Agroforestry Practices, Runoff, and Nutrient Loss: A Paired Watershed Comparison

Ranjith P. Udawatta,* J. John Krstansky, Gray S. Henderson, and Harold E. Garrett

ABSTRACT

A paired watershed study consisting of agroforestry (trees plus grass buffer strips), contour strips (grass buffer strips), and control treatments with a corn (*Zea mays* L.)–soybean [*Glycine max* (L.) Merr.] rotation was used to examine treatment effects on runoff, sediment, and nutrient losses. During the (1991–1997) calibration and subsequent three-year treatment periods, runoff was measured in 0.91- and 1.37-m H-flumes with bubbler flow meters. Composite samples were analyzed for sediment, total phosphorus (TP), total nitrogen (TN), nitrate, and ammonium. Calibration equations developed to predict runoff, sediment, and nutrients losses explained 66 to 97% of the variability between treatment watersheds. The contour strip and agroforestry treatments reduced runoff by 10 and 1% during the treatment period. In both treatments, most runoff reductions occurred in the second and third years after treatment establishment. The contour strip treatment reduced erosion by 19% in 1999, while erosion in the agroforestry treatment exceeded the predicted loss. Treatments reduced TP loss by 8 and 17% on contour strip and agroforestry watersheds. Treatments did not result in reductions in TN during the first two years of the treatment period. The contour strip and agroforestry treatments reduced TN loss by 21 and 20%, respectively, during a large precipitation event in the third year. During the third year of treatments, nitrate N loss was reduced 24 and 37% by contour strip and agroforestry treatments. Contour strip and agroforestry management practices effectively reduced nonpoint-source pollution in runoff from a corn–soybean rotation in the clay pan soils of northeastern Missouri.

Agroforestry is a land management program that intersperses agricultural crops with trees (Krstansky et al., 1997). It has long been used in tropical climates but has only recently received attention in temperate zones. Agroforestry practices have been proposed as being more environmentally friendly for agricultural production in temperate North America (Gold and Hanover, 1987; Garrett and Buck, 1997). However, despite intense interest in agroforestry practices worldwide, the knowledge base for implementation is only just developing (Sanchez, 1995; Garrett and Buck, 1997).

Agroforestry farming practices provide multiple benefits including high productivity and additional income while maintaining soil health (Kang et al., 1984). Vegetative filter strips established in the form of agroforestry or contour grass buffer strips have the potential to improve water quality, wildlife abundance, biodiversity, and aesthetic value. Filter strips of permanent vegetation that reduce runoff and trap sediment can be used to greatly reduce nonpoint-source (NPS) pollution (Robinson et al., 1996; Cooper and Lipe, 1992). Nonpoint-source pollution is a landscape-scale phenomenon and its diffuse nature complicates mitigation (Verchot et al., 1998), but vegetative filter strips have distinct advantages over other erosion control technologies (Robinson et al., 1996). Normally, interest in the use of agroforestry practices and contour grass strips for various environmental benefits relates to their potential to increase infiltration, reduce runoff, and reduce NPS pollution.

Limited research suggests that properly established filter strips can reduce runoff, sediment load, and NPS pollution (Dillaha et al., 1989) although the design factors of such practices have not been clearly defined. The increased infiltration found under natural forests also occurs in agroforestry and with other vegetative filter practices. Multistrata systems, combined with litter cover and dense root systems, hold runoff when it first reaches the surface and subsequently promote infiltration. The vegetation in an agroforestry practice serves two major purposes: (i) the fine root system holds soil in place, reducing susceptibility to erosion, and (ii) plant stems decrease the flow velocity, enhancing sedimentation. Tree roots can also take up nutrients that would otherwise be lost by leaching (van Noordwijk et al., 1996). Moreover, the addition of organic matter from trees in agroforestry improves soil physical properties, chemical properties, and infiltration, thus reducing runoff, NPS pollution, and sediment loss (Young, 1997). Nonpoint-source pollution removal capability of vegetative strips depends on the nature of the pollutant, hydrology of the area, soil properties, and nature of the trees. Site-specific interrelationships of these factors contribute to large variability in pollution control effectiveness.

Schmitt et al. (1999) compared grass versus grass–shrub–tree buffers to test their effectiveness in protecting waterways from contaminants. The grass–shrub–tree combination produced significantly lower sediment and total nitrogen runoff than the grass buffer strips. Numerous other studies have shown that forest vegetation removes significant quantities of NPS pollution from agricultural runoff (Cooper and Gilliam, 1987; Lowrance et al., 1984; Peterjohn and Correll, 1984). It might therefore be assumed that incorporation of trees and grass in upland buffer designs would enhance water quality. However, sufficient quantitative information from direct experimental studies designed to evaluate the relative filtering performance of agroforestry and grass strips is lacking to provide guidance in designing effective agroforestry buffer strip practices.

The watershed practices in our study are complex, so to investigate their influences we had to minimize extraneous effects. No reported studies on the effects of agroforestry and contour grass strips on water quality that included a calibration and a treatment period were

---

School of Natural Resources, Univ. of Missouri, Columbia, MO 65211. Received 24 Aug. 2001. *Corresponding author (udawattar@missouri.edu).

Published in J. Environ. Qual. 31:1214–1225 (2002).

Abbreviations: NPS, nonpoint source; TN, total nitrogen; TP, total phosphorus.

found in the literature. The research we are reporting is unique as it employs paired watersheds to examine the effect of agroforestry while eliminating extraneous effects. In a paired watershed study, watersheds need not be identical (Hewlett and Pienaar, 1973). However, areas should be small enough to obtain uniform treatment over the entire watershed; watersheds should be similar in size, slope, location, and land cover; and watersheds should be in a steady state prior to the study (USEPA, 1993). Since climatic and hydrologic differences are statistically described using calibration data, water quality changes can be attributed to treatments.

Although several benefits of agroforestry have been claimed, in fact, there has been little research to demonstrate these benefits, thereby restricting the systematic selection of best management practices suitable for the region. This is especially true with highly erodible landscapes dominated by corn–soybean agriculture as is practiced in northern Missouri and much of the U.S. Midwest. The primary objective of this study was to examine whether agroforestry is a viable and effective land management strategy when protection of water sources from sediment and nutrients is a goal. Specific objectives were to (i) quantify runoff, sediment loss, and nutrient loss from typical corn–soybean rotations; (ii) determine the effectiveness of agroforestry when employed in conjunction with corn–soybean agriculture in reducing runoff, sediment, and nutrient losses; and (iii) examine the specific mechanisms of agroforestry and contour grass filter strips in reducing sediment and nutrient loss from watersheds.

