapeake Bay and evaluate nitrate transport (Lindsey and others, 2003). Multiple age-dating techniques typically are employed to provide greater confidence in the resultant ages. In the Big Spring Run Basin, three types of age-dating methods were used - chlorofluorocarbons (CFCs), sulfur hexafluoride (SF6), and tritium/helium ($^3H/^3He$). Each method has advantages and disadvantages (Plummer and others, 1993; Cook and Solomon, 1997; Solomon and Cook, 1999; Plummer and Busenberg, 1999), the discussion of which is beyond the scope of this report (the interested reader may find the following references useful: For dating with CFCs – Busenberg and Plummer, 1992; Plummer and Busenberg, 1999. For dating with SF6 – Busenberg and Plummer, 2000. For dating with $^3H/^3He$ – Schlosser and others 1988, 1989; Poreda and others, 1988; Solomon and Sudicky, 1991; Solomon and others, 1993; Ekwurzel and oth-

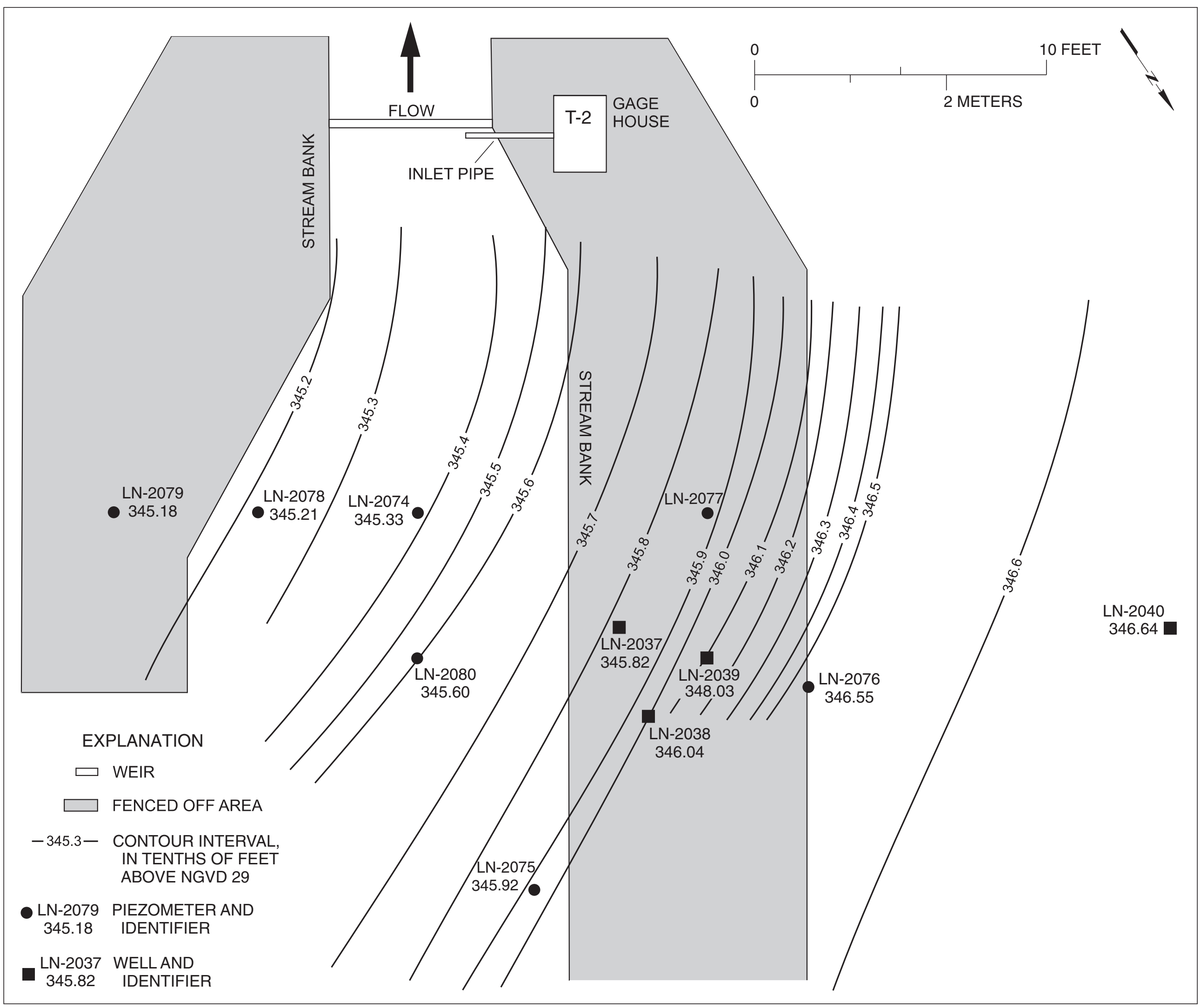


Figure 58. Potentiometric surface for piezometers, ground-water wells, and surface water at surface-water site T-2 site on March 18, 1999 in the Big Spring Run Basin, Lancaster County, Pa.

Table 30. Age of water for samples collected during summer 1999 from wells, piezometers, and springs in the Big Spring Run Basin, Lancaster County, Pa.

[CFCs, chlorofluorocarbons; SF6, sulfur hexafluoride; $^3H/^3He$, tritium/helium; ND, not determined]

Identification number	CFCs (age of water in years/ CFC dated)	SF6 (years)	$^3H/^3He$ (years)	Denitrification	Recommended age (years)
		Wells			
LN-2038	25.8-26.3/CFC-11 7.3- 9.3/CFC-12	3.8	ND	Yes	None
LN-2039	Contaminated	Contaminated	Contaminated	No	None
LN-2040	23.8-25.8/CFC-11	3.8	1.5	Yes	None
LN-2042	40.3-41.8/CFC-11 33.8-34.8/CFC-12	- 2	.6-1.0	Yes	33.8-34.8
LN-2044	22.8/CFC-11 15.3/CFC-12	-.2	ND	Yes	15.3
		Piezometers			
LN-2071	50.8/CFC-11 36.8/CFC-12	ND	ND	Yes	36.8
LN-2073	46.8/CFC-11 39.3/CFC-12 44.3/CFC-113	ND	ND	Yes	39.3
LN-2076	24.3/CFC-11 5.6/CFC-12	Contaminated	ND	Yes	None
		Springs			
SP-17	Contaminated	-.2	1.1-3.1	No	2.1
SP-72	Contaminated	.3	2.8-3.6	No	3.2
SP-73	Contaminated	3.3	ND	No	None
SP-76	Contaminated	-.2	Contaminated	No	None

ers, 1994; Solomon and Cook, 1999). In general, N. Plummer and E. Busenberg (U. S. Geological Survey, written commun., 1999) found the ground water in Big Spring Run Basin was extensively contaminated by CFCs and this contamination adversely affected many of their estimated dates. The ages of water are listed in table 30 and ages were determined to be young for the springs sampled and significantly older in wells and piezometers. This divergence of ages suggests the flow paths taken by water that reaches springs were relatively short and direct, differing significantly from the flow paths of ground water intercepted by wells and piezometers.

Plummer and Busenberg (written commun.,1999) also looked at nitrate and denitrification. The four springs exhibited no signs of denitrification, and this was probably the result of the waters young age. With the exception of well LN-2039, the wells and piezometers contained an excess amount of N, which suggests these waters have undergone denitrification and this is in accordance with their older ages.

Water-Level Fluctuations

Water-level data were collected in each of the eight wells (figs. 59 and 60) for the length of the study. The altitude of water levels varied by well, precipitation events, and season. For the eight monitor wells in the study basin, deep well LN-2039 at surface-water site T-2 exhibited the greatest variation in water levels with a range of 8.91 ft; shallow well LN-2041 at surface-water site T-1 exhibited the smallest variation in water levels with a range of 1.78 ft.

Large precipitation events (total precipitation that, on average, exceeded 1.0 in. in a 24-hour period) were examined in detail to determine the relation and response time between the onset of rainfall and rise in water levels in the ground-water wells and stream gages at T-1 and T-2. Water levels in each well typically increased in response to precipitation events. In five of the six shallow wells and the two deep wells, the timing of this increase was closely correlated (1 to 2 hours) to the rise in the water elevation of the stream. For well LN-2044, the response was slightly longer, about 3 to 4 hours. Water levels in the shallow and deep wells commonly peaked within 2 hours after the stream peaked, although the peaks were broader and not as distinctive in wells LN-2039, LN-2043, and LN-2044.

The precipitation and water-level data for wells and streams at T-1 and T-2 for July 22 through July 24, 1997, are shown in figure 61 and were determined to be fairly representative of dry summer conditions (prior to the storm event on July 23, 1997). The initial precipitation pulse of 0.05 in. on July 22 had no effect on water levels in the wells or on the stream gages at either site. Relatively more consistent and intense precipitation began on July 23 at about 0430 hours. By 0600 hours, a total of 0.25 in. of precipitation had fallen, and water levels began to rise in the stream gage at T-1 and wells LN-2041 through LN-2043. By 0700 hours, a total of 0.32 in. of precipitation had fallen, and water levels began to rise in the stream gage at T-2 and wells LN-2037 through LN-2040. By 1000 hours, after a total precipitation of 0.52 in. had fallen, the water level in well LN-2044 finally began to rise. Water levels in wells LN-2037 and LN-2038, and the stream gage at T-2 peaked at about 1400 hours, whereas wells LN-2039 and LN-2040 peaked about an hour later. At T-1, the stream gage peaked at about 1500 hours, and wells LN-2041 and LN-2042 peaked an hour later. Wells LN-2043 and LN-2044 did not show distinctive peaks from the first storm event, but exhibited a considerable change in slope at about 1800 hours.

Analysis of the July 23 and 24, 1997, storm event indicates hydrologic differences as well as similarities existed at each site and between sites. Prior to the storm event, the stream segment at T-2 was a source of ground-water recharge. During, and for a number of days afterward, the stream segment at T-2 was converted to an area of ground-water discharge. At T-1, however, the stream remained a source of ground-water recharge. On the basis of when the streams peaked and when water levels in the adjacent wells peaked, it would appear most of the shallow wells have good connection to the streams at T-1 and T-2. This interpretation, however, may not be entirely accurate as water levels generally rise as a wetting front generated as precipitation moves through the soil, regolith, and fractured bedrock. This wetting front will cause an increase in the interstitial pore pressure that commonly results in a rise in water levels in wells as it progresses through the ground-water system. Wells LN-2039, LN-2043, and LN-2044 do not appear to be as well-connected to the streams because their peaks were delayed several hours and were not distinctive.

As previously mentioned, seasonal water-level differences exist between the well networks at T-1 and T-2, despite the fact these wells were completed in the same aquifer, at similar depths, and similar distance from the surface streams. Excluding the effects of storm events, water levels in wells at T-1 exhibited only minor changes (less than 1 ft) between summer and winter. This muted, seasonal response may be the result of location. Analysis of ground-water flow paths using wells and piezometers indicated the adjacent stream segment contributed recharge to the ground-water system at T-1. At T-2, however, the stream segment was determined to be an area of ground-water discharge where much larger changes (3-5 ft) in water levels between summer and winter occurred. The greatest changes in water levels at T-2 between summer and winter were during drier years (1995, 1997-1999) (fig. 60). The abrupt declines in water levels from 1997 through 1999 could have been magnified by the establishment and growth of grasses and shrubs within the fenced area, but this could not affect water levels in LN-2044, which was outside the fenced area. During the summer, evapotranspiration is at a maximum so very little water remains in the regolith and water levels in the wells fall. On the basis of the large seasonal decline (4-5 ft) in water levels for deep well LN-2039, it appears at least some of the water-producing fractures penetrated by this well have short lengths that correspond to limited secondary storage, and are well connected to the regolith.

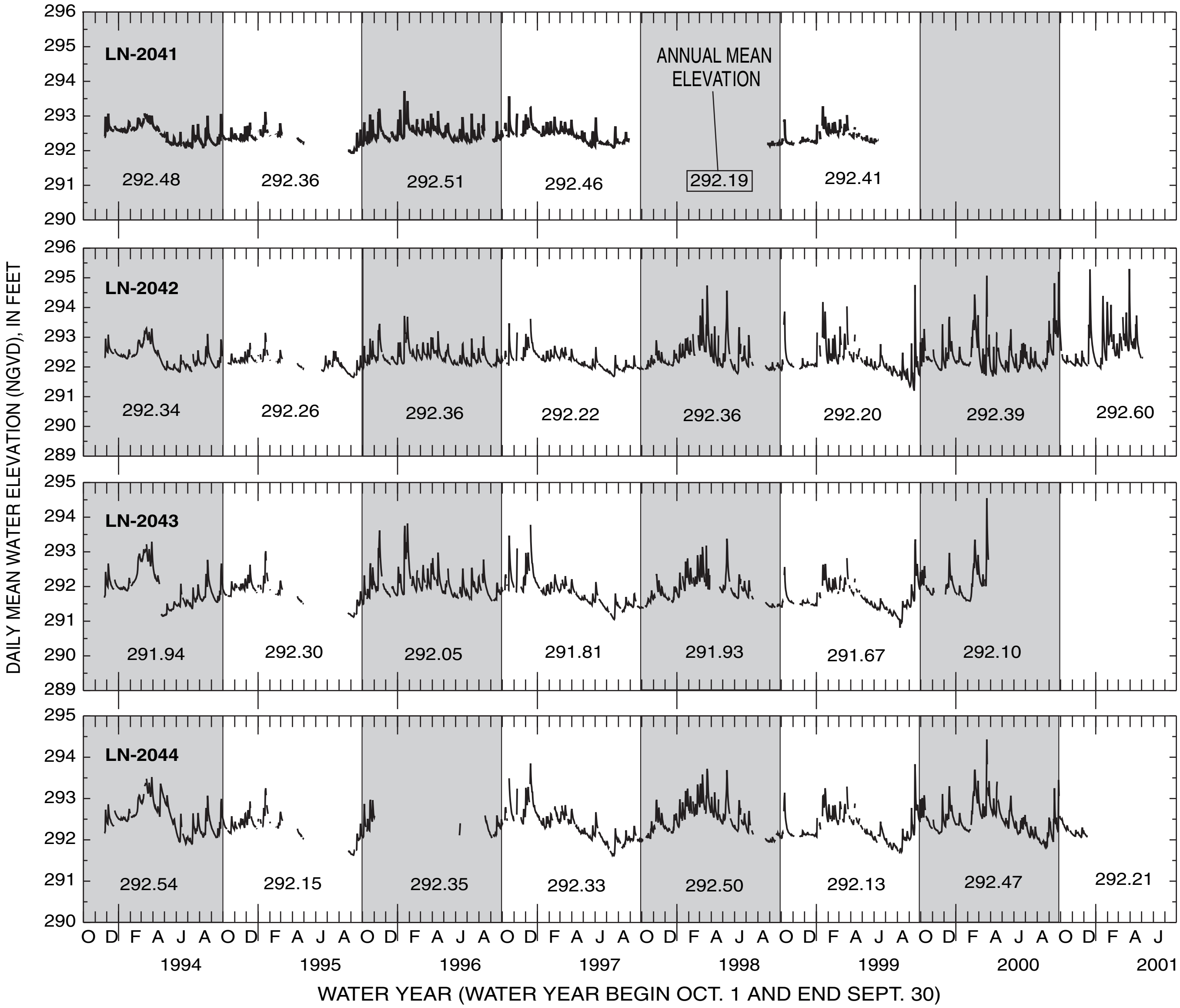


Figure 59. Graphs of daily and annual mean water-level elevations for the four ground-water wells at surface-water site T-1 from November 1993 through June 2001 in the Big Spring Run Basin, Lancaster County, Pa.

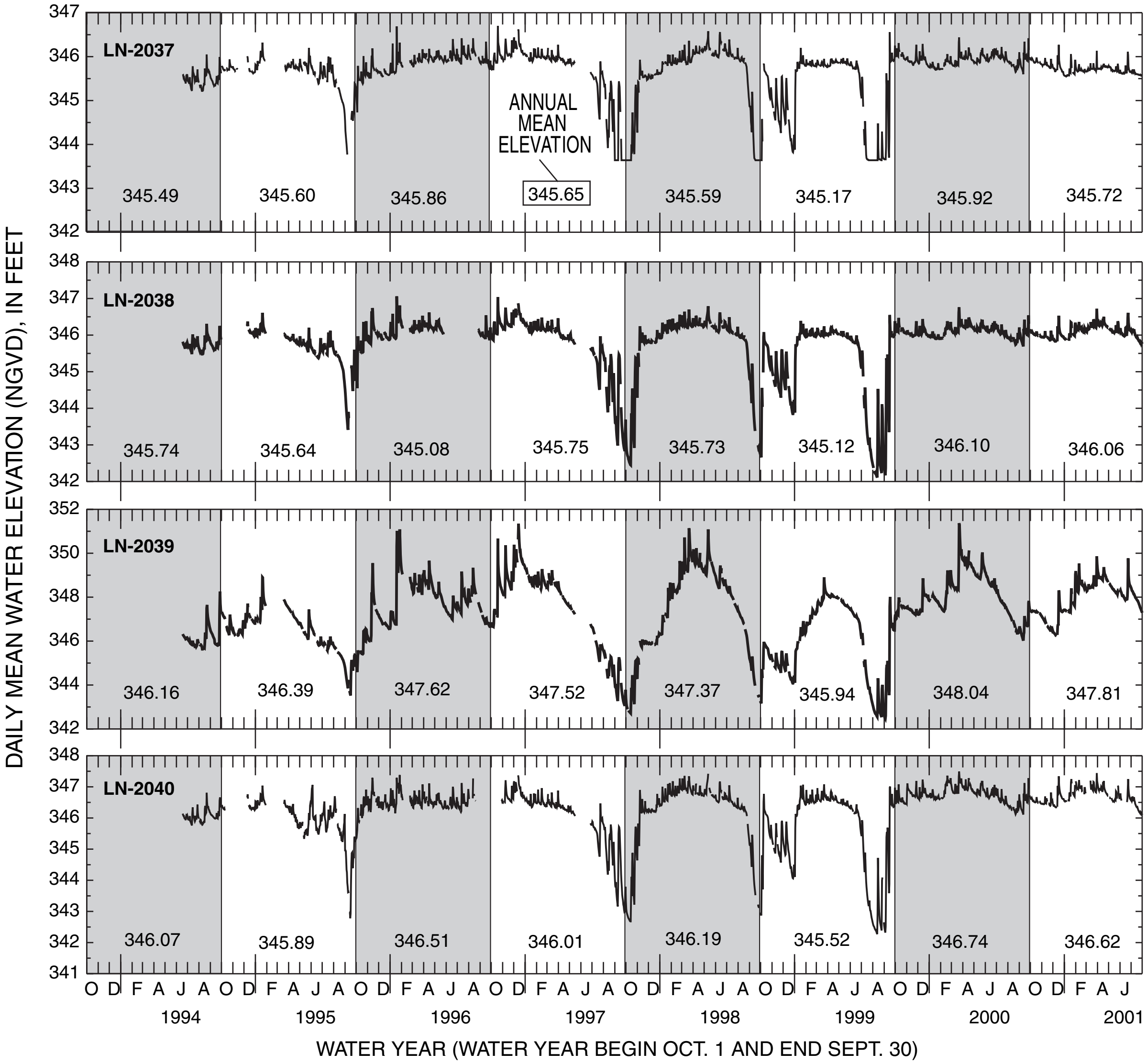


Figure 60. Graphs of daily and annual mean water-level elevations for the four ground-water wells at surface-water site T-2 from November 1993 through June 2001 in the Big Spring Run Basin, Lancaster County, Pa.

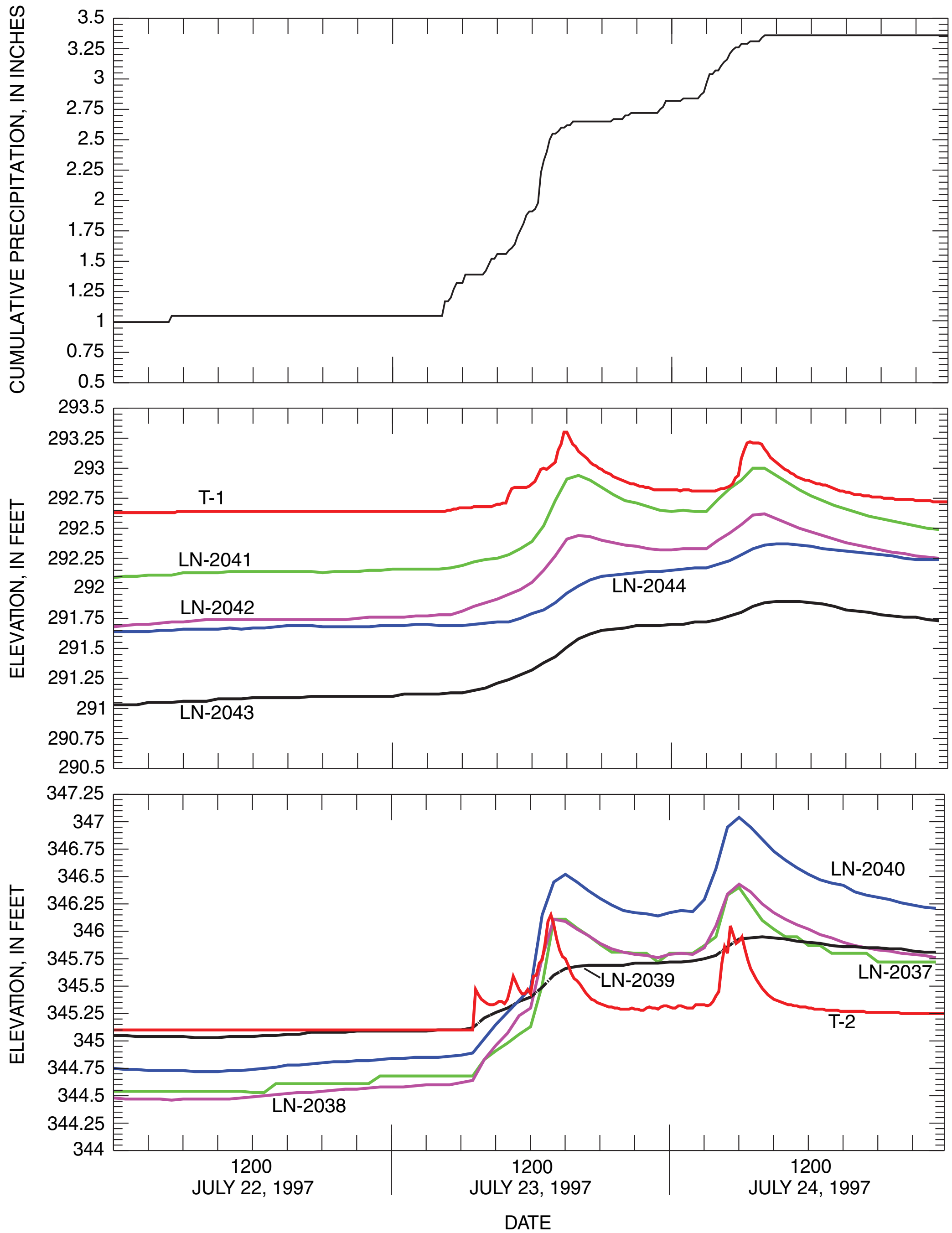


Figure 61. Cumulative precipitation and water-level elevations for selected surface-water stations and the eight ground-water wells located in the Big Spring Run Basin, Lancaster County, Pa., from July 22-24, 1997.

External influences on water levels in the wells also were evident. Water levels in the shallow wells also were affected by pumping of one or more unknown adjacent wells. Pumping may occur throughout the year but was most noticeable during the months of May to October, during which water levels fluctuate about 0.1 to 0.2 ft/d. Despite the measurable daily fluctuations, no long-term changes in water levels were observed.

For the duration of the study, with the exception of major storm events, the unnamed tributary to Big Spring Run at surface-water site T-1 was a source of ground-water recharge. For the years 1994, 1996, and 2000, the unnamed tributary to Big Spring Run at surface-water site T-2 was receiving ground-water discharge. For the summer months of years 1995, 1997, 1998, and 1999, the unnamed tributary at T-2 was a source of ground-water recharge.

Water Quality

Eight wells were sampled regularly for alkalinity, field characteristics (DO, SC, pH, and water temperature), dissolved nitrite, dissolved nitrate plus nitrite, DKN, dissolved ammonia, dissolved phosphorus, dissolved orthophosphate, and fecal streptococcus. After the first year of study, the sampling frequency for dissolved ammonia, DKN, dissolved nitrite, dissolved phosphorus, and dissolved orthophosphorus decreased from monthly to quarterly. Dissolved orthophosphate was not frequently measured above detection limits and will be not be discussed further. Samples were separated into several groups: (1) non-control and control wells, (2) differences between wells at T-1 and T-2, and (3) pre-treatment and post-treatment periods. Within each group, a number of variables such as amount of precipitation, time between precipitation events, depth to water, recharge versus base flow, and DO concentration were evaluated to determine which had a significant effect on water quality. Significant differences were identified within and between groups and for most of the variables analyzed. These differences, however, were not consistent between wells or for each site, making it difficult to fully understand which variable played an important role in affecting ground-water quality during the pre- and post-treatment periods.

Description of Data

Field pH was measured in 613 ground-water samples. Field pH ranged from 5.9 in well LN-2042 to 9.3 in LN-2038. Median pH values (medians reported are either for the pre- or post-treatment period) ranged from 7.0 to 7.4 (figs. 62d and 63d). Most of the pH values that exceeded 8.3 were measured during the first year and were probably related to grout contamination during the installation of the well. Seasonal effects were evident; pH characteristically was lower in the summer months and higher in the fall months. The pH in water from well LN-2043, however, generally was higher in the late summer and early spring.

Field SC was measured in 622 ground-water samples. Field SC ranged from 450 μS/cm in well LN-2039 to 1,540 μS/cm in well LN-2044. Median SC values (medians reported are either for the pre- or post-treatment period) ranged from 642 to 944 μS/cm (figs. 62c and 63c), which is considerably greater than the median SC of 570 μS/cm reported for the Conestoga Formation by Low and others (2002). The higher SC values commonly were measured during the first year and the months immediately following the passage of the remnants of Hurricane Floyd in early September 1999. Seasonal affects on SC were observed in all wells. Typically, the lowest SC was measured in July and the highest in October. Wells LN-2043 and LN-2044 commonly exhibited the highest SC in the spring.

