doi:10.2489/jswc.63.6.475

Long-term stream chemistry trends in the southern Georgia Little River Experimental Watershed

G.W. Feyereisen, R. Lowrance, T.C. Strickland, D.D. Bosch, and J.M. Sheridan

Abstract: Long-term stream water quality data may provide opportunities to study the effectiveness of conservation practices. The first three decades of data for the Little River in southwestern Georgia were analyzed for trends as part of the Conservation Effects Assessment Project. Concentrations and loads for chloride, ammonium-N, nitrate plus nitrite-N, total Kjeldahl N, total P, and dissolved molybdate reactive phosphorus were determined from 1974 through 2003 for eight nested subwatersheds in the Little River Experimental Watershed. There was a statistically significant downward trend for annual mean total phosphorus concentration in five subwatersheds and an upward trend for chloride in three subwatersheds. The decrease in total phosphorus concentration occurred primarily in winter. Trends in phosphorus and chloride concentrations did not appear to be related to land use. There were no statistical differences in annual streamflow or nutrient loads expressed on a per area basis among the nested subwatersheds. Annual and seasonal flow-weighted mean concentrations were different among the subwatersheds for nitrate-N and chloride. The larger subwatersheds had significantly higher nitrate-N in winter and spring. The nutrient loads and concentrations from these subwatersheds were an order of magnitude less compared to other agricultural watersheds. Conservation practices were implemented on 11% of the watershed area from 1980 to 2003; however, the affects of the practices on watershed water quality was not clear. Earlier short-term studies attributed the low levels of nutrient transport to the presence of extensive riparian forests and the general prevalence of forest in these mixed land use watersheds.

Key words: Conservation Effects Assessment Project (CEAP)—Coastal Plain—nutrients—stream chemistry—water quality—watershed data record

Water resource quantity and quality issues will continue to be of concern throughout the United States as the population expands, land development continues, and food and bio-energy production demands grow. Flooding, drought, and potential water quality impairment by anthropogenic activities or natural processes will be significant topics for water resource managers, researchers, and policy makers for the foreseeable future. Of particular interest currently to agricultural policy makers are the effects that various agricultural land and water conservation measures have on water resources. A large body of research shows that nutrients originating from agricultural activities are associated with increased levels of nitrate-N in groundwater, degraded stream and lake water quality, and environmental problems in estuaries and coastal waterways (Ritter and Bergstrom 2001). The sources of the nutrients include fertilizers, animal wastes, mineralized soil organic matter, and municipal waste solids (Ritter and Bergstrom 2001). Nutrient losses to surface and ground waters occur via the processes of surface runoff, leaching below the root zone, lateral flow to riparian areas or streams, and artificial subsurface drainage. These processes are affected by climate, soil properties, topography, management practices, and crop rotation. Reducing the negative effects of nutrients on ground, surface, and coastal waterways has been a major focus of agricultural research in the past decades (Mostaghimi et al. 2001).

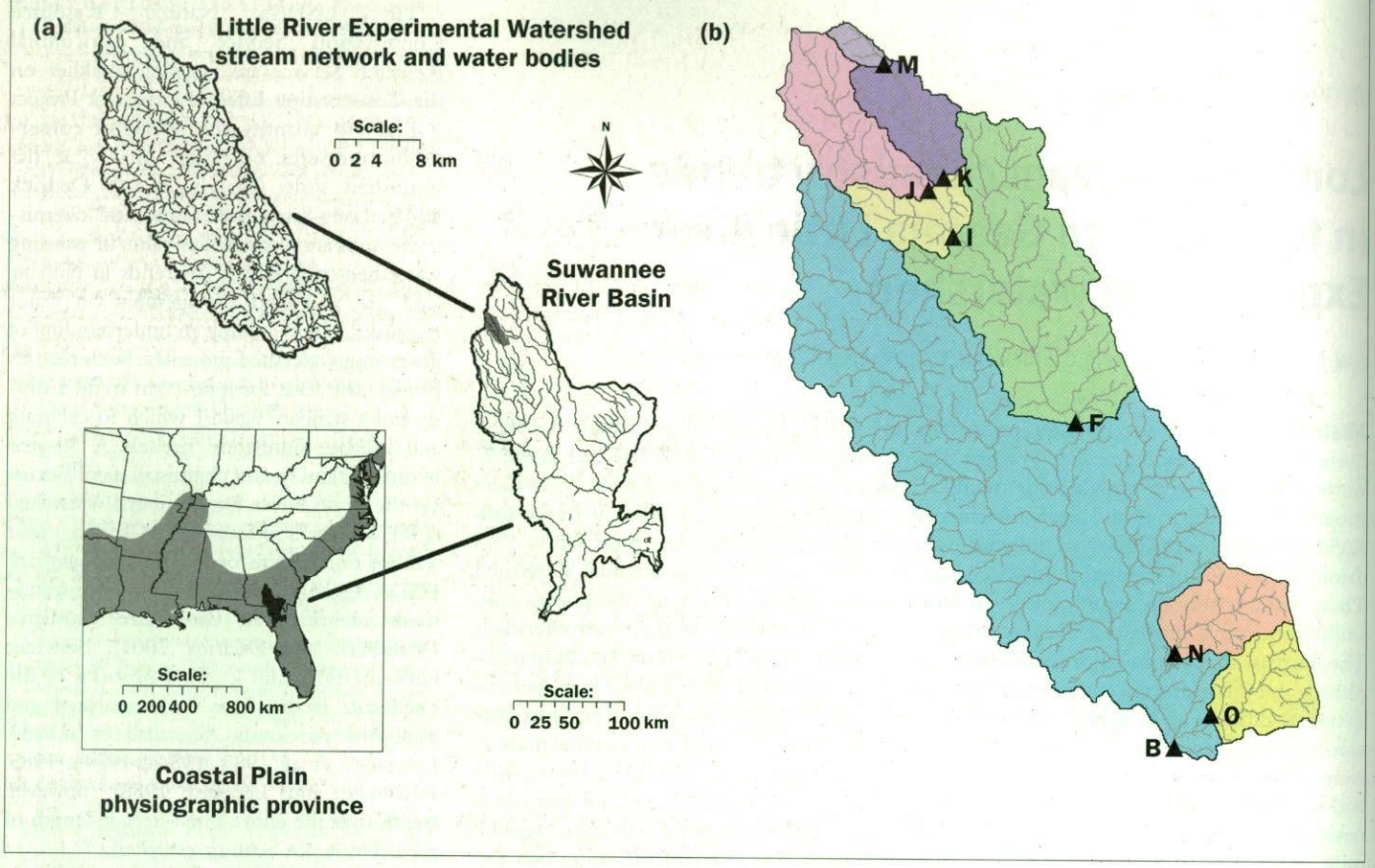
The USDA's Natural Resource Conservation Service and Agricultural Research Service are working together on the Conservation Effects Assessment Project (CEAP) to quantify the effects of conservation measures on water quality at the watershed scale (Mausbach and Dedrick 2004). Long-term streamflow and chemistry records are of particular value in assessing watershed-scale effects of trends in climate, land use, and land management on water resources, and in gaining an understanding of governing watershed processes. Such records provide the basis for long-term trend analysis and a standard against which to calibrate and validate simulation models. A 30-year hydrology and stream chemistry record exists for the Little River Experimental Watershed (LREW) in southwestern Georgia, near Tifton, one of the original 12 benchmark USDA Agricultural Research Service watersheds identified in the CEAP initiative (Mausbach and Dedrick 2004). Nutrient concentrations and loads of portions of the chemistry record have been analyzed and published previously (Sheridan et al.1983; Lowrance et al. 1983, 1984a, 1984b, 1985; Lowrance and Leonard 1988); however, trends over the entire three-decade length of record have not been investigated.