MATERIALS AND METHODS

The study was conducted at the University of Missouri-Greenley Memorial Research Center in Knox County, Missouri, USA (40°01′ N, 92°11′ W). Three adjacent north-facing watersheds were instrumented in early 1991 (Fig. 1). Treatments were randomly assigned and implemented in 1997. The control (east) watershed is 1.65 ha, the agroforestry watershed (center) is 4.44 ha, and the contour strip watershed (west) is 3.16 ha. Each watershed is drained by a grass waterway that leads into a concrete approach structure and an H-flume. The control watershed is instrumented with a 0.91-m (3-ft.) flume while the other two watersheds are instrumented with 1.37-m (4.5-ft.) flumes. Contour grass–legume strips (4.5 m [15 ft.] wide) consisting of redtop (*Agrostis gigantea* Roth), brome grass (*Bromus* spp.), and birdsfoot trefoil (*Lotus corniculatus* L.) were established at 36.5 m (some in lower slope positions were 22.8 m apart) intervals on the agroforestry and contour strip watersheds in June 1997 (Fig. 1). Pin oak (*Quercus palustris* Muenchh.), swamp white oak (*Q. bicolor* Willd.), and bur oak (*Q. macrocarpa* Michx.) were planted 3 m apart down the center of the grass–legume strips of the agroforestry watershed in November 1997. Trees had almost 100% survival in spite of deer damage and a severe drought in 1999. Only one pin oak tree was replaced, in May 2000. In 1999, welded wire fences (5-cm mesh, 1 m in diameter) were installed to protect trees from deer damage.

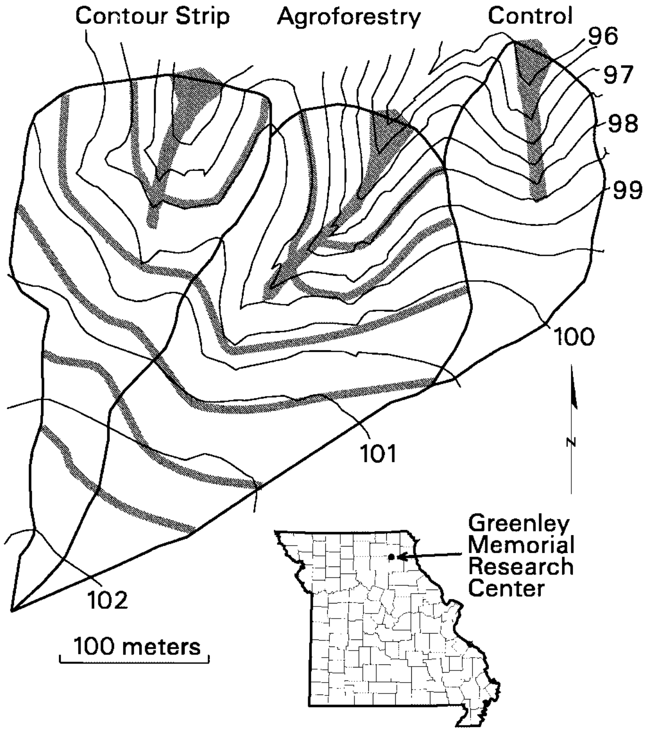

Fig. 1. Study site location and 0.5-m-interval topographic maps of contour strip, agroforestry, and control watersheds. Broad gray areas represent grass strips (contour strip), trees and grass strips (agroforestry), and grass waterways (contour strip, agroforestry, and control).

The watersheds are underlain by glacial and loess material. Soils are mapped as Putnam silt loam (fine, smectitic, mesic Vertic Albaqualf) and Kilwinning silt loam (fine, smectitic, mesic Vertic Albaqualf) (Watson, 1979). Putnam occurs on nearly level (0–1%) slope portions of the catchments while Kilwinning occurs downslope from the Putnam on 2 to 5% slopes. Generally, the upper (south) end of the watersheds had the most gentle slope. Minor portions of the watersheds are in Armstrong loam (fine, smectitic, mesic Aquertic Hapludalf), where 5 to 9% slopes occur. The predominantly clay B horizon in both soils severely restricts vertical soil water percolation and these soils produce copious surface runoff during periods of saturation in the spring and early summer. Detailed soil sampling revealed the depth to the B horizon to be quite variable (4 to 37 cm) depending on past erosion and landscape position. Soil acidity varied between pH 7.1 and 4.3 for the sampled 1-m soil profile. Most surface soil pH values were greater than 6.4. Organic matter in the surface (0 to 5 cm) varied between 1.3 and 2.2%.

Thirty-year mean annual precipitation in the region is 920 mm yr^{-1}, of which more than 66% falls from April through September (Owenby and Ezell, 1992). Mean annual air temperature is approximately 11.7°C with an average monthly low of −6.6°C in February and an average monthly high of 31.4°C in July (Owenby and Ezell, 1992). Snowfall averages about 590 mm per year and can stay on the ground for extended periods.

Agricultural activities since 1991 are summarized in Table 1. Prior to 1991, the field containing these watersheds was in a corn–soybean rotation with cultivation along straight rows conforming to the field boundaries. Since 1992, cultivation has been on the contour, unless otherwise noted.

Each year, ISCO (Lincoln, NE) bubbler flow measuring devices (these replaced float recorders in August 1995) and ISCO 3700 samplers are installed at the site in late February or early March to record flow rate and sampling times and collect samples. These units are removed from the field in December to protect them from possible damage due to freezing. Thus, the sample collection period extends from March to December. Generally, the ground is frozen during the winter

Table 1. Summary of agricultural activities on three watersheds during calibration and following three years of treatments (contour grass strips, agroforestry, and control).

Field preparation planting method	Crop	Variety	Planting date	Yield†	Fertilizer	Herbicide	Rate	Active ingredient	Rate
				kg ha^{-1}			L ha^{-1}		kg ha^{-1}
				1991					
No-till, off contour	corn	‡	20 May	‡	‡	Roundup	3.50	glyphosate	1.28
				1992					
Field cultivate, off contour	soybean	‡	8 June	1 680 (25)	none	Squadron	3.50	pendimethalin	0.89
								imazquin	0.14
						Galaxy	3.50	bentazon	1.26
								acifluorfen	0.42
				1993					
Off contour	corn	Pioneer 3394	1 June	8 152 (130)	160–50–100	Roundup	1.75	glyphosate	0.63
						2,4-D	0.58	2,4-D	0.28
						Bicep	5.84	atrazine	2.15
								metolachlor	1.68
				1994					
No-till, off contour	soybean	Pioneer 9362	18 May	3 695 (55)	none	Galaxy	3.50	bentazon	1.26
								acifluorfen	0.42
				1995					
No-till, off contour	soybean	GH	21 June	2 849 (42.4)	0–40–120	Roundup	1.75	glyphosate	0.63
		DK		2 130 (31.7)		2,4-D	0.58	2,4-D	0.28
		MH		6 119 (24.1)					
				1996					
No-till, on contour	corn	Lewis 4503	25 April, 5 June (replant)	10 660 (170)	NH_4NO_3 (179 kg/ha)	Bicep	5.61	atrazine	2.06
								metolachlor	1.62
						2,4-D	0.58	2,4-D	0.28
						Roundup	1.75	glyphosate	0.63
						Dual	1.17	*s*-metolachlor	1.06
				1997					
No-till, on contour	soybean	Pioneer 9363	16 May	2 822 (42)	none	Steel	3.50	pendimethalin	0.84
								imazquin	0.07
								imazethapyr	0.07
						Galaxy	3.50	bentazon	1.26
								acifluorfen	0.42
				1998					
No-till, on contour	corn	Pioneer 3335	18 May	5 017 (80)	160–50–100	Bicep	5.84	atrazine	2.15
								metachlor	1.68
						2,4-D	0.58	2,4-D	0.28
				1999					
No-till, on contour	soybean	MFA 5385CN	2 June	1 881 (28)	none	Roundup	1.75	glyphosate	0.63
						2,4-D	0.58	2,4-D	0.28
								pendimethalin	0.84
						Steel	3.50	imazquin	0.07
								imazethapyr	0.07
						Blazer (post)	1.17	acifluorfen	0.28
				2000					
Field cultivate, on contour	corn	Garst 8342	14 April	10 153 (162)	160–50–100	Bicep II	5.84	atrazine	2.33
								metolachlor	1.75
						Magnum	5.84	*s*-metolachlor	2.54
						2,4-D	0.58	2,4-D	0.28

† Values in parentheses are bushels per acre.
‡ Not available.

months and little or no runoff is observed from these watersheds. Flow measuring devices engage the sampler to withdraw a 135-mL sample of runoff after each 25 m^3 of flow occurs. Thus, samples are flow-weighted and collected for individual storms. For some consecutive events (maximum two) samples were not separated by event. Some precipitation events did not generate sufficient runoff to activate the sampler. After runoff events, recording devices were interrogated to save flow rate, water level, and sample intake time.