During the study period, 540 ground-water samples were collected and analyzed for DO. The amount of DO in the samples ranged from 0.1 to 16 mg/L. Median concentrations of DO (medians reported are either for the pre- or post-treatment period) ranged from 1.1 mg/L at well LN-2040 to 8.75 mg/L at well LN-2043 (figs. 62c and 63c); this indicates the limestone aquifer is open to the atmosphere. For most wells, DO was lowest in the warmer months. In well LN-2043, DO tended to be lower in the spring and fall.

The temperature of ground water was measured in 615 samples. The minimum and maximum ground-water temperatures were 2.6 and 21.4 °C, respectively, and were measured in well LN-2041. Median temperatures (medians reported are either for the pre- or post-treatment period) ranged from 10.8 (LN-2040) to 13.3 °C (LN-2043) (figs. 62c and 63c). Shallow wells had the greatest temperature variations. Strong seasonal effects were observed in all wells. The lowest temperatures typically were in February and the warmest in August or September.

Alkalinity was measured in 458 ground-water samples. Alkalinity ranged from 168 mg/L in well LN-2037 to 444 mg/L in well LN-2041. The ground water is slightly to moderately alkaline; median alkalinities (medians reported are either for the pre- or post-treatment period) ranged from 240 mg/L (LN-2038) to 342 mg/L (LN-2043) (figs. 62d and 63d). This is characteristic of ground water in a limestone aquifer. Seasonal effects were relatively minor; the lowest alkalinities generally were in mid- to late-winter and the highest in mid- to late-summer or early fall.

DKN was measured in 267 ground-water samples. DKN concentrations as N ranged from the detection level to 2.9 mg/L; the highest concentration was measured in well LN-2041. Median concentrations (medians reported are either for the pre- or post-treatment period) ranged from 0.10 in well LN-2039 to 1.15 mg/L as N in well LN-2041 (figs. 62b and 63b).

Concentrations of dissolved ammonia ranged from the detection limit (0.02 mg/L) to 2.04 mg/L as N; the highest concentration was measured in well LN-2041. Fifty percent of the 314 ground-water samples collected contained dissolved ammonia as N at concentrations of 0.50 mg/L or less (figs. 62a and 63a).

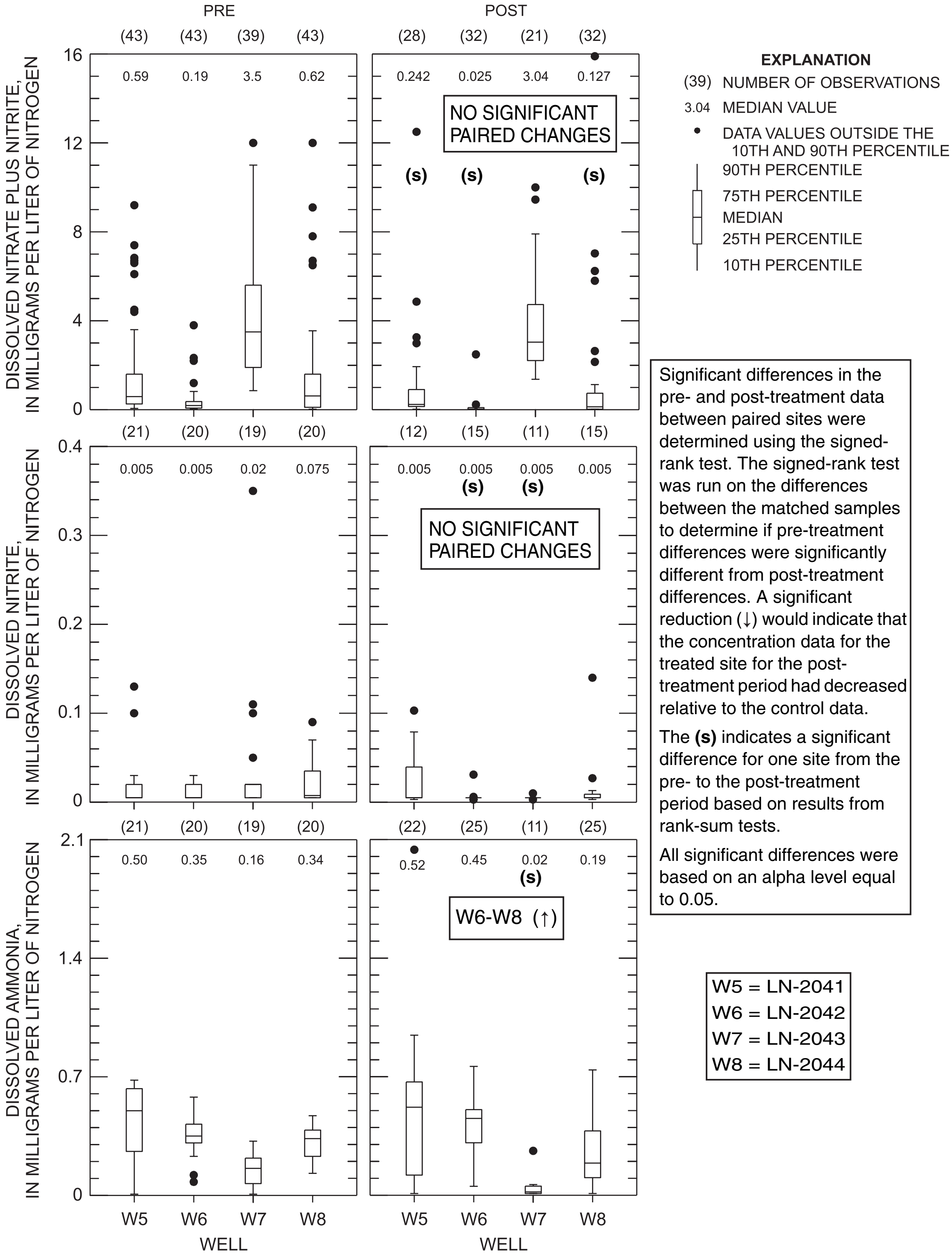


Figure 62a. Ranges of constituents for ground-water samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at the T-1 well nest located in the Big Spring Run Basin, Lancaster County, Pa.

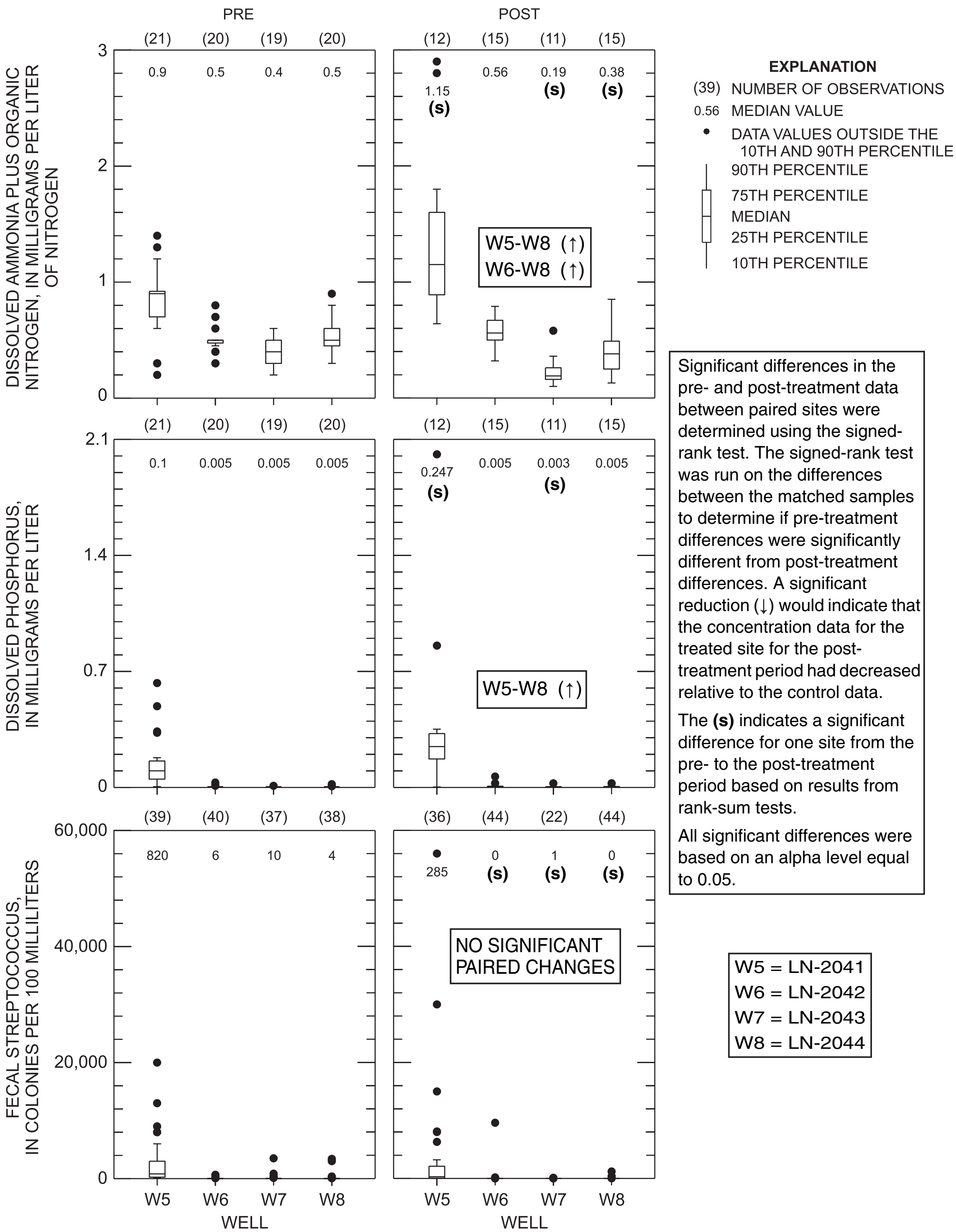


Figure 62b. Ranges of constituents for ground-water samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at the T-1 well nest located in the Big Spring Run Basin, Lancaster County, Pa.—Continued

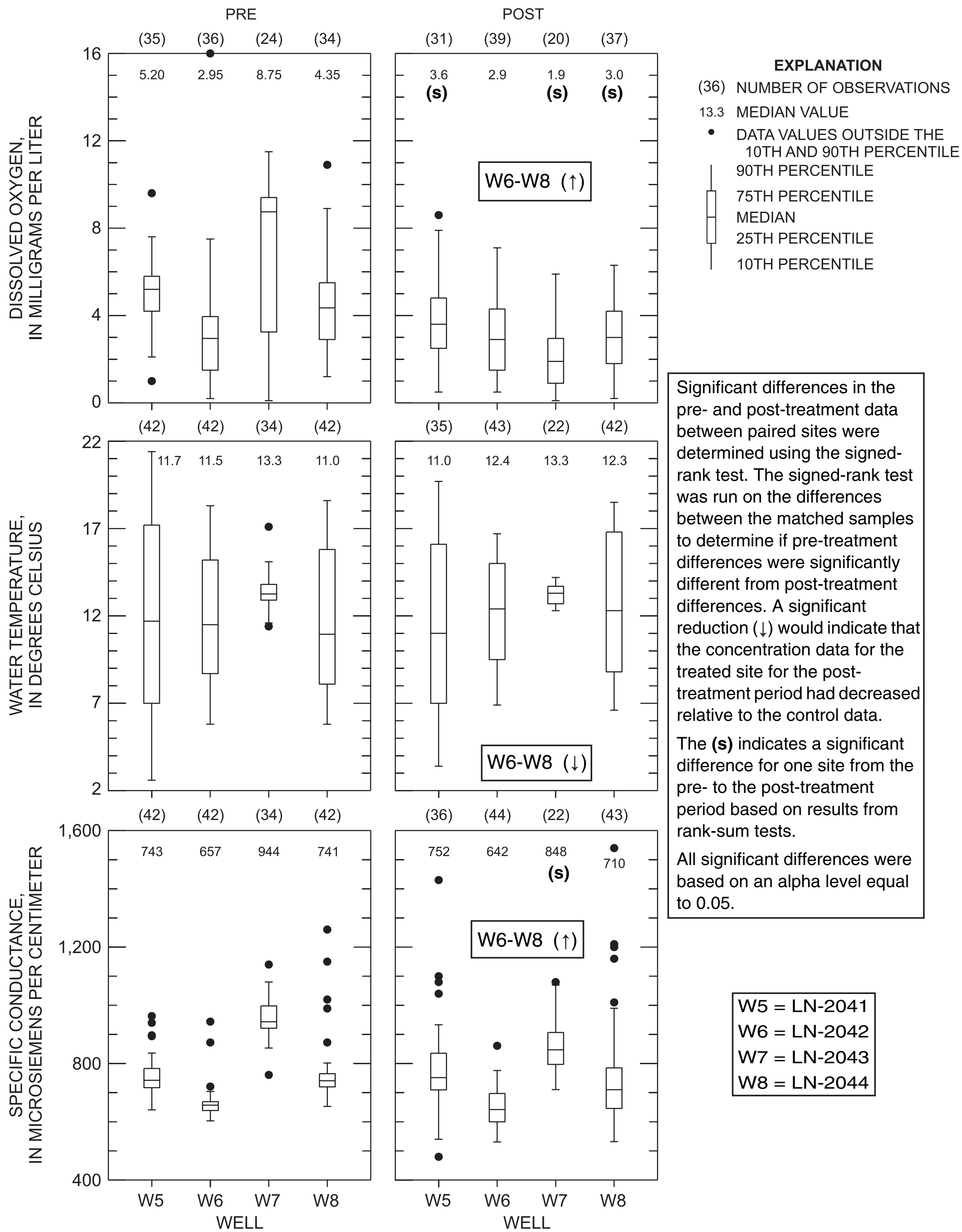


Figure 62c. Ranges of constituents for ground-water samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at the T-1 well nest located in the Big Spring Run Basin, Lancaster County, Pa.—Continued

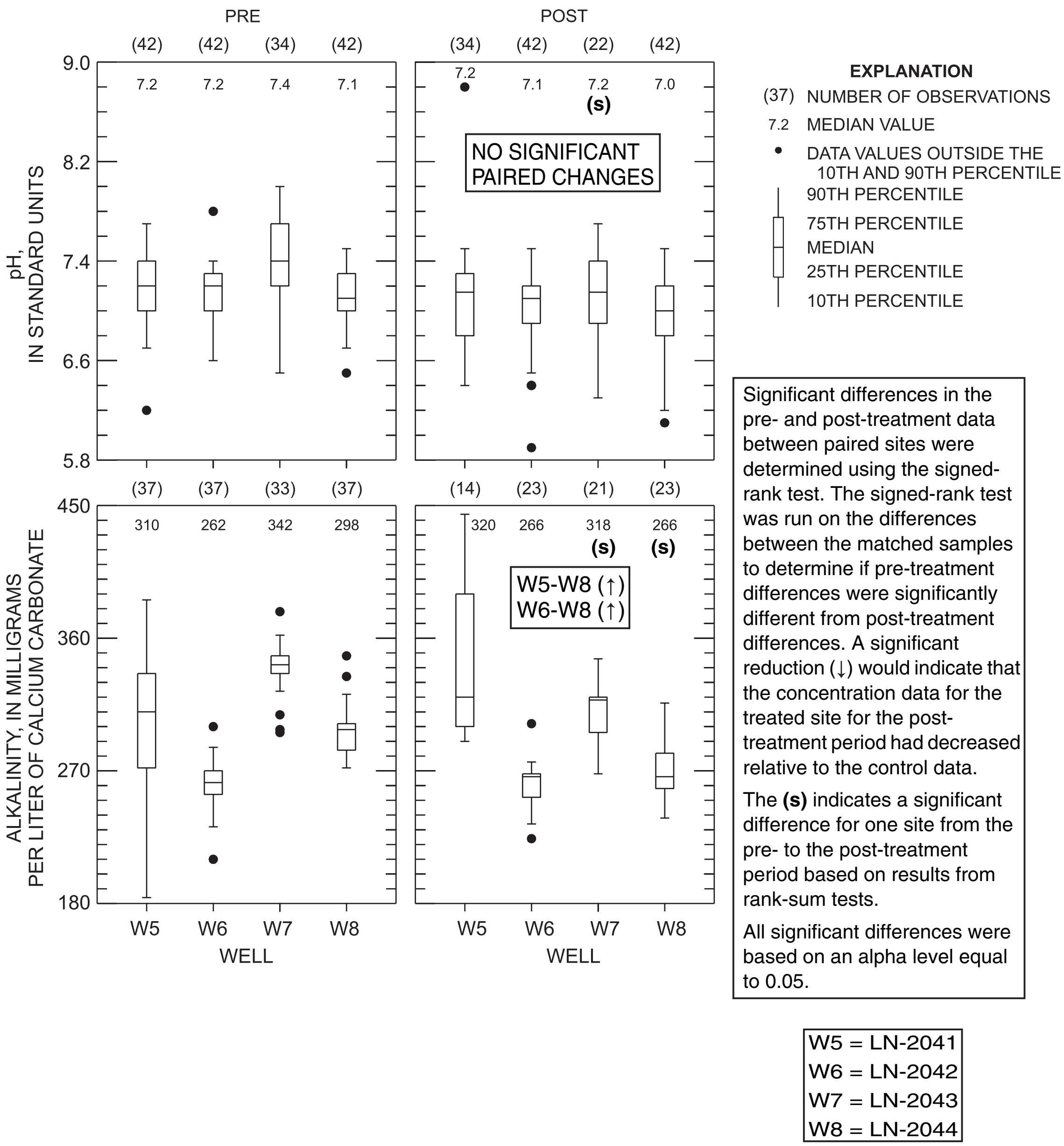


Figure 62d. Ranges of constituents for ground-water samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at the T-1 well nest located in the Big Spring Run Basin, Lancaster County, Pa.—Continued

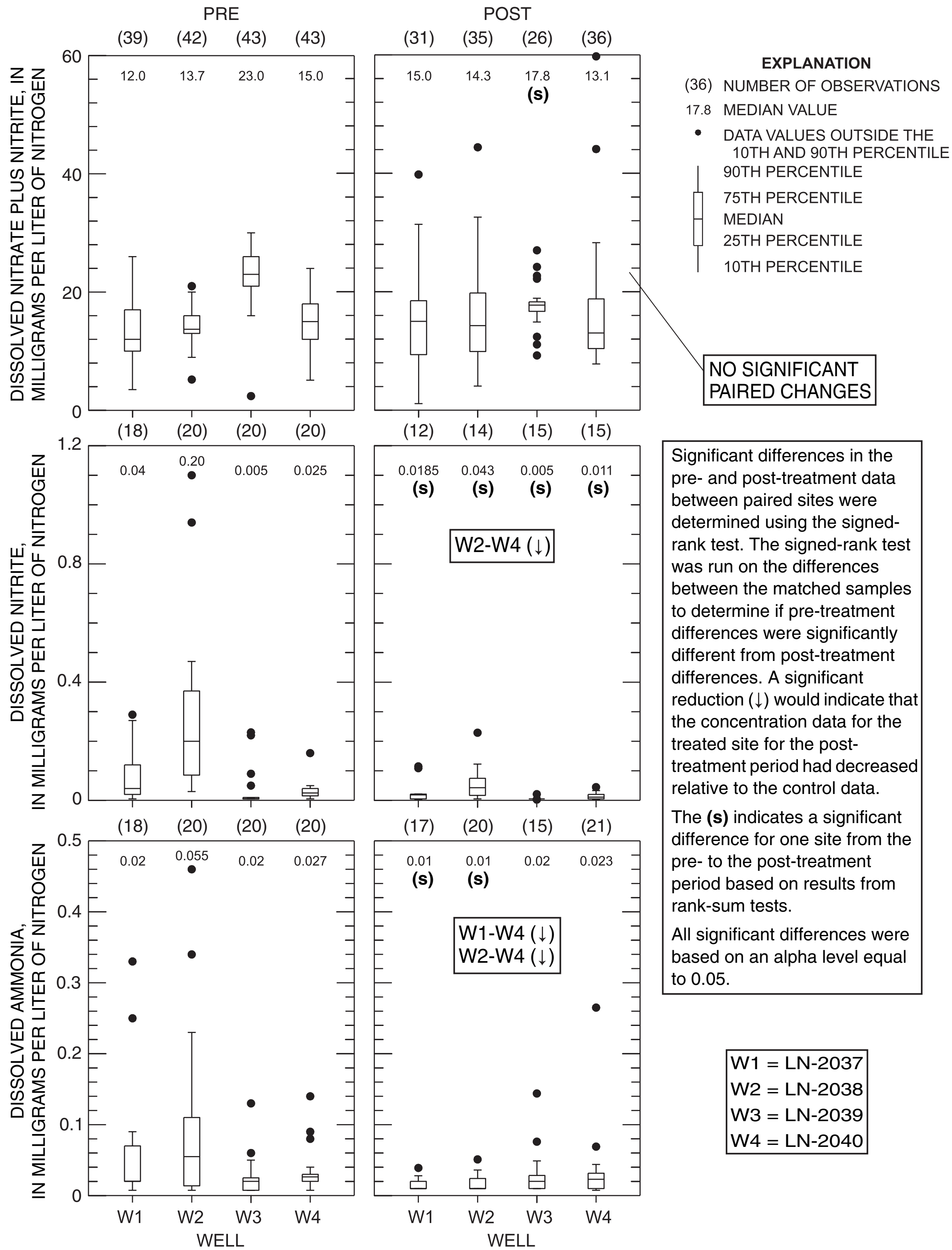


Figure 63a. Ranges of constituents for ground-water samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at the T-2 well nest located in the Big Spring Run Basin, Lancaster County, Pa.

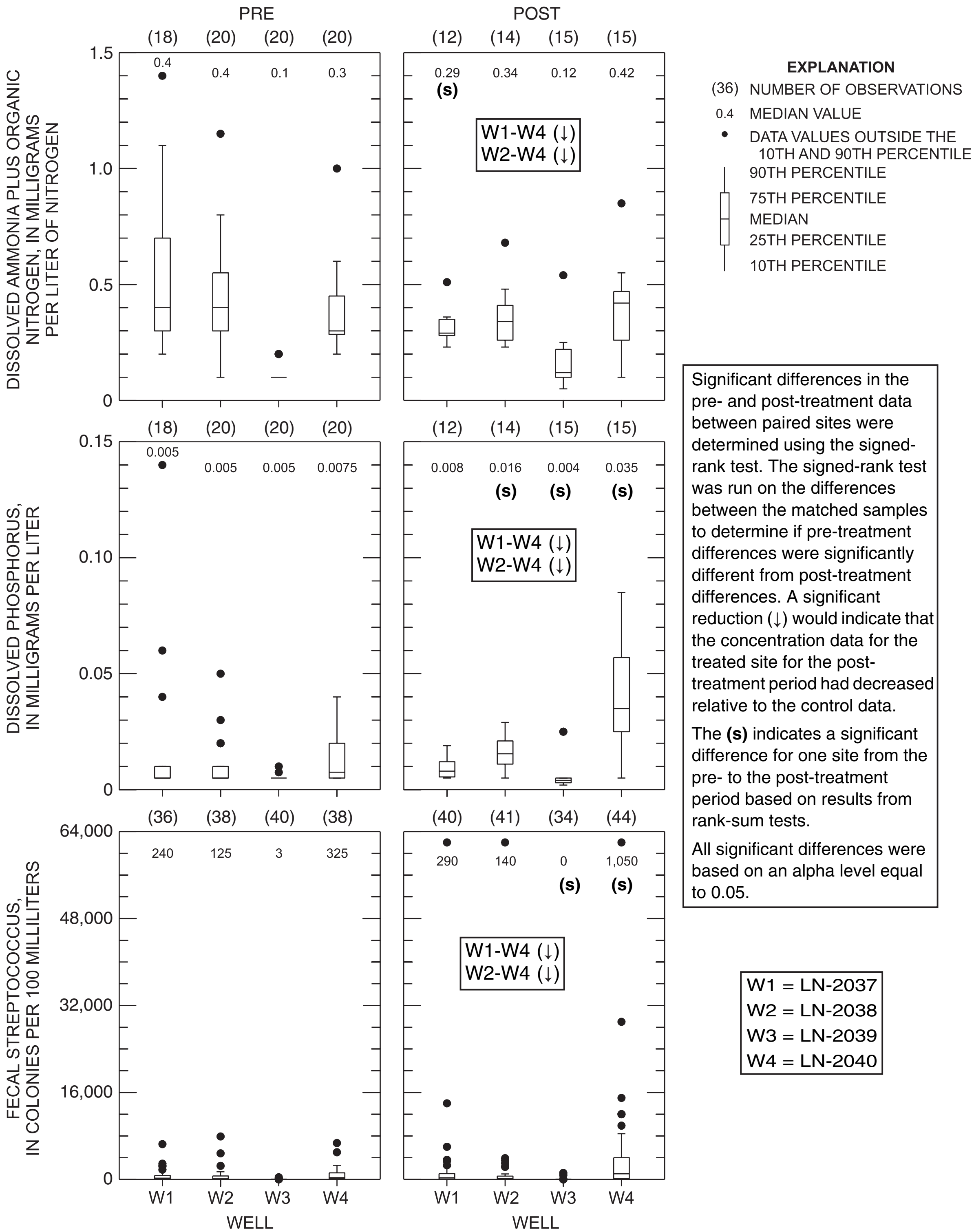


Figure 63b. Ranges of constituents for ground-water samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at the T-2 well nest located in the Big Spring Run Basin, Lancaster County, Pa.—Continued