The LREW is a 334 km^2 (129 mi^2) drainage in the headwaters of the Suwannee River Basin. The headwaters of the Little River are located approximately 9 km (6 mi) west of Ashburn, Georgia, and the instrumented study area extends to the southeast approximately 36 km (22 mi) to 6 km (4 mi) west of Tifton, Georgia (figure 1a). The entire LREW (Little River Basin [LRB]) contains seven other subwatersheds (LRF, LRI, LRJ, LRK, LRM, LRN, and LRO) ranging in size from 9 to 115 km^2 (3.5 to 44 mi^2) (figure 1b). Stream slopes range from 0.1% to 0.5%; upland slopes are gentle and typically less than 5%. The mean sea level elevation in the LREW ranges from 146 m (479 ft) at the

Gary W. Feyereisen is a research hydrologist for the Pasture Systems and Watershed Management Research Unit, USDA Agricultural Research Service, University Park, Pennsylvania. Richard Lowrance is an ecologist, Timothy C. Strickland is a supervisory soil scientist, David D. Bosch is a research hydraulic engineer, and Joseph M. Sheridan is a research hydraulic engineer for the Southeast Watershed Research Laboratory, USDA Agricultural Research Service, Tifton, Georgia.

Figure 1

(a) Little River Experimental Watershed location in the Suwannee River Basin. (b) Little River Experimental Watershed subwatersheds.



top of the watershed to 81 m (266 ft) at the outlet, LRB (Sheridan 1997).

The following is a review of research findings that involve the stream nutrient records at the LREW. Sheridan et al. (1983) developed nitrate-N, dissolved molybdate reactive phosphorus (DMRP), and chloride (Cl) budgets for the 1,665-ha (4,114-ac) LRK from the stream water chemistry and flow data collected from 1975 to 1978. They reported that average annual streamflow losses of the three nutrients for this period were 0.3, 0.15, and 37.1 kg ha^{-1} y^{-1} (0.27, 0.13, and 33.1 lb ac^{-1} yr^{-1}), respectively. Streamflow nutrient loads were used to develop nutrient budgets for the entire 1,568-ha (3,875-ac) LRN and for the riparian zone of the subwatershed with 1979 data (Lowrance et al. 1983, 1984a, 1984b). Of the nitrate-N entering the riparian zone from upland water discharge and bulk precipitation, only about 1/3 (3.9 kg N ha^{-1} y^{-1}) (3.5 lb N ac^{-1} yr^{-1}) was discharged through streamflow. This annual streamflow load of total N was about 29% of the input to the watershed via precipitation. The nitrate-N concentration was highest during the winter months (January, February, March) and lowest during the summer months (July, August, September) (Lowrance et al. 1984a). Annual streamflow loads of nitrate-N, ammonium-N, organic-N, total N, DMRP, total P, and Cl for LRJ, LRK, LRN, and LRO for the years 1979 to 1981 were used to determine watershed nutrient budgets (Lowrance et al. 1985). Streamflow loads of N and Cl were higher for LRN and LRO than for LRJ and LRK, but loads of DMRP and total P did not show this same trend. Land use in LRN and LRO averaged 52% row crops and pasture while LRJ and LRK averaged 38% row crops and pasture. Although the differences in land use did not affect the ratio of streamflow runoff to precipitation, the larger areas in crop and pasture in LRN and LRO did increase the N load from the subwatersheds (Lowrance et al. 1985).

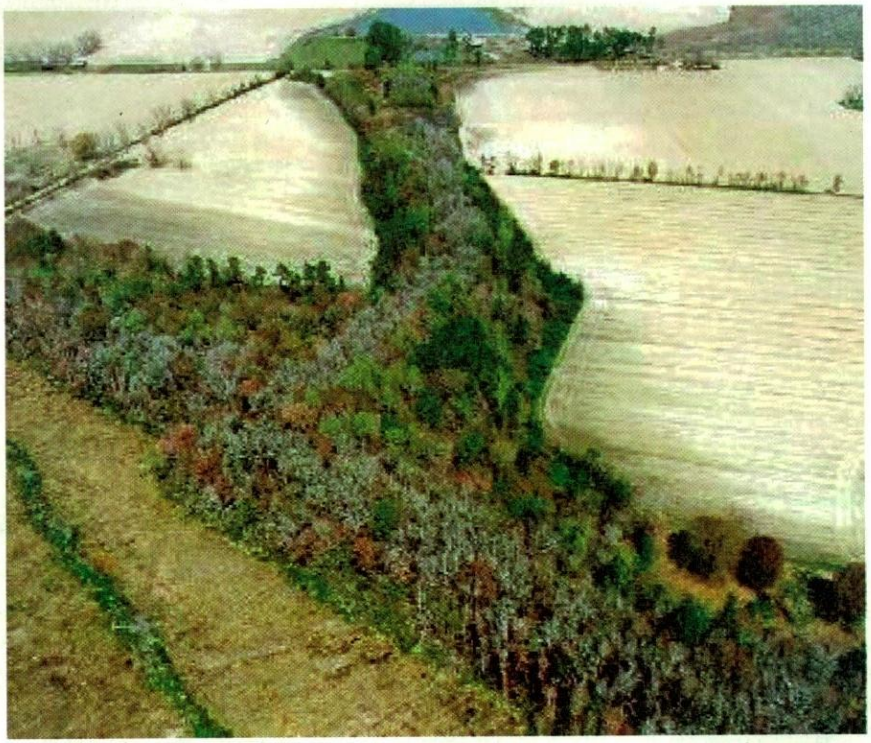
Streamflow nutrient concentrations and loads were determined for LRM, LRK, LRJ, LRI, and LRF, in a study that spanned four years, from spring 1982 to spring 1986 (Lowrance and Leonard 1988). Analysis of storm event data indicated that streamflow nitrate-N concentration was enriched and Cl concentration was diluted during stormflow relative to baseflow. The results for P were mixed. Dissolved molybdate reactive phosphorus concentration was enriched in LRJ, while total P concentration was diluted in LRJ and LRI. The other P concentrations showed no effect. Mean concentrations and loads obtained by sampling at 3-h and 12-h intervals were not significantly different from one another. The study results showed that streamflow nutrient loads were not correlated to percent of land in spring-planted row crops. However, there was correlation between nutrient loads and area in winter wheat, a crop that was typically fertilized in late fall or early winter just prior to the period of highest rainfall, lowest evapotranspiration and highest leaching potential in this region (Lowrance and Leonard 1988). Other studies in the LREW conducted on representative upland areas estimated surface and subsurface losses of nitrate-N (Hubbard and Sheridan 1983) and identified the division of surface and subsurface N losses by nitrate, ammonium, and organic N (Lowrance 1992).

This paper adds to the earlier reports on LREW using stream nutrient concentrations and loads for LREW for the first 30 years of data collection. Our objectives were (1) to determine if there are long-term annual or seasonal trends in stream nutrient concentrations or loads; (2) to determine if there are seasonal differences in concentrations and loads; and (3) to determine if there are differences in nutrient concentrations or loads among the subwatersheds of the LREW.

Materials and Methods

Watershed Soils and Climate. The soils of the uplands are typically low-fertility sands, loamy sands, and sandy loams with high infiltration capacity, underlain at a depth of 1 to 2 m (3.3 to 6.6 ft) with a plinthic layer of dense clay to sandy clay loam with low hydraulic conductivity. The soils are developed from the underlying Hawthorne formation, a formation of Miocene age consisting chiefly of interbedded sand, clay, claystone, and limestone or from small areas of deep sands located on the eastern side of major stream channels. The major part of the Hawthorne formation forms the confining layer for the underlying principal artesian aquifer (Stringfield 1966). The Hawthorne formation, and soil plinthic layers, limit deep recharge of the underlying principal artesian aquifer and divert percolating soil water to the dendritic stream channel network through lateral subsurface flow. Surface runoff from agricultural uplands occurs both when intense rainfall exceeds infiltration rates and when water tables are near the surface and soil available water storage is minimal. The alluvial lowland soils are poorly drained and generally support wetland vegetation. The riparian forests occupying these wetland soils are a dominant feature of the watersheds (figure 2). The alluvial soils have seasonal high water tables with more extensive swamps along larger streams, which are flooded for much of the winter and spring. Precipitation for the 30-year period from 1974 to 2003 averaged 1,213 mm (48 in). The ratio of annual streamflow to annual precipitation for the entire watershed (LRB) has been 0.27 over this period of time and has remained relatively stable (Bosch et al. 2006). Two distinct runoff periods have been noted for the LREW (Sheridan 1997). On average, 73% of the annual streamflow was measured at the LREW outlet during January through April when, on average, 37% of average annual rainfall occurred. During those four months, 54% of precipitation became streamflow whereas only 12% became streamflow during the remainder of the year.