Chemical and physical analyses of composite water samples were performed in the Forest Hydrology Laboratory at the University of Missouri. A known volume of a well-mixed sample was filtered through a preweighed 934-AH glass microfiber filter (Whatman, Maidstone, UK) using a vacuum pump to estimate sediment concentration (American Public Health Association, American Water Works Association, and Water Environment Federation, 1992, p. 2–56). These filters were dried at 105°C to a constant weight and their dry weights were recorded. The difference between two dry weights and the filtered sample volume were used to estimate total suspended sediment concentration.

Unprocessed samples were refrigerated at 4°C until analysis. From 1991 to 1998, total nitrogen (TN), nitrate, and ammonium were determined with a Technicon (Terrytown, NY) autoanalyzer. This method determined total Kjeldhal nitrogen in ammonium form. Nitrate N concentrations of those samples were added to total Kjeldhal nitrogen to estimate total nitrogen of samples analyzed by the Technicon autoanalyzer. A

Lachat (Milwaukee, WI) Quick-Chem 8000 Analyzer was purchased in 1998 and subsequent TN, nitrate, and ammonium analyses were performed on this new instrument. Total nitrogen was determined using cadmium reduction on unfiltered samples following potassium perusulfate digestion. Total phosphorus (TP) was determined by ascorbic acid–molybdate procedure on unfiltered samples following ammonium peroxidisulfate digestion. Total phosphorus, TN, nitrate N (cadmium reduction), and ammonium (phenolate) were determined as outlined by Lachat Quickchem methods 10-115-01-1-F, 10-107-04-1-C, 10-107-04-1-B, and 10-107-6-1-A, respectively. The detection limit for the four methods was ≤ 0.002 mg L^{-1}. Quality control for the Lachat analyzer was maintained by randomly positioning three control standards with differing concentrations, four duplicate samples, and one quality control sample in each tray (90 samples). All samples with suspect concentrations and trays with unacceptable concentrations were reanalyzed.

Calibration relationships between control and treatment watersheds were developed using runoff and chemistry data. Runoff calibration was developed using 110 data pairs between the control and contour strip and control and agroforestry watersheds. For nutrients and sediment, 64 and 71 data pairs were used to develop calibration regressions. A few (maximum three) suspect or out of range data pairs were not used when developing these relationships. The treatment period started in June 1997. Treatment effects on runoff were examined using 44 events. However, only 17 runoff events produced sufficient runoff to collect samples for sediment and nutrient analysis, which allows us to compare sediment and nutrient losses due to treatments.

Approximately 9% of the runoff data collected during the calibration period on the control and agroforestry watersheds was lost due to sampler and flow meter failures caused by lightening damage, electronic malfunction, missed samples, and pump tube loosening or presence of debris in water samples. The contour strip watershed suffered a similar loss of 12% of the sampling events. Runoff and nutrient data for the control and one remaining treatment (when only one was available) were used to develop calibration for that treatment even though the other treatment data were lost. However, if a flowmeter or a sampler recorded any malfunction at any time during the treatment period, runoff and nutrient data for the entire event were discarded.

Statistical analyses of the data were performed using Statistical Analysis Systems (SAS Institute, 1999). Statistical relationships between the control and treatments were developed using regression. Our study contained a sufficient number of observations to determine a 10% change in discharge (USEPA, 1993). The procedure also explains testing the significance of the regression, testing the significance of overall regression, testing for a significant worthwhile difference, testing residual error, and evaluating the range of values obtained.

RESULTS AND DISCUSSION

Precipitation and Runoff Events

Weather conditions differed markedly among years and growing seasons (Table 2). During 1993 and 1998, the study area received 42 and 52% above normal precipitation, respectively. Two years (1991 and 1995) had 15% more annual precipitation than the long-term mean. During the 10-yr study period, four years had below normal precipitation and in 1999, the year the least rainfall was recorded, the area received 15% below normal precipitation. In 1997, the study area received long-term average precipitation. The annual precipitation at the Greenley Center during the 10-yr study period averaged 1010 mm, which is 9.8% above the long-term mean.

Table 2. Greenley Center annual precipitation, precipitation deviation from the long-term mean, and number of runoff events during the study period.

Year	Precipitation	Percent deviation from the mean precipitation	Number of runoff events
	mm	%	
1991	1064.0	+15.7	6
1992	1037.8	+11.6	18
1993	1307.6	+42.1	30
1994	863.1	−6.2	17
1995	1060.5	+15.3	18
1996	888.2	−3.5	12
1997	920.8	–	19
1998	1396.7	+51.8	27
1999	778.3	−15.4	6
2000	787.1	−14.4	2

Yearly, seasonal, and within growing season variations in the frequency and intensity of precipitation influenced runoff (Table 2). In general, most of the runoff occurred in the spring, early summer, and late fall when ground cover was at a minimum. Exceptions were recorded when abundant rainfall occurred in the midsummer of 1998. On average, each watershed produced 28 runoff events in years 1993 and 1998, when record high precipitation occurred. Two consecutive years (1999 and 2000) with below normal precipitation significantly reduced the number of measurable runoff events. This was especially true in 2000 when the intervals between precipitation events were longer and we recorded the least number of runoff events. During the 10-yr duration of the study, watersheds produced an average of 15 runoff events per year. Watersheds did not produce runoff after every rainfall event and some precipitation events caused very little runoff. Small runoffs during these periods failed to activate the sampler to collect water samples.

Calibration Parameters

Highly significant ($R^2 = 0.97$ and $p = 0.0001$) relationships for discharge (m^3 ha^{-1}) exist between the control and two treatments for the calibration period (Table 3).

Table 3. Regression relationships for runoff and nutrients during the calibration period. $p = 0.0001$ for all relationships.