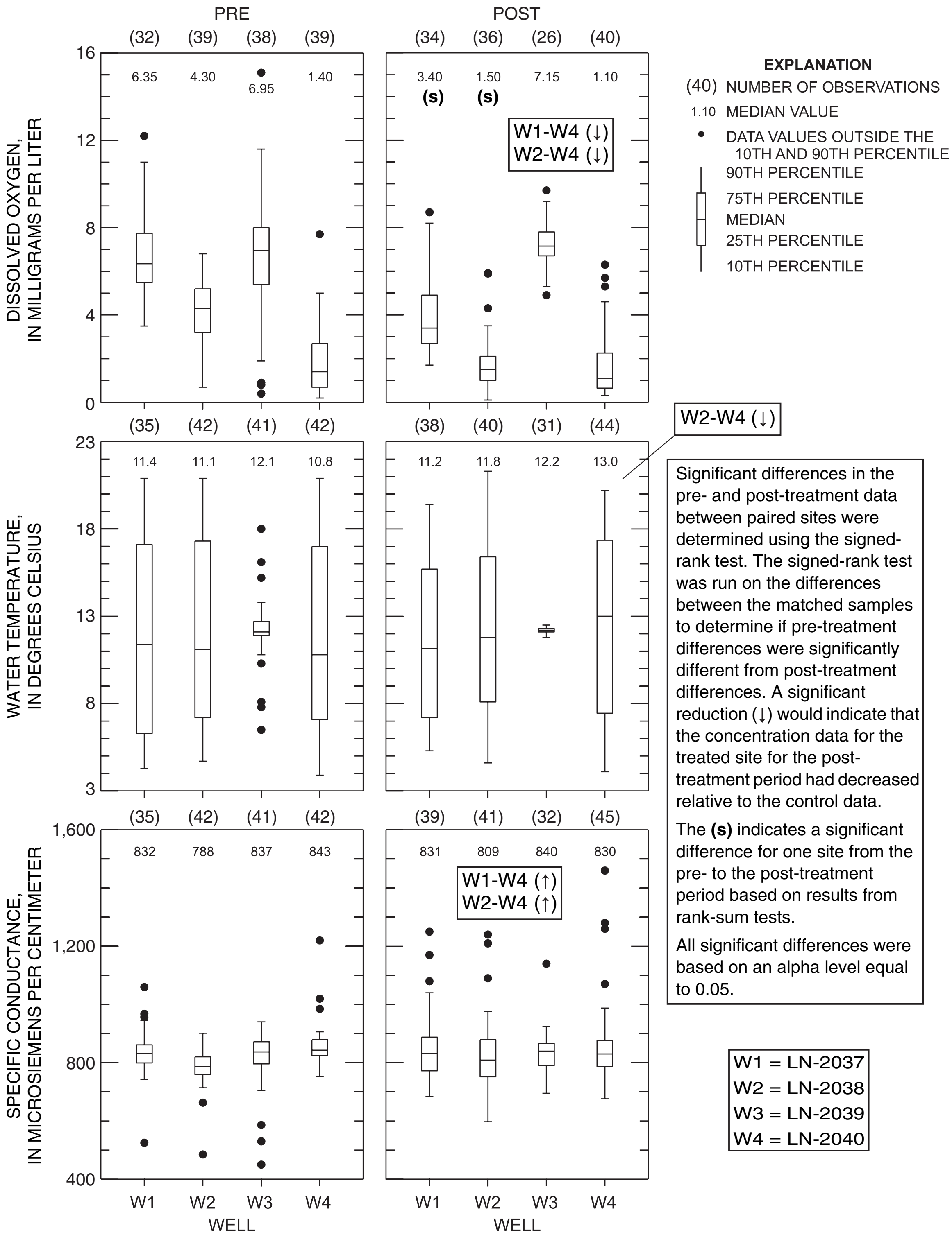


Figure 63c. Ranges of constituents for ground-water samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at the T-2 well nest located in the Big Spring Run Basin, Lancaster County, Pa.—Continued

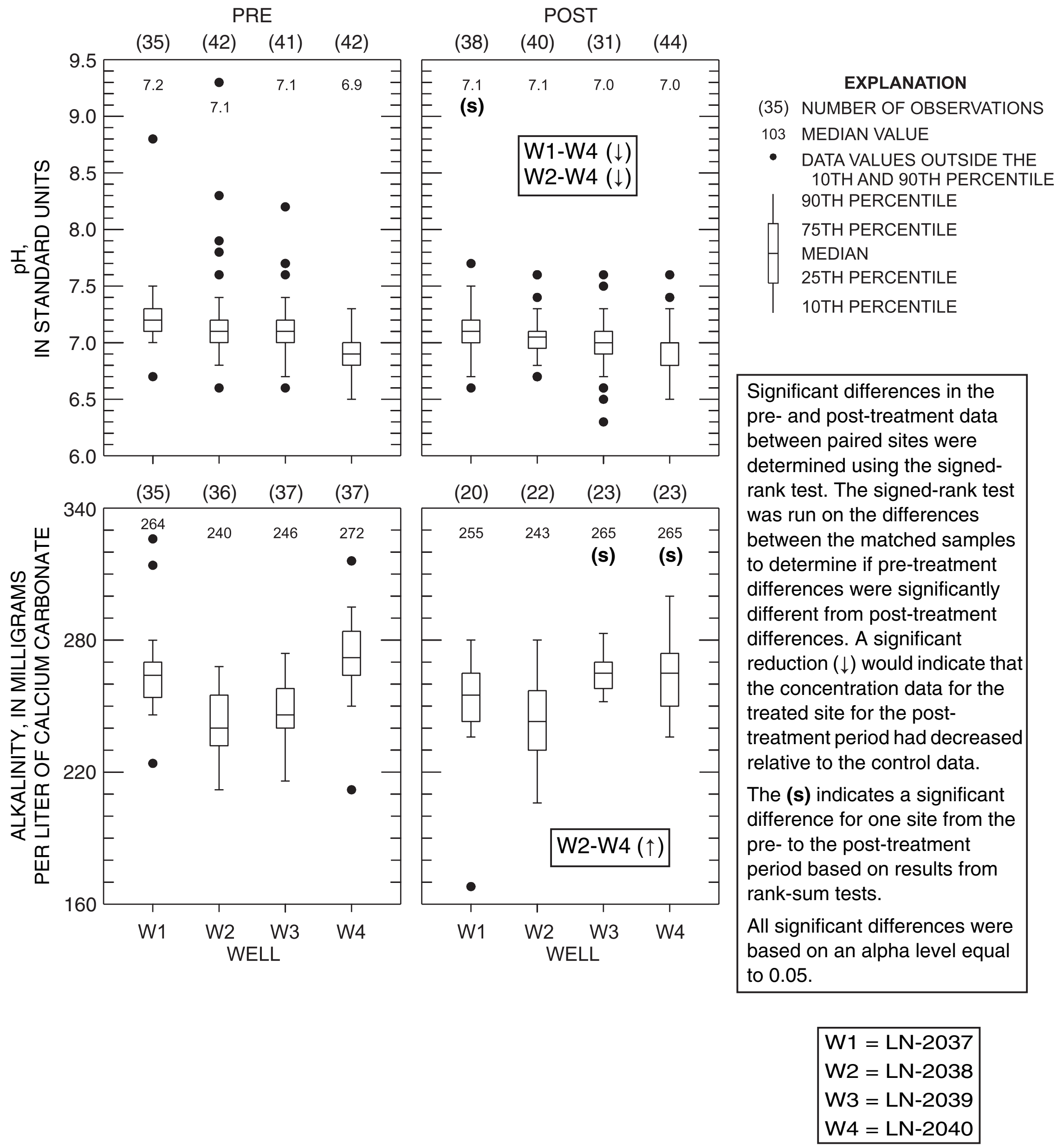


Figure 63d. Ranges of constituents for ground-water samples collected during the pre- and post-treatment periods from October 1993 through July 2001 at the T-2 well nest located in the Big Spring Run Basin, Lancaster County, Pa.—Continued

Almost all the ground-water samples were analyzed for dissolved nitrate plus nitrite. Because concentrations of dissolved nitrite were consistently very small compared to nitrate concentrations, the analyzed concentrations of nitrate plus nitrite are referred to as dissolved nitrate for all samples. Concentrations of dissolved nitrate in 576 samples ranged from the detection level to 59.8 mg/L as N; the highest concentration was measured in well LN-2040. Median concentrations (medians reported are either for the pre- or post-treatment period) ranged from 0.025 in well LN-2042 to 23.0 mg/L as N in well LN-2039 (figs. 62a and 63a). Relatively strong seasonal effects were observed at wells that contained water with elevated concentrations of dissolved nitrate. Commonly, concentrations of dissolved nitrate were lower in the summer and higher in the winter. For well LN-2039, concentrations of dissolved nitrate tended to be lowest in the spring and highest in the fall, but for well LN-2043, the reverse was true.

In general, ground water unaffected by contaminant sources contains less than 2 mg/L of nitrate as N (U.S. Geological Survey, 1999, p. 34). Ground water with concentrations of nitrate N greater than 2 mg/L may indicate anthropogenic sources such as fertilizer. Concentrations of nitrate N greater than 2 mg/L were consistently measured in the ground-water samples from all the wells at T-2 and in about 75 percent of the ground-water samples collected from well LN-2043 at T-1. Almost all the water with nitrate concentrations greater than the USEPA maximum contaminant level (MCL) of 10 mg/L as N were collected from wells at T-2. On the basis of N isotope samples collected from the Big Spring Run Basin, the major source of N, and ultimately nitrate, was animal manure.

Nitrite represents a short-lived intermediate oxidation product in the nitrification of ammonia to nitrate. Dissolved nitrite was measured in 267 ground-water samples. Concentrations of dissolved nitrite ranged from the detection limit to 1.1 mg/L as N (LN-2038). Except for well LN-2038, most wells contained dissolved nitrite at concentrations of 0.1 mg/L as N or less (figs. 62a and 63a).

Concentrations of dissolved P ranged from the detection limit to 2.01 mg/L in well LN-2041. More than 75 percent of the 267 ground-water samples collected contained dissolved phosphorus at concentrations of 0.04 mg/L or less (figs. 62b and 63b).

Fecal-streptococcus bacteria were detected at some time in all the wells. The highest counts (62,000 col/100 mL) out of 611 ground-water samples collected were reported in the water from wells LN-2037, LN-2038, and LN-2040 (figs. 62b and 63b). A weak seasonal pattern was identified in the fecal-streptococcus bacteria count. Typically, counts were greatest in the summer and lowest in the winter.

Well installation appeared to have an effect on ground-water quality. Although all of the monitor wells were installed by November 11, 1993, elevated concentrations of alkalinity, dissolved ammonia, DKN, dissolved phosphorus, and SC were measured into the fall of 1994. During the pre-treatment period, 75 percent of the maximum values reported for the analyzed constituents were during the first year of monitoring.

Major summer droughts occurred during the summers of 1995, 1997, 1998, and 1999; the summer drought of 1998 extended into the winter. Water levels in wells at T-2 indicated that during these periods the stream was a source of ground-water recharge. Under such dry field conditions, concentrations of various forms of N as well as P can build up in the soil. For example, organic N oxidizes to ammonia, which can then sorb to unsaturated-zone clay particles in the form of ammonium. Sorbed ammonium oxidizes to soluble nitrate that moves easily with infiltration water through the soil column to the water table. Also during dry periods, organic N in soil can be mineralized to nitrate. If a major storm event moves through after a prolonged dry period, a “flushing” effect can occur. This effect was observed following the passage of the remnants of Hurricane Floyd in September 1999, at which time 6.7 in. of precipitation were recorded in the study basin after 7 consecutive months of below-normal precipitation. For the chemical constituents of the post-treatment period, 40 percent of the maximum values were within 6 months of Floyd’s passing.

Relation to Agricultural Activities

The transport of nutrients contained in manure to the water table is a dynamic and complex process and is dominated by three variables: (1) amount of N, (2) soil type, and (3) amount of water. Bacterial processes that convert organic N to ammonium, nitrate, and N gas are also important but are difficult to quantify.

Manure Application

The amount of N deposited in the study area from manure was determined by combining the reported application rates by farmers with the amount deposited by animals in pasture. Organic N, which enters the soil through animal manure, eventually can be mineralized to nitrate and/or ammonium. Nitrate is the most common form of dissolved N in ground water in this study area. Nitrate is very soluble, moving freely with ground water under most environmental conditions (Freeze and Cherry, 1979).

Not all the N from applied manure, however, reaches the ground-water system. Hall and Risser (1993) estimated about 40 percent of the surface-applied N to a field site in Lancaster County, Pa., was lost through ammonia volatilization. Parsons and others (1995) also could explain some of the N losses or gains in ground water through overland and subsurface runoff processes. Also, any biological uptake of nitrate or ammonia would inhibit transport to the ground-water system.

Although nitrate from manure applications is transported with recharge to the ground water throughout the year, there does appear to be a seasonal effect. Concentrations of dissolved nitrate in the study area generally were greater in the winter than in the summer. This difference may in part be the result of when the manure was applied. In the study area, the greatest amount of manure was applied in the winter. Another factor is that biological uptake of nitrate is greater in the summer months, hence less available nitrate N to transport to ground-water systems. Also, recharge to ground-water systems typically is less in the summer because of evapotranspiration. Decreased recharge helps to keep nitrate in the soil matrix, with subsequent leaching of nitrate to ground-water systems once recharge is increased. Moog and Whiting (2002) found a similar pattern for ground water; Koerkle and others (1996) and Unangst (1992) found concentrations of nitrate N to be higher in the winter for surface-water base flow.

Soil Type

Soils in the study area act as a sink for the temporary storage of N. Non-biologically derived ammonia forms through a mineralization process in the soil. 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 N 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; however, plants can also uptake ammonium (Larcher, 1983).

N in soils exists in three phases: in soil air, bound to soil particles and organic compounds, and in soil water. Koerkle and others (1996), Gerhart (1986), and Shuford and others (1977) have suggested that limestone soils can be viewed as having dual porosity, which consists of micropores and macropores. Soils that have slow infiltration rates are typically clay rich and dominated by micropores that impede nitrate movement, which can result in lower than expected nitrate concentrations in ground water over the short term. Slow water drainage from these soils corresponds to longer periods of saturation than faster-drained soils. Saturated soils tend to be more oxygen deficient, potentially leading to denitrification. Through tilling or by permitting vegetation to grow, macropores can develop. Once a macropore has been developed, it may expand through subsurface erosion associated with rapidly moving infiltration water. Macropore movement of nitrate to ground water would lead to less lag time between manure applications and nitrate reaching the ground-water system.

P, unlike nitrate, is bound in the soil and is not easily transported to the ground-water system unless macropore flow exists. The soil binds P until binding sites are saturated with orthophosphate or other anions (Nagpal, 1986). Once all available sites are bound with P, unbound P is leached from the soil matrix and eventually this P-rich water recharges the ground-water system. Also, if P is bound to soil, it can be transported to ground water through macropores. P tends to adsorb to particles that are more easily transported through the macropores than the micropores (Addiscott and others, 2000).

Relation to Storm Events

The ground-water sampling design was not geared to sample storm events. However, a general review of the water-quality data and storm events suggested a possible correlation with ammonia and nitrate. To evaluate the effect of storm events on ground-water-quality data, precipitation records were reviewed to identify samples collected within 24 hours of storm events that produced 0.5 in. or greater of rainfall. The data set was limited to three storm events, one during the pre- and two during the post-treatment period. Concentrations of ammonia significantly greater than the median were reported for all three events for well LN-2042. Concentrations of dissolved nitrate were greater than the median for the two post-treatment storm events at wells LN-2037, LN-2038, and LN-2039 but were not significantly different for the storm during the pre-treatment period.

To increase the sample population, storm events that produced 0.5 in. or more of precipitation up to 5-days prior to ground-water-quality sampling were reviewed. Several trends were observed. First, concentrations of dissolved ammonia greater than the median value occurred infrequently (table 31). Second, concentrations of dissolved nitrate commonly exceeded median values. The lower concentrations of dissolved ammonia could be caused by ammonia volatilization or ammonium adsorption to soil colloids (Meisinger and Jokela, 2000). Ammonia tends to volatilize if exposed to the atmosphere, so it may not accumulate in the soil matrix as nitrate would. Also, as water moves through the subsurface zones, ammonium can be adsorbed to soil colloids that have available cation exchange sites. Nitrate can form through mineralization of organic N. During periods with no precipitation, nitrate can accumulate in the soil, and eventually this nitrate can be transported to the water table when recharge occurs. Nitrate does not bind to soil colloids to the extent that ammonium does; thus, nitrate is known as a conservative ion.

Table 31. Precipitation events in the Big Spring Run Basin, Lancaster County, Pa., that produced a minimum of 0.5 inch up to 5 days prior to sampling and the percent of ammonia and dissolved nitrate for ground-water wells that exceeded the median value.

[SS/CC - (number of samples that exceeded median well value)/(number of samples collected per well); NH_3, dissolved ammonia; NO_3+NO_2, dissolved nitrate plus nitrite]

Well identification number	NH_3 (SS/CC)	NO_3+NO_2 (SS/CC)	Percent exceedance	
			NH_3	NO_3+NO_2
LN-2037	3/12	12/21	25	57
LN-2038	4/14	11/21	29	52
LN-2039	5/12	8/19	42	42
LN-2040	5/13	10/21	28	48
LN-2041	4/14	14/20	29	70
LN-2042	8/14	10/22	57	45
LN-2043	4/10	8/15	40	53
LN-2044	7/14	17/22	50	77

Relation to Water Levels

Burkart and others (1999) found that seasonally high water-table depth had the strongest relation to nitrate concentration. The positive correlation suggests the lowest nitrate concentrations generally were associated with the shallowest water-table depth. Shallow water tables generally reflect poorly drained soils and anaerobic conditions. Under these conditions, denitrification of nitrate can occur in the presence of organic carbon and denitrifying bacteria.They noted also that other factors such as land use, hydrogeology, and climate are also needed to explain the transporting of nitrate to ground water. Goolsby and others (2001) found concentrations of nitrate tended to be highest in the spring when stream flow was highest and associated it with leaching of nitrate from the soil during periods of high rainfall and subsequent recharge to the water table.

To evaluate water levels and their effect on selected nutrients, the data from well LN-2042 were separated into base-flow and recharge groups. Forty-three samples were categorized as base flow, and 44 samples were categorized as recharge. Samples categorized as base flow were collected at least 2 weeks after any major ground-water recharge event. A major recharge event was defined as being a rise of 0.5 ft in the shallow well. No significant increase (or decrease) in nutrients or other water-quality constituents in well LN-2042 was observed. Additional water-quality analyses were performed on all wells to compare the water level in wells to water quality, and again no significant relations were identified.

Relation to Streambank Fencing

Samples were grouped as pre-treatment if they were collected prior to July 15, 1997, and post-treatment if collected after that date. On the basis of the location of the fence and the placement of wells, it is possible to evaluate wells at T-1 and T-2 under two monitoring designs: (1) paired-wells, and (2) before and after fence installation.

Paired wells

The goal of using a paired-well design was to establish a relation between a control well (well outside of the fenced area) and treatment wells (wells within the fenced area) and to determine if a change has occurred because of the treatment method. During the calibration period, land use at both sites remained relatively constant. At the end of the calibration period, the treatment method (fencing) was implemented.

The paired-well design at T-1 utilized well LN-2044 as the control well and wells LN-2041 and LN-2042 as the treatment wells. Well LN-2043 was excluded from the paired-well design because it was completed much deeper than the other wells (LN-2043 was 100 ft deep and the other wells at T-1 were 6 to 12 ft deep (table 3)) and its water chemistry was determined to be significantly different from the water in wells LN-2041, LN-2042, and LN-2044 for many of the characteristics measured (pH, SC, alkalinity, and DO). There was a caveat, however, to this pairing. Water-level data indicated wells LN-2041 and LN-2042 were receiving recharge from the creek (fig. 57).

Well LN-2040 was the control well for wells at T-2 and wells LN-2037 and LN-2038 were the treatment wells. Water levels for wells at T-2 indicated ground-water flowed towards the creek and water sampled from well LN-2040 had yet to reach the treatment area (fig. 58). Well LN-2039 was excluded from the paired-well design also because it was completed much deeper than the other wells (LN-2039 was 63 ft deep and the other wells at T-2 were 6 to 8 ft deep (table 3)) and because water-level altitudes in this well were greater (higher hydrologic head), which indicated it was not open to the same flow system as the shallow wells (closed or confined versus unconfined) at T-2.

Prior to any statistical analysis, the data were screened to eliminate elevated outliers. This decision was based on (1) the effects of well construction-during the pre-treatment period, 75 percent of the maximum values reported for the analyzed constituents were during the first year of monitoring, and (2) the effects of the remnants of Hurricane Floyd - during the post-treatment period, 40 percent of the maximum values reported for the analyzed constituents were within 6 months of Floyd’s passing. Data that exceeded the 90th percentile were considered elevated outliers and eliminated from statistical analysis of the paired-well design.

After the initial screening to eliminate elevated outliers, ANCOVA was performed. Regression lines relating the treatment data to the control data were generated for each of the paired wells. The general form of the equation is given in equation 1. A difference in the regression lines between paired wells indicated a change in water quality because of land treatment (Grabow and others, 1999). For this analysis, a confidence interval of 0.10 was set as the criteria for model acceptance. If a significant model was identified for the relation, fencing did change water quality. Predicted (least-square) mean values for the paired wells were also determined and were based on the regression relation to the control data. The least-square means for pre- and post-treatment data were then used to determine a percentage change between paired wells for specific constituents.

The results of the ANCOVA and the least-square means are presented in table 32, which shows fence installation affected ground-water quality. Shaded areas indicate treatment was determined not to be a significant factor in the change of ground-water quality. Considerable variation existed between wells at each site despite their close proximity and similar treatment method (table 32).

The relative decrease in water temperature from the pre- to post-treatment period in all treatment wells (table 32) was the result of the shading and insulating capacities of the grass cover. More shade would tend to reduce summer soil temperatures, which could equate to cooler, shallow well water, and more insulation from grass would also reduce the overall temperature extremes in summer and winter.

Table 32. Paired well comparisons and percentage change in constituent values in treatment wells derived from analysis of covariance for the lower 90 percent of data (values greater than the 90th percentile were removed because of outliers relating to well installation) between pre- and post-treatment periods in the Big Spring Run Basin, Lancaster County, Pa. (Wells LN-2037 and LN-2038 were the treatment wells for site T-2, wells LN-2041 and LN-2042 were the treatment wells for site T-1; wells LN-2040 and LN-2044 were the respective control wells)

[Shaded, no treatment affect; DKN, dissolved ammonia plus organic nitrogen]

Constituent	LN-2041/LN-2044	LN-2042/LN-2044	LN-2037/LN-2040	LN-2038/LN-2040
Dissolved oxygen	25.3	32.7	-40.8	-59.5
Specific conductance	1.02	31.5	1.22	4.59
Temperature	-8.50	-2.84	-3.60	-4.53
pH	.42	.22	-2.10	-.95
Alkalinity	9.90	8.90	-.25	2.10
Dissolved ammonia	38.7	36.3	-49.0	-52.3
DKN	30.6	31.3	-66.0	-16.6
Dissolved nitrate plus nitrite	46.8	-16.2	4.05	-8.60
Dissolved nitrite	36.8	17.1	-52.5	-69.7
Dissolved phosphorus	14.9	-30.4	-62.9	128
Fecal streptococcus	21.5	-9.80	-79.9	-31.6

The relative decrease in DO concentrations and pH in the treatment wells at site T-2 from the pre- to post-treatment period (table 32) was probably the result of increased plant respiration within the fenced area. According to Larcher (1983), oxygen in the soil is consumed by the respiration of the plant roots, soil animals, and aerobic micro-organisms. After the air in the soil is consumed, anaerobic micro-organisms begin to multiply, which may result in the formation of acids, hence, explaining the decrease in pH.

SC and alkalinity showed relative increases from the pre- to post-treatment period for at least one treatment well at both locations (table 32). These changes may be the result of changes in ground-water flow paths from the pre- to post-treatment period. At T-2, the treatment wells intercepted ground water that was down slope of well LN-2040 with correspondingly longer flow paths, so it may be that during the post-treatment period and the reduced amounts of recharge, ground-water flow paths changed enough so that the pre-treatment relation between wells LN-2037 and LN2038 to well LN-2040 was changed to another flow-path regime that transported more dissolved ions to the treated wells as opposed to the control well. This would also explain the slightly higher (but significant) increase in alkalinity from the pre- to post-treatment period at well LN-2038 relative to well LN-2040. The SC and alkalinity increases evident for treatment wells at T-1 may also be related to changes in the ground-water flow paths; however, these wells were found to be in an area that usually acts as a ground-water recharge area (fig. 57). Therefore, it is difficult to say if the treatment wells (LN-2041 and LN-2042) were hydrologically connected to well LN-2044. On the basis of contours on figure 57, there does not appear to be a direct connection. That is, water passing through the treatment wells appears to bypass LN-2044, and vice versa. Thus, if there were changes in ground-water flow paths from the pre- to post-treatment period (and this likely occurred during the drought periods identified earlier), the changes could have caused a relative increase in SC at treated wells and alkalinity (for well LN-2038) from the pre- to post-treatment period. SC data for low-flow samples collected at T-1 did not show a significant increase from the pre- to post-treatment period (fig. 8e); thus, the surface-water system did not appear to cause the relative SC increase at wells LN-2041 and LN-2042.

The relative decrease in dissolved ammonia, DKN, and dissolved nitrite from the pre- to post-treatment period for the treatment wells at T-2 can be explained in part by the reduction in manure and urine deposition near the wells relative to the pre-treatment period. From table 32, it appears that all or most of the relative reduction in DKN could be attributed to a reduction in dissolved ammonia.

Significant changes in concentrations of dissolved nitrate only occurred at T-2 (table 32); however, the treatment wells showed opposite trends from the pre- to post-treatment period relative to the control well. None of the shallow wells at T-2 showed a significant change in concentrations of dissolved nitrate from the pre- to post-treatment period (fig. 63a). The relative increase in the concentration of dissolved nitrate at well LN-2037 may be related to changes in the ground-water flow paths from the pre- to post-treatment period (see discussion above). The relative decrease in the concentration of dissolved nitrate at well LN-2038 could be somewhat related to well depths. Well LN-2038 was completed 0.6 ft deeper than well LN-2037 (table 3). Data from well LN-2039 (the deep well at T-2) indicated that concentrations of dissolved nitrate significantly decreased from the pre- to post-treatment period (fig. 63a). Thus, it may be that well LN-2038 intercepted some of this deeper water, which did have lower concentrations of nitrate (relative to the pre-treatment period), and this could have caused the slight relative reduction in nitrate concentrations from the pre- to post-treatment period for well LN-2038.