Figure 2
Aerial photo showing riparian forest typical of the Little River Experimental Watershed.



Land Use. Land use in the LREW is primarily forest and agricultural, with less than 10% of the area being attributed to urban and water (table 1) (Bosch et al. 2006). Using LandSat images, Bosch et al. (2006) estimated that over the period from 1975 to 2003 forested area ranged from 41% to 56% of the watershed and that the tilled land ranged from 31% to 52%; pasture accounted for 2% to 10% (figure 3). During the 1980 to 1985 time frame, some forest was converted to agricultural land use, but the areas were relatively small. Although land will remain in forest for long periods of time, there were some relatively large forest tracts harvested during the record period. For example, Lowrance and Leonard (1988) noted that 240 ha (593 ac) of timber was clear cut in LRI in the fall of 1982. During the 1990s and continuing to present, there has been a trend to plant pine trees on previously cropped land as a conservation measure. Over the 1974 to 2003 period, the National Agricultural Statistics Service data for Turner County, which contains 59% of LREW, indicated major trends in proportion of total acres planted for corn, soybean, and wheat (decrease), and for cotton (increase) (figure 4) (Feyereisen et al. 2006). During these decades peanut was a major crop, accounting for 30% to 55% of the planted acreage.

Conservation Practices. Sullivan and Batten (2007) published information about the implementation of conservation practices in the LREW from 1980 to 2006, including a summary of statistics about the area having conservation practices applied and the types of practices used (figure 5) (Sullivan and Batten 2007). From 1980 to 2003, conservation practices had been implemented on approximately 11% of the area in the LREW (Sullivan and Batten 2007). Using the cumulative acreage of conservation practices implemented by 2003 as a baseline, adoption rates were generally <3% per year from 1980 to 1990 and rose steadily from 1990 to 1999. Adoption rates declined until they were <1% per year for 2002 to 2003, prior to new activity occurring in response to the 2002 Farm Bill. The most prevalent conservation practices identified were: nutrient management, pest management, grassed waterways, contour farming, seasonal residue

Table 1

Land-use classifications derived from the LandSat images collected from 1975 to 2003 for Little River Experimental Watershed (data adapted from Bosch et al. 2006).

	Percentage of total watershed acreage					
Classification	**June 4, 1975**	**July 11, 1980**	**July 2, 1985**	**June 14, 1990**	**Aug. 15, 1995**	**July 20, 2003**
Fallow	NC*	16%	10%	18%	7%	4%
General agriculture	34%	20%	42%	22%	33%	27%
Total tilled†	34%	36%	52%	40%	40%	31%
Water	0%	1%	1%	2%	2%	2%
Pasture	5%	2%	6%	3%	5%	10%
Riparian forest	23%	29%	22%	24%	28%	28%
Upland forest	26%	27%	19%	24%	20%	22%
Total forest‡	49%	56%	41%	48%	48%	50%
Urban	12%	5%	1%	6%	5%	7%

* NC = land use not included in the classification.

† Total tilled area includes both fallow and general agriculture.

‡ Total forest area includes both upland and riparian forest.

Figure 3

Results of the land-use classification of the Landsat images (adapted from Bosch et al. 2006).

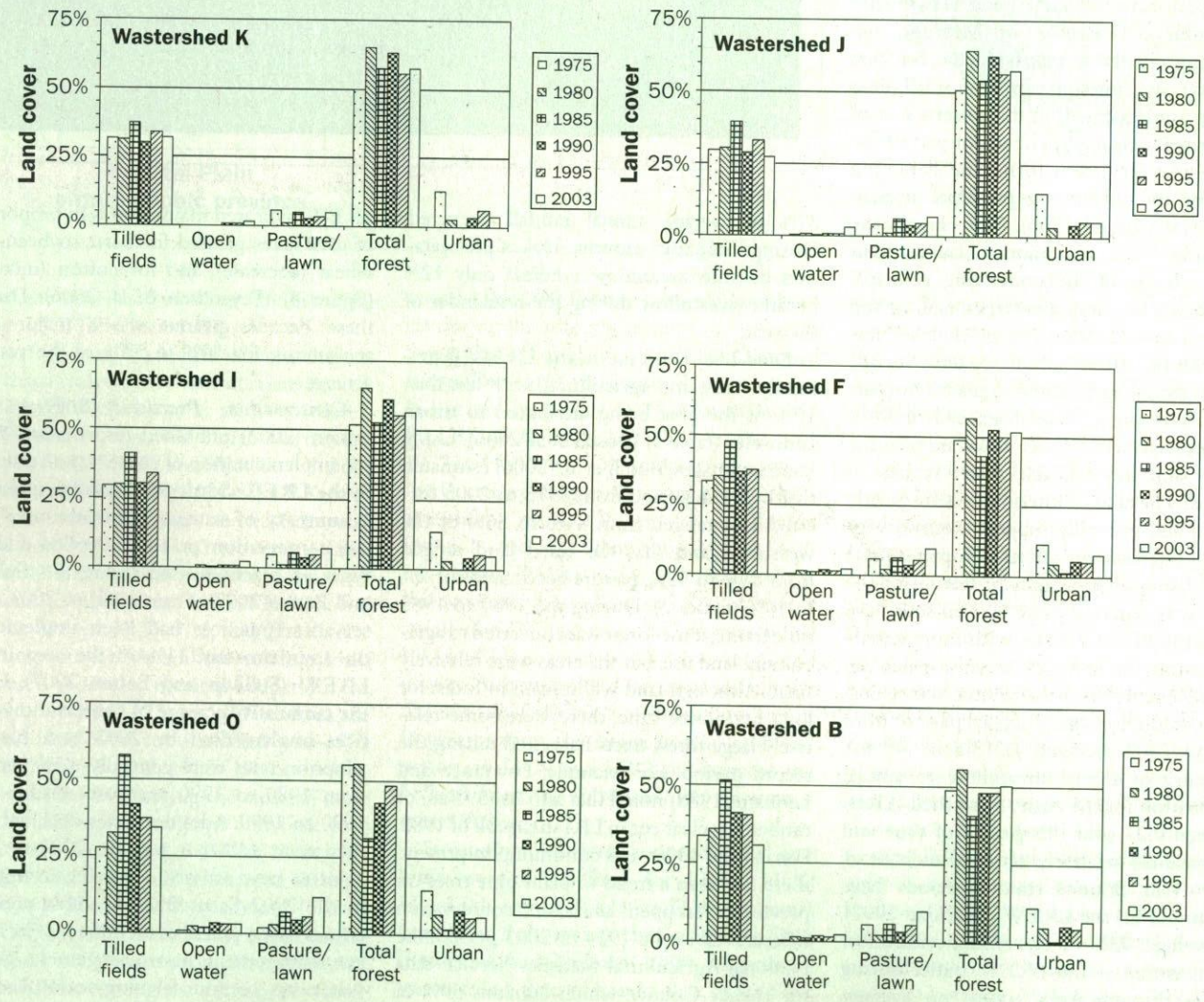


Figure 4
Periods of differing cropping systems in Turner County, (a) 1985 to 1990 and (b) 1998 to 2003 (adapted from Feyereisen et al. 2006).