Variable	R^2	Slope	n
Runoff			
Contour strip vs. control	0.97	0.999	110
Agroforestry vs. control	0.97	0.770	110
Sediment			
Contour strip vs. control	0.77	1.030	67
Agroforestry vs. control	0.88	0.973	71
Total P			
Contour strip vs. control	0.74	1.134	66
Agroforestry vs. control	0.69	0.931	70
Total N			
Contour strip vs. control	0.96	0.780	66
Agroforestry vs. control	0.95	0.636	68
Nitrate			
Contour strip vs. control	0.92	0.763	67
Agroforestry vs. control	0.92	0.577	70
Ammonia			
Contour strip vs. control	0.96	1.164	64
Agroforestry vs. control	0.94	0.544	69

Regression relationships for sediment and nutrients (kg ha^{-1}) from treatment watersheds and the control were significant during the calibration period ($p = 0.0001$). Total nitrogen, nitrate, and ammonium calibrations had an R^2 greater than 0.92 for both treatments. Agroforestry possessed a better relationship for sediment ($R^2 = 0.88$) than the contour strip treatment ($R^2 = 0.77$). The lowest regression coefficients for calibration were found for total phosphorus (0.74 and 0.69 for contour strip and agroforestry, respectively). Calibration slopes for each relationship of the contour strip treatment were always greater than the slopes of the agroforestry treatment (Table 3). This could be due to watershed morphology differences, inherent soil–site characteristics, relative effectiveness of the grass waterways, or other unknown factors.

Runoff

Treatments resulted in a 1 and 10% reduction in runoff during the treatment period on agroforestry and contour strip watersheds, respectively. The difference between observed and predicted losses averaged 18 and 230 m^3 ha^{-1} annually for agroforestry and contour strip treatments, during the three years of treatment. The total runoff during the treatment period was lower from both treatment watersheds compared with the predicted runoff. The agroforestry and contour strip watersheds produced only 76 and 90% of the runoff produced by the control watershed. During 1997, the same year treatments were applied, runoff reductions were not apparent. In fact, both watersheds produced more runoff than their respective predictions. Initial soil disturbance and reduced evapotranspiration in the developing contour strips may have caused the larger runoff volumes on treatment watersheds.

While Fig. 2 shows the runoff reduction trends on both treatments through the three-year time period, the initial 11 runoff events in 1998 on both agroforestry and contour strip treatments produced more discharge than predicted. As trees started to transpire, and vegetation buffer strips became better established, watersheds began showing discharge reductions (Fig. 2). The first reductions on both treatments appeared on 3 July 1998 (event 25), about one year after the grass–legume strips were established. The agroforestry and contour strip treatments resulted in 23 and 10% runoff reductions based on calibrations for this single event. Of the subsequent eight events (26th to 33rd events), only three generated runoff on the control. In October 1998, when fall precipitation commenced, both treatment watersheds consistently produced less runoff than predicted. Based on calibration relationships, agroforestry and contour strip treatments reduced runoff by 0.2 and 6% during the 1998 sampling period.

Plotting the deviations between predicted and observed runoff shows that most of the runoff decreases occurred in late 1998 and in 1999. The agroforestry and

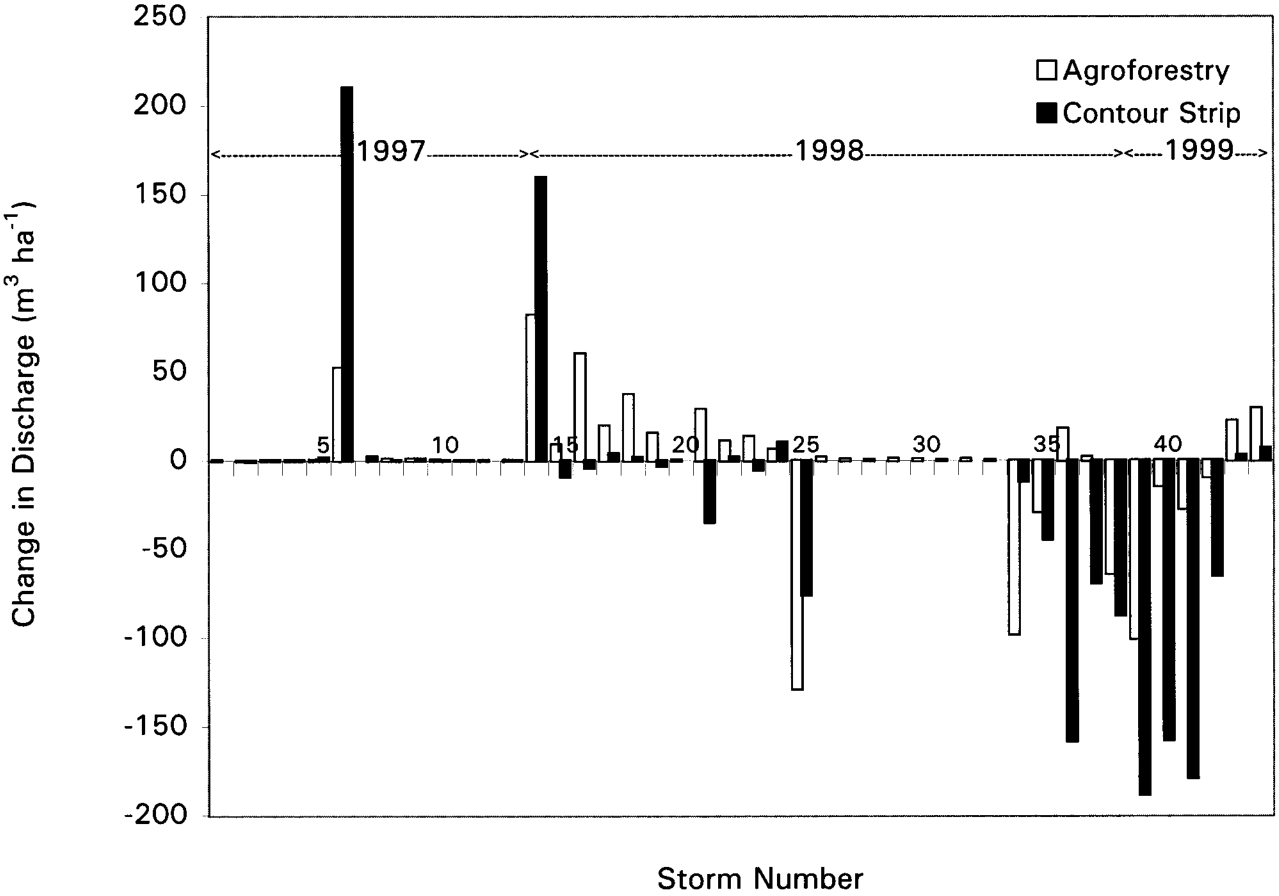

Fig. 2. Observed deviation from predicted (observed minus predicted) runoff on agroforestry and contour strip watersheds during the treatment period.

contour strip treatments reduced runoff by 7 and 33% during 1999. In terms of volume, the agroforestry and contour strip watersheds recorded 100 and 580 m^3 ha^{-1} reductions, respectively, compared with the predicted loss. Although both treatments reduced runoff compared with the predicted values, most events showed greater reductions in the contour strip than in the agroforestry treatment. In addition to this general trend, during the last four events of the treatment period, the agroforestry treatment showed consistently lower reductions in discharge compared with the contour strip treatment. At this point we cannot explain this difference but it may be related to natural differences that have occurred in the densities of the grass strips in the two treatment watersheds.