Differences in concentrations of dissolved P identified between paired wells from the pre- to post-treatment period had different causes at each site. Treated wells at both sites showed increases and decreases relative to control wells (table 32). At well T-1, LN-2041 showed a significant increase in the concentration of dissolved P from the pre- to post-treatment period (fig. 62b). This may have been caused by problems associated with well completion. The grout used to complete the well may have acted as a sink. That is, possibly the grout particles may have adsorbed P from water, possibly acting like a receptor for anions, and maybe, over time, this dissolved P was released during the post-treatment period. At T-2, wells LN-2038 and LN-2040 showed significant increases from the pre- to post-treatment period (fig. 63b), and it appeared the agricultural fields upgradient of the wells were acting as the source. Historically, the field upgradient of the T-2 well nest was planted in corn with rotational crops of alfalfa. It is believed soil in this field was nearing or had reached a maximum adsorption capacity for P; therefore, as more manure was applied to the field, retention sites for P were becoming limited, and some of the P in manure began to be transported to subsurface zones where it eventually was transported downgradient to the shallow wells at T-2.

The relative increase in fecal streptococcus from the pre- to post-treatment period at site T-1 for well LN-2041 may be the result of poor well construction. The water collected in this well was constantly cloudy and the filters (fecal-streptococcus samples were processed by vacuum filtrating water through a holding device with a gridded piece of 0.45 micron filter paper, with the filter paper placed on prepared agar and allowed to incubate) used to plate the bacteria samples clogged up almost immediately unless the sample water was diluted.

The treatment wells at T-2 showed decreases in fecal-streptococcus colonies relative to the control well during the post-treatment period; however, only the decrease at wells LN-2038 was related to fence installation (table 32). Decreased numbers of fecal streptococcus at the treated wells was somewhat expected because the cows in the pasture could not excrete waste near the wells inside the fence. There was ample visual evidence that the control well at T-2 outside the fence was visited by the dairy herd after fence installation.

Pre- and Post-Treatment Comparisons

The goal of using a pre- and post-treatment comparison was to evaluate the effects of establishing a fence and the development of a riparian border. Possible variations in well construction and ground-water-flow paths that existed between paired wells would thus be eliminated.

The establishment of a riparian border along the stream channel at site T-1 had minimal effect on pH at three of the four wells, and only the water in deep well LN-2043 changed significantly from the pre- to post-treatment period. The pH change (and any other change) at well LN-2043 can not be attributed to fence installation because the well depth was 100 ft and it was likely the contributing area to the well extended well beyond the fence boundaries. The pH decreased as well location increased in distance away from the stream, and this occurred under both pre- and post-treatment periods. The elevated pH of 8.8 at well LN-2041 appeared to be related to grout contamination; however, this was measured several years into the post-treatment period (fig. 62d).

The establishment of a riparian border affected pH at wells at site T-2. Values of pH significantly decreased at well LN-2037 during the post-treatment period (fig. 63d). As stated earlier, this pH decrease may be related to increased plant respiration in the fenced area. Elevated pH values in wells LN-2037 and LN-2038 (8.8 and 9.3, respectively) were measured during the first year of monitoring. These elevated pH values were probably a by-product of well installation and possible grout contamination.

SC decreased in three of the four wells at site T-1 during the post-treatment period. The greatest change was at deep well LN-2043 where the median significantly decreased from 944 to 848 μS/cm (fig. 62c). SC values from samples collected in well LN-2042 were consistently lower than in the other wells for both periods (table 33). SC measurements from wells LN-2042 and LN-2044 closely track those obtained from surface-water samples after the fence was installed and until the arrival of the remnants of Hurricane Floyd in September 1999. After September 1999, SC in the water from well LN-2044 increased, most noticeably in the year 2001.

SC exhibited only minor variation at wells at site T-2 between the pre- and post-treatment periods (fig. 63c). The greatest change was at well LN-2038 where the median increased from 788 to 809 μS/cm. In many studies, SC is used as a relative indicator to determine residence times or length of ground-water flow between wells. At site T-2, the post-treatment period increase of SC at well LN-2038 suggests a change to a longer flow path. Under pre-treatment conditions, SC values for well LN-2038 were significantly lower than in wells LN-2039 and LN-2040 during the pre-treatment period; however, during the post-treatment period, no significant differences were found, which also supports a change in flow paths (table 34).

Water temperatures for both well nests did not show any significant differences between wells within each nest for either the pre- or post-treatment period (thus, no table is presented for water temperature). Before the fence was established, ground-water temperatures in shallow wells increased towards the creek at both sites (fig. 62c and fig. 63c). During the post-treatment period, ground-water temperatures in shallow wells at site T-2 decreased as flow moved towards

Table 33. Significant relations for pH, specific conductance, dissolved oxygen, and alkalinity for wells at surface-water site T-1 in the Big Spring Run Basin, Lancaster County, Pa., for the pre- and post-treatment periods.

[G, greater than; L, less than; NS, not significant; table is read from top down. Example, the specific conductance from well LN-2041 was significantly less than the specific conductance from well LN-2043; all significant differences were based on an alpha level equal to 0.05]

Well number	LN-2041	LN-2042	LN-2043	Well number	LN-2041	LN-2042	LN-2043
pH (pre-treatment)				pH (post-treatment)			
LN-2041				LN-2041			
LN-2042	NS			LN-2042	NS		
LN-2043	L	L		LN-2043	NS	NS	
LN-2044	NS	NS	G	LN-2044	NS	NS	NS
Specific conductance (pre-treatment)				Specific conductance (post-treatment)			
LN-2041				LN-2041			
LN-2042	G			LN-2042	G		
LN-2043	L	L		LN-2043	L	L	
LN-2044	NS	L	G	LN-2044	NS	L	G
Dissolved oxygen (pre-treatment)				Dissolved oxygen (post-treatment)			
LN-2041				LN-2041			
LN-2042	G			LN-2042	G		
LN-2043	L	L		LN-2043	G	NS	
LN-2044	NS	L	G	LN-2044	NS	NS	L
Alkalinity (pre-treatment)				Alkalinity (post-treatment)			
LN-2041				LN-2041			
LN-2042	G			LN-2042	G		
LN-2043	L	L		LN-2043	NS	L	
LN-2044	NS	L	G	LN-2044	G	NS	G

the creek. For wells at T-1, the temperature trend was not as distinctive, but the overall average water temperature (average of the two median values for wells LN-2041 and LN-2042 for the post-treatment period) for the two shallow wells adjacent to the creek was lower than the median value for well LN-2044 during the post-treatment period (fig. 62c). It should be noted that even though this form of analysis did not detect significant differences in water temperature between wells for either the pre- or post-treatment period, ANCOVA results did detect significant changes from the pre- to post-treatment period between treatment and control wells; thus, the relation between wells was significantly affected, even though the distributions overlapped enough such that no significant differences were detected when conducting grouped comparison tests.

From the pre- to post-treatment period, DO decreased significantly for three of the four shallow treatment wells (figs. 62c and 63c). This was likely the result of plants consuming oxygen in the soil through respiration. Both treatment wells at T-2 showed a significant decrease; the control well showed no significant change from the pre- to post-treatment period (fig. 63c). Under pre- and post-treatment conditions for wells at site T-2, DO tended to increase as ground water flowed toward the creek. Normally, because of longer residence times, one would expect the DO to decrease towards the creek. Because the opposite occurred at T-2, the increase in DO suggests the depth to ground water was very shallow and the soils and regolith were highly permeable with a good connection to the atmosphere. The shallow treatment wells at T-1 showed a varied DO response during the post-treatment period. Well LN-2042 showed a significant increase in DO concentration relative to well LN-2044 during the post-treatment period, and well LN-2041 did not show a significant increase relative to well LN-2044; however, DO concentrations for well LN-2041 decreased significantly from the pre- to post-treatment period (fig. 62c). In general, differences evident in DO concentrations between these shallow wells at T-1 during the pre-treatment period were not as evident during the post-treatment period. The median DO concentrations for the shallow wells at T-1 during the pre-treatment period ranged from 2.95 to 5.20 mg/L; the post-treatment median concentrations ranged from 2.9 to 3.6 mg/L. Thus, it appears that either

Table 34. Significant relations for pH, specific conductance, dissolved oxygen, and alkalinity for wells at surface-water site T-2 in the Big Spring Run Basin, Lancaster County, Pa., for the pre- and post-treatment periods.

[G, greater than; L, less than; NS, not significant; table is read from top down. Example, the pH from well LN-2037 was significantly greater than the pH from well LN-2040; all significant differences were based on an alpha level equal to 0.05]

Well number	LN-2037	LN-2038	LN-2039	Well number	LN-2037	LN-2038	LN-2039
pH (pre-treatment)				pH (post-treatment)			
LN-2037				LN-2037			
LN-2038	G			LN-2038	NS		
LN-2039	G	NS		LN-2039	NS	NS	
LN-2040	G	G	G	LN-2040	G	G	NS
Specific conductance (pre-treatment)				Specific conductance (post-treatment)			
LN-2037				LN-2037			
LN-2038	G			LN-2038	NS		
LN-2039	NS	L		LN-2039	NS	NS	
LN-2040	NS	L	NS	LN-2040	NS	NS	NS
Dissolved oxygen (pre-treatment)				Dissolved oxygen (post-treatment)			
LN-2037				LN-2037			
LN-2038	G			LN-2038	G		
LN-2039	NS	L		LN-2039	L	L	
LN-2040	G	G	G	LN-2040	G	NS	G
Alkalinity (pre-treatment)				Alkalinity (post-treatment)			
LN-2037				LN-2037			
LN-2038	G			LN-2038	NS		
LN-2039	G	NS		LN-2039	L	L	
LN-2040	L	L	L	LN-2040	NS	L	NS

changes in ground-water flow paths or effects from fencing were muting any differences evident in the DO concentrations between these wells.

Changes in alkalinities were measured between pre- and post-treatment conditions at all eight wells. For wells at site T-2, the alkalinity in deep well LN-2039 increased significantly, but decreased significantly in control well LN-2040 during the post-treatment period (fig. 62d). For wells at site T-1, the alkalinity in deep well LN-2043 and control well LN-2044 decreased significantly after the riparian buffer was established. Any changes in alkalinity could be related to changes in ground-water flow paths, or be related to changes in pH which could consume alkalinity in a buffering process. It was noted earlier that plant respiration could cause a pH decrease due to acid formation, and any pH decrease caused by this process could also consume alkalinity.

Dissolved Ammonia

Median concentrations of dissolved ammonia decreased in five of the eight wells after fencing was installed (figs. 62a and 63a). For wells LN-2037, LN-2038 (T-2) and LN-2043 (T-1), the decrease in ammonia was statistically significant. Concentrations of dissolved ammonia decreased in ground water with distance of the well away from the creek. The significant decrease for LN-2037 and LN-2038 can be attributed to fence installation and subsequent lack of access of the dairy herd to the land surface near wells LN-2037 and LN-2038. The order of magnitude decrease in dissolved-ammonia concentrations during the post-treatment period at well LN-2043 was unexpected, and probably related to plant uptake and the very shallow reservoir that supplies well LN-2043 with ground-water recharge. For all wells, dissolved-ammonia concentrations at or below the detection level ranged from 10 to 22 percent during the pre-treatment period and from 43 to 88 percent during the post-treatment period. As a side note, concentrations of ammonia for wells at T-1 were about an order of magnitude greater under pre- and post-treatment periods than for wells at T-2. This strongly indicates a difference (between the well nests at the different sites) in (1) ground-water flow paths, (2) geochemical environments, or (3) both.

Dissolved Ammonia Plus Organic Nitrogen (DKN)

Median concentrations of DKN increased in four of the eight wells after fencing was installed (figs. 62b and 63b). The significant increase in DKN at well LN-2041 and significant decrease in wells LN-2043 and LN-2044 at site T-1 were unexpected and indicate that ground-water flow paths are not well understood at site T-1. For wells at site T-2, concentrations of DKN increased during the post-treatment period at wells LN-2039 and LN-2040, but not significantly. DKN concentrations decreased in wells LN-2037 and LN-2038, the former representing a significant decrease. Concentrations of DKN were significantly lower in deep well LN-2039 than in the other wells at site T-2 under pre- and post-treatment conditions (table 35). This suggests that either 1) as water moved from the shallow to the deeper ground-water system at site T-2, there was sufficient time for most of the organic N to be effectively removed through mineralization and (or) bacterial consumption, and/or 2) LN-2039 was capturing water from a deeper regional system where other factors besides fencing were contributing to the overall chemistry of the well. DKN for wells at site T-2 during the pre-treatment period show that DKN actually increased towards the stream, with the reverse true under post-treatment conditions. This latter effect could also be attributed to dairy cows not being able to access the land surface at wells LN-2037 and LN-2038 during the post-treatment period.

Table 35. Significant relations for dissolved nitrate plus nitrite, nitrate, nitrite, ammonia, and ammonia plus organic nitrogen for wells at surface-water site T-2 in the Big Spring Run Basin, Lancaster County, Pa., for the pre- and post-treatment periods.

[G, greater than; L, less than; NS, not significant; DKN, dissolved ammonia plus organic nitrogen; table is read from top down. Example, the concentration of dissolved nitrate from well LN-2037 was significantly less than the concentration from well LN-2039 during the pre-treatment period; all significant differences were based on an alpha level equal to 0.05]

Well number	LN-2037	LN-2038	LN-2039	Well number	LN-2037	LN-2038	LN-2039
Dissolved nitrate plus nitrite (pre-treatment)				Dissolved nitrate plus nitrite (post-treatment)			
LN-2037				LN-2037			
LN-2038	NS			LN-2038	NS		
LN-2039	L	L		LN-2039	L	L	
LN-2040	NS	NS	G	LN-2040	NS	NS	G
Dissolved nitrate (pre-treatment)				Dissolved nitrate (post-treatment)			
LN-2037				LN-2037			
LN-2038	NS			LN-2038	NS		
LN-2039	L	L		LN-2039	NS	L	
LN-2040	NS	NS	G	LN-2040	NS	NS	G
Dissolved nitrite (pre-treatment)				Dissolved nitrite (post-treatment)			
LN-2037				LN-2037			
LN-2038	L			LN-2038	NS		
LN-2039	G	G		LN-2039	G	G	
LN-2040	NS	G	L	LN-2040	NS	G	L
Dissolved ammonia (pre-treatment)				Dissolved ammonia (post-treatment)			
LN-2037				LN-2037			
LN-2038	NS			LN-2038	NS		
LN-2039	NS	G		LN-2039	NS	NS	
LN-2040	NS	NS	NS	LN-2040	L	NS	NS
DKN (pre-treatment)				DKN (post-treatment)			
LN-2037				LN-2037			
LN-2038	NS			LN-2038	NS		
LN-2039	G	G		LN-2039	G	G	
LN-2040	NS	NS	L	LN-2040	NS	NS	L

Dissolved Nitrate

Median concentrations of dissolved nitrate decreased in six of the eight wells sampled after fencing was installed (fig. 62a and 63a). All wells at site T-1 exhibited a decrease in nitrate, with significant decreases in wells LN-2041, LN-2042, and LN-2044. The general pattern, of higher dissolved-nitrate concentrations in well LN-2043 and lower dissolved-nitrate concentrations in well LN-2042 at site T-1 did not change between pre- and post-treatment periods (table 36). The concentrations of dissolved nitrate for shallow wells at site T-1 were one to two orders of magnitude lower than the concentration for shallow wells at site T-2. This indicated the shallow hydrogeologic system between the two sites was not similar. The deeper systems at the two sites (as evident by the deep wells at T-1 and T-2) also indicated that the deeper hydrogeologic systems were not similar. The median concentrations of dissolved nitrate for all samples collected LN-2039 and LN-2043 were 21.0 and 3.34 mg/L, respectively.

For wells at site T-2, dissolved-nitrate concentrations increased in wells LN-2037 and LN-2038 during the post-treatment period (fig. 63a); however, dissolved-nitrate concentrations decreased in wells LN-2039 and LN-2040, with the decrease in the deep well (LN-2039) being significant. Figure 63a indicates that dissolved nitrate tended to decrease as shallow ground water flowed towards the stream under pre-treatment conditions, but increased under post-treatment conditions at site T-2. The changes observed in dissolved nitrate for the

Table 36. Significant relations for dissolved nitrate plus nitrite, nitrate, nitrite, ammonia, and ammonia plus organic nitrogen for wells at surface-water site T-1 in the Big Spring Run Basin, Lancaster County, Pa., for the pre- and post-treatment periods.

[G, greater than; L, less than; NS, not significant; DKN, dissolved ammonia plus organic nitrogen; table is read from top down. Example, the concentration of dissolved nitrate from well LN-2041 was significantly less than the concentration from well LN-2043; all significant differences were based on an alpha level equal to 0.05]

Well number	LN-2041	LN-2042	LN-2043	Well number	LN-2041	LN-2042	LN-2043
Dissolved nitrate plus nitrite (pre-treatment)				Dissolved nitrate plus nitrite (post-treatment)			
LN-2041				LN-2041			
LN-2042	G			LN-2042	G		
LN-2043	L	L		LN-2043	L	L	
LN-2044	NS	L	G	LN-2044	NS	L	G
Dissolved nitrate (pre-treatment)				Dissolved nitrate (post-treatment)			
LN-2041				LN-2041			
LN-2042	G			LN-2042	G		
LN-2043	L	L		LN-2043	L	L	
LN-2044	NS	L	G	LN-2044	NS	L	G
Dissolved nitrite (pre-treatment)				Dissolved nitrite (post-treatment)			
LN-2041				LN-2041			
LN-2042	NS			LN-2042	NS		
LN-2043	NS	NS		LN-2043	NS	NS	
LN-2044	NS	NS	NS	LN-2044	NS	NS	NS
Dissolved ammonia (pre-treatment)				Dissolved ammonia (post-treatment)			
LN-2041				LN-2041			
LN-2042	NS			LN-2042	NS		
LN-2043	G	G		LN-2043	G	G	
LN-2044	G	NS	L	LN-2044	G	G	L
DKN (pre-treatment)				DKN (post-treatment)			
LN-2041				LN-2041			
LN-2042	G			LN-2042	G		
LN-2043	G	G		LN-2043	G	G	
LN-2044	G	NS	L	LN-2044	G	G	L

treatment area were unexpected, but may be the result of increased ground-water discharge related to the rainfall associated with the remnants of Hurricane Floyd and other large storm events that followed a major drought (so called "flushing" effect).

Dissolved Nitrite

Detection levels for dissolved nitrite varied throughout the duration of the study. For some wells non-detects also represented a significant and in some cases the majority (plus 90 percent) of the reported results, hence making comparisons between pre- and post-treatment periods difficult. Despite the changing detection limits and large number of non-detects, post-treatment data were significantly different than pre-treatment data for a number of wells. Concentrations of dissolved nitrite were significantly lower from the pre- to post-treatment period in the water from wells LN-2042 and LN-2043 at T-1 (fig. 62a) and from all four wells at site T-2 (fig. 63a) during the post-treatment period. The decreasing trend in dissolved-nitrite concentrations at T-1 was not as evident as for T-2 wells. The significant decrease for wells at T-1 appeared to be more related to changes in detection limits. For non-detect data (for this and any other parameter collected for either surface water or ground water), the estimated concentration was determined by dividing the non-detect value by 2; thus, changes for T-1 could have been related to this estimation procedure. For shallow wells at T-2, dissolved-nitrite data had less non detects than T-1 wells. Wells LN-2041 and LN-2042 had over 50 percent of the samples at concentrations of dissolved nitrite of 0.005 mg/L as N or lower, with 0.005 mg/L the detection limit at least for part of the study period. For wells at T-2, wells LN-2037 and LN-2038 had over 10 percent (and less than 25 percent) and over 25 percent (and less than 50 percent), respectively, of samples that did not exceed 0.005 mg/L of dissolved nitrite as N. Thus, the decreasing trend in dissolved nitrite for the wells at T-2 appeared to be more reliable than the trend at T-1.

Dissolved Phosphorus

Evaluating changes in the concentrations of dissolved P between pre- and post-treatment periods for wells at sites T-1 and T-2 was difficult as a result of the large number of nondetects. Eighty to 95 percent of the samples collected for wells LN-2042, LN-2043, and LN-2044 and analyzed for dissolved-P concentrations were below the minimum detection level during the pre-treatment period. For well LN-2041, however, only 5 percent of the samples collected during the pre-treatment period were below the detection level. For wells at site T-2, the percent of nondetects during the pre-treatment period ranged from a low of 50 percent at well LN-2040 to a high of 90 percent at well LN-2039. During the post-treatment period for wells at site T-2, the range in nondetects went from a low of 7 percent at well LN-2040 to a high of 73 percent at well LN-2039. Figures 62b and 63b show that concentrations of dissolved P increased significantly from the pre- to post-treatment period in the water from well LN-2041 at site T-1 and wells LN-2038 and LN-2040 at site T-2. LN-2041 had significantly greater concentrations of dissolved P than all wells at site T-1 during the pre- and post-treatment periods (table 37). This could be caused by problems associated with well completion. For wells at site T-2, the shallow wells showed an increase in dissolved P relative to the deep well (LN-2039) from the pre- to post-treatment period (table 38). This indicated that there was a shallow source of dissolved P during the post-treatment period at site T-2. Increased concentrations of dissolved P was also evident in the surface water for low-flow samples collected at site T-2 during the post-treatment period (fig. 8d). Explanation for the increased dissolved-P concentra-

Table 37. Significant relations for dissolved phosphorus and fecal streptococcus for wells at surface-water site T-1 in the Big Spring Run Basin, Lancaster County, Pa., for the pre- and post-treatment periods.

[G, greater than; L, less than; NS, not significant; table is read from top down. Example, the concentration of dissolved phosphorus from well LN-2041 was significantly greater than the concentration from well LN-2043; all significant differences were based on an alpha level equal to 0.05]

Well number	LN-2041	LN-2042	LN-2043	Well number	LN-2041	LN-2042	LN-2043
Dissolved phosphorus (pre-treatment)				Dissolved phosphorus (post-treatment)			
LN-2041				LN-2041			
LN-2042	G			LN-2042	G		
LN-2043	G	NS		LN-2043	G	NS	
LN-2044	G	NS	NS	LN-2044	G	NS	NS
Fecal streptococcus (pre-treatment)				Fecal streptococcus (post-treatment)			
LN-2041				LN-2041			
LN-2042	G			LN-2042	G		
LN-2043	G	NS		LN-2043	G	NS	
LN-2044	G	NS	NS	LN-2044	G	NS	NS

Table 38. Significant relations for dissolved phosphorus and fecal streptococcus for wells at surface-water site T-2 in the Big Spring Run Basin, Lancaster County, Pa., for the pre- and post-treatment periods.

[G, greater than; L, less than; NS, not significant; table is read from top down. Example, The concentration of dissolved phosphorus from well LN-2039 was significantly greater than the concentration from well LN-2040; all significant differences were based on an alpha level equal to 0.05]

Well number	LN-2037	LN-2038	LN-2039	Well number	LN-2037	LN-2038	LN-2039
Dissolved phosphorus (pre-treatment)				Dissolved phosphorus (post-treatment)			
LN-2037				LN-2037			
LN-2038	NS			LN-2038	L		
LN-2039	NS	NS		LN-2039	G	G	
LN-2040	NS	NS	L	LN-2040	L	L	L
Fecal streptococcus (pre-treatment)				Fecal streptococcus (post-treatment)			
LN-2037				LN-2037			
LN-2038	NS			LN-2038	NS		
LN-2039	G	G		LN-2039	G	G	
LN-2040	NS	L	L	LN-2040	L	L	L

tion for low flow during the post-treatment period at site T-2 are given in the surface-water discussion.

Summary

The Big Spring Run Basin lies in a geologic zone that is structurally complex and characterized by repeated deformation, faulting, and folding. The Lower to Middle Cambrian rocks that underlie the basin are dominated by limestone; siliciclastics rocks comprise about 10 percent of the bedrock. The ground-water/surface-water system that has developed in the bedrock and regolith is complex and poorly understood. On the basis of water levels, flow directions, age dating, and chemical quality, the ground-water flow system appears to be controlled by the bedrock geology. The system itself, however, is driven by the timing, duration, and intensity of precipitation events.

Water-level altitudes for the well nest at site T-1 indicated this section of Big Spring Run is a losing stream, recharging the shallow and deep ground-water systems. Water-quality data, however, suggested that the stream is not in good hydraulic connection with the shallow or deep ground-water system and that water in the deep well (LN-2043) was significantly different from the water in the shallow wells (LN-2041, LN-2042, LN-2044). Water levels altitudes for the well nest at site T-2 indicated that this tributary to Big Spring Run gains water from its eastern bank, but loses water on its western bank, indicating this reach has more dynamic ground-water flow than at site T-1. Water-quality data at site T-2 indicated that the shallow wells (LN-2037, LN-2038, LN-2040) were in fairly good connection with the stream, but the deep well (LN-2039) was not. Water-quality data from both sites also indicated that a “flushing” effect can occur (40 percent of the maximum analyte values occurred within 6 months of the remnants of Hurricane Floyd’s passing) if a prolonged period of dry weather (drought) is followed by an intense storm event such as heavy rains from tropical storms.

Samples were grouped as pre-treatment if they were collected prior to July 15, 1997 and post-treatment if collected after that date. On the basis of the location of the fence and the placement of wells, it was possible to evaluate wells at sites T-1 and T-2 under two monitoring designs: (1) paired-wells and (2) and pre- (before) and post- (after) treatment (fence installation).