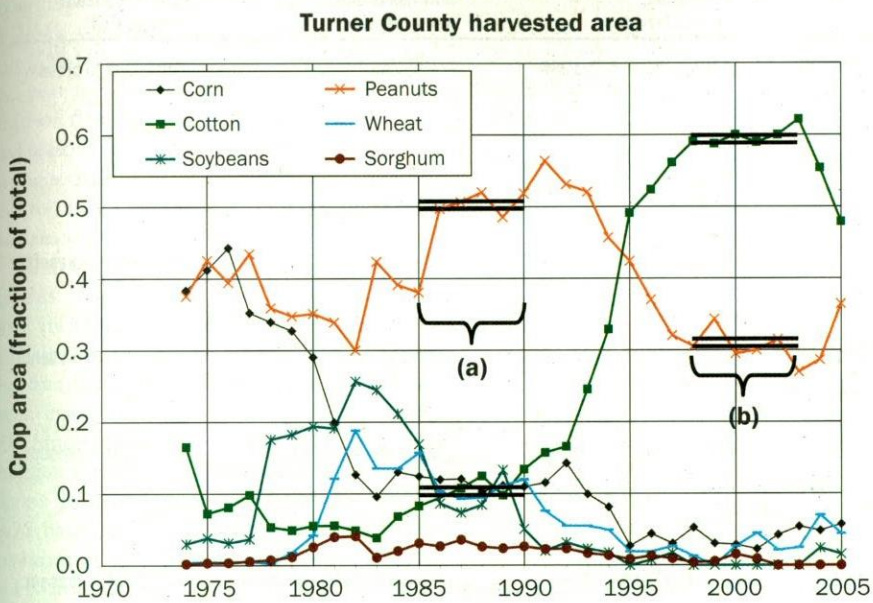
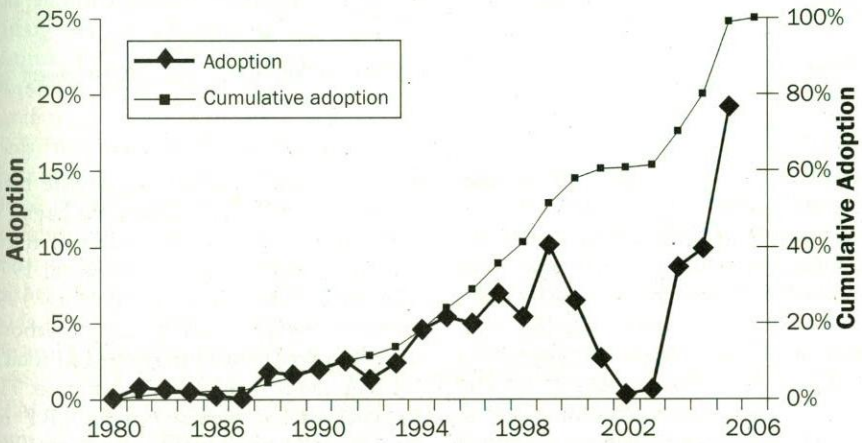


Figure 5
Overall trends in USDA Natural Resources Conservation Service assisted conservation practice adoption from 1980 through 2006.



Notes: Data along the primary y axis depict the annual conservation practice adoption as a percentage of the total acreage in conservation practice for the entire period. Data along the secondary y axis depict the cumulative percentage in conservation adoption over time (adapted from Sullivan and Batten 2007).

management, and terraces. The map of conservation practices developed by Sullivan and Batten (2007) indicated that implementation of conservation measures were focused on LRN and LRO, which have a higher percent agricultural land use, but which unfortunately have an interrupted water quality and hydrology measurement record.

Stream Discharge Measurements. The eight LREW flow measurement structures were installed between 1967 and 1971 (Yates 1976) (figure 1b). Flow measurements at LRB, LRF, LRI, LRJ, and LRK have been continuous since their establishment. Flow measurements at LRM, LRN, and LRO are for shorter periods with a gap of approximately 20 years in the record. The flow measurement structures are horizontal broad-crested concrete weirs, each with a center v-notch. The structures at LRB, LRF, and LRI are located at highway bridges; the remaining five structures are located at highway box culverts. A digital, punched-tape recording system was used from 1967 thru 1992 to record the water levels every five minutes upstream and downstream from the weirs. From 1993 to the present, pressure transducers and electronic recording equipment have been used to measure and record water levels. The water level readings were converted to rate-of-flow values calculated from rating equations developed for each weir. The five-minute rate-of-flow values were summed to provide total flow data on a daily basis. Additional details of the flow measurement and instrumentation system, the associated data management system, and the quality and availability of the data can be found in Sheridan et al. (1995) and Bosch and Sheridan (2007).

An initial network of 52 rain gauges, spaced at approximately 2- to 8-km (1.2- to 5.0-mi) intervals, was established over the LREW to measure and record precipitation. The network was altered in 1982 to reduce data collection and processing requirements. Currently there are 46 gauges and three meteorological stations in the LREW. Additional details about the precipitation monitoring system are found in Sheridan et al. (1995), Sheridan (1997), and Bosch et al. (2007).

Water Chemistry Measurements. The methods of water sample collection were modified over the 30-year period, reflecting changes in research needs, specific study objectives, laboratory funding levels, and technological advances. Samples were taken weekly or more often by one of the following methods: manual grab (MG); automated, timed, discrete (ATD); automated, flow-weighted composite (AFWC), non-refrigerated; and automated, flow-weighted composite, refrigerated. Table 2 identifies the timeframes the various sampling methods were used on each subwatershed.

Manual grab samples during the 1974 to 1978 period were collected from the flow over the weir. Manual grab samples collected after that were obtained from the stilling area immediately upstream of the control weir at each gauging station. All pumped samples were also taken from the stilling area immediately upstream of the control weir at the

Table 2
Dates of flow measurements and various sampling methods for each subwatershed in the Little River Experimental Watershed.

Subwatershed	Flow measurements	Manual grab	Automated, timed, discrete	Automated, flow composite, nonrefrigerated	Automated, flow composite, refrigerated
LRM	1974 to 1986 2002 to present	8/15/1974 to 12/28/1977 1/25/1979 to 4/24/1981 1/8/1982 to 4/9/1982 12/1986 1/7/2002 to 4/29/2002	4/13/1982 to 4/2/1986	–	5/6/2002 to present
LRK	1974 to present	1/24 to 8/15/1974 4/2/1993 to 1/12/1995	8/21/1974 to 3/26/1993	1/20/1995 to 12/30/2002	1/6/2003 to present
LRJ	1974 to present	1/24/1974 to 8/18/1978 1/12/1990 to 12/30/1994	12/28/1978 to 1/2/1990	1/20/1995 to 12/30/2002	1/6/2003 to present
LRI	1974 to present	1/24/1974 to 4/2/1982 12/12/1986 to 1/12/1995 1/14/2002 to 3/31/2003	4/6/1982 to 3/30/1986	1/20/1995 to 12/29/2003	1/5/2004 to present
LRF	1974 to present	2/1/1974 to 3/26/1982 12/5/1986 to 1/12/1995	4/2/1982 to 4/2/1986	1/20/1995 to 12/29/2003	1/5/2004 to present
LRN	1974 to 1981 2002 to present	1/28/1974 to 12/14/1978 1/7/2002 to 4/29/2002	12/22/1978 to 12/25/1981	–	5/6/2002 to present
LRO	1974 to 1981 1993 to present	1/28/1974 to 12/22/1978 2/11/1993 to 1/12/1995	12/23/1978 to 12/30/1981	1/20/1995 to 12/30/2002	1/6/2003 to present
LRB	1974 to present	1/28/1974 to 1/12/1995	–	1/20/1995 to 12/30/2002	1/6/2003 to present

gauging stations. Analyte concentrations measured for the MG and ATD pumped samples represent instantaneous concentrations. The ATD samples were selected for analysis depending on the goals of particular studies. For 1979 to 1986, ATD samples were collected at 12-hour intervals. All samples during a storm event (rising or falling limb of hydrograph) were analyzed, and selected baseflow samples were analyzed (Lowrance et al. 1985; Lowrance and Leonard 1988). For one study period (December 19, 1985, to February 27, 1986), ATD samples were taken every three hours on LRJ and LRI and all samples were analyzed (Lowrance and Leonard 1988). Additional details of ATD sampling methods have been published by Lowrance and Leonard (1988). Automated, flow-weighted composite samples have been obtained by programming automated water quality samplers to draw sample quantities at intervals of equal streamflow volume. Thus, the AFWC samples represent the mean concentration over the sampling period. The composite samples are collected weekly. Each composite sample consists of 14 to 180 subsamples depending on the flow rate. The AFWC samplers were converted to refrigerated samplers in 2002 and 2003.

Concentrations of dissolved $NO_3 + NO_2$-N, NH_4-N, Cl, and DMRP were determined using US Environmental Protection Agency approved colorimetric techniques (American Public Health Association 1976; Clesceri et al. 1998). Total Kjeldahl N and total P were determined on digestates of unfiltered samples (Technicon 1977; Lachat Instruments 1997). From the beginning of the record period through 1986, these analyses were conducted on a Technicon Autoanalyzer II instrument. Beginning in 1987 through the present, these analyses were conducted using Lachat flow injection analyzers. Nitrate-N monitoring began in 1974; however, data from the 1992 to 1997 timeframe were not usable because of an issue with instrument detection limits. Dissolved molybdate reactive phosphorus and Cl analysis was begun in 1975. Ammonium-N, total Kjeldahl nitrogen, and total P analysis was begun in 1979. The total P data from the 1992 to 1996 timeframe were not usable because a laboratory detection limit issue resulted in total P concentrations being recorded as zero. The laboratory Cl concentrations for LRF for 1975 to 1978 were approximately an order of magnitude lower than expected and were therefore eliminated from the analysis.