The largest decreases in runoff from both watersheds were associated with the largest runoff events. The largest deviations occurred on 15 Apr. 1999. This indicates the effect of grass strips before tree leaf out and before crop planting. During this period, the ground had the maximum exposure, which resulted in more runoff, but the grass strips reduced runoff by 20 and 29% on agroforestry and the contour strip treatments, respectively. Studying runoff control mechanisms, Robinson et al. (1996) reported that vegetative filter strips encourage infiltration of water into the soil. They found that the first 3 m of the vegetative filter strip was the most effective in decreasing the runoff volume. However, other studies have indicated that 4.6-m grass strips were more effective in controlling runoff and sediment loss than more narrow ones (Schmitt et al., 1999). Grass strips in this study are 4.56 m wide and reduced runoff significantly. In spite of the differences, this study confirms that agroforestry and contour grass strips markedly reduce runoff in corn–soybean rotation watersheds in the Midwest.

The largest runoff event during the treatment period occurred on 4 Oct. 1998 and accounted for 16, 14, and 17% of the total runoff on the control, agroforestry, and contour strip watersheds, respectively. The second and third largest events occurred on 3 July 1998 and 29 Oct. 1998. Averaged over the three watersheds, the largest two and three runoff events accounted for 26 (24.6 to 28.1) and 36 (33 to 37.2)%, respectively, of the total runoff during the treatment period. Some major precipitation events generate greater runoff in severe storms than in a more gradual rainfall event. Studies suggest that control of runoff and erosion caused by such catastrophic events is more important for water quality improvement than regulation of smaller events (Edwards and Owens, 1991). During the largest three runoff events, the contour strip and agroforestry treatments reduced runoff by 10 and 11%, respectively, on the two watersheds compared with the predicted runoff losses.

Sediment Loss

During the treatment period, the control, agroforestry, and contour strip treatments lost 200, 264, and 242 kg ha^{-1} sediment, respectively. The predicted losses, based on calibrations on the agroforestry and contour strip treatments, were 195 and 206 kg ha^{-1}. The agroforestry and contour strip treatments lost respectively 35 and 17% more than the predicted amounts. The first two runoff events during the treatment period on average caused 45 and 36 kg ha^{-1} sediment loss on agroforestry and contour strip treatments, respectively. Soon after treatments were established, both treatments lost more sediment. This could be due to soil disturbance during tree planting and grass strip establishment.

However, Fig. 3 shows that sediment loss has declined

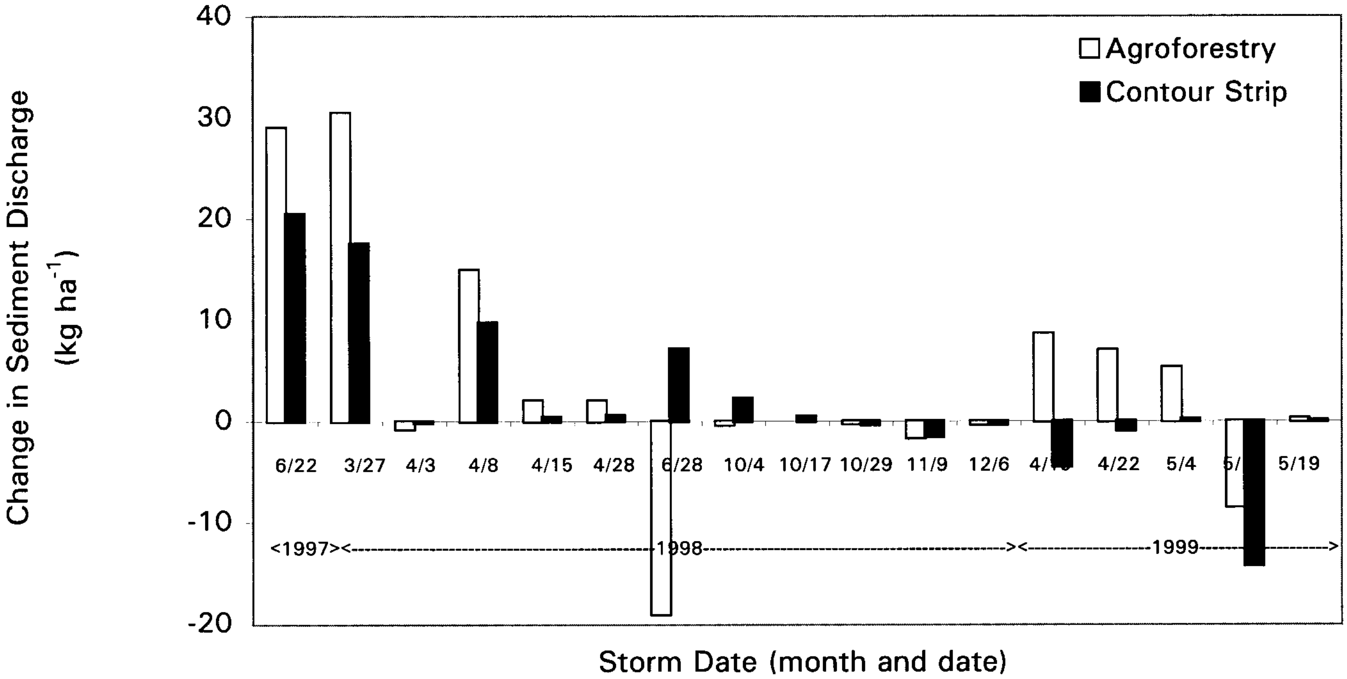

Fig. 3. Observed deviation from predicted (observed minus predicted) sediment loss on agroforestry and contour strip watersheds during the treatment period.

over the treatment period. As the grass strip and permanent vegetation became established and started to transpire and infiltration improved, runoff loss was reduced and associated losses declined. During the first two years (1997 and 1998) of sampling, the treatments did not result in reductions of sediment loss. However, more recently, both the agroforestry and contour strip watersheds have shown trends toward declining differences. At this time we cannot speculate whether these treatment differences will continue to trend lower.

The contour strip treatment showed a 19% reduction in sediment loss during 1999. Although the agroforestry treatment has consistently indicated a trend toward progressively smaller differences between the observed and predicted losses, it is still generating larger losses than the predictions would suggest. The three consecutive precipitation events in 1999 between April and May resulted in above-predicted losses of sediment on the agroforestry treatment. The study area received less than the normal precipitation during the early months of 1999. Large precipitation events in April (164 mm) and May (108.5 mm) caused severe losses on the agroforestry treatment before crops were planted when the ground had minimum cover. Soon after these three severe losses, sediment loss on the agroforestry watershed appeared to be effectively controlled by the treatment. The contour strip treatment, however, consistently reduced sediment loss from the beginning of the treatment period. Certain small runoff events resulted in more than the predicted loss on the contour strip treatment, but these did not yield significant losses.

Figure 3 also shows that soil loss was largely determined by individual precipitation events and the ground condition. The greatest sediment loss by an individual runoff event on the three watersheds was caused by different precipitation events. The precipitation event on 27 Mar. 1998 caused the greatest loss on agroforestry and contour strip treatments while the precipitation event on 15 Apr. 1999 caused the greatest loss on the control watershed. The second and third largest sediment losses on the agroforestry and contour strip watersheds, respectively, occurred on 2 June 1998. The large sediment losses on the treatment watersheds in the beginning of the treatment period indicate that grass and tree components of these two watersheds were not well enough established to be effective in controlling sediment loss. As time progressed and the buffers became better established, treatment effects significantly improved. Results from these initial three years of the study show that grass strips without trees were more effective than grass strips with trees (agroforestry).