The paired-well design utilized well LN-2044 as the control well for site T-1 with LN-2041 and LN-2042 as the treatment wells. Well LN-2040 was the control well for site T-2 with LN-2037 and LN-2038 as the treatment wells. ANCOVA results from the paired-well comparison indicated that fence installation significantly affected shallow ground-water quality (table 39). Overall, when combining paired ANCOVA results for both of the treatment wells at each site, the concentrations of dissolved ammonia, DKN, dissolved nitrate, dissolved nitrite, along with pH, DO, and fecal-streptococcus counts decreased in the two shallow wells at site T-2 but increased in the shallow wells at site T-1 relative to control wells during the post-treatment period. The likely reason for the opposite trends in these constituents relative to control sites during the post-treatment period was the difference in shallow ground-water flow paths at both sites. The well nest at T-2 was in a zone where the stream was gaining water from the shallow ground-water system, while the opposite (stream loses to ground water) occurred at T-1. Therefore, at the T-2 well nest, the shallow ground water moves from outside the fence, through the fenced area, and into the stream. Typically at the T-1 well nest, water is lost from the stream, so it moves from the stream and into the fenced riparian zone where the “treatment” wells are located. The opposite trends in water-

Table 39. Paired well comparison summary showing percent change in constituent values during the post-treatment period based on analysis of covariance for shallow well pairs in the treatment basin of the Big Spring Run Basin, Lancaster County, Pa., (values greater than the 90th percentile were removed because of outliers relating to well installation) with wells LN-2037 and LN-2038 as the treatment wells for site T-2, LN-2041 and LN-2042 as the treatment wells for site T-1; LN-2040 and LN-2044 were the respective control wells.

[DKN, dissolved ammonia plus organic nitrogen; a positive value indicates that the treatment wells increased relative to the control well during the post-treatment period]

Constituent	T-1 well pairs	T-2 well pairs
Dissolved oxygen	29	-50
Specific conductance	16	2.9
Temperature	-5.7	-4.1
pH	.32	-1.5
Alkalinity	9.4	.92
Dissolved ammonia	38	-51
DKN	31	-41
Dissolved nitrate plus nitrite	15	-2.3
Dissolved nitrite	27	-61
Dissolved phosphorus	-7.8	33
Fecal streptococcus	5.8	-56

quality constituents for shallow wells closest to the stream for these two well nests suggest that, at T-1, the wells inside the fenced area, really were not treatment wells considering that water was moving from the stream into the shallow ground-water system. The reductions apparent for the shallow wells at the T-2 well nest would be somewhat expected given that the cows were not allowed to access the area near these wells after fence installation.

The only similarities between the treatment wells at the two well nests were the relative increases in SC and alkalinity and a relative decrease in water temperature during the post-treatment period (table 39). Establishment of the riparian zone in the treatment basin could cause water temperature to decrease in shallow wells at both nests, even though flow paths showed differences. The stream also showed a decrease in water temperature during the post-treatment period at both T-1 and T-2. The relative increase in SC and alkalinity for both well nests during the post-treatment period may be related to changes in shallow ground-water flow paths at each site due to decreased amounts of precipitation.

The one constituent which showed relative increases at the T-2 well nest and not at T-1 during the post-treatment period was dissolved P (table 39). It appeared that the source of the P was an upgradient field where animal manure had historically been applied. This source of P in the shallow ground-water system was also evident in low-flow stream samples collected at T-2.

Overall, it can be stated that streambank fencing had a positive impact on water quality in the shallow ground-water system that was contributing water to the stream system. The well depths at this site were no greater than 7 ft. Improvements were detected in water temperature, DO, N species, and fecal streptococcus.

Conclusions

This study in the Big Spring Run Basin of the Mill Creek Watershed was designed to determine the effects of streambank fencing of pasture land adjacent to stream channels on water quality and benthic macroinvertebrates using a paired-basin and upstream/downstream approach. Data were collected from 1993 to 2001 at eight surface-water sites and eight ground-water wells during a calibration and post-treatment period (each about 4 years in duration). Approximately 2 miles of stream were fenced in the treatment basin, with all near-stream pastures fenced in the treatment basin.

Besides fence installation, other changes occurred from the pre- to post-treatment period that could have affected the results of this study. Precipitation during the post-treatment period averaged about 5 in. less per year and streamflow about 56-63 percent less than the pre-treatment period. Also, agricultural activity in the study basin did show some changes from the pre- to post-treatment period. There were 27 and 33 percent decreases in the estimated amount of N and P applied, respectively, to the land in the treatment basin as inorganic and organic fertilizers from the pre- to the post-treatment period. For the control basin, however, there was a 3 percent decrease and 7 percent increase in the estimated amount of N and P applied, respectively, over the same period. In both study basins, the number of pastured cows decreased during the post-treatment period, primarily during the latter part of the study. The control basin showed an approximate 50 percent decrease in numbers of cows pastured from WY1999 to WY2001, while the treatment basin showed a similar decrease from WY2000 to WY2001. The change in precipitation from the pre- to post-treatment period was accounted for due to the nested experimental design and types of statistical analyses. Changes in the number of cows pastured and nutrient applications during the post-treatment period could have affected results of this study. The effects of changes in nutrient applications are difficult to evaluate in karst terrains due to potentially large differences in time of application and when nutrients may reach the water table (and eventually the stream). The decrease in cow density in both basins also could have affected the results, but given that similar decreases were seen in both basins and the change occurred in the control basin earlier, one would expect any effect from this to reduce any treatment effects (relative to control sites).

It should be noted that even though cow density did decrease in the post-treatment period, the number of pastures remained the same.

Results from this study indicated that streambank fencing and the establishment of a 5- to 12-ft wide buffer strip along the 2 miles of stream resulted in decreases in N-species, total-P, and suspended-sediment concentrations and yields at the outlet (drainage area of 1.42 mi^2) of the treatment basin relative to untreated sites; however, dissolved-P concentrations and yields increased. The dissolved-P increase was thought to be caused by subsurface movement of dissolved P from an upgradient crop field. It is not possible to determine what the effects of fencing would be on dissolved P if this upgradient field were not acting as a source. It does emphasize that nutrient management, even in concert with streambank fencing, is important in helping to control nutrient loadings to streams in this agricultural setting.

Another site upstream in the treatment basin (drainage area of 0.36 mi^2) showed improvement only in suspended-sediment concentrations and yields. This site appeared to be directly downgradient from the field that was the source of dissolved P. The lack of improvement in other constituents was related to microscale processes occurring at this upstream site. Data indicate that streambank fencing effects should be studied at a large as scale as possible since microscale influences on water quality as drainage area decreases can mute fencing impacts.

Benthic-macroinvertebrate data collected during this study indicated that streambank fencing had a positive influence on benthic macroinvertebrates and their habitat. More improvement in both benthic-macroinvertebrate community structure and habitat were detected at the outlet of the treatment basin than at the upstream sites; however, in this case, there also was improvement at upstream sites. Numerous biological metrics were used to determine the effects of fencing, and most of them did indicate that fencing caused improvement. Probably the most important biological metric, taxa richness, indicated greater number of benthic-macroinvertebrate taxa at treated relative to control sites after fencing.

Results for shallow ground-water wells in the treatment basin indicated that fencing did improve shallow ground-water quality (except for dissolved P, which again was affected by the upgradient field acting as a P source) as noted by decreased concentrations of N species and fecal-streptococcus counts. This improvement in shallow wells in the treatment basin only occurred at the well nest for which the ground-water flow path dictated that water moved from the shallow ground-water system into the stream (a gaining stream reach).

For this study, given the small buffer width within the fenced area (5 to 12 ft), it was unclear to what extent water-quality changes would occur. Results of the study indicated that even a small buffer width can have a positive influence on surface-water quality, benthic macroinvertebrates, and near stream shallow ground-water quality. The results do show, however, that streambank fencing in itself cannot alleviate excessive nutrient inputs that may be transported through subsurface zones into the stream system. Overland runoff processes that move suspended sediment to the stream can be controlled (or reduced) to some extent by establishment of a vegetative buffer inside the fenced area.

Acknowledgments

Many land owners in the study area were kind enough to allow access to their property and to agree to work with project personnel in order to get fencing installed and also to provide pertinent agricultural data during the study period. A few land owners also allowed for instrumentation to be installed on their property and allowed unlimited access to project personnel to maintain the instruments and collect necessary samples.

The project could not have been conducted without the help of several agencies and individuals. Many technical meetings to discuss project direction and results were conducted with cooperative agencies, including PaDEP, USEPA, NRCS, and LCCD. Barbara Lathrop of PaDEP provided invaluable help with logistics over the course of the study.

Fence installation in the treatment basin could not have been conducted without the help of NRCS, LCCD, the U.S. Fish and Wildlife Service, and USGS personnel. Ducks Unlimited, along with NRCS (specifically Frank Lucas) also provided land owners with sapling trees for planting in newly fenced areas. Frank Lucas was the primary person in the field who insured that the fence installation was completed properly, and he installed much of the fencing.

The intensive collection of water-quality samples during the project was made possible only through the work of numerous USGS employees, including Edward Koerkle, Todd Creamer, Scott Kline, Todd Spanish, Michelle Koch, and Samuel Fishel. Agricultural data were collected primarily by USGS personnel and by Omer Brubaker of NRCS.

The authors would like to thank the technical reviewers of the report including Daniel Line of the North Carolina State University's Cooperative Extension Service, Barbara Lathrop of PaDEP, Douglas Chichester and Kent Crawford of the USGS, and Michael Bilger of the USGS who reviewed the benthic macroinvertebrate taxonomic list. Other USGS personnel involved were Kim Otto, who provided help in compiling the report and in editorial guidance; and James Bubb, who helped to generate numerous graphics included in the report.

References

Addiscott, T.M., Brockie, D., Catt, J.A., Christian, D.G., Harris, G.L., Howse, K.R., Mirza, N.A., and Pepper, T.J., 2000, Phosphate losses through field drains in a heavy cultivated soil: Journal of Environmental Quality, v. 29, no. 2, p. 522-532.

Alderfer, R.B., and Robinson, R.R., 1949, Runoff from pastures in relation to grazing intensity and soil compaction: Journal of American Society of Agronomy, v. 39, p. 948-958.

Algone.com, 2004, Nitrites in the aquarium: accessed April 13, 2003, at *http://www.algone.com/nitrites.htm*.

Andrews, E.D., 1978, Present and potential sediment yields in the Yampa River Basin, Colorado and Wyoming: U.S. Geological Survey Water-Resources Investigations Report 78-105, 33 p.

Barbour, M.T., Gerritsen, J., Snyder, B.D., and Stribling, J.B., 1999, Rapid bioassessment protocols for use in streams and wadable rivers—Periphyton, benthic macroinvertebrates and fish, 2nd ed.: Washington, D.C., U.S. Environmental Protection Agency, EPA-841-B-99-002, 213 p. + appendices.

Barden, C.J., 2001, Establishing riparian buffers: Kansas, Kansas State University, 4 p.

Barker, J.L., 1978, Bacteriological water quality of Tulpehocken Creek basin, Berk and Lebanon Counties, Pennsylvania: U.S. Geological Survey Water-Resources Investigations 78-53, 25 p.

Berg, T.M., and Dodge, C.M., eds., 1981, Atlas of preliminary geologic quadrangle maps of Pennsylvania: Pennsylvania Topographic and Geological Survey, 4th ser., Map 61, 636 p.

Berg, T.M., Edmunds, W.E., Geyer, A.R., Glover, A.D., Hoskins, D.M., MacLachlan, D.B., Root, S.I., Sevon, W.D., and Socolow, A.A., comps., 1980, Geologic map of Pennsylvania: Pennsylvania Geological Survey, 4th ser., Map 1, scale 1:250,000, 3 sheets.

Bryant, F.T., Blaser, R.E., and Peterson, J.R., 1972, Effect of trampling by cattle on bluegrass yield and soil compaction of a Meadowville Loam: Agronomy Journal, v. 64, p. 331-334.

Burkart, M.R., Kolpin, D.W., Jaquis, R.J., and Cole, K.J., 1999, Agrichemicals in ground water of the midwestern USA-Relations to soil characteristics: Journal of Environmental Quality, v. 28, no. 6, p. 1,908-1,915.

Busenberg, E., and Plummer, L.N., 1992, Use of chlorofluorocarbons (CCl_3F and CCl_2F_2) as hydrologic trackers and age-dating tools - Example - the alluvium and terrace system of central Oklahoma: Water Resources Research, v. 28, p. 2,257-2,284.

___, 2000, Dating young groundwater with sulfur hexafluoride - Natural and anthropogenic sources of sulfur hexafluoride: Water Resources Research, v. 36, p. 3,011-3,030.

Chesapeake Bay Program, 1992, Progress report of the bay-wide nutrient reduction reevaluation: U.S. Environmental Protection Agency, 68 p.

Clausen, J.C., Jokela, W.E., Potter III, F.I., and Williams, J.W., 1996, Paired watershed comparison of tillage effects on runoff, sediment, and pesticide losses: Journal of Environmental Quality, v. 25, p. 1,000-1,007.

Clausen, J.C., and Spooner, J., 1993, Paired watershed study design: Washington, D.C., U.S. Environmental Protection Agency, Office of Water, USEPA 841-F-93-009, 8 p.

Connecticut River Joint Commissions, 1998, River, banks, and buffers - Buffers for agriculture, no. 5: accessed Nov. 21, 2003, at *http://www.crjc.org/buffers/Buffers%20for%20Agriculture.pdf*.

Cook, P.G., and Solomon, D.K., 1997, Recent advances in dating young groundwater: chlorofluorocarbons, $^3H/^3He$, and ^{85}Kr: Journal of Hydrology, v. 191, p. 245-265.

Cooper, J.R., Gilliam, J.W., Daniels, R.B., and Robarge, W.P., 1987, Riparian areas as filters for agricultural sediment: Soil Science Society of Ameria Journal, v. 51, p. 416-420.

Custer, B.H., 1985, Soil survey of Lancaster county, Pennsylvania: U.S. Department of Agriculture, Soil Conservation Service, 152 p.

Dillaha, T.A., Reneau, R.B., Mostaghimi, S., and Lee, D., 1989, Vegetative filter strips for agricultural nonpoint source pollution control: TRANSACTIONS of the ASAE, v. 32, p. 513-519.

Drever, J.I., 1982, The geochemistry of natural waters: Englewood Cliffs, N.J., Prentice-Hall, Inc., 388 p.

Durlin, R.R., and Schaffstall, W.P., 1996, Water resources data, Pennsylvania, water year 1994, Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-94-2, 418 p.

___, 1997a, Water resources data, Pennsylvania, water year 1995, Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-95-2, 518 p.

___, 1997b, Water resources data, Pennsylvania, water year 1996, Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-96-2, 280 p.

___, 1998, Water resources data, Pennsylvania, water year 1997, Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-97-2, 439 p.

___, 1999, Water resources data, Pennsylvania, water year 1998, Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-98-2, 456 p.

___, 2000, Water resources data, Pennsylvania, water year 1999, Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-99-2, 444 p.

___, 2001, Water resources data, Pennsylvania, water year 2000, Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-00-2, 458 p.

___, 2002, Water resources data, Pennsylvania, water year 2001, Susquehanna and Potomac River Basins: U.S. Geological Survey Water-Data Report PA-01-2, 441 p.

Edwards, D.R., Daniel, T.C., Scott, H.D., Murdoch, J.F., Habiger, M.J., and Burks, H.M., 1996, Stream quality impacts of best management practices in a northwestern Arkansas basin: Water Resources Bulletin, v. 32, p. 499-509.

Ehlke, T.A., Ehrlich, G.G., Greeson, P.E., Kirkland, R.T., and Mallard, G.E., 1987, Part 1 - Description of methods - Bacteria, in Britton, L.J., and Greeson, P.E., eds., Methods for collection and analysis of aquatic biological and microbiological samples: U.S. Geological Survey Techniques of Water-Resources Investigations, book 5, chap. A4, p. 3-98.

Ekwurzel, B., Schlosser, P., Smethie, W., Jr., Plummer, L.N., Busenberg, E., Michel, R.L., Weppernig, R., and Stute M., 1994, Dating of shallow groundwater - Comparison of the transient tracers, $^{3}H/^{3}He$, chlorofluorocarbons and ^{85}Kr: Water Resources Research, v. 30, p. 1,693-1,708.

Fennessey, M.S., and Cronk, J.K., 1997, The effectiveness and restoration potential of riparian ecotones for the management of nonpoint source pollution, particularly nitrate: Critical Reviews in Environmental Science and Technology, v. 27, no. 4, p. 285-317.

Fishman, M.J., and Friedman, L.C., eds., 1989, Methods for determination of inorganic substances in water and fluvial sediments: U.S. Geological Survey Techniques of Water-Resources Investigations, book 5, chap. A1., 545 p.

Focazio, M.J., Plummer, L.N., Bôhlke, J.K., Busenberg, E., Bachman, L.J., and Powars, D.S., 1998, Preliminary estimates of residence times and apparent ages of ground water in the Chesapeake Bay Watershed, and water quality from a survey of springs: U.S. Geological Survey Water-Resources Investigations Report 97-4225, 75 p.

Fokkens, B., and Weijenberg, J., 1968, Measuring the hydraulic potential of ground water with the hydraulic potential probe: Journal of Hydrology, v. 6, p. 306-313.

Freeze, R.A., and Cherry, J.A., 1979, Groundwater: Englewood Cliffs, N.J., Prentice-Hall, Inc., 604 p.

Friedman, L.C., and Erdmann, D.E., 1982, Quality assurance practices for the chemical and biological analyses of water and fluvial sediments: U.S. Geological Survey Techniques of Water-Resources Investigations, book 5, chap. A6, 181 p.

Gauch, H.G., Jr., 1982, Multivariate analysis in community ecology: New York, Cambridge University Press, 298 p.

Gere, J.M., and Weaver, W., Jr., 1965, Matrix algebra for engineers: New York, D. Van Nostrand Company, 168 p.

Gerhart, J.M., 1986, Ground-water recharge and its effects on nitrate concentrations beneath a manured field site in Pennsylvania: Ground Water, v. 24, no. 4, p. 483-489.

Gerhart, J.M., and Lazorchick, G.J., 1988, Evaluation of the ground-water resources of the Lower Susquehanna River Basin, Pennsylvania and Maryland: U.S. Geological Survey Water-Supply Paper 2284, 128 p.

Goolsby, D.A., Battaglin, W.A., Aulenbach, B.T., and Hooper, R.P., 2001, Nitrogen input to the Gulf of Mexico: Journal of Environmental Quality, v. 30, no. 2, p. 329-336.

Grabow, G.L., Spooner, J., Lombardo, L.A., and Line, D.E., 1999, Detecting water quality changes before and after BMP implementation: Use of SAS for statistical analysis: NWQEP notes, The NCSU Water Quality Group Newsletter, North Carolina State University, Raleigh, North Carolina, p. 1-11.

Guy, H.P., 1969, Laboratory theory and methods for sediment analysis: U.S. Geological Survey Techniques of Water-Resources Investigations, book 5, chap. C1, 58 p.

Hall, D.W., and Risser, D.W., 1993, Effects of agricultural nutrient management on nitrogen fate and transport in Lancaster County, Pennsylvania: Water Resources Bulletin, v. 29, no. 1, p. 55-76.

Helsel, D.R., and Hirsch, R.M., 1992, Statistical methods in water resources: Amsterdam, Elsevier Science Publishing Co., 522 p.

Hem, J.D., 1985, Study and interpretation of the chemical characteristics of natural water: U.S. Geological Survey Water-Supply Paper 2254, third edition, 263 p.

Jacobs, T.C., and Gilliam, J.W., 1985, Riparian losses of nitrate from agricultural drainage waters: Journal of Environmental Quality, v. 14, no. 4, p. 472-478.

Jongman, R.H.G., ter Braak, C.J.F., and Van Tongeren, O.F.R., eds., 1995, Data analysis in community and landscape ecology: New York, Cambridge University Press, 299 p.

Kauffman, J.B., Krueger, W.C., and Vavra, M., 1983, Impacts of cattle on streambanks in northeastern Oregon: Journal of Range Management, v. 36, p. 683-685.

Klemm, D.J., Lewis, P.A., Fulk, F., and Lazorchak, J.M., 1990, Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface waters: Cincinnati, Ohio, EPA/600/4-90/030, Aquatic Biology Branch and Development and Evaluation Branch, Quality Assurance Research Division, Environmental Monitoring Systems Laboratory, U.S. Environmental Protection Agency, 256 p.

Koerkle, E.H., Fishel, D.K., Brown, M.J., and Kostelnik, K.M., 1996, Evaluation of agricultural best-management practices in the Conestoga River headwaters, Pennsylvania - Effects of nutrient management on water quality in the Little Conestoga Creek headwaters, 1983-89: U.S. Geological Survey Water-Resources Investigations Report 95-4046, 49 p.

Koerkle, E.H., and Gustafson-Minnich, L.C., 1997, Surface-water quality changes after 5 years of nutrient management in the Little Conestoga Creek headwaters, Pennsylvania, 1989-91: U.S. Geological Survey Water-Resources Investigations Report 97-4048, 43 p.

Langland, M.J., and Reed, L.A., 1996, Nutrient and triazine-herbicide concentrations in streams of the Chickies Creek Basin, south-central Pennsylvania, during low-flow conditions: U.S. Geological Survey Water-Resources Investigations Report 96-4073, 6 p.

Larcher, W., 1983, Physiological plant ecology (2nd ed.): New York, Springer-Verlag, 303 p.

Lee, D.R., and Cherry, J.A., 1978, A field exercise on groundwater flow using seepage meters and mini-piezometers: Journal of Geological Education, v. 27, p. 6-10.

Lietman, P.L., Gustafson-Minnich, L.C., and Hall, D.W., 1997, Evaluation of agricultural best-management practices in the Conestoga River headwaters, Pennsylvania - Effects of pipe-outlet terracing on quantity and quality of surface runoff and ground water in a small carbonate-rock basin near Churchtown, Pennsylvania, 1983-89: U.S. Geological Survey Water-Resources Investigations Report 94-4206, 62 p.

Lietman, P.L., Ward, J.R., and Behrendt, T.E., 1983, Effects of specific land uses on nonpoint sources of suspended sediment, nutrients, and herbicides - Pequea Creek Basin, Pennsylvania, 1979-80: U.S. Geological Survey Water-Resources Investigations Report 83-4113, 96 p.

Lindsey, B.D., Phillips, S.W., Donnelly, C.A., Speiran, G.K., Plummer, L.N., Bôhlke, J.K., Focazio, M.J., Burton, W.C., and Busenberg, E., 2003, Residence times and nitrate transport in ground water discharging to streams in the Chesapeake Bay Watershed: U.S. Geological Survey Water-Resources Investigations Report 03-4035, 201 p.

Low, D.J., Hippe, D.J., and Yannacci, D., 2002, Geohydrology of southeastern Pennsylvania: U.S. Geological Survey Water-Resources Investigations Report 00-4166, 347 p.

Lowrance, R., 1992, Groundwater nitrate and denitrification in a coastal plain riparian forest: Journal of Environmental Quality, v. 21, p. 401-405.

Lystrom, D.J., Rinella, F.A., Rickert, D.A., and Zimmerman, L., 1978, Multiple regression modeling approach for regional water quality management: U.S. Environmental Protection Agency 600/7-78-198, 68 p.

Mandaville, S.M., 2002, Benthic macroinvertebrates in freshwater - Taxa tolerance values, metrics, and protocols: Canada, accessed March 10, 2004, at *http://lakes.chebucto.org/H-1/tolerance.pdf*.

McLeod, R.V., and Hegg, R.O., 1984, Pasture runoff quality from application of inorganic and organic nitrogen sources: Journal of Environmental Quality, v. 13, no.1, p. 122-126.

Meisinger, J.J., and Jokela, W.E., 2000, Ammonia volatilization from dairy and poultry manure, *in* Managing nutrients and pathogens from animal agriculture: Ithaca, N.Y., Natural Resource, Agriculture, and Engineering Service (NRAES) NRAES-130, p. 334-354.

Meisler, H., and Becher, A.E., 1971, Hydrogeology of the carbonate rocks of the Lancaster 15-Minute Quadrangle, southeastern Pennsylvania: Pennsylvania Topographic and Geologic Survey, 4th ser., Ground Water Report 26, 149 p.

Merritt, R.W., and Cummins, K.W., eds., 1984, An introduction to the aquatic insects of North America (2d ed.): Dubuque, Iowa, Kendall/Hunt Publishing Company, 722 p.

Moog, D.B., and Whiting, P.J., 2002, Climatic and agricultural factors in nutrient exports from two watersheds in Ohio: Journal of Environmental Quality, v. 31, no.1, p. 72-83.

Mostaghimi, S., McClellan, P.W., Tim, U.S., Carr, J.C., Byler, R.K., Dillaha, T.A., Shanholtz, V.O., and Pratt, J.R., 1989, Watershed/water quality monitoring for evaluating animal waste BMP effectiveness - Owl Run watershed: Richmond, Virginia, Pre-BMP final report, Report No. O-P1-8906, Virginia Department of Conservation and Historic Resources, Division of Soil and Water Conservation.

Nagpal, N.K., 1986, Effect of soil and effluent characteristics on phosphorus sorption in dosed columns: Journal of Environmental Quality, v. 15, no. 1, p. 73-78.

National Oceanic and Atmospheric Administration, 1994, Climatological Data - Annual Summary - Pennsylvania 1994: Asheville, N.C., National Climate Data Center.

Nelson, W.M., Gold, A.J., and Groffman, P.M., 1995, Spatial and temporal variation in groundwater nitrate removal in a riparian forest: Journal of Environmental Quality, v. 24, p. 691-699.

Orr, H.K., 1960, Soil porosity and bulk density on grazed and protected Kentucky bluegrass range in the Black Hills: Journal of Range Management, v. 13, p. 80-86.

Osmond, D.L., and Gilliam, J.W., 2002, Soil facts - Agricultural riparian buffers: North Carolina, Department of Soil Science, North Carolina Cooperative Extension Service, p. 2, accessed Nov. 24, 2003, at *http://www.neuse.ncsu.edu/riparian.pdf*.

Osmond, D.L., Line, D.E., and Spooner, J., 1995, Section 319 National Monitoring Program—An overview: Raleigh, North Carolina, North Carolina State University Water Quality Group, Biological and Agricultural Engineering Department, North Carolina State University, 13 p.

Ott, A.N., Takita, C.S., Edwards, R.E., and Bollinger, S.W., 1991, Loads and yields of nutrients and suspended sediment transported in the Susquehanna River Basin, 1985-89: Harrisburg, Pennsylvania, Susquehanna River Basin Commission Publication No. 136, 254 p.