Additional details about the LREW water chemistry measurement system and availability of concentration and load data are found in Feyereisen et al. (2007).

Data Analysis. Streamflow nutrient loads (kg ha^{-1}) were calculated by summing the product of the nutrient concentration and volume of streamflow corresponding to the sampling period. Flow-weighted mean concentrations (mg L^{-1}) were calculated both for hydrologic years and for each season of

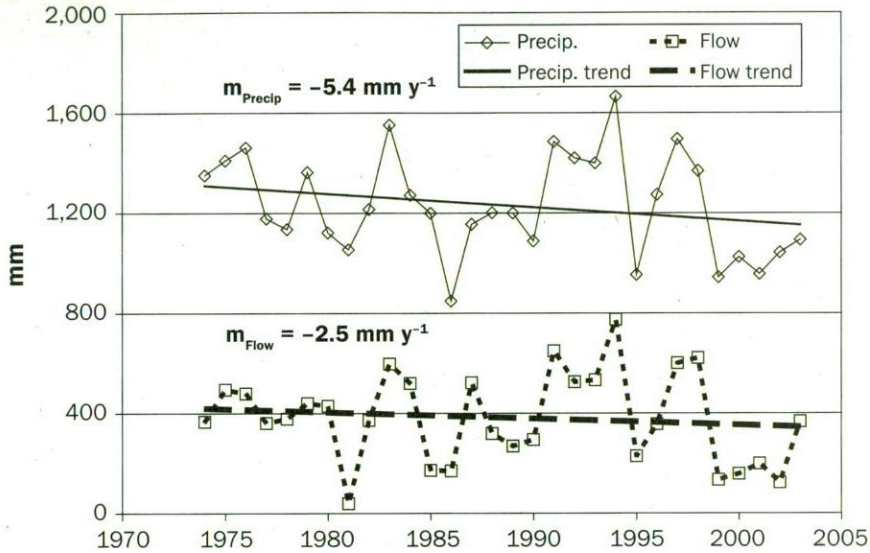
each hydrologic year. These will be referred to as annual or seasonal flow-weighted mean (FWM) concentrations. The FWMs were calculated by summing nutrient loads over the year or season and dividing by the volume of flow over the same period. Annual and seasonal FWM concentrations were used in the statistical analyses to determine trends and to test for differences among watersheds and among seasons.

All of the Littler River sites are intermittent streams. Historically in the LREW, precipitation increases, evapotranspiration decreases, and streamflow begins in December. Thus, the data were analyzed on a hydrologic year basis, from December 1 through November 30. Since nitrate-N stream concentration measurements began January 1, 1974, daily nitrate loads were estimated for December 1973 based on December concentration versus monthly flow for the years 1974 to 2003.

Hydrologic year streamflow and nutrient concentrations versus year were analyzed using the Statistical Analysis System (SAS) (SAS Institute 2002) General Linear Models (GLM) procedure. Trends were identified at the $\alpha = 0.05$ and $\alpha = 0.10$ levels of significance. Flow-weighted mean concentrations and annual nutrient loads were compared among the subwatersheds using SAS (SAS Institute 2002) GLM Procedure with Duncan's multiple range test to determine significant differences in concentrations and loads among subwatersheds. Shorter stream chemistry records are available for LRM, LRN, and LRO. Since comparison of means needs to be made on records with equivalent years, we excluded these three subwatersheds in the analysis of mean concentrations and loads. The subwatersheds were included in the analyte concentration trend analysis.

Statistical analysis was performed with the GLM procedures using the Duncan's Multiple Range Test (SAS Institute 2002) on seasonal means of streamflow and FWM concentrations by subwatershed (LRB, LRF, LRI, LRJ, and LRK) and among watersheds. The seasons were defined as winter (December, January, February), spring (March, April, May), summer (June, July, August), and fall (September, October, November). Trend analysis was performed on FWM concentrations by season for all eight subwatersheds using the SAS GLM procedure (SAS Institute 2002). As with the annual analysis, trends were identified at the $\alpha = 0.05$ and $\alpha = 0.10$ levels of significance.

Figure 6
Annual precipitation and streamflow for LRK, 1974 to 2003.



Results and Discussion

Annual Precipitation and Streamflow Quantity and Quality. There were no statistically significant trends for annual streamflow from 1974 to 2003 for any of the subwatersheds, nor were there significant differences in mean annual streamflow among the watersheds. However, the annual mean precipitation has decreased over this period, for example, on average by 5.4 mm y^{-1} (0.21 in yr^{-1}) and streamflow by 2.5 mm y^{-1}, (0.10 in yr^{-1}), respectively, in LRK (figure 6). The trend is driven by five years of below average precipitation from 1999 to 2003.

Streamflow, means of the annual FWM concentrations and number of years of data for each analyte (*n*) for LRB, LRF, LRI, LRJ, and LRK are shown in table 3. Although there was not a statistical difference in flow among the watersheds, flow tended to be greater for the smaller watersheds. The smaller, nested watersheds are in the upland, have steeper slope and a higher percentage of forested land use. The nitrate-N concentration of LRF was higher than the smaller LRJ and LRK. Chloride concentrations at the LREW outlet, LRB, and at the largest of the nested subwatersheds, LRF, were significantly higher than the concentration at LRJ. Also, Cl concentrations for LRF were higher than for LRK. Ammonium-N, total Kjeldahl nitrogen, DMRP, and total P concentrations were statistically the same across subwatersheds. The concentrations are low for agricultural watersheds. For comparison purposes, Jaynes et al. (2004) reported annual average FWM nitrate-N concentrations of 11.3 and 16.0 mg N L^{-1} (ppm) from two subbasins in a 400-ha (988-ac) tile-drained subbasin in central Iowa, nearly two orders of magnitude greater than the concentrations in the LREW. Mean annual nutrient loads on a subwatershed per hectare basis are shown in table 4. There were no significant nutrient load differences among subwatersheds. On a per hectare basis, nitrate-N load from these Coastal Plain watersheds, ranging from 0.58 to 0.94 kg N ha^{-1} y^{-1} (0.52 to 0.84 lb N ac^{-1} yr^{-1}), are a small fraction of that observed in agricultural watersheds in the Midwest. Again, for example, Jaynes et al. (1999) measured annual stream loads of 4 to 66 kg N ha^{-1} y^{-1} (3.6 to 58.9 lb N ac^{-1} yr^{-1}) (average of 29 kg N ha^{-1} y^{-1} [average of 25.9 lb N ac^{-1} yr^{-1}]), in the 5,130-ha (12,680-ac) Walnut Creek watershed in central Iowa over the years 1992 to 1995. Past research in the LREW has shown that a relatively large amount of N, 56 kg N ha^{-1} y^{-1} (50 lb N ac^{-1} yr^{-1}), is either retained in the watershed or lost through denitrification (Lowrance et al. 1985).

Trend analysis results showed a significant trend for three of seven analytes tested (table 5). At the $\alpha = 0.10$ level, the analysis indicated that annual FWM total P concentrations decreased in four of the five major subwatersheds (LRB, LRF, LRI, and LRJ) plus LRO (table 5). An increasing trend in Cl concentration was identified in LRJ, LRN, and LRO. Chloride is not subject to biological reactions in the riparian zone and stream

Table 3
Mean annual streamflow and means of annual flow-weighted mean nutrient concentrations for five subwatersheds.