Historical data shows that grass buffer strips improve infiltration and trap sediment. In France, 5.7- and 11.1-m-wide grass filter strips reduced suspended sediment in runoff by 69 to 90% and 69 to 97%, respectively (Patty et al., 1997). Studying soil erosion on corn, wheat, and meadow watersheds, Edwards and Owens (1991) observed 92, 8, and 0% soil loss on their watersheds, respectively. During the third year of this study soil loss was reduced by 19% on the contour strip watershed. We attribute these reductions mainly to the grass strips. However, as much as 87 and 93% reductions of sediment in runoff on grass strips and grass–shrub–tree strips, respectively, have been reported by Schmitt et al. (1999). Research has also shown that most of the sediment and nutrients are trapped within the first 4 to 7.5 m of the strip and, thereafter, increasing width results in marginal improvements in retention (Schmitt et al., 1999). Our study design consists of several grass strips and trees within grass strips and, therefore, we anticipate higher percentage reductions in sediment as runoff travels through these buffers.

Total Phosphorus (TP) Loss

During the three-year treatment period, the control, agroforestry, and contour strip treatments lost 3.11, 2.41, and 3.26 kg TP ha^{-1}. The predicted losses based on calibration relationships for the agroforestry and contour strip were 2.89 and 3.53 kg TP ha^{-1}, respectively. Treatments have reduced TP loss by 17 and 8% on the agroforestry and contour strip watersheds, respectively.

Annual total phosphorus loss from the control, agroforestry, and contour strip watersheds averaged 1.0, 0.8, and 1.1 kg TP ha^{-1} during the three-year treatment period. However, TP reductions generally did not begin to occur until fall 1998 (Fig. 4). No significant reductions in TP loss were noticed for either treatment watershed in 1997. The agroforestry watershed showed greater reductions (18%) than the contour strip treatment (3.7%) in 1998. In fact, most of the reductions in TP loss occurred in 1998 when watersheds produced the greatest number of runoff events during the treatment period. During 1999, the agroforestry and contour strip treatments produced 14 and 26% reductions, respectively, in TP loss based on calibration relationships. The contour strip watershed indicated a marked improvement in reduction of P in runoff from 1998 to 1999. In contrast, the agroforestry watershed did not indicate a similar improvement. However, it appears that loss of P in runoff varies from year to year and from watershed to watershed.

Loss of P in runoff occurs in dissolved and particulate forms sorbed by soil and organic particles (Sharpley et al., 1994). Jordan et al. (1997) noticed that loss of P from some watersheds was found to be related to rates of erosion and these relationships changed from year to year. Total phosphorus concentration in runoff is significantly affected by particle size and amount of P in the sediment (Quinton et al., 2001). As sediment concentration increases in runoff water, silt-size particles with lower P increase in proportion to clay-size particles that have higher concentrations of P (Sharpley et al., 1992; Wall et al., 1996). On our study watersheds (Fig. 1), runoff water travels through grass waterways before it reaches the flume. The control watershed has the shortest grass waterway while the agroforestry has the longest. The length of the grass waterway might influence size distribution of sediments and thus measured TP loss during a storm.

The largest TP losses on each watershed were associated with heavier precipitation events and open ground

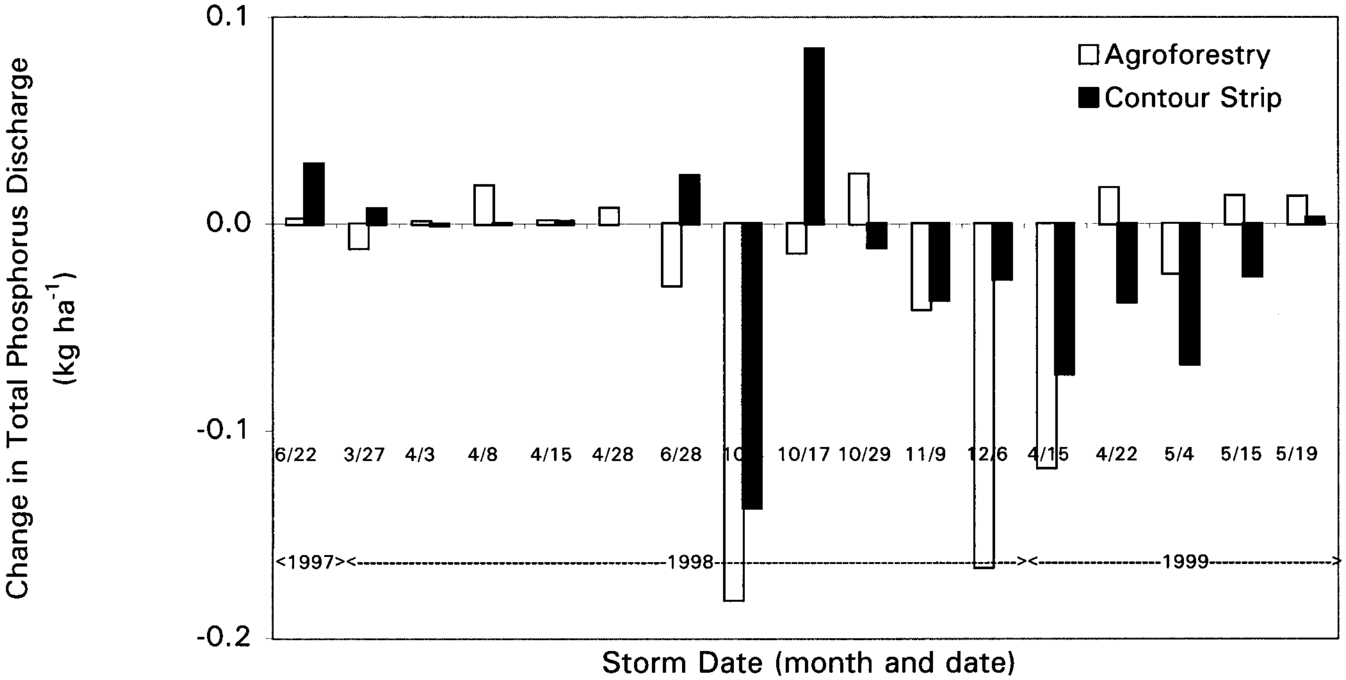

Fig. 4. Observed deviation from predicted (observed minus predicted) total phosphorus loss on agroforestry and contour strip watersheds during the treatment period.

conditions. The greatest losses occurred on 4 and 29 Oct. and 9 Nov. 1998 on the three watersheds. These losses occurred after crops had been harvested and above-average fall precipitation occurred. The largest single TP loss accounted for 19 to 22% of the total P loss from the watersheds during the three-year treatment period. Since the heaviest rainfall events result in the greater P losses, grass buffers with or without trees are especially important in reducing TP losses from agricultural fields and their subsequent effect on water quality. Permanent vegetation such as grass and trees actively transpires water, increases infiltration, and traps sediment, thereby reducing TP loss.