Parsons, J.E., Daniels, R.B., Gilliam, J.W., and Dillaha, T.A., 1994a, Reduction in sediment and chemical load agricultural field runoff by vegetative filter strips: Raleigh, North Carolina, Water Resources Research Institute of the University of North Carolina, Report No. 286.

Parsons, J.E., Gilliam, J.W., Munoz-Carpena, R., Daniels, R.B., and, Dillaha, T.A., 1994b, Nutrient and sediment removal by grass and riparian buffers: Proceedings of the second environmentally sound agriculture conference, p. 147-154.

Parsons, R.L., Pease, J.W., and Bosch, D.J., 1995, Simulating nitrogen losses from agricultural land — Implications for water quality and protection policy: Water Resources Bulletin, v. 31, no. 6, p. 1,079-1,087.

Pearce, R.A., Trlica, M.J., Leininger, W.C., Smith, J.L., and Frasier, G.W., 1997, Efficiency of grass buffer strips and vegetation height on sediment filtration in laboratory rainfall simulations: Journal of Environmental Quality, v. 26, p. 139-144.

Pennak, R.W., 1953, Fresh-water invertebrates of the United States: New York, The Ronald Press Company, 769 p.

Pennsylvania Department of Environmental Protection, 2002, Pennsylvania's Chesapeake Bay nutrient reduction strategy: Harrisburg, Pennsylvania, Office of the Chesapeake Bay Watershed, 46 p.

Pennsylvania Department of Environmental Resources, 1986, Field application of manure: Harrisburg, Pennsylvania, Commonwealth of Pennsylvania, 13 p.

Pit, R., Lantrip, J., Harrison, R., Henry, C.L., Xue, D., and O'Connor, T.P., 1999, Infiltration through disturbed urban soils and compost-amended soil effects on runoff quality and quantity: Cincinnati, Ohio, U.S. Environmental Protection Agency, EPA/600/R-00/016, 70 p. + appendices.

Plafkin, J.L., Barbour, M.T., Porter, K.D., Gross, S.K., and Hughes, R.M., 1989, Rapid bioassessment protocols for use in streams and rivers—Benthic macroinvertebrates and fish: Washington, D.C., U.S. Environmental Protection Agency, Office of Water Regulations and Standards, EPA 440-4-89-001, 196 p.

Plummer, L.N., Michel, R.L., Thurman, E.M., and Glynn, P.D., 1993, Environmental tracers for age-dating young ground water, *in* Alley, W.M., ed., Regional ground-water quality: New York, Van Nostrand Reinhold, p. 255-294.

Plummer, L.N., and Busenberg, E., 1999, Chlorofluorocarbons, *in* Cook, P., and Herczeg, A., eds., Environmental tracers in subsurface hydrology: Boston, Massachusetts, Kluwer Academic Publishers, p. 441-478.

PondGuy, 2004, Nitrites: accessed April 13, 2004, at *http://www.pondsplus.com/Infofiles/arto5.htm*.

Poreda, R.J., Cerling, T.E., and Solomon, D.K., 1988, Tritium and helium isotopes as hydrologic tracers in a shallow unconfined aquifer: Journal of Hydrology, v. 103, p. 1-9.

Porter, S.D., Cuffney, T.F., Gurtz, M.E., and Meador, M.R., 1993, Methods for collecting algal samples as part of the National Water-Quality Assessment Program: U.S. Geological Survey Open-File Report 93-409, 39 p.

Poth, C.W., 1977, Summary ground-water resources of Lancaster County, Pennsylvania: Pennsylvania Topographic and Geological Survey, 4th ser., Water Resource Report 43, 80 p.

Pritt, J.W., and Raese, J.W., 1995, Quality assurance/quality control manual—National Water Quality Laboratory: U.S. Geological Survey Open-File Report 95-443, 35 p.

Rauzi, F., and Hanson, C.L., 1966, Water intake and runoff as affected by intensity of grazing: Journal of Range Management, v. 19, p. 351-356.

Rogers, R.D., and Schumm, S.A., 1991, The effect of sparse vegetative cover on erosion and sediment yield: Journal of Hydrology, v. 123, p. 19-24.

SAS Institute Inc., 1999, SAS/STAT User's Guide: accessed Feb. 1, 2005, at *http://v8doc.sas.com/sashtml/*.

Schlosser, P., Stute, M., Dorr, H., Sonntag, C., and Munnich, K.O., 1988, Tritium/^{3}He dating of shallow groundwater: Earth and Planetary Science Letters, v. 89, p. 353-362.

Schlosser, P., Stute, M., Sonntag, C., and Munnich, K.O., 1989, Tritiogenic ^{3}He in shallow groundwater: Earth and Planetary Sciences Letters, v. 94, p. 245-256.

Shaffer, L.L., 1991, The limestone stream of Pennsylvania: Pennsylvania Fish and Boat Commission, 4 p.

Sholar, C.J., and Shreve, E.A., 1998, Quality-assurance plan for the analysis of fluvial sediment by the Northeastern region, Kentucky district sediment laboratory: U.S. Geological Survey Open-File Report 98-384, 20 p.

Shuford, J.W., Fritton, D.D., and Baker, D.E., 1977, Nitrate-nitrogen movement through undisturbed field soil: Journal of Environmental Quality, v. 6, p. 255-258.

Shultz, C.H., 1999, ed., The geology of Pennsylvania: Pennsylvania Topographic and Geologic Survey, 4th ser., and the Pittsburgh Geological Society, Special Publication 1, 888 p.

Solomon, D.K., and Cook, P.G., 1999, ^{3}H and ^{3}He, *in* Cook, P., and Herczeg, A., eds., Environmental tracers in subsurface hydrology: Boston, Massachusetts, Kluwer Academic Publishers, p. 397-424.

Solomon, D.K., Schiff, S.L., Poreda, R.J., and Clark, W.B., 1993, A validation of the ^{3}H/^{3}He method for determining groundwater recharge: Water Resources Research, v. 29, p. 2,951-2,962.

Solomon, D.K., and Sudicky, E.A., 1991, Tritium and helium 3 isotope ratios for direct estimation of spatial variations in groundwater recharge: Water Resources Research, v. 27, p. 2,309-2,319.

Spooner, J., Maas, R.P., Dressing, S.A., Smolen, M.D., and Humenik, F.J., 1985, Appropriate designs for documenting water quality improvements from agricultural NPS control programs, *in* Perspectives on Nonpoint Source Pollution: EPA 440/5-85-001, p. 30-34.

Stiteler, W.M., 1979, Multivariate statistics with applications in statistical ecology, *in* Orloci, L., Rao, C.R., and Stiteler, W.M., eds., Multivariate methods in ecological work: Fairfield, Maryland, International Co-operative Publishing House, p. 279-300.

ter Braak, C.J.F., 1986, Canonical correspondence analysis—A new eigenvector technique for multivariate direct gradient analysis: Ecology, v. 67, no. 5, p. 1,167-1,179.

___, 1987, The analysis of vegetation—Environment relationships by canonical correspondence analysis: Vegetatio, v. 69, p. 69-77.

ter Braak, C.J.F., and Verdonschot, P.F.M., 1995, Canonical correspondence analysis and related multivariate methods in aquatic ecology: Aquatic Sciences, v. 57, no. 3, p. 254-289.

ter Braak, C.J.F., and Smilauer, P., 1998, CANOCO reference manual and user's guide to canoco for windows - Software for canonical community ordination, version 4: Ithaca, New York, Microcomputer Power, 351 p.

Tjaden, R.L., and Weber, G.M., 1998, Riparian buffer management - Riparian buffer systems: Maryland, Maryland Cooperative Extension, University of Maryland, accessed Nov. 21, 2003, at *http://www.riparianbuffers.umd.edu/PDFs/FS733.pdf*.

Unangst, D., chair., Pennsylvania State Rural Clean Water Program Coordinating Committee, 1992, The Conestoga Headwaters Project 10-Year Report of the Rural Clean Water Program: U.S. Department of Agriculture, Agricultural Stabilization and Conservation Service, 329 p.

University of Georgia, 2003, Tools for quality growth - Riparian buffers: accessed Feb., 27, 2004, at *http:/outreach.ecology.uga.edu/tools/fact_sheet/facts_riparian.html*.

U.S. Environmental Protection Agency, 2002a, Guidance for quality assurance project plans: Washington, D.C., EPA QA/G-5, 58 p.

___, 2002b, Consumer factsheet on nitrates/nitrites: accessed April 14, 2004, at *http://www.epa.gov/safewater/dwh/c-ioc/nitrates.html*.

U.S. Geological Survey, 1999, The quality of our nation's waters - Nutrients and pesticides: U.S. Geological Survey Circular 1225, 82 p.

Viessman, W., Jr., Knapp, J.W., Lewis, G.L., and Harbaugh, T.E., 1977, Introduction to hydrology: New York, Harper and Row, Publishers, 704 p.

Voshell, J.R., Jr., 2002, A guide to common freshwater invertebrates of North America: Blacksburg, Virginia, The MacDonald and Woodward Publishing Company, 442 p.

Walker, J.F., Grackles, D.J., Cores, S.R., Owens, D.W., and Whirl, J.A., 1995, Evaluation of nonpoint-source contamination, Wisconsin - Land-use and best-management-practices inventory, selected stormwater-quality data, urban-watershed quality assurance and quality control, constituent loads in rural streams, and snowmelt-runoff analysis, water year 1994: U.S. Geological Survey Open-File Report 95-320, 21 p.

Ward, J.R., 1987, Surface-water quality in Pequea Creek Basin, Pennsylvania, 1977-79: U.S. Geological Survey Water-Resources Investigations Report 85-4250, 66 p.

Wise, D.U., 1970, Multiple deformation, geosynclinal transitions, and the Martic Problem in Pennsylvania, *in* Fisher, G.W., Pettijohn, F.J., Reed, J.C., Jr., and Weaver, K.N., eds., Studies of Appalachian geology - Central and southern: New York, Interscience Publishers, p. 317-333.

Witt, E.C., Hippe, D.J., and Giovannitti, R.M., 1992, Evaluation of nutrient quality-assurance data for Alexanders and Mount Rock Springs Basins, Cumberland County, Pennsylvania: U.S. Geological Survey Open-File Report 92-165, 31 p.

Young, R.A., Huntrods, T., and Anderson, W., 1980, Effectiveness of vegetated buffer strips in controlling pollution from feedlot runoff: Journal of Environmental Quality, v. 9, p. 483-487.

Appendix 1.

Habitat assessment field data sheet used to conduct rapid bioassessment during benthic-macroinvertebrate sampling in the Big Spring Run Basin, Lancaster County, Pennsylvania. [>, greater than; cms, cubic meters per second; cfs, cubic feet per second; <, less than; m/s, meters per second; m, meters; w/, with]

[1]site ______________ date______________

	Category			
Habitat parameter	**Excellent**	**Good**	**Fair**	**Poor**
1. Bottom substrate available cover	Greater than 50% rubble, gravel, submerged logs, undercut banks, or other stable habitat.	30-50% rubble, gravel or other stable habitat. Adequate habitat.	10-30% rubble, gravel or other stable habitat. Habitat availability less than desirable.	Less than 10% rubble gravel or other stable habitat. Lack of habitat is obvious.
	16-20	11-15	6-10	0-5
2. Embeddedness	Gravel, cobble, and boulder particles are between 0 and 25% surrounded by fine sediment.	Gravel, cobble, and boulder particles are between 25 and 50% surrounded by fine sediment.	Gravel, cobble, and boulder particles are between 50 and 75% surrounded by fine sediment.	Gravel, cobble, and boulder particles are over 75% surrounded by fine sediment.
	16-20	11-15	6-10	0-5
3. <0.15 cms (5cfs) - flow at rep. low flow or >0.15 cms (5 cfs) - velocity/depth	Cold >0.05 cms (2 cfs) Warm >0.15 cms (5 cfs) or Slow (<0.3 m/s), deep (>0.5m); slow, shallow (<0.5m); fast (>0.3 m/s), deep; fast, shallow; habitats all present.	0.03-0.05 cms (1-2 cfs) 0.05-0.15 cms (2-5 cfs) or Only 3 or the 4 habitat categories present (missing riffles or runs receive lower score than missing pools.)	0.01-0.03 cms (.5-1 cfs) 0.03-0.05 cms (1-2 cfs) or Only 2 of the 4 habitat categories present (missing riffles/runs receive lower score).	<0.01 cms (.5 cfs) <0.03 cms (1 cfs) or Dominated by one velocity/depth category (usually pool).
	16-20	11-15	6-10	0-5
4. channel alteration	Little or no enlargement of islands or point bars, and/or no channelization.	Some new increase in bar formation, mostly from coarse gravel; and/or some channelization present.	Moderate deposition of new gravel, coarse sand on old and new bars; pools partially filled w/silt; and/or embankments on both banks.	Heavy deposits of fine material, increased bar development; most pools filled w/silt; and/or extensive channelization.
	12-15	8-11	4-7	0-3
5. Bottom scouring and deposition	Less than 5% of the bottom affected by scouring and deposition.	5-30% affected. Scour at constrictions and where grades steepen. Some deposition in pools.	30-50% affected. Deposits and scour at obstruction, constrictions and bends. Some filling of pools.	More than 50% of the bottom changing nearly year long. Pools almost absent due to deposition only large rocks in riffle exposed.
	12-15	8-11	4-7	0-3

[1]site ______________ date_____________

	Category			
Habitat parameter	**Excellent**	**Good**	**Fair**	**Poor**
6. Pool/riffle, run/bend ratio (distance between riffles divided by stream width)	5-7. Variety of habitat. Deep riffles and pools.	7-15. Adequate depth in pools and riffles. Bends provide habitat.	15-25. Occasional riffle or bend. Bottom contours provide some habitat.	>25. Essentially a straight stream. Generally all flat water or shallow riffle. Poor habitat.
	12-15	8-11	4-7	0-3
7. Bank stability	Stable. No evidence of erosion or bank failure. Side slopes generally <30%. Little potential for future problem.	Moderately stable. Infrequent, small areas of erosion mostly healed over. Side slopes up to 40% on one bank. Slight potential in extreme floods.	Moderately unstable. Moderate frequency and size of erosional areas. Side slopes up to 60% on some banks. High erosion potential during extreme high flow.	Unstable. Many eroded areas. Side slopes >60% common. "Raw" areas frequent along straight sections and bends.
	9-10	6-8	3-5	0-2
8. Bank vegetative stability	Over 80% of the streambank surfaces covered by vegetation, gravel or boulders, and cobble.	50-79% of the streambank surfaces covered by vegetation, gravel, or larger material.	25-49% of the streambank surfaces covered by vegetation, gravel, or larger material.	Less than 25% of the streambank surfaces covered by vegetation, gravel, or larger material.
	9-10	6-8	3-5	0-2
9. Streamside cover	Dominant vegetation is shrub.	Dominant vegetation is of tree form.	Dominant vegetation is grass or forbes.	Over 50% of the streambank has no vegetation and dominant material is soil, rock, bridge materials, culverts, or mine tailings.
	9-10	6-8	3-5	0-2

[1] From Plafkin and others, 1989.

Appendix 2

Rapid Bioassessment Protocol (RBP) habitat scores for benthic-macroinvertebrate samples collected in the Big Spring Run Basin, Lancaster County, Pennsylvania.

Site name	Date	Bottom substrate available cover	Embed-dedness	Velocity to depth ratio	Channel alteration	Bottom scouring and deposition	Pool/riffle, run/bend ratio	Bank stability	Bank vegetative stability	Stream-side cover	Total habitat score
C-1	9/22/1993	17	15	15	8	11	11	9	9	4	99
T-1	9/22/1993	15	5	5	2	6	5	7	9	4	58
T1-3	9/22/1993	7	7	6	2	5	7	3	8	4	49
T2-3	9/22/1993	5	11	6	9	8	6	8	8	4	65
C-1	5/10/1994	10	10	12	5	10	10	6	9	5	77
T-1	5/10/1994	12	9	10	13	6	9	4	7	4	74
T1-3	5/10/1994	15	16	10	7	4	9	2	9	5	77
T2-3	5/10/1994	8	8	8	4	5	5	4	9	5	56
C-1	9/14/1994	16	15	10	12	15	11	10	10	5	104
T-1	9/14/1994	15	13	10	12	15	6	5	6	5	87
T1-3	9/14/1994	13	8	13	3	15	8	6	10	5	81
T2-3	9/14/1994	15	12	14	12	12	10	7	9	5	96
C-1	5/16/1995	11	11	16	15	11	12	9	10	5	100
T-1	5/16/1995	15	12	16	12	13	10	6	6	5	95
T1-3	5/16/1995	11	12	16	13	4	9	5	3	5	78
T2-3	5/16/1995	15	13	16	13	11	8	5	3	3	87
C-1	9/21/1995	12	7	16	12	7	10	7	9	5	85
T-1	9/21/1995	12	13	16	12	7	8	7	9	5	89
T1-3	9/21/1995	6	7	16	10	3	9	8	8	5	72
T2-3	9/21/1995	13	13	16	12	10	8	6	9	5	92
C-1	5/20/1996	5	5	6	3	7	3	6	8	5	48
T-1	5/20/1996	10	5	11	11	11	7	5	7	5	72
C1-2	5/20/1996	11	10	14	13	7	11	5	5	5	81
T1-3	5/20/1996	5	5	1	3	4	7	2	2	5	34
T2-3	5/20/1996	12	10	12	11	10	8	5	5	5	78
C-1	9/10/1996	16	8	14	12	14	8	7	9	5	93
T-1	9/10/1996	12	9	10	11	7	8	5	8	5	75
C1-2	9/10/1996	12	10	16	12	12	15	4	6	5	92
T1-3	9/10/1996	15	10	14	13	12	10	7	9	5	95
T2-3	9/10/1996	12	9	15	12	13	10	5	9	5	90
C-1	5/20/1997	16	11	10	7	12	8	4	6	5	79
T-1	5/20/1997	12	11	10	12	12	5	6	9	5	82
C1-2	5/20/1997	15	15	16	13	7	14	5	5	5	95
T1-3	5/20/1997	10	11	7	12	12	9	5	5	5	76
T2-3	5/20/1997	6	7	6	12	12	9	8	6	5	71
C-1	9/8/1997	12	10	12	13	13	6	8	9	5	88
T-1	9/8/1997	13	11	10	12	12	8	9	10	4	89
C1-2	9/8/1997	12	15	15	12	10	11	8	6	5	94

Site name	Date	Bottom substrate available cover	Embed-dedness	Velocity to depth ratio	Channel alteration	Bottom scouring and deposition	Pool/riffle, run/bend ratio	Bank stability	Bank vegetative stability	Stream-side cover	Total habitat score
T1-3	9/8/1997	11	5	6	5	6	5	5	5	4	52
T2-3	9/8/1997	10	15	5	11	9	5	8	8	3	74
C-1	5/12/1998	12	9	13	12	9	9	5	8	5	82
T-1	5/12/1998	12	12	15	15	15	11	9	10	5	104
C1-2	5/12/1998	11	8	16	12	12	11	5	5	2	82
T1-3	5/12/1998	6	5	10	12	13	8	7	4	3	68
T2-3	5/12/1998	10	11	11	13	11	8	9	9	5	87
C-1	9/8/1998	10	8	11	13	7	7	6	9	3	74
T-1	9/8/1998	10	12	6	13	15	5	10	10	5	86
C1-2	9/8/1998	12	11	13	12	11	10	6	6	5	86
T1-3	9/8/1998	6	6	6	13	12	7	8	9	5	72
T2-3	9/8/1998	15	11	11	15	11	9	9	9	5	95
C-1	5/1/1999	10	6	10	12	9	7	6	8	5	73
T-1	5/1/1999	16	5	10	11	8	7	8	9	5	79
C1-2	5/1/1999	15	13	16	12	8	12	5	5	5	91
T1-3	5/1/1999	10	4	11	13	11	10	6	5	5	75
T2-3	5/1/1999	12	11	11	9	8	9	7	7	5	79
C-1	9/8/1999	11	11	11	13	14	5	9	9	5	88
T-1	9/8/1999	14	11	11	14	14	10	6	8	5	93
C1-2	9/8/1999	16	15	14	11	7	9	4	4	3	83
T1-3	9/8/1999	11	11	10	12	12	7	8	9	5	85
T2-3	9/8/1999	15	14	15	13	13	7	8	8	5	98
C-1	5/8/2000	16	10	16	15	13	8	9	9	5	101
T-1	5/8/2000	16	10	16	10	14	8	9	10	5	98
C1-2	5/8/2000	11	12	16	11	12	9	5	5	3	84
T1-3	5/8/2000	13	15	11	8	12	8	3	4	3	77
T2-3	5/8/2000	11	10	16	10	11	11	5	8	5	87
C-1	9/5/2000	12	11	16	14	13	7	8	9	5	95
T-1	9/5/2000	15	15	11	6	12	4	9	9	5	86
C1-2	9/5/2000	12	12	16	14	14	11	8	8	5	100
T1-3	9/5/2000	11	6	6	14	12	7	3	6	5	70
T2-3	9/5/2000	12	6	15	12	12	7	8	9	5	86
C-1	5/9/2001	16	13	14	13	13	8	8	9	5	99
T-1	5/9/2001	10	11	11	3	8	8	6	8	5	70
C1-2	5/9/2001	16	15	15	11	13	11	5	6	3	95
T1-3	5/9/2001	11	11	15	12	12	10	5	8	4	88
T2-3	5/9/2001	17	15	15	13	14	9	8	6	5	102

Appendix 3.

List of benthic macroinvertebrates identified for May and September sampling events from September 1993 through May 2001 in the Big Spring Run Basin, Lancaster County, Pennsylvania.

Table 3-1. Benthic macroinvertebrates identified at C-1 (station 01576521) site in May for years 1994-2001.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C-1 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Platyhelminthes								
Turbellaria								
Tricladida								
Planariidae	-	-	-	1	1	-	4	1
Annelida								
Clitellata								
Hirudinea								
Arhynchobdellida								
Erpobdellidae	-	-	-	-	-	-	2	-
Erpobdella	-	-	-	-	-	-	-	1
Oligochaeta								
Haplotaxida								
Enchytraeidae	-	-	1	-	-	-	-	1
Naididae	-	-	-	1	-	32	-	-
Nais	7	-	-	10	2	155	8	10
Ophidonais serpentina	-	-	-	1	-	8	2	-
Tubificidae	-	3	5	5	12	-	4	-
Limnodrilus	-	-	8	1	9	-	-	1
Lumbriculida								
Lumbriculidae	-	-	-	-	1	-	-	-
Mollusca								
Bivalvia								
Veneroida								
Pisidiidae	-	-	-	-	-	-	1	-
Arthropoda								
Chelicerata								
Arachnida								
Acariformes								
Prostigmata								
Hygrobatidae								
Hygrobates	1	4	-	11	2	-	-	2
Lebertiidae								
Lebertia	1	-	-	-	-	-	-	-
Sperchonidae								
Sperchon	-	-	-	-	-	-	-	1
Crustacea								
Malacostraca								
Isopoda								
Asellidae								
Caecidotea	-	-	-	-	1	-	1	-
Amphipoda								

Table 3-1. Benthic macroinvertebrates identified at C-1 (station 01576521) site in May for years 1994-2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C-1 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Gammaridae	50	106	37	111	127	-	13	72
Ostracoda	1	-	-	-	-	-	-	2
Hexapoda								
Insecta								
Ephemeroptera								
Pisciforma								
Baetidae	1	9	5	7	5	-	4	3
Coleoptera								
Adephaga								
Dytiscidae								
Agabus	2	-	1	-	-	-	1	-
Polyphaga								
Elmidae								
Dubiraphia	-	2	3	-	-	-	-	1
Optioservus	1	14	3	1	2	-	-	2
Promoresia	-	2	-	-	-	-	-	-
Stenelmis	1	5	-	2	-	-	1	-
Trichoptera								
Hydropsychidae								
Cheumatopsyche	-	1	-	-	1	-	4	4
Hydropsyche	-	3	-	1	2	-	1	-
Hydroptilidae	-	-	-	-	-	-	-	1
Hydroptila	2	-	2	3	1	-	-	1
Orthotrichia	-	-	-	-	-	-	-	2
Diptera								
Nematocera								
Ceratopogonidae	-	-	-	-	-	-	1	-
Culicoides	-	-	-	-	-	-	-	1
Chironomidae								
Tanypodinae								
Pentaneurini								
Conchapelopia	1	-	1	-	1	-	-	-
Thienemannimyia gr.	-	2	-	-	-	-	1	-
Diamesinae								
Diamesini								
Diamesa	1	-	-	-	1	-	-	-
Pagastia	-	-	6	-	-	-	-	-
Prodiamesinae								
Prodiamesa	-	4	7	-	1	-	-	-
Orthocladiinae	2	-	-	-	-	-	1	-
Cricotopus	19	6	7	1	7	2	10	6

Table 3-1. Benthic macroinvertebrates identified at C-1 (station 01576521) site in May for years 1994-2001.—Continued

[Taxonomy was checked on March 1, 2006]

	Site C-1 May benthic-macroinvertebrate numbers by year							
Benthic Macroinvertebrates	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**
Cricotopus trifascia gr.	-	-	-	-	-	-	7	-
Eukiefferiella	-	2	-	1	2	-	1	2
Orthocladius	41	3	45	14	19	-	9	20
Parakiefferiella	2	-	-	-	-	-	-	13
Parametriocnemus	-	2	-	1	4	-	1	2
Paratrichocladius	-	-	-	-	-	-	-	1
Psectrocladius	-	-	-	-	1	-	-	-
Tvetenia	-	4	-	-	1	-	-	-
Chironominae	-	-	-	1	-	-	1	-
Chironomini								
Chironomus	-	-	5	-	1	-	-	-
Dicrotendipes	13	15	59	9	4	-	3	1
Microtendipes	-	-	-	-	1	-	-	-
Micropsectra	9	21	2	4	12	-	4	7
Paracladopelma	-	1	1	-	-	-	-	-
Paratanytarsus	1	-	-	-	-	-	-	-
Paratendipes	2	9	3	-	-	-	-	-
Phaenopsectra	-	-	-	-	4	-	-	-
Polypedilum	2	1	-	4	-	-	5	2
Rheotanytarsus	-	-	-	-	-	-	-	2
Stictochironomus	-	-	6	-	-	-	-	-
Tanytarsus	-	-	-	-	1	-	-	-
Zavrelia	1	-	-	-	-	-	-	-
Simuliidae								
Simulium	22	3	3	2	7	2	120	49
Brachycera-Orthorrhapha								
Empididae								
Hemerodromia	-	-	-	-	-	-	-	2