Subwatershed	Flow (mm)	*n*	NO_3-N (mg L^{-1})	*n*	NH_4-N (mg L^{-1})	*n*	TKN (mg L^{-1})	*n*	DMRP (mg L^{-1})	*n*	Total P (mg L^{-1})	*n*	Cl (mg L^{-1})	*n*
LRB	333a	30	0.21ab	24	0.087a	25	1.57a	25	0.040a	29	0.27a	20	9.75ab	29
LRF	363a	30	0.23a	24	0.077a	25	1.66a	25	0.048a	29	0.20a	20	10.09a	25
LRI	419a	30	0.16ab	24	0.063a	25	1.48a	25	0.036a	29	0.20a	20	8.95bc	29
LRJ	405a	30	0.15b	24	0.066a	25	1.67a	25	0.037a	29	0.19a	20	8.55c	29
LRK	383a	30	0.14b	24	0.065a	25	1.70a	25	0.036a	29	0.19a	20	8.65bc	29

Notes: *n* = number of annual observations. TKN = total Kjeldahl nitrogen. DMRP = dissolved molybdate reactive phosphorus. Different letters within a column indicate differences for $\alpha = 0.05$.

Table 4
Mean annual nutrient loads on five subwatersheds.

	Load (kg ha^{-1})											
Subwatershed	NO_3-N	*n*	NH_4-N	*n*	TKN	*n*	DMRP	*n*	Total P	*n*	Cl	*n*
LRB	0.84a	24	0.32a	25	5.58a	25	0.12a	29	0.59a	20	31.1a	29
LRF	0.94a	24	0.31a	25	6.50a	25	0.16a	29	0.64a	20	34.0a	25
LRI	0.76a	24	0.31a	25	7.06a	25	0.14a	29	0.69a	20	35.5a	29
LRJ	0.70a	24	0.32a	25	7.90a	25	0.15a	29	0.64a	20	32.6a	29
LRK	0.58a	24	0.30a	25	7.11a	25	0.14a	29	0.56a	20	30.4a	29

Notes: *n* = number of annual loads used in calculations. TKN = total Kjeldahl nitrogen. DMRP = dissolved molybdate reactive phosphorus. Different letters within a column indicate differences for $\alpha = 0.05$.

Table 5
Results of analysis for trends in annual flow-weighted mean concentrations.

Parameter	B	F	I	J	K	M	N	O
NO_3-N	—	—	—	—	—	—	—	—
NH_4-N	—	—	—	i	i	—	—	—
TKN	—	—	—	—	—	—	—	—
Total N	—	—	—	—	—	—	—	—
DMRP	—	—	—	—	—	—	—	—
Total P	d	D	d	D	—	—	—	D
Cl	—	—	—	I	—	—	I	I

Notes: TKN = total Kjeldahl nitrogen. DMRP = dissolved molybdate reactive phosphorus. — = not significant. D = significant decreasing trend. I = significant increasing trend for $\alpha = 0.05$. Lowercase letters indicate significance for $\alpha = 0.10$.

channel, and concentrations of Cl have been shown to decrease during stormflow and years with higher flow rates (Lowrance and Leonard 1988). Thus, the Cl trend is to be expected as precipitation, and flow rates were lower in the last five years of the study period. For $\alpha = 0.10$, an increasing trend in ammonium-N concentration was observed in LRJ and LRK.

Seasonal Streamflow Quantity and Quality. Streamflow by subwatershed for the 30-year study period was not statistically different among the five subwatersheds analyzed on a seasonal basis (table 6). However, for each subwatershed, streamflow was significantly higher in winter and spring than in summer and fall. Although the mean winter flow was greater than mean spring flow in every subwatershed, and mean summer flow was greater than mean fall flow, there were not significant differences between winter and spring or summer and fall flows.

There were significant differences among watersheds by season for mean nitrate-N (table 7) and Cl concentrations (table 8). During the winter, nitrate-N concentrations were higher in LRF than in the smaller, nested LRI, LRJ, and LRK, and during the spring were higher in LRB than in LRK. Through summer and fall, seasons with the lowest flow, the highest uptake of water and nutrients by plants, and the highest rates of biological processing, mean nitrate-N concentrations were the same among the watersheds. During winter and spring, seasons with higher flow rates, lower evapotranspiration demand, and lower biological processing rates, higher stream nitrate-N concentrations may be expected. This was indeed the case for LRB and LRF, but not for the smaller nested LRI, LRJ, and LRK, for which nitrate-N concentrations were not significantly different among seasons. Chloride concentration was higher in fall than in at least one other season for each subwatershed (table 8). Results were similar for LRB and LRF; fall concentrations were higher than those for spring and summer. For LRJ and LRK, Cl concentrations were higher in fall than for all three other seasons. Since Cl is a conservative ion for which biological uptake is insignificant, it is expected that concentrations would be higher during periods of low flow, e.g. fall, and lower during periods of high flow, e.g. spring. The mixed results among the subwatersheds for winter and summer seem to support a lag or delay in Cl stream concentration changes

Table 6
Mean seasonal streamflow for five nested subwatersheds, 1974 to 2003, $n = 30$.

Subwatershed	Flow (mm) Winter	Spring	Summer	Fall
LRB	x145.0	x130.8	y34.1	y23.5
LRF	x157.7	x137.7	y37.8	y29.6
LRI	x182.9	x156.1	y43.9	y36.5
LRJ	x178.7	x153.3	y41.6	y32.2
LRK	x165.0	x147.1	y39.8	y30.7

Notes: Different letters within a row indicate differences for $\alpha = 0.05$. There are no significant differences among watersheds for any of the seasons.

Table 7
Means of seasonal flow-weighted mean concentrations for NO_3-N on five watersheds.

Subwatershed		Winter	Spring	Summer	Fall
LRB	Concentration (mg L^{-1})	x0.19ab	xy0.16a	xy0.12a	y0.09a
	n	30	30	27	26
LRF	Concentration (mg L^{-1})	x0.26a	y0.14ab	y0.13a	y0.08a
	n	30	30	27	25
LRI	Concentration (mg L^{-1})	x0.15b	x0.10ab	x0.08a	x0.08a
	n	30	30	25	23
LRJ	Concentration (mg L^{-1})	x0.14b	x0.10ab	x0.09a	x0.08a
	n	30	30	24	22
LRK	Concentration (mg L^{-1})	x0.13b	x0.09b	x0.12a	x0.14a
	n	29	30	27	25

Notes: n = number of observations. The letters a and b indicate differences within a column for $\alpha = 0.05$. The letters x and y indicate differences within a row for $\alpha = 0.05$.

Table 8
Means of seasonal flow-weighted mean concentrations for Cl on five watersheds.

Subwatershed		Winter	Spring	Summer	Fall
LRB	Concentration (mg L^{-1})	xy10.45ab	y9.51ab	y9.47ab	x11.62a
	n	29	29	27	26
LRF	Concentration (mg L^{-1})	xy10.96a	y9.66a	y10.10a	x12.29a
	n	25	25	22	20
LRI	Concentration (mg L^{-1})	xy9.36bc	y8.37bc	xy9.16ab	x10.64a
	n	29	29	25	23
LRJ	Concentration (mg L^{-1})	y9.01c	y8.06c	y8.32b	x11.51a
	n	29	29	24	22
LRK	Concentration (mg L^{-1})	y9.03c	y7.93c	y8.74ab	x11.56a
	n	28	29	27	25

Notes: n = number of observations. The letters a, b, and c indicate differences within a column for $\alpha = 0.05$. The letters x and y indicate differences within a row for $\alpha = 0.05$.

between annual high (fall) and low (spring) concentrations for the larger watersheds (LRB, LRF, and LRI). Analysis by season among watersheds shows that fall Cl concentrations are statistically the same among the watersheds, as was the case with nitrate-N concentrations. LRF had the highest concentration in every season and was greater than LRI, LRJ, and LRK in winter and spring and LRJ in summer.

There were no statistical differences among seasons by subwatershed or among watersheds by season, for ammonium-N or total Kjeldahl nitrogen. The only difference observed in DMRP concentrations was for fall, when LRB was greater than LRI, LRJ, and LRK.

Trend analysis results for seasonal FWM concentrations showed a decreasing trend for total P and an increasing trend for Cl, similar to the trends observed in annual FWM concentrations (table 9). For these two analytes, all subwatersheds that had a significant annual trend also had a significant trend for the winter season. In addition, two subwatersheds (LRK and LRN) that did not exhibit an annual total P concentration trend did exhibit a decreasing trend for the winter season. Fall season trends were similar but not identical. Six of the eight subwatersheds showed a significant increase ($\alpha = 0.05$) in Cl concentration in summer, potentially a consequence of lower-than-average precipitation and streamflow for 1999 to 2003. There was a significant decrease in total P concentration in LRJ for summer and LRN for spring. There were no other concentration trends in spring. Figure 7a depicts a time series of annual FWM Cl concentration for LRB for spring and fall. The fall concentration exhibited an increasing trend while spring concentration showed no trend. Mean Cl concentration over the study period was greater for fall than for spring, 11.6 to 9.5 mg L^{-1} (ppm), respectively. The decreasing trends in total P concentration for LRB shown in figure 7b are statistically significant for winter and not significant for spring.