In findings reported by Dillaha et al. (1989), orchard grass filter strips of 9.1 and 4.6 m in width removed 79 and 61% P in runoff, respectively. Strips of varying widths have been shown to remove 22 to 89% TP (Sharpley et al., 1992; Patty et al., 1997; Schmitt et al., 1999). Wider strips compared with narrower strips and more established buffers compared with newer ones are normally regarded as being more effective. The reduced TP loss from treatment watersheds in our study may have resulted from greater infiltration and less interaction of runoff with surface soil. Surface coverage in the filter strips would logically reduce runoff and enhance settling of material from the runoff. Also, roots of the vegetation should effectively remove portions of the inorganic P in the soil solution through uptake, causing increased P adsorption capacity of soils and increased P retention (Lyons et al., 1998). Subsurface losses of agricultural P in some soils are important (Gburek and Sharpley, 1998) and, therefore, management strategies should consider minimization of such losses (Sims et al., 1998). However, in the northern Missouri soybean–corn areas, most soils have a restrictive argillic horizon, which inhibits deep subsurface flow and promotes copious runoff. Our study demonstrates that contour grass strips and agroforestry treatments can be used to effectively control TP in runoff from row-cropped fields.

Total Nitrogen (TN) Loss

During the treatment period, the control, agroforestry, and contour strip watersheds lost 11.3, 10.1, and 9.7 kg TN ha^{-1}, respectively. On average, control, agroforestry, and contour strip watersheds lost 3.77, 3.37, and 3.24 kg TN ha^{-1} per year. Compared with predicted losses based on calibrations, TN loss was not reduced in 1997 or 1998 but small reductions occurred in 1999. During the 1998 and 1999 sampling years, TN loss was reduced for only three storms on the agroforestry watershed. However, on the contour strip watershed, TN loss was generally reduced in October 1998 and thereafter (Fig. 5). The storm on 4 Apr. 1999 had the greatest reduction in TN loss associated with the treatments. On this day, the agroforestry and contour strip treatments reduced loss by 21 and 20%, respectively, compared with their predicted discharges.

The largest losses of TN occurred on 28 June 1998, 4 Oct. 1998, and 4 Apr. 1999. The control, agroforestry, and contour strip treatments lost 2.8, 4.2, and 2.7 kg N ha^{-1}, respectively, during the largest event. This single largest loss accounted for 25, 41, and 28% of the total loss on the control, agroforestry, and contour strip watersheds, respectively. The study area received 52% above the normal precipitation during 1998. Annual fertilizer application had been completed by 18 May and field cultivation had been completed by 27 June, loosening the surface soil. These may have been contributing factors to the large nitrogen runoff on the following day (28 June).

The largest two and three TN losses accounted for

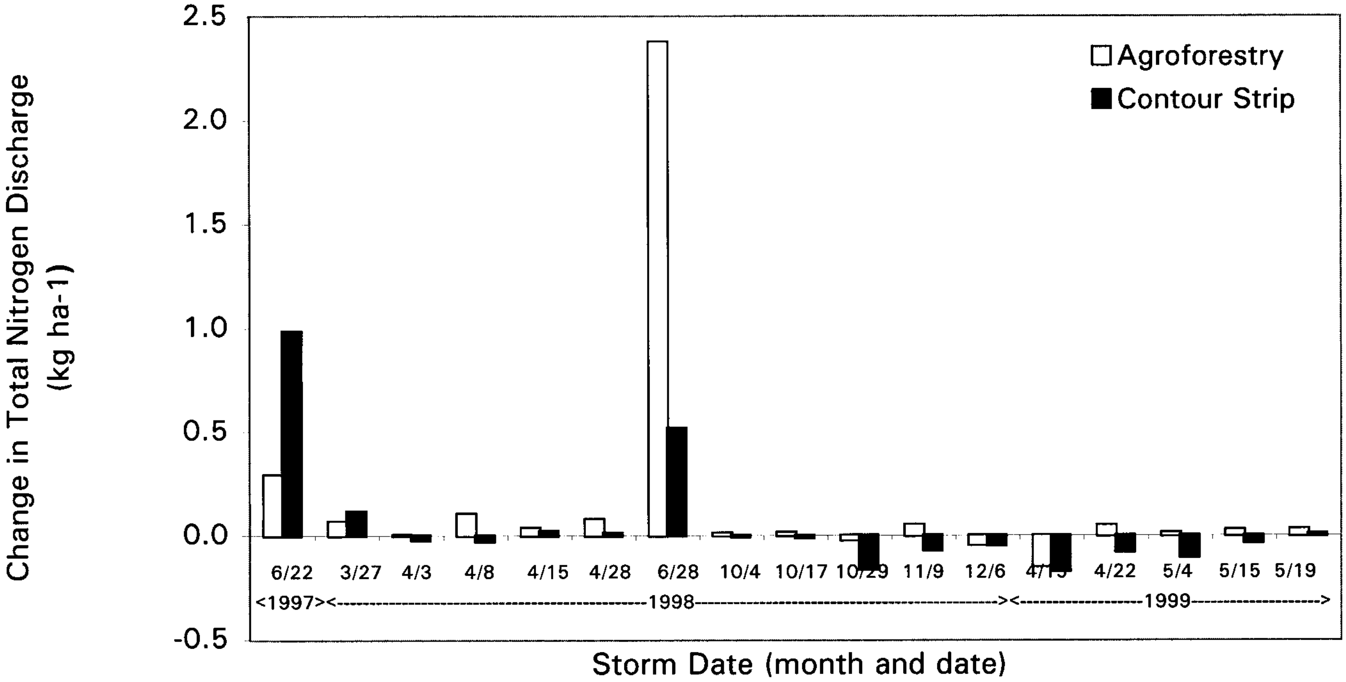

Fig. 5. Observed deviation from predicted (observed minus predicted) total nitrogen loss on agroforestry and contour strip watersheds during the treatment period.

42 and 50%, respectively, of the total loss averaged over the three watersheds. This finding is consistent with those from other studies in which a few large storms have been shown to account for most of the annual loss (Edwards and Owens, 1991). These losses occurred early in the second year of treatments, with the largest loss being associated with the first precipitation event in the second year and the second largest associated with the second event. While additional data are required to determine if TN losses resulting from large precipitation events can be effectively reduced by contour strip and agroforestry treatments, our data suggest a trend toward reduced discharges following the initial establishment period for the strips and trees.

The presence of stem, thatch, and roots effectively slows runoff, promotes sediment removal, and increases uptake of N and other nutrients. Other studies have also shown decreased nitrogen loss due to grass strips and agroforestry (Dillaha et al., 1989; Lowrance et al., 1984; Schmitt et al., 1999). Plant uptake is the single most important process that removes N during the growing season (Lowrance et al., 1984), while dentrification is the dominant process in the winter months (Gilliam, 1994). On our treatment watersheds, contour grass strips were established in June 1997. Treatment effects during the first year were not observed. However, during the second year, especially after October, trends were observed that suggest both treatments have the potential to reduce TN loss from the site (Fig. 5).

Burwell et al. (1976) reported that in northwestern Missouri, subsurface discharge of NO_3–N accounted for up to 84% of the total annual stream discharge. Comparing flow components on two watersheds, one with riparian buffer strips and the second with grass filter strips (in northeastern Missouri), Schmitt (1999) found that shallow subsurface and surface flow were the primary storm flow processes for each. He attributed the differences to vegetation, infiltration, flow resistance, antecedent moisture, and season. The location of the grass strip, number of strips, and the width of the strip appear to influence proportion of surface and subsurface losses. In our study, each treatment watershed has several grass strips parallel to the contours. Therefore, we would expect greater reductions in nutrient losses due to flow resistance, interception, infiltration, and vegetation uptake.