Table 3-2. Benthic macroinvertebrates identified at C-1 (station 01576521) site in September for years 1993-2000.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C-1 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Platyhelminthes								
Turbellaria								
Tricladida								
Dugesiidae								
Cura	-	9	-	-	-	-	-	-
Planariidae	-	-	1	-	1	-	20	2
Phagocata	10	-	-	-	-	-	-	-
Annelida								
Clitellata								
Hirudinea								
Arhynchobdellida								
Erpobdellidae	4	2	-	1	-	-	1	-
Oligochaeta								
Branchiobdellida								
Branchiobdellae	-	1	-	-	-	-	-	-
Haplotaxida								
Enchytraeidae	-	1	1	-	-	-	1	-
Naididae	-	-	1	-	-	3	-	-
Nais	-	1	-	-	-	-	3	-
Pristina	-	-	-	-	-	1	-	-
Pristinella	-	-	-	-	-	-	1	-
Tubificidae	2	1	19	-	-	24	20	-
Limnodrilus	2	-	1	-	15	-	2	2
Tubifex	-	-	-	-	37	-	-	-
Mollusca								
Gastropoda	-	-	-	-	-	-	1	-
Bivalvia	-	-	-	-	-	-	-	2
Veneroida								
Pisidiidae	-	-	-	-	2	1	-	-
Pisidium	-	-	-	-	-	-	2	-
Arthropoda								
Chelicerata								
Arachnida								
Acariformes								
Prostigmata								
Hygrobatidae								
Hygrobates	-	2	3	-	6	4	-	-
Lebertiidae								
Lebertia	-	-	-	-	3	1	2	1
Crustacea								
Malacostraca								

Table 3-2. Benthic macroinvertebrates identified at C-1 (station 01576521) site in September for years 1993-2000.—Continued

[Taxonomy was checked on March 1, 2006]

	Site C-1 September benthic-macroinvertebrate numbers by year							
Benthic Macroinvertebrates	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**
Isopoda								
Asellidae								
Caecidotea	6	2	-	-	-	2	-	-
Amphipoda	-	-	-	-	-	-	-	-
Gammaridae	14	23	86	46	31	12	-	16
Ostracoda	-	-	1	-	-	-	-	33
Hexapoda								
Insecta								
Ephemeroptera								
Pisciforma								
Baetidae	4	10	21	21	6	15	1	9
Odonata								
Zygoptera								
Coenagrionidae	-	-	-	-	-	1	-	-
Enallagma	-	-	-	-	-	-	2	2
Hemiptera								
Heteroptera								
Corixidae	-	-	-	-	-	-	13	-
Palmacorixa	-	-	-	-	-	-	4	-
Sigara	-	-	-	-	-	-	2	-
Coleoptera								
Adephaga								
Dytiscidae								
Agabus	-	-	-	-	-	-	-	1
Polyphaga								
Elmidae	-	-	5	-	1	2	-	-
Dubiraphia	14	-	3	-	1	1	-	3
Optioservus	10	19	25	9	-	17	-	1
Promoresia	2	-	-	-	-	-	-	-
Stenelmis	4	4	7	9	1	4	-	1
Trichoptera								
Hydropsychidae	-	8	-	-	3	4	1	7
Cheumatopsyche	-	4	12	9	2	3	4	6
Hydropsyche	-	15	8	15	2	13	-	-
Hydroptilidae								
Hydroptila	-	8	-	-	2	2	2	8
Diptera	-	-	-	-	-	1	-	-
Nematocera								
Chironomidae								
Tanypodinae								
Pentaneurini								

Table 3-2. Benthic macroinvertebrates identified at C-1 (station 01576521) site in September for years 1993-2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C-1 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Ablabesmyia	-	-	-	1	-	-	-	-
Thienemannimyia gr.	-	1	-	5	5	4	1	1
Orthocladiinae	-	-	1	-	-	-	-	-
Cricotopus/Orthocladius	-	-	-	-	-	-	3	-
Cricotopus	-	7	4	5	8	2	4	-
Eukiefferiella	-	1	1	2	-	2	-	-
Orthocladius	-	22	-	-	14	26	-	-
Parakiefferiella	8	-	-	-	-	-	-	-
Parametriocnemus	4	1	1	3	3	14	3	3
Thienemanniella	-	-	-	1	5	-	2	-
Tvetenia	-	1	-	1	21	8	2	-
Chironominae	-	-	-	1	3	-	-	1
Chironomini								
Chironomus	-	-	-	3	-	-	-	-
Cladotanytarsus	-	-	-	-	-	-	-	2
Cryptochironomus	-	-	-	-	2	-	2	2
Dicrotendipes	-	3	5	4	8	4	30	27
Microtendipes	-	-	-	-	-	1	-	-
Micropsectra	4	4	-	-	-	-	11	-
Paracladopelma	-	-	-	-	2	-	-	-
Paratendipes	-	-	-	-	1	3	13	7
Phaenopsectra	-	1	-	-	-	-	1	-
Polypedilum	-	4	1	10	-	-	6	-
Rheotanytarsus	102	16	3	6	-	4	23	21
Stictochironomus	-	-	-	1	-	-	1	-
Tanytarsus	-	1	3	2	3	-	31	10
Tribelos	-	-	1	-	-	-	-	-
Psychodidae	-	-	-	-	1	-	-	-
Simuliidae								
Simulium	10	24	-	38	8	5	5	33
Tipulidae								
Tipulinae								
Tipula	-	-	-	-	-	-	1	-
Limoniinae								
Antocha	-	-	-	-	-	1	-	-
Dicranota	-	-	-	-	-	1	-	-
Brachycera-Orthorrhapha								
Empididae								
Hemerodromia	-	1	-	-	-	1	-	2

Table 3-3. Benthic macroinvertebrates identified at C1-2 (station 01576519) site in May for years 1996-2001.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C1-2 May benthic-macroinvertebrate numbers by year					
	1996	1997	1998	1999	2000	2001
Platyhelminthes						
Turbellaria						
Tricladida						
Planariidae	-	-	-	-	2	-
Annelida						
Clitellata						
Oligochaeta						
Haplotaxida						
Enchytraeidae	-	-	-	-	1	-
Naididae	-	-	1	-	-	-
Nais	-	2	2	89	16	-
Tubificidae	-	1	1	3	11	-
Limnodrilus	1	-	-	-	-	-
Lumbriculida						
Lumbriculidae	3	-	-	-	-	-
Mollusca						
Gastropoda						
Basommatophora						
Physidae	1	-	-	-	-	-
Bivalvia						
Veneroida						
Pisidiidae	-	-	-	1	-	-
Arthropoda						
Chelicerata						
Arachnida						
Acariformes						
Prostigmata						
Hygrobatidae						
Hygrobates	-	1	-	-	-	-
Lebertiidae						
Lebertia	-	-	1	-	-	-
Crustacea						
Malacostraca						
Amphipoda						
Gammaridae	49	156	178	26	107	163
Hexapoda						
Insecta						
Ephemeroptera						
Pisciforma						
Baetidae	-	15	-	-	-	-
Coleoptera						

Table 3-3. Benthic macroinvertebrates identified at C1-2 (station 01576519) site in May for years 1996-2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C1-2 May benthic-macroinvertebrate numbers by year					
	1996	1997	1998	1999	2000	2001
Adephaga						
Dytiscidae						
Agabus	-	-	1	-	-	-
Polyphaga						
Elmidae	-	-	-	1	-	-
Dubiraphia	3	1	-	1	-	7
Optioservus	1	1	-	5	-	4
Promoresia	-	-	-	1	-	-
Stenelmis	3	5	2	5	-	2
Trichoptera						
Hydropsychidae						
Cheumatopsyche	1	-	-	-	-	-
Hydropsyche	-	1	-	-	-	-
Hydroptilidae	-	-	-	1	-	-
Hydroptila	-	-	-	1	-	-
Diptera						
Nematocera						
Chironomidae						
Tanypodinae						
Pentaneurini						
Thienemannimyia gr.	-	-	-	-	1	-
Diamesinae						
Diamesini						
Diamesa	-	-	-	23	-	-
Pagastia	1	-	-	-	-	-
Prodiamesa	1	-	1	-	-	-
Orthocladiinae	-	1	-	1	-	-
Orthocladiini						
Cricotopus	10	4	5	12	14	6
Cricotopus trifascia gr.	-	-	-	-	1	5
Orthocladius	120	8	6	37	7	5
Parakiefferiella	-	-	-	-	-	3
Parametriocnemus	3	1	-	-	1	-
Pseudosmittia	1	-	-	-	-	-
Rheocricotopus	-	-	-	1	-	-
Tvetenia	-	-	1	-	-	-
Chironominae						
Chironomini						
Chironomus	1	-	-	-	-	-
Dicrotendipes	6	3	3	-	2	3
Microtendipes	-	1	-	-	-	-

Table 3-3. Benthic macroinvertebrates identified at C1-2 (station 01576519) site in May for years 1996-2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C1-2 May benthic-macroinvertebrate numbers by year					
	1996	1997	1998	1999	2000	2001
Paratendipes	-	-	-	-	1	2
Phaenopsectra	-	1	-	-	-	-
Polypedilum	-	2	-	-	-	-
Stictochironomus	1	-	-	-	-	-
Tanytarsini						
Micropsectra	-	-	1	-	1	-
Tanytarsus	-	-	-	3	1	-
Simuliidae						
Simulium	-	7	1	13	41	1
Tipulidae						
Limoniinae						
Pseudolimnophila	-	-	1	-	-	-

Table 3-4. Benthic macroinvertebrates identified at C1-2 (station 01576519) site in September for years 1996-2000.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C1-2 September benthic-macroinvertebrate numbers by year				
	1996	1997	1998	1999	2000
Platyhelminthes					
Turbellaria					
Tricladida					
Planariidae	-	-	27	36	2
Annelida					
Clitellata					
Oligochaeta					
Haplotaxida					
Enchytraeidae	-	-	-	1	-
Haplotaxidae	-	-	-	1	-
Naididae					
Nais	-	-	-	1	-
Pristinella	-	-	-	9	-
Tubificidae	-	7	-	16	-
Isochaetides curvisetos	1	-	-	2	-
Mollusca					
Gastropoda					
Basommatophora					
Physidae					
Physa	-	-	-	1	-
Arthropoda					
Chelicerata					
Arachnida					
Acariformes					
Prostigmata					
Hygrobatidae					
Hygrobates	1	-	-	-	-
Lebertiidae					
Lebertia	1	-	1	2	-
Crustacea					
Malacostraca					
Isopoda					
Asellidae					
Caecidotea	-	-	-	1	-
Amphipoda					
Gammaridae	159	188	114	85	153
Ostracoda	-	-	-	-	1
Hexapoda					
Insecta					
Collembola					
Isotomidae					

Table 3-4. Benthic macroinvertebrates identified at C1-2 (station 01576519) site in September for years 1996-2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C1-2 September benthic-macroinvertebrate numbers by year				
	1996	1997	1998	1999	2000
Isotomurus	3	1	-	-	-
Ephemeroptera					
Pisciforma					
Baetidae	17	-	12	-	-
Odonata					
Anisoptera					
Aeschnidae					
Boyeria vinosa	-	1	-	-	-
Zygoptera					
Coenagrionidae	-	-	-	1	-
Hemiptera					
Heteroptera					
Corixidae					
Sigara	-	2	-	-	-
Trichocorixa	2	-	-	-	-
Coleoptera					
Adephaga					
Haliplidae					
Haliplus	-	-	1	-	-
Polyphaga					
Elmidae	-	1	-	1	-
Dubiraphia	-	-	-	3	-
Optioservus	1	7	3	14	4
Stenelmis	2	1	8	16	6
Trichoptera					
Hydropsychidae	-	-	-	1	-
Cheumatopsyche	2	1	-	-	-
Hydropsyche	1	1	-	-	-
Diptera					
Nematocera					
Chironomidae					
Tanypodinae					
Conchapelopia	4	-	-	-	-
Thienemannimyia gr.	-	-	4	-	1
Procladiini					
Procladius	-	-	-	1	-
Prodiamesinae					
Prodiamesa	-	1	1	-	-
Orthocladiinae	-	-	-	-	2
Orthocladiini					
Cricotopus	11	-	4	-	-

Table 3-4. Benthic macroinvertebrates identified at C1-2 (station 01576519) site in September for years 1996-2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site C1-2 September benthic-macroinvertebrate numbers by year				
	1996	1997	1998	1999	2000
Eukiefferiella	1	-	-	-	-
Orthocladius	3	1	12	5	-
Parakiefferiella	-	-	-	1	-
Parametriocnemus	-	-	3	1	-
Tvetenia	-	1	2	-	-
Chironominae					
Chironomus	1	-	-	-	-
Dicrotendipes	1	2	6	8	16
Harnischia	-	-	1	-	-
Paratendipes	1	-	-	-	-
Polypedilum	10	-	-	-	-
Tanytarsini					
Micropsectra	1	-	-	-	-
Paratanytarsus	-	-	-	1	-
Rheotanytarsus	1	-	-	1	-
Tanytarsus	-	-	1	-	1
Simuliidae					
Simulium	9	-	2	-	-
Brachycera-Orthorrhapha					
Empididae					
Hemerodromia	-	-	-	-	1
Brachycera-Cyclorrhapha					
Muscidae	1	1	1	-	-

Table 3-5. Benthic macroinvertebrates identified at T-1 (station 01576529) site in May for years 1994-2001.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Platyhelminthes								
Turbellaria								
Tricladida								
Planariidae	-	-	1	-	6	3	4	9
Annelida								
Clitellata								
Hirudinea	-	1	2	-	-	-	-	-
Arhynchobdellida	-	-	-	-	4	-	-	-
Erpobdellidae	1	-	-	-	-	-	-	-
Oligochaeta								
Haplotaxida								
Enchytraeidae	-	1	-	-	-	-	-	-
Naididae	-	-	-	-	-	-	-	1
Nais	-	79	-	11	-	133	31	4
Ophidonais serpentina	-	-	-	77	-	-	1	-
Tubificidae	-	22	9	1	-	-	1	3
Ilyodrilus templetoni	5	-	-	-	-	-	-	-
Limnodrilus	14	-	24	-	-	20	-	4
Mollusca								
Gastropoda								
Basommatophora								
Physidae								
Physinae	-	-	-	1	6	-	-	-
Bivalvia								
Veneroida								
Pisidiidae	-	-	-	-	-	3	-	-
Arthropoda								
Chelicerata								
Arachnida	-	-	-	-	4	-	-	-
Acariformes								
Prostigmata								
Hygrobatidae								
Hygrobates	1	7	-	3	-	-	-	-
Lebertiidae								
Lebertia	-	2	-	-	-	-	-	-
Sperchonidae								
Sperchon	-	-	-	1	-	-	-	1
Crustacea								
Malacostraca								
Isopoda								
Asellidae								

Table 3-5. Benthic macroinvertebrates identified at T-1 (station 01576529) site in May for years 1994-2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Caecidotea	-	-	-	-	-	-	12	6
Amphipoda								
Gammaridae	-	-	-	1	12	-	38	6
Maxillopoda								
Copepoda	-	-	-	-	-	2	-	-
Hexapoda								
Insecta								
Ephemeroptera								
Pisciforma								
Baetidae	5	-	13	31	24	2	1	-
Furcatergalia								
Caenidae								
Caenis	-	-	-	1	-	-	-	-
Setisura								
Heptageniidae								
Stenonema mediopunctatum	-	-	-	1	-	-	-	-
Odonata								
Zygoptera								
Coenagrionidae	-	-	-	-	-	-	1	-
Argia	-	-	1	1	-	-	-	-
Enallagma	-	-	-	-	-	-	-	1
Coleoptera								
Adephaga								
Dytiscidae								
Agabus	1	-	2	-	2	1	5	1
Deronectes	1	-	-	-	-	-	-	-
Polyphaga								
Psephenidae								
Ectopria	-	-	-	-	-	-	-	2
Elmidae	-	-	-	-	-	1	-	-
Dubiraphia	1	5	1	1	2	-	-	3
Optioservus	-	2	-	-	-	2	3	4
Promoresia	1	-	-	1	-	-	-	-
Stenelmis	1	8	15	3	24	6	4	10
Trichoptera								
Hydropsychidae								
Cheumatopsyche	-	-	1	-	6	-	1	-
Hydropsyche	1	1	6	-	-	-	-	-
Hydroptilidae								
Hydroptila	-	-	-	-	2	-	-	2
Diptera								

Table 3-5. Benthic macroinvertebrates identified at T-1 (station 01576529) site in May for years 1994-2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Nematocera								
Ceratopogonidae								
Bezzia	1	-	-	-	-	-	-	-
Probezzia	-	-	-	-	4	-	-	-
Chironomidae								
Tanypodinae	-	-	-	-	-	1	-	-
Coelotanypodini								
Clinotanypus	-	-	-	-	2	-	-	-
Pentaneurini								
Conchapelopia	-	-	1	-	-	-	-	-
Thienemannimyia gr.	1	1	-	1	-	1	-	2
Orthocladiinae	1	-	-	-	-	-	-	-
Orthocladiini								
Cardiocladius	1	-	-	-	-	-	-	-
Cricotopus trifascia gr.	-	-	-	-	-	-	-	3
Cricotopus	9	26	75	11	38	-	15	3
Eukiefferiella	-	-	-	2	-	-	-	-
Orthocladius	95	10	41	33	16	9	13	32
Parakiefferiella	13	14	5	18	8	4	-	18
Parametriocnemus	1	-	2	-	20	1	2	5
Smittia	-	-	1	-	-	-	-	-
Thienemanniella	1	-	1	-	2	-	-	-
Tvetenia	-	2	1	1	14	-	-	-
Chironominae								
Chironomini								
Chironomus	1	-	7	-	-	-	-	-
Cryptochironomus	1	-	-	-	-	-	1	1
Dicrotendipes	5	18	5	14	38	-	5	2
Microtendipes	1	-	-	-	-	-	-	-
Paracladopelma	-	-	-	1	-	-	-	-
Paratendipes	1	1	-	-	-	2	2	1
Phaenopsectra	1	-	-	-	6	-	-	-
Polypedilum	4	2	22	-	24	-	14	1
Stictochironomus	1	-	1	-	-	-	3	1
Tribelos	-	2	-	-	-	-	-	-
Tanytarsini								
Micropsectra	1	7	5	1	88	7	4	6
Rheotanytarsus	-	-	9	-	6	1	1	3
Subletta	-	-	1	-	-	-	-	-
Tanytarsus	-	-	-	-	-	1	-	-
Simuliidae								

Table 3-5. Benthic macroinvertebrates identified at T-1 (station 01576529) site in May for years 1994-2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Simulium	40	2	10	1	12	1	31	61
Brachycera-Orthorrhapha								
Empididae								
Hemerodromia	-	2	-	-	2	-	-	1

Table 3-6. Benthic macroinvertebrates identified at T-1 (station 01576529) site in September for years 1993-2000.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Cnidaria								
Hydrozoa								
Hydroida								
Hydridae	-	-	1	-	-	1	-	-
Platyhelminthes								
Turbellaria								
Tricladida								
Dugesiidae	-	-	3	1	5	8	4	5
Cura	-	2	-	-	-	-	-	-
Annelida								
Clitellata								
Hirudinea								
Arhynchobdellida								
Erpobdellidae	2	-	-	1	7	1	2	-
Rhynchobdellida								
Glossiphoniidae								
Helobdella stagnalis	1	-	-	-	2	-	-	-
Oligochaeta								
Haplotaxida								
Naididae	2	-	1	-	-	-	-	-
Nais	-	-	1	-	-	-	-	-
Tubificidae	-	9	48	-	1	7	38	-
Limnodrilus	2	-	4	-	-	-	-	17
Mollusca								
Gastropoda	-	1	-	-	-	-	-	-
Basommatophora								
Physidae	-	-	-	-	-	-	-	1
Physinae	-	-	-	-	-	2	-	-
Physella	-	-	-	-	76	-	3	-
Bivalvia								
Veneroida								
Pisidiidae								
Pisidium	-	-	-	-	1	-	-	-
Arthropoda								
Chelicerata								
Arachnida								
Acariformes								
Prostigmata								
Hygrobatidae								
Hygrobates	-	2	-	1	-	1	-	-
Lebertiidae								

Table 3-6. Benthic macroinvertebrates identified at T-1 (station 01576529) site in September for years 1993-2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Lebertia	-	2	2	-	-	3	-	-
Sarcoptiformes								
Oribatei								
Hydrozetidae								
Hydrozetes	-	-	-	-	1	-	-	-
Crustacea								
Malacostraca								
Isopoda								
Asellidae								
Caecidotea	-	-	-	-	-	1	6	1
Amphipoda	-	-	-	-	-	-	9	1
Gammaridae	-	-	-	1	-	2	1	2
Talitridae								
Hyalella azteca	76	-	-	-	12	13	50	-
Ostracoda	-	-	-	-	-	-	-	15
Hexapoda								
Insecta								
Collembola	-	-	3	-	-	-	-	-
Ephemeroptera								
Pisciforma								
Baetidae	16	51	12	112	16	31	-	1
Callibaetis	-	-	-	-	14	-	-	-
Centroptilum	-	-	-	-	1	-	-	-
Cloeon	-	-	3	-	-	-	-	-
Siphlonuridae	2	-	-	-	1	-	2	-
Furcatergalia								
Caenidae								
Caenis	-	1	1	-	1	-	-	-
Setisura								
Heptageniidae	-	-	-	-	1	1	-	2
Stenacron interpunctatum	-	-	-	-	-	2	-	-
Odonata								
Anisoptera								
Libellulidae	-	-	-	-	-	-	1	-
Zygoptera								
Coenagrionidae	-	-	5	-	2	2	4	-
Argia	-	-	1	1	4	-	1	-
Enallagma	2	-	1	-	2	1	2	-
Hemiptera								
Heteroptera								
Corixidae	11	-	8	4	2	-	6	18

Table 3-6. Benthic macroinvertebrates identified at T-1 (station 01576529) site in September for years 1993-2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Dasycorixa	-	-	-	-	2	-	-	-
Palmacorixa	-	-	-	-	6	-	-	-
Sigara	-	-	-	-	-	1	1	-
Trichocorixa	2	-	1	-	6	-	-	-
Coleoptera	-	-	-	-	-	1	-	-
Adephaga								
Haliplidae								
Haliplus	-	-	-	-	-	-	-	2
Peltodytes	6	-	-	-	-	-	1	-
Dytiscidae	-	-	-	-	1	-	-	-
Polyphaga								
Hydrophilidae								
Berosus	-	1	1	-	2	-	-	-
Psephenidae								
Ectopria	1	-	-	-	1	4	1	4
Psephenus herricki	-	-	-	-	-	1	-	-
Scirtidae	1	-	-	-	-	1	1	-
Elmidae	-	-	-	2	1	-	-	-
Dubiraphia	6	1	3	-	1	4	1	4
Optioservus	-	-	1	-	2	10	1	3
Promoresia	-	-	-	-	-	1	-	-
Stenelmis	13	13	19	5	6	40	15	7
Megaloptera								
Sialidae								
Sialis	-	-	-	-	1	-	-	-
Trichoptera								
Hydropsychidae	1	2	-	5	1	9	-	-
Cheumatopsyche	-	2	7	25	11	6	4	9
Hydropsyche	-	5	-	16	2	5	-	-
Hydroptilidae								
Hydroptila	-	14	-	-	3	2	-	-
Orthotrichia	-	-	-	-	-	-	2	-
Diptera								
Nematocera								
Ceratopogonidae	1	-	-	-	-	-	-	-
Bezzia	-	-	-	-	-	-	1	-
Ceratopogon	-	1	-	-	-	-	-	-
Probezzia	-	-	2	-	-	1	-	-
Chironomidae								
Tanypodinae	-	-	-	-	1	-	-	-
Coelotanypodini								

Table 3-6. Benthic macroinvertebrates identified at T-1 (station 01576529) site in September for years 1993-2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Clinotanypus	1	-	-	-	1	-	28	-
Coelotanypus	-	-	-	-	1	-	-	-
Pentaneurini								
Thienemannimyia gr.	-	2	-	4	6	2	2	6
Procladiini								
Procladius	-	-	-	-	4	1	-	3
Tanypus	-	-	-	-	1	-	-	-
Orthocladiinae	-	1	-	-	-	-	-	-
Corynoneurini								
Corynoneura	-	-	-	-	-	-	-	1
Orthocladiini								
Cricotopus	1	32	9	7	10	1	-	-
Eukiefferiella	-	-	-	-	-	-	-	1
Nanocladius	2	1	-	-	1	1	-	-
Orthocladius	-	17	-	-	-	2	-	-
Parakiefferiella	6	-	-	-	-	-	-	-
Parametriocnemus	-	-	-	1	-	2	-	2
Thienemanniella	-	2	2	-	-	2	1	-
Tvetenia	-	-	-	1	-	1	-	-
Chironominae	-	-	-	-	1	-	-	-
Chironomini								
Chironomus	-	8	22	2	1	6	-	7
Cryptochironomus	-	1	-	2	1	1	-	1
Dicrotendipes	1	2	16	3	9	1	-	-
Paratendipes	1	1	-	1	-	1	-	3
Phaenopsectra	-	-	-	1	-	-	-	-
Polypedilum	1	3	3	13	5	6	4	1
Tanytarsini								
Cladotanytarsus	1	2	-	-	-	1	-	-
Micropsectra	-	-	1	-	-	-	-	-
Paratanytarsus	-	1	-	-	-	-	-	-
Rheotanytarsus	27	4	26	3	1	13	19	36
Subletta	-	-	-	1	1	-	-	-
Tanytarsus	4	3	2	3	2	1	-	-
Culicidae								
Anopheles	-	-	1	-	1	-	-	-
Simuliidae								
Simulium	6	2	-	20	3	1	-	43
Tipulidae								
Tipulinae								
Tipula	-	-	-	-	1	1	-	-