The changes in sampling and analysis methodology over the past 30 years of sampling Little River are detailed in table 2. The major changes in analysis that took place were verified to provide equivalent results when the changeover occurred from segmented flow by Technicon auto-analyzer to flow injection using Lachat instrumentation. The major difference in sampling methodology was the change from grab sampling to fixed intake sampling. Fixed intakes were placed 15 to 30 cm (6 to 12 in) from the stream bottom while grab sampling was typically done near the top of the water columns. Under low flow conditions, these would be roughly equivalent, but under high flow conditions, there could be large differences in where the sample was taken within the water column. For LRK, which has the most observations, comparison of the grab sampling to an automated timed discrete sampling period before and to an automated flow composite nonrefrigerated sampling period after showed

Table 9
Results of analysis for trends in seasonal flow-weighted mean concentration.

Parameter	B	F	I	J	K	M	N	O
Winter								
NO_3-N	–	–	–	–	–	–	–	–
NH_4-N	–	–	–	–	–	–	i	–
TKN	–	–	–	–	–	–	–	–
DMRP	–	–	–	–	–	–	–	–
Total P	D	D	D	D	D	–	D	D
Cl	–	–	–	i	i	I	I	I
Spring								
NO_3-N	–	–	–	–	–	–	–	–
NH_4-N	–	–	–	–	–	–	–	–
TKN	–	–	–	–	–	–	–	–
DMRP	–	–	–	–	–	–	–	–
Total P	–	–	–	–	–	–	D	–
Cl	–	–	–	–	–	–	–	–
Summer								
NO_3-N	–	–	–	–	–	–	–	–
NH_4-N	–	–	–	–	–	–	–	–
TKN	–	–	–	–	i	i	–	–
DMRP	–	D	–	–	–	–	–	–
Total P	–	–	–	D	–	–	–	–
Cl	I	I	I	I	–	–	I	I
Fall								
NO_3-N	–	–	–	–	–	–	–	–
NH_4-N	–	–	–	–	–	–	–	–
TKN	–	–	–	–	–	–	–	i
DMRP	–	–	–	–	–	–	–	–
Total P	–	D	D	–	–	–	–	d
Cl	I	–	i	–	–	–	–	I

Notes: TKN = total Kjeldahl nitrogen. DMRP = dissolved molybdate reactive phosphorus. – = not significant. D = significant decreasing trend. I = significant increasing trend for $\alpha = 0.05$. Lowercase letters indicate significance for $\alpha = 0.10$.

no consistent trends. Chloride was lower for grab samples, but NH_4 was the same for the three sampling regimes. Ortho-P was higher for grab samples. The grab sampling period on LRK had detection limit problems for nitrate-N and total P, and they were not available for comparison. Overall, there seemed to be no consistent patterns related to the sampling technique.

The trends noted in Cl and total P concentration do not appear to be related to changes in land use (figure 3). Forest and tilled fields represent over 75% of the land use in all of the subwatersheds. Although the tilled fields land use increased and forest land use decreased during the middle of the study timeframe, each returned to a level near that of the initial years of the study. Thus, land use patterns do not seem to match those of Cl and total P concentrations or loads. Over the three decades under consideration, there was a marked decrease in tilled land use in corn. There is a potential relationship between this decrease and the decline in total P concentration and load. However, the decrease in corn acreage tended to be offset by cotton acreage. There are differences in fertilizer requirements between these crops, but it would be difficult to determine whether the change in cropping system was responsible in part for the decrease in total P or the increase in Cl.

Beginning in 2004, conservation practice adoption increased dramatically in the LREW, with 40% of the cumulative adoption occurring over the 1980 to 2006 timeframe taking place in 2004 to 2006. Depending on any lag time required for these conservation practices to affect watershed-scale water quality changes, the next few years should present an opportunity to measure post implementation benefits. In LRN, the subwatershed with the highest percent land use in agriculture and the highest percent of land area with conservation practices applied, comparison with past data will be hampered by the relatively short record period. The conservation practice information presented by Sullivan and Batten (2007) has been recently compiled, and additional research to investigate relationships between water quality parameters and applied conservation practices, the essence of CEAP, is ongoing.

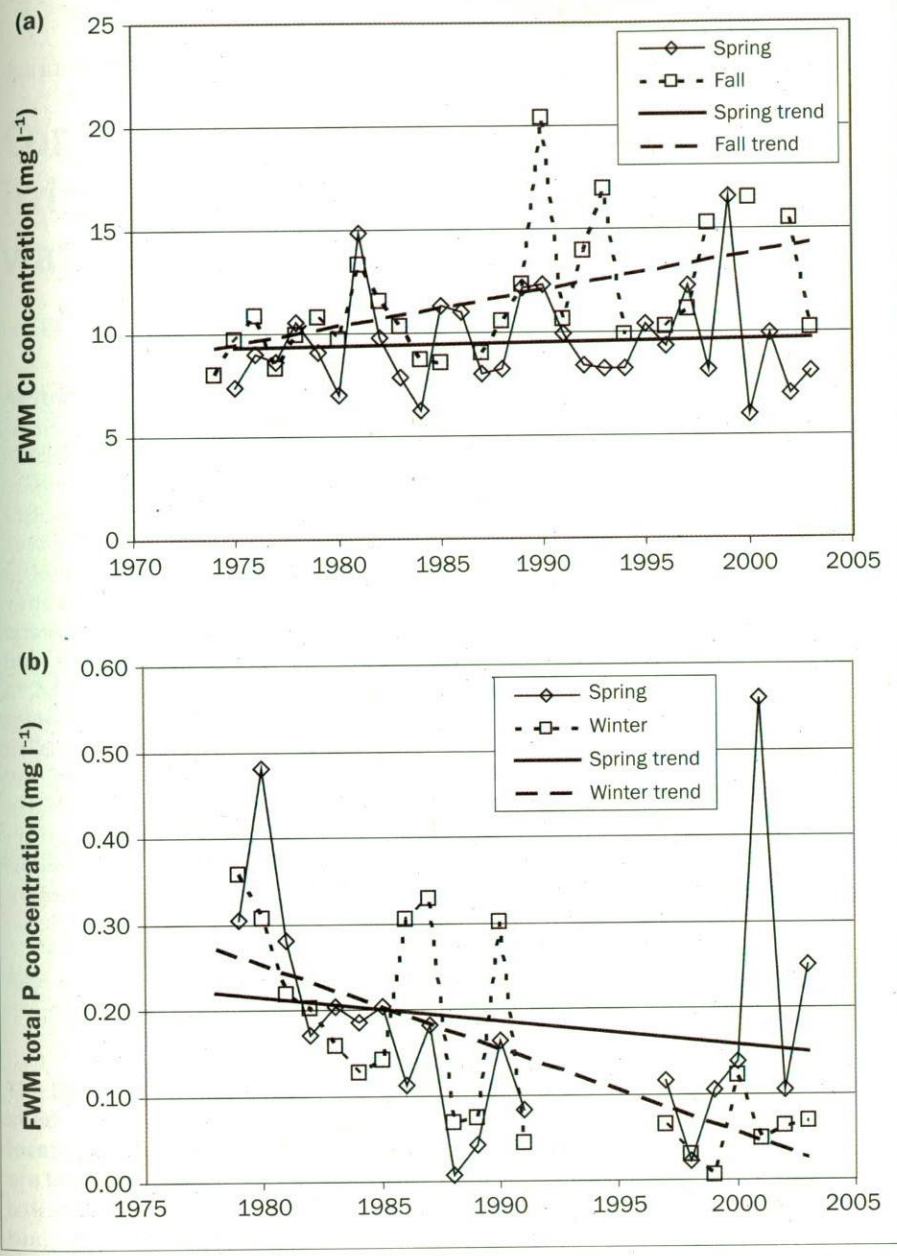
Summary and Conclusions

As part of CEAP, we analyzed streamflow and water chemistry in the LREW for the first three decades of record. Annual precipitation and streamflow showed no statistically significant trends over the period 1974 to 2003. During these years, total P annual FWM concentration has decreased significantly on four of the five major subwatersheds and on one of the three subwatersheds with shorter stream chemistry records. Seasonal analysis showed that the total P trend occurred primarily in winter. An increase in annual FWM Cl concentration was observed in LRJ, LRN, and LRO. Seasonal Cl concentration increases were noted in winter in the smaller LRJ, LRK, LRM, LRN, and LRO, and in six of eight subwatersheds in summer. Potential hypotheses for the total P trend include reduction in animal agriculture in the watershed and less erosion. Potential hypotheses for increased Cl concentrations include several years of below-normal precipitation toward the end of the record and additional Cl input through tropical storm rainfall.

Differences in annual nitrate-N and Cl concentrations were seen between LRF and LRJ and LRK; however, annual nutrient loads were not significantly different among the five major subwatersheds for any analyte. Streamflow by season was also statistically the same for the subwatersheds for each season.

On the larger watersheds, LRB and LRF, mean seasonal nitrate-N concentration was significantly higher in winter than

Figure 7
(a) LRB, flow-weighted mean Cl concentration: spring trend significant for $\alpha = 0.05$. (b) LRB, flow-weighted mean total P concentration: winter trend significant for $\alpha = 0.05$.



in fall; however, there were no differences in nitrate-N concentration among seasons on the smaller LRI, LRJ, and LRK. During winter, mean nitrate-N concentration was significantly higher in LRF than LRI, LRJ, and LRK, and during spring was higher in LRB than LRK. Mean seasonal nitrate-N concentrations were equivalent among the subwatersheds in summer and fall. Mean fall Cl concentration was highest among seasons in each subwatershed, as would be expected, and was significantly higher than spring concentration in each case. Mean fall Cl concentrations were equivalent among watersheds. Winter and spring Cl concentrations were higher in LRB and LRF than in LRJ and LRK.

Different trends in concentration differences among seasons between the larger and smaller subwatersheds may indicate process differences at the scales represented and have potential implications with respect to measuring the effects of conservation practices on stream chemistry. The concentrations and loads represented in the LREW are relatively low for an agricultural watershed and confirm the role of the riparian forest in reducing impact of agricultural practices on stream water quality in the southeastern Coastal Plain.

References

American Public Health Association. 1976. Standard methods for the examination of water and wastewater, 14th ed. Washington, DC: American Public Health Association.

Bosch, D.D., and J.M. Sheridan. 2007. Stream discharge database, Little River Experimental Watershed, Georgia, United States. Water Resources Research 43, W09473, doi:10.1029/2006WR005833.

Bosch, D.D., J.M. Sheridan, and L. Marshall. 2007. Precipitation, soil moisture, and climate database, Little River Experimental Watershed, Georgia, United States. Water Resources Research 43, W09472, doi:10.1029/2006WR005834.

Bosch, D.D., D.G. Sullivan, and J.M. Sheridan. 2006. Hydrologic impacts of land-use changes in Coastal Plain watersheds. Transactions of the ASABE 49(2):423-432.

Clesceri, L.S., A.E. Greenburg, and D.A. Eaton, ed. 1998. Standard Methods for the Examination of Water and Wastewater 20th Edition. Washington, DC: American Public Health Association, USA, 1. 2998.

Feyereisen, G.W., R. Lowrance, T.C. Strickland, D.D. Bosch, J.M. Sheridan, and D.G. Sullivan. 2006. Long-Term nitrogen load from the Little River Experimental Watershed on the Coastal Plain of Southwest Georgia. ASABE Paper No. 062201. St. Joseph, MI: ASABE.

Feyereisen, G.W., R. Lowrance, T.C. Strickland, J.M. Sheridan, R.K. Hubbard, and D.D. Bosch. 2007. Long-term water chemistry database, Little River Experimental Watershed, southeast Coastal Plain, United States. Water Resources Research 43, W09474, doi:10.1029/2006WR005835.

Hubbard, R.K., and J.M. Sheridan. 1983. Water and nitrate-nitrogen losses from a small, upland Coastal Plain watershed. Journal of Environmental Quality 12(2):291-295.

Jaynes, D.B., J.L. Hatfield, and D.W. Meek. 1999. Water quality in Walnut Creek watershed: Herbicides and nitrate in surface waters. Journal of Environmental Quality 28(1):45-59.

Jaynes, D.B., D.L. Dinnes, D.W. Meek, D.L. Karlen, C.A. Cambardella, and T.S. Colvin. 2004. Using the late spring nitrate test to reduce nitrate loss within a watershed. Journal of Environmental Quality 33(2):669-677.

Lachat Instruments. 1997. Determination of total Kjeldahl N and total P by flow injection analysis colorimetry (block digestion method). QuickChem Method 10-107-06-2-E.

Lowrance, R. 1992. Nitrogen outputs from a field-sized agricultural watershed. Journal of Environmental Quality 21(4):602-607.

Lowrance, R., and R. Leonard. 1988. Streamflow nutrient dynamics on Coastal Plain watersheds. Journal of Environmental Quality 17(4):734-740.

Lowrance, R.R., R.A. Leonard, and L.E. Asmussen. 1985. Nutrient budgets for agricultural watersheds in the southeastern coastal plain. Ecology 66(1):287-296.

Lowrance, R.R., R.L. Todd, and L.E. Asmussen. 1983. Waterborne nutrient budgets for the riparian zone of an agricultural watershed. Agricultural Ecosystems and Environment 10:371-384.

Lowrance, R.R., R.L. Todd, and L.E. Asmussen. 1984a. Nutrient cycling in an agricultural watershed: II. Streamflow and artificial drainage. Journal of Environmental Quality 13(1):27-32.

Lowrance, R., R. Todd, F. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1984b. Riparian forests as nutrient filters in agricultural watersheds. BioScience 34(6):374-377.

Mausbach, M.J., and A.R. Dedrick. 2004. The length we go - measuring environmental benefits of conservation practices. Journal of Soil and Water Conservation 59(5):96-103.

Mostaghimi, S., K.M. Brannan, T.A. Dillaha, and A.C. Bruggeman. 2001. Chapter 10: Best management practices for nonpoint source pollution control. *In* Agricultural Nonpoint Source Pollution, 59-89. Boca Raton, FL: Lewis Publishers.

Ritter, W.F., and L. Bergstrom. 2001. Chapter 3: Nitrogen and water quality. *In* Agricultural Nonpoint Source Pollution, 59-89. Boca Raton, FL: Lewis Publishers.

SAS Institute. 2002. SAS for Windows Version 9.1. Cary, NC: SAS Institute.

Sheridan, J.M. 1997. Rainfall-streamflow relations for Coastal Plain watersheds. Applied Engineering in Agriculture 13(3):333-344.

Sheridan, J.M., W.C. Mills, and M.L. Hester. 1995. Data management for experimental watersheds. Applied Engineering in Agriculture 11(2):249-259.

Sheridan, J.M., P. Yates, and L.E. Asmussen. 1983. Hydrology and nutrient and sediment movement from Southeastern Coastal Plain Watersheds. *In* Nutrient Cycling in Agricultural Ecosystems. ed. R. Lowrance, R. Todd, and R. Leonard. Univ. of Ga. College of Agricultural Experiment Stations, Special Publication 23. 1983.

Stringfield, V.T. 1966. Artesian water in tertiary limestone in the southeastern states. Geological Survey Professional Paper 517. Washington, DC: US Government Printing Office.

Sullivan, D.G., and H.L. Batten. 2007. Little River Experimental Watershed, Tifton, Georgia, United States: A historical geographic database of conservation practice implementation. Water Resources Research 43, W09475, doi:10.1029/2007WR006143.

Technicon. 1977. Technicon industrial method No. 376-75 W/B. Tarrytown, NY: Technicon Industrial Instruments.

Yates, P. 1976. Design, construction, and operation of streamflow-measuring facilities in the Little River Watershed, Georgia. Agricultural Research Service ARS-S-148. Washington, DC: USDA Agricultural Research Service.