Table 3-6. Benthic macroinvertebrates identified at T-1 (station 01576529) site in September for years 1993-2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T-1 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Limoniinae								
Antocha	-	-	-	-	-	1	-	-
Lipsothrix	-	-	-	-	-	1	-	-
Brachycera-Orthorrhapha								
Empididae								
Hemerodromia	-	-	-	-	-	1	-	2
Tabanidae								
Chrysops	-	-	-	-	-	-	1	-

Table 3-7. Benthic macroinvertebrates identified at T1-3 (station 01576528) site in May for years 1994- 2001.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T1-3 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Platyhelminthes								
Turbellaria								
Tricladida								
Planariidae	-	-	-	-	-	3	1	1
Annelida								
Clitellata								
Hirudinea	1	-	-	-	-	-	-	-
Arhynchobdellida	-	-	-	-	2	-	-	-
Erpobdellidae	-	-	-	-	-	-	2	-
Rhynchobdellida								
Glossiphoniidae	-	1	-	-	-	-	-	-
Oligochaeta								
Branchiobdellida								
Branchiobdellae	-	1	-	-	-	-	-	-
Haplotaxida								
Enchytraeidae	-	1	-	-	2	-	-	-
Naididae	-	2	-	-	-	-	-	1
Nais	-	52	-	75	-	115	6	4
Ophidonais serpentina	-	-	-	81	-	16	-	-
Tubificidae	-	12	-	5	40	27	7	1
Ilyodrilus templetoni	1	-	-	-	-	-	-	-
Limnodrilus	4	1	-	2	10	-	-	2
Arthropoda								
Chelicerata								
Arachnida								
Acariformes								
Prostigmata								
Hygrobatidae								
Hygrobates	-	1	-	-	-	-	-	-
Crustacea								
Malacostraca								
Isopoda								
Asellidae								
Caecidotea	-	-	-	-	-	1	10	3
Amphipoda								
Gammaridae	9	1	-	2	88	8	109	139
Maxillopoda								
Copepoda	-	-	-	-	-	4	-	-
Ostracoda	2	-	-	-	-	-	-	-
Hexapoda								
Insecta								

Table 3-7. Benthic macroinvertebrates identified at T1-3 (station 01576528) site in May for years 1994- 2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T1-3 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Ephemeroptera								
Pisciforma								
Baetidae	5	-	4	12	2	-	-	-
Siphlonuridae	-	-	-	-	2	-	-	-
Odonata								
Zygoptera								
Coenagrionidae								
Enallagma	-	-	-	-	2	-	-	-
Hemiptera								
Heteroptera								
Corixidae	-	-	-	-	4	-	-	-
Coleoptera								
Adephaga								
Dytiscidae								
Agabus	1	-	-	-	-	2	2	1
Polyphaga								
Psephenidae								
Ectopria	-	-	-	-	-	-	-	1
Elmidae	-	-	-	-	-	4	-	-
Dubiraphia	1	2	2	4	8	3	1	3
Optioservus	-	-	1	-	2	4	4	4
Stenelmis	-	1	3	1	6	1	2	7
Trichoptera								
Hydropsychidae								
Hydropsyche	-	-	1	-	2	-	-	-
Hydroptilidae								
Hydroptila	-	-	2	1	2	1	-	1
Diptera								
Nematocera								
Ceratopogonidae								
Bezzia	-	-	-	-	-	-	1	-
Probezzia	-	1	-	-	-	-	-	-
Chironomidae								
Tanypodinae	-	-	-	-	2	-	-	-
Natarsiini								
Natarsia	-	-	-	-	2	-	-	-
Pentaneurini								
Thienemannimyia gr.	-	2	-	-	-	-	-	-
Procladiini								
Procladius	-	-	1	-	-	-	3	-
Diamesinae								

Table 3-7. Benthic macroinvertebrates identified at T1-3 (station 01576528) site in May for years 1994- 2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T1-3 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Diamesini								
Diamesa	-	-	1	-	-	-	-	-
Pagastia	-	-	6	-	-	-	-	-
Orthocladiinae	1	-	-	-	-	-	-	-
Orthocladiini								
Acricotopus	-	-	-	-	-	2	-	-
Cricotopus	6	9	94	4	44	-	8	-
Eukiefferiella	2	1	-	1	2	-	-	-
Orthocladius	32	4	48	22	20	3	6	4
Parakiefferiella	1	11	1	1	-	-	-	2
Parametriocnemus	-	-	-	-	-	-	2	-
Rheocricotopus	-	-	-	-	-	-	1	-
Stilocladius	-	-	-	-	-	1	-	-
Thienemanniella	-	-	1	-	2	1	-	-
Tvetenia	1	-	-	-	4	-	-	-
Chironominae								
Chironomini								
Chironomus	1	1	3	-	4	-	4	-
Cryptochironomus	-	-	-	-	2	-	-	-
Dicrotendipes	4	89	2	2	26	1	8	-
Polypedilum	1	1	7	-	-	-	-	-
Stictochironomus	1	-	1	-	8	-	4	6
Tanytarsini								
Micropsectra	-	10	5	-	76	4	8	-
Paratanytarsus	-	-	-	-	10	-	-	-
Rheotanytarsus	-	-	6	-	2	-	-	1
Simuliidae								
Simulium	135	3	2	-	22	4	15	12
Brachycera-Orthorrhapha								
Empididae								
Hemerodromia	-	1	-	-	-	-	-	-
Brachycera-Cyclorrhapha								
Muscidae	1	-	-	-	-	-	-	-

Table 3-8. Benthic macroinvertebrates identified at T1-3 (station 01576528) site in September for years 1993- 2000.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T1-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Platyhelminthes								
Turbellaria								
Tricladida								
Dugesiidae								
Cura	-	21	-	-	-	-	-	-
Planariidae	-	-	4	-	4	2	2	-
Phagocata	2	-	-	-	-	-	-	-
Annelida								
Clitellata								
Hirudinea								
Arhynchobdellida								
Erpobdellidae	2	2	1	-	-	-	-	-
Rhynchobdellida								
Glossiphoniidae								
Gloiobdella elongata	-	-	-	-	-	-	1	-
Oligochaeta	-	3	-	-	-	-	-	-
Haplotaxida								
Enchytraeidae	-	-	-	-	-	-	1	-
Naididae								
Nais	-	-	-	1	-	-	-	-
Ophidonais serpentina	-	-	-	-	-	1	-	-
Pristinella	-	-	-	-	-	-	1	-
Tubificidae	-	-	-	-	9	5	5	-
Aulodrilus	-	-	-	-	-	-	1	-
Limnodrilus	16	-	-	-	-	-	1	5
Tubifex	4	-	-	-	-	-	-	-
Lumbriculida								
Lumbriculidae	-	-	-	-	-	10	-	1
Mollusca								
Gastropoda								
Basommatophora								
Lymnaeidae								
Fossaria	-	-	-	-	-	-	2	-
Pseudosuccinea columella	-	-	-	-	-	-	4	-
Physidae	-	-	1	-	-	-	-	-
Physa	-	-	-	-	7	-	20	-
Planorbidae								
Armiger crista	-	-	-	-	-	1	-	-
Bivalvia								
Veneroida								
Pisidiidae	2	-	-	-	3	1	-	-

Table 3-8. Benthic macroinvertebrates identified at T1-3 (station 01576528) site in September for years 1993- 2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T1-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Pisidium	-	-	-	-	-	1	-	-
Arthropoda								
Chelicerata								
Arachnida								
Acariformes								
Prostigmata								
Lebertiidae								
Lebertia	-	-	-	-	2	-	1	-
Crustacea								
Malacostraca								
Isopoda								
Asellidae								
Caecidotea	2	-	-	-	-	-	2	-
Amphipoda								
Gammaridae	2	2	-	7	22	75	8	50
Talitridae								
Hyalella azteca	-	-	-	-	-	32	2	-
Maxillopoda								
Copepoda								
Cyclopidae	-	1	-	-	-	-	-	-
Ostracoda	-	-	-	-	-	-	-	1
Hexapoda								
Insecta								
Ephemeroptera								
Pisciforma								
Baetidae	14	3	-	49	-	1	-	7
Callibaetis	-	-	-	-	6	-	-	-
Siphlonuridae	-	-	-	-	-	1	19	-
Furcatergalia								
Caenidae								
Caenis	-	-	-	-	-	1	1	-
Odonata								
Anisoptera								
Aeshnidae								
Boyeria	-	-	-	-	-	-	1	-
Zygoptera								
Coenagrionidae	-	-	-	-	3	11	-	-
Argia	-	-	-	-	-	1	-	-
Enallagma	-	-	3	-	-	2	41	1
Ischnura	-	-	-	-	-	2	-	-
Hemiptera								

Table 3-8. Benthic macroinvertebrates identified at T1-3 (station 01576528) site in September for years 1993- 2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T1-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Heteroptera								
Corixidae	-	-	2	-	6	2	10	-
Dasycorixa	-	-	-	-	2	-	-	-
Sigara	-	-	-	-	-	1	6	-
Trichocorixa	-	-	-	-	3	1	-	-
Coleoptera								
Adephaga								
Haliplidae								
Peltodytes	2	-	-	-	-	-	-	-
Dytiscidae								
Agabetes	2	-	-	-	-	-	-	-
Laccophilus	-	-	1	-	-	-	-	-
Polyphaga								
Hydrochidae								
Hydrochus	-	-	-	-	-	-	1	-
Hydrophilidae	-	-	-	-	-	1	2	-
Berosus	-	1	3	-	2	3	-	-
Helochares	-	-	-	-	-	-	5	-
Tropisternus	-	-	-	-	-	-	1	-
Dryopidae								
Helichus	-	-	-	-	-	-	1	-
Scirtidae	-	-	-	-	-	1	-	-
Elodes	-	-	-	-	-	-	2	-
Elmidae	-	-	-	5	-	-	-	6
Dubiraphia	-	-	7	1	10	9	2	37
Optioservus	-	-	2	-	4	-	2	17
Stenelmis	4	2	8	2	12	6	3	33
Trichoptera								
Hydropsychidae	-	5	-	16	1	-	-	-
Cheumatopsyche	-	1	3	28	5	2	-	3
Hydropsyche	-	3	3	37	-	-	-	-
Hydroptilidae								
Hydroptila	-	1	-	1	2	-	-	1
Oxyethira	-	-	1	-	-	-	-	-
Diptera								
Nematocera								
Ceratopogonidae								
Bezzia	-	-	-	-	1	-	1	1
Probezzia	1	-	5	-	-	-	1	-
Chironomidae								
Tanypodinae								

Table 3-8. Benthic macroinvertebrates identified at T1-3 (station 01576528) site in September for years 1993- 2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T1-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Coelotanypodini								
Clinotanypus	-	-	-	-	-	-	6	-
Macorpelopiini								
Psectrotanypus	2	-	-	-	-	-	-	-
Natarsiini								
Natarsia	-	-	-	-	2	1	-	-
Pentaneurini								
Ablabesmyia	-	1	-	-	-	-	2	-
Conchapelopia	6	-	-	-	-	-	-	-
Paramerina	-	-	-	-	1	-	-	-
Thienemannimyia gr.	4	3	-	9	7	6	4	6
Zavrelimyia	-	-	-	-	-	-	2	-
Procladiini								
Procladius	-	2	-	-	5	-	31	-
Tanypodini								
Tanypus	-	1	-	-	1	-	1	-
Orthocladiinae	-	-	-	-	-	1	1	-
Corynoneurini								
Corynoneura	-	1	-	-	-	2	-	-
Orthocladiini								
Cardiocladius	-	-	-	1	-	-	-	-
Cricotopus/Orthocladius	4	-	-	-	-	-	-	-
Cricotopus	4	-	12	1	3	-	-	-
Cricotopus trifascia gr.	-	-	-	-	-	-	1	-
Nanocladius	-	-	-	-	-	1	-	-
Orthocladius	-	-	7	-	2	2	-	-
Parakiefferiella	-	-	-	-	2	-	3	-
Parametriocnemus	-	-	-	-	1	-	-	3
Thienemanniella	4	-	-	-	3	1	-	-
Tvetenia	-	1	-	2	-	-	-	-
Chironominae	-	-	-	-	-	-	-	2
Chironomini								
Chironomus	-	1	19	6	26	-	3	1
Cryptochironomus	2	-	-	1	3	-	1	1
Cryptotendipes	-	-	-	-	1	-	-	-
Dicrotendipes	-	23	15	25	31	1	-	1
Microtendipes	-	-	-	-	1	-	-	-
Paratendipes	-	-	-	-	1	2	-	2
Phaenopsectra	-	-	2	1	-	-	-	-
Polypedilum	18	15	1	27	-	1	1	2
Stictochironomus	-	-	-	1	-	-	-	-

Table 3-8. Benthic macroinvertebrates identified at T1-3 (station 01576528) site in September for years 1993- 2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T1-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Tribelos	-	-	-	-	-	1	-	-
Tanytarsini								
Cladotanytarsus	-	-	2	-	-	-	-	-
Micropsectra	2	2	1	-	-	-	2	-
Paratanytarsus	-	-	-	-	3	2	-	-
Rheotanytarsus	38	66	56	3	2	3	4	13
Tanytarsus	-	-	5	4	12	1	-	1
Culicidae								
Aedes	-	-	-	-	-	-	1	-
Anopheles	-	-	-	1	-	1	3	-
Psychodidae	-	-	1	-	-	-	-	-
Similiidae								
Simulium	44	53	7	28	-	-	-	-
Tipulidae								
Limoniinae								
Antocha	-	-	-	1	-	-	-	-
Brachycera-Orthorrhapha								
Empididae								
Hemerodromia	-	-	-	-	-	-	-	3
Tabanidae	-	-	-	-	2	-	-	-
Brachycera-Cyclorrhapha								
Ephydridae	2	-	-	-	2	-	-	-
Muscidae	-	-	-	-	1	-	-	-

Table 3-9. Benthic macroinvertebrates identified at T2-3 (station 01576526) site in May for years 1994-2001.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T2-3 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Platyhelminthes								
Turbellaria								
Tricladida								
Planariidae/Dugesiidae								
Dugesia/*Cura*	2	-	-	-	-	-	-	-
Annelida								
Clitellata								
Hirudinea								
Arhynchobdellida								
Erpobdellidae	-	-	1	-	-	-	-	-
Mooreobdella	-	-	-	2	-	-	-	-
Oligochaeta	6	-	-	-	-	-	-	-
Haplotaxida								
Enchytraeidae	3	3	-	-	-	8	1	-
Naididae								
Nais	11	45	-	22	5	11	8	8
Ophidonais serpentina	-	-	1	-	-	-	-	-
Pristinella	1	-	-	-	-	-	-	-
Tubificidae	-	2	5	5	1	41	5	3
Ilyodrilus templetoni	18	-	-	-	-	-	-	-
Limnodrilus	6	-	2	-	10	2	-	3
Mollusca								
Gastropoda								
Basommatophora								
Physidae								
Physinae	-	-	-	-	1	-	-	-
Bivalvia								
Veneroida								
Pisidiidae	-	-	-	-	1	-	-	-
Arthropoda								
Chelicerata								
Arachnida								
Acariformes								
Prostigmata								
Hygrobatidae								
Hygrobates	-	3	-	1	-	-	-	-
Sperchonidae								
Sperchon	-	-	-	1	-	-	-	-
Crustacea								
Malacostraca								
Isopoda								

Table 3-9. Benthic macroinvertebrates identified at T2-3 (station 01576526) site in May for years 1994-2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T2-3 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Asellidae								
Caecidotea	-	-	-	-	-	1	3	1
Amphipoda								
Gammaridae	15	-	122	48	103	85	130	67
Ostracoda	1	13	-	-	-	-	-	-
Hexapoda								
Insecta								
Collembola								
Isotomidae	-	-	-	-	-	1	-	-
Isotomurus	-	1	1	-	-	-	-	-
Ephemeroptera								
Pisciforma								
Baetidae	1	-	1	17	1	-	3	-
Coleoptera								
Adephaga								
Haliplidae								
Peltodytes	-	-	-	-	-	1	-	-
Dytiscidae								
Agabetes	1	1	2	-	1	-	5	3
Polyphaga								
Hydrophilidae	-	-	-	-	1	-	-	-
Elmidae	-	-	-	2	-	1	-	-
Dubiraphia	-	3	8	-	5	1	1	8
Optioservus	-	1	6	2	2	1	3	7
Promoresia	-	-	-	-	-	1	-	-
Stenelmis	-	1	3	-	-	1	2	7
Trichoptera								
Hydropsychidae								
Hydropsyche	-	-	2	1	-	-	-	-
Diptera								
Nematocera								
Ceratopogonidae								
Bezzia	-	-	-	-	-	2	-	-
Probezzia	-	1	2	-	1	-	-	-
Chironomidae								
Tanypodinae								
Natarsiini								
Natarsia	-	-	-	-	-	-	-	1
Pentaneurini								
Thienemannimyia gr.	-	1	-	-	-	-	-	-
Conchapelopia	-	-	1	-	-	-	-	-

Table 3-9. Benthic macroinvertebrates identified at T2-3 (station 01576526) site in May for years 1994-2001.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T2-3 May benthic-macroinvertebrate numbers by year							
	1994	1995	1996	1997	1998	1999	2000	2001
Diamesinae								
Diamesini								
Diamesa	3	-	-	2	1	2	-	-
Pagastia	-	-	3	-	-	-	-	-
Orthocladiinae	-	-	-	1	-	-	-	-
Orthocladiini								
Cricotopus	28	4	22	15	15	4	8	5
Cricotopus trifascia gr.	-	-	-	-	-	-	-	1
Eukiefferiella	2	-	-	-	-	-	-	-
Orthocladius	52	-	57	65	5	22	4	27
Parakiefferiella	1	2	-	-	-	-	-	3
Parametriocnemus	-	-	3	1	-	1	-	-
Rheocricotopus	-	-	-	1	-	-	3	1
Stilocladius	-	-	-	-	-	4	-	-
Tvetenia	1	-	-	1	2	2	-	-
Chironominae	-	1	-	-	-	-	-	-
Chironomini								
Chironomus	-	-	-	-	1	-	1	-
Dicrotendipes	1	111	4	-	5	2	2	9
Einfeldia	-	-	-	-	-	-	-	4
Paracladopelma	-	-	-	-	-	1	-	-
Paratendipes	-	1	1	-	-	2	2	6
Polypedilum	-	-	-	1	2	-	-	-
Stictochironomus	-	-	1	-	-	3	-	27
Tanytarsini								
Micropsectra	-	14	12	-	41	2	2	2
Paratanytarsus	-	1	-	-	-	-	-	-
Rheotanytarsus	-	-	1	-	-	-	-	-
Subletta	-	-	-	-	1	-	-	-
Tanytarsus	-	-	-	-	-	1	-	-
Similiidae								
Simulium	45	1	1	1	8	3	16	-
Brachycera-Orthorrhapha								
Empididae								
Hemerodromia	-	3	-	2	-	-	-	-
Brachycera-Cyclorrhapha								
Ephydridae	-	-	-	-	-	1	-	-

Table 3-10. Benthic macroinvertebrates identified at T2-3 (station 01576526) site in September for years 1993- 2000.

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T2-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Cnidaria								
Hydrozoa								
Hydroida	-	1	-	-	-	-	-	-
Platyhelminthes								
Turbellaria								
Tricladida								
Dugesiidae								
Cura	-	7	-	-	-	-	-	-
Planariidae	-	-	2	-	-	-	1	-
Phagocata	2	-	-	-	-	-	-	-
Annelida								
Clitellata								
Hirudinea								
Arhynchobdellida								
Erpobdellidae	2	1	1	1	1	-	2	-
Rhynchobdellida								
Glossiphoniidae								
Helobdella stagnalis	8	1	-	-	-	-	1	-
Oligochaeta								
Haplotaxida	-	13	20	-	17	6	5	-
Naididae								
Chaetogaster	-	-	-	-	1	-	-	-
Tubificidae								
Limnodrilus	92	-	-	-	-	-	1	4
Tubifex	20	-	-	-	-	-	-	-
Mollusca								
Gastropoda								
Basommatophora								
Lymnaeidae								
Pseudosuccinea columella	-	-	-	-	-	1	-	-
Physidae	4	-	-	-	1	-	-	-
Physinae	-	-	-	-	-	2	-	-
Physa	-	-	-	-	-	-	1	-
Physella	-	-	-	1	-	-	-	-
Bivalvia								
Veneroida								
Pisidiidae	-	3	-	-	-	-	-	-
Pisidium	-	-	-	-	2	-	1	-
Arthropoda								
Chelicerata								
Arachnida								

Table 3-10. Benthic macroinvertebrates identified at T2-3 (station 01576526) site in September for years 1993- 2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T2-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Acariformes								
Prostigmata								
Hygrobatidae								
Hygrobates	-	1	11	-	-	-	-	-
Lebertiidae								
Lebertia	-	-	1	-	-	-	-	-
Crustacea								
Malacostraca								
Isopoda								
Asellidae								
Caecidotea	-	-	-	-	1	4	11	-
Amphipoda								
Gammaridae	24	3	10	40	8	125	149	152
Maxillopoda								
Copepoda								
Cyclopidae	-	2	-	-	-	-	-	-
Ostracoda	-	34	-	-	-	-	-	-
Hexapoda								
Insecta								
Collembola	-	-	1	-	-	-	-	-
Isotomidae								
Isotomurus	-	-	-	-	3	-	-	-
Ephemeroptera								
Pisciforma								
Baetidae	10	19	4	73	-	3	1	6
Callibaetis	-	-	-	-	-	1	-	-
Centroptilum	-	-	-	-	4	-	-	-
Cloeon	-	-	2	-	-	-	-	-
Siphlonuridae	6	2	-	-	1	-	-	-
Odonata								
Zygoptera								
Coenagrionidae	-	2	3	-	7	-	-	1
Argia	-	-	1	-	1	-	-	-
Enallagma	-	-	3	-	-	-	-	-
Ischnura	-	-	-	-	2	-	-	-
Hemiptera								
Heteroptera								
Corixidae	-	-	12	-	-	-	3	-
Palmacorixa	-	-	-	-	-	-	1	-
Sigara	-	-	7	-	-	-	2	-
Trichocorixa	-	-	2	-	-	-	-	-

Table 3-10. Benthic macroinvertebrates identified at T2-3 (station 01576526) site in September for years 1993- 2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T2-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Coleoptera	-	-	-	-	-	1	-	-
Adephaga								
Haliplidae								
Peltodytes	6	-	-	-	-	-	-	-
Dytiscidae	-	-	-	-	-	-	-	1
Polyphaga								
Hydroptilidae								
Hydrochus	-	-	-	-	-	-	1	-
Hydrophilidae								
Berosus	-	-	2	2	4	-	-	-
Hydrobius	-	-	1	-	-	-	-	-
Elmidae	-	-	-	-	-	-	1	-
Dubiraphia	4	-	16	3	15	2	11	7
Optioservus	8	-	2	1	-	8	3	14
Promoresia	-	-	1	1	3	1	-	-
Stenelmis	4	2	5	3	3	8	4	7
Trichoptera								
Hydropsychidae	-	-	-	4	1	1	-	-
Cheumatopsyche	-	-	9	4	-	6	1	5
Hydropsyche	-	-	-	30	1	3	-	-
Hydroptilidae								
Hydroptila	-	-	1	2	-	1	-	-
Diptera								
Nematocera								
Ceratopogonidae								
Probezzia	-	-	-	-	1	-	-	-
Chironomidae								
Tanypodinae								
Coelotanypodini								
Clinotanypus	-	-	-	-	-	-	1	-
Natarsiini								
Natarsia	-	1	-	-	-	-	-	1
Pentaneurini								
Conchapelopia	-	-	-	1	-	-	-	-
Thienemannimyia gr.	-	6	-	-	1	2	1	1
Paramerina	-	-	-	-	5	-	-	-
Procladiini								
Procladius	-	2	-	-	1	1	-	-
Orthocladiinae	-	-	-	-	1	-	-	-
Corynoneurini								
Corynoneura	-	-	-	-	-	3	-	1

Table 3-10. Benthic macroinvertebrates identified at T2-3 (station 01576526) site in September for years 1993- 2000.—Continued

[Taxonomy was checked on March 1, 2006]

Benthic Macroinvertebrates	Site T2-3 September benthic-macroinvertebrate numbers by year							
	1993	1994	1995	1996	1997	1998	1999	2000
Orthocladiini								
Cricotopus	8	1	17	12	9	5	-	-
Nanocladius	-	-	-	-	-	1	-	-
Orthocladius	-	11	21	1	6	2	-	-
Parakiefferiella	-	2	2	-	4	-	-	-
Parametriocnemus	-	-	1	-	-	1	-	-
Thienemanniella	-	-	-	-	1	-	-	-
Tvetenia	-	-	-	1	-	-	-	-
Chironominae	-	-	-	-	2	1	-	3
Chironomini								
Chironomus	-	2	-	4	28	9	1	-
Cryptochironomus	2	-	1	1	3	-	-	-
Dicrotendipes	-	58	32	4	8	-	1	-
Paracladopelma	-	-	1	-	3	1	-	-
Paratendipes	-	-	-	-	2	9	7	1
Phaenopsectra	-	-	-	1	-	-	-	-
Polypedilum	-	2	5	14	4	1	-	1
Pseudochironomini								
Pseudochironomus	-	-	-	1	-	-	-	-
Tanytarsini								
Cladotanytarsus	-	-	-	1	-	-	-	-
Micropsectra	-	3	-	-	-	-	-	-
Paratanytarsus	-	-	-	-	3	-	-	-
Stictochironomus	-	-	-	-	-	-	6	-
Rheotanytarsus	4	2	4	6	1	2	-	-
Subletta	-	-	-	-	2	-	-	-
Tanytarsus	-	1	2	-	4	-	1	-
Culicidae								
Anopheles	-	-	1	-	2	-	-	-
Culex	-	-	-	-	1	-	-	-
Simuliidae								
Simulium	2	-	2	5	-	-	-	2
Tipulidae								
Limoniinae								
Molophilus	-	-	-	-	-	-	1	-