In riparian studies, subsurface water appears to be the dominant pathway of nitrate flux between croplands and riparian forests (Peterjohn and Correll, 1984). In a 208-ha forested watershed in Pennsylvania, subsurface flow accounted for more than 95% of storm flow (DeWalle et al., 1988). In an area where intact permanent vegetation with widely distributed root systems exists, one might expect it to be capable of taking up most of the subsurface nutrients before they leave the watershed. Reduction in losses results from a combination of reduction in runoff and utilization of nutrients by the vegetation. Although our watersheds are dominated by surface flow discharge, we anticipate that the subsurface flow component on the agroforestry watershed may account for a greater portion of storm flow as the trees grow (their current age is three years) and exert their influence on filtration and uptake.

Nitrate Loss

The measured loss from the control, agroforestry, and contour strip watersheds averaged 1.9, 1.8, and 1.5 kg ha^{-1} yr^{-1} nitrate N, respectively, during the first three years of treatment. Only one runoff event occurred in 1997, after treatments were established. However, during the 1998 sampling year, 11 runoff events occurred

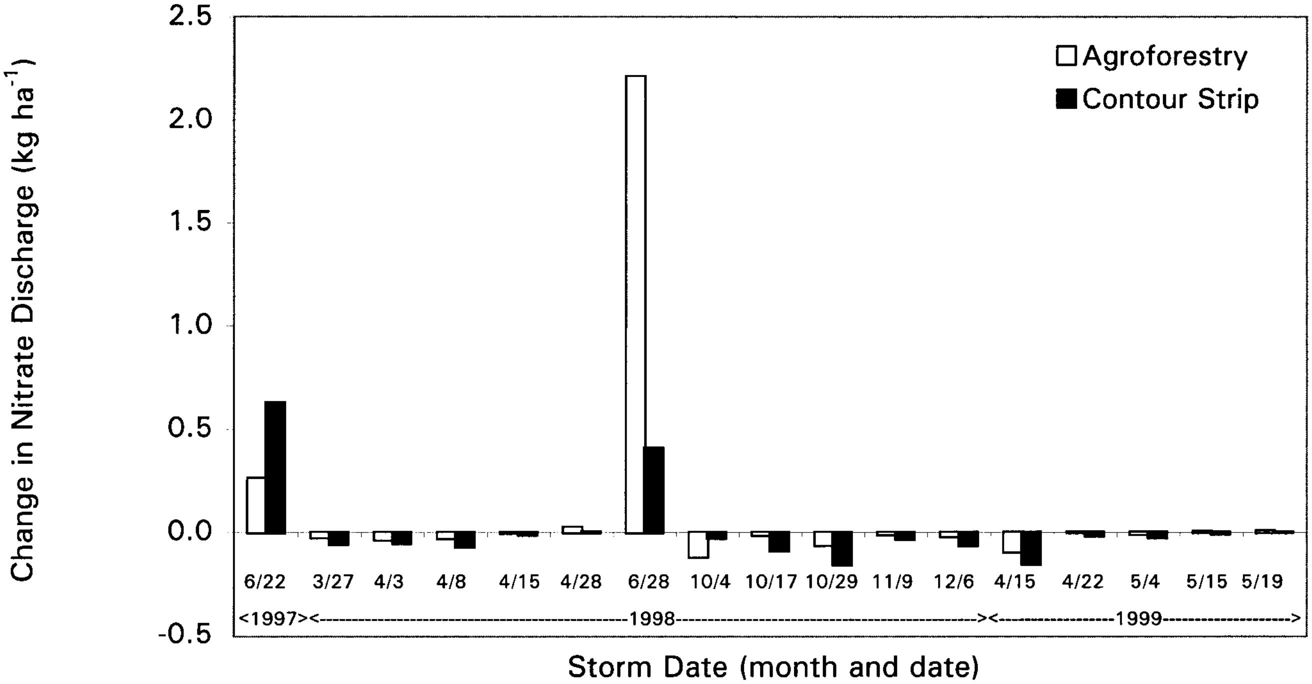

Fig. 6. Observed deviation from predicted (observed minus predicted) nitrate N loss on agroforestry and contour strip watersheds during the treatment period.

on the three watersheds that produced samples (Fig. 6). The control, agroforestry, and contour strip treatments lost 4.35, 4.43, and 3.17 kg ha^{-1} yr^{-1} nitrate N, respectively, in 1998. The contour strip treatment had a 0.15 kg ha^{-1} reduction in nitrate loss while the agroforestry treatment lost 1.92 kg ha^{-1} more than the predicted losses. The reduction in nitrate loss on the contour strip represents 4% of the predicted loss.

In the third year (1999) of sampling, both the agroforestry and the contour strip treatments reduced nitrate loss compared with the predicted loss. During the five sampling events, control, agroforestry, and contour strip watersheds lost 0.72, 0.32, and 0.35 kg ha^{-1} yr^{-1}, respectively. During these five runoff events, a 0.10 kg ha^{-1} reduction in nitrate loss was observed on the agroforestry plots while a 0.20 kg ha^{-1} reduction was observed on the contour strip plots. This indicated that contour grass strips and agroforestry treatments reduced nitrate loss by 36 and 24%, respectively, during the third year of treatments.

The largest single nitrate loss occurred on 28 June 1998 (2.57, 3.69, and 2.37 kg ha^{-1} loss on the control, agroforestry, and contour strip watersheds, respectively). This accounted for 46, 70, and 52% of the total loss during the treatment period for the control, agroforestry, and contour strip treatments. Again, timing of fertilizer application, land preparation, and heavy precipitation are believed to be related to these losses. This largest single loss was associated with 2.2 and 0.41 kg ha^{-1} more nitrate N loss on the agroforestry and contour strip plots than was predicted. With the exclusion of the nitrate N loss during this one event, agroforestry and contour strip treatments show a 0.13 kg ha^{-1} nitrate reduction over the treatment period. The second largest loss on treatment watersheds occurred soon after treatments were established (22 June 1997). On the control treatment, it occurred on 8 Apr. 1999. During this period heavy precipitation occurred when the ground was fully recharged and free of vegetation. Nitrate loss that occurred on our watersheds suggests the importance of nutrient management, timing of fertilizer application, ground conditions, and precipitation. Heavy precipitation events are especially important as they accounted for the greater percentage of the total loss of nitrate in our study.

Our results indicate that during the first year of our treatment period, soon after treatments were established, grass strips and agroforestry treatments were ineffective in controlling nitrate loss. However, during the second year, the treatments began showing an effect. If the largest single loss on both the agroforestry and contour strip plots was excluded, they would have had 28 and 41% reductions in nitrate N losses, respectively, during the second year. The benefits from having grass strips and trees became even more apparent in the third year of sampling (Fig. 6). Furthermore, during the early part of 1999, when heavy rains occurred and the ground was fully charged, a positive effect was found from having the grass strips and trees. This occurred in spite of the existence of the worst combination of conditions on the site. Even though our buffers are only three years old, they are already showing good potential as a management practice to reduce sediment and nutrient losses from row-cropped fields. With the exclusion of the largest single loss, agroforestry and contour strip treatments reduced nitrate N loss by 26 and 39%, respectively, on average during the second and third years of treatments.

Ammonium Loss

During the treatment period, the control, agroforestry, and contour strip treatments lost 0.5, 0.3, and 0.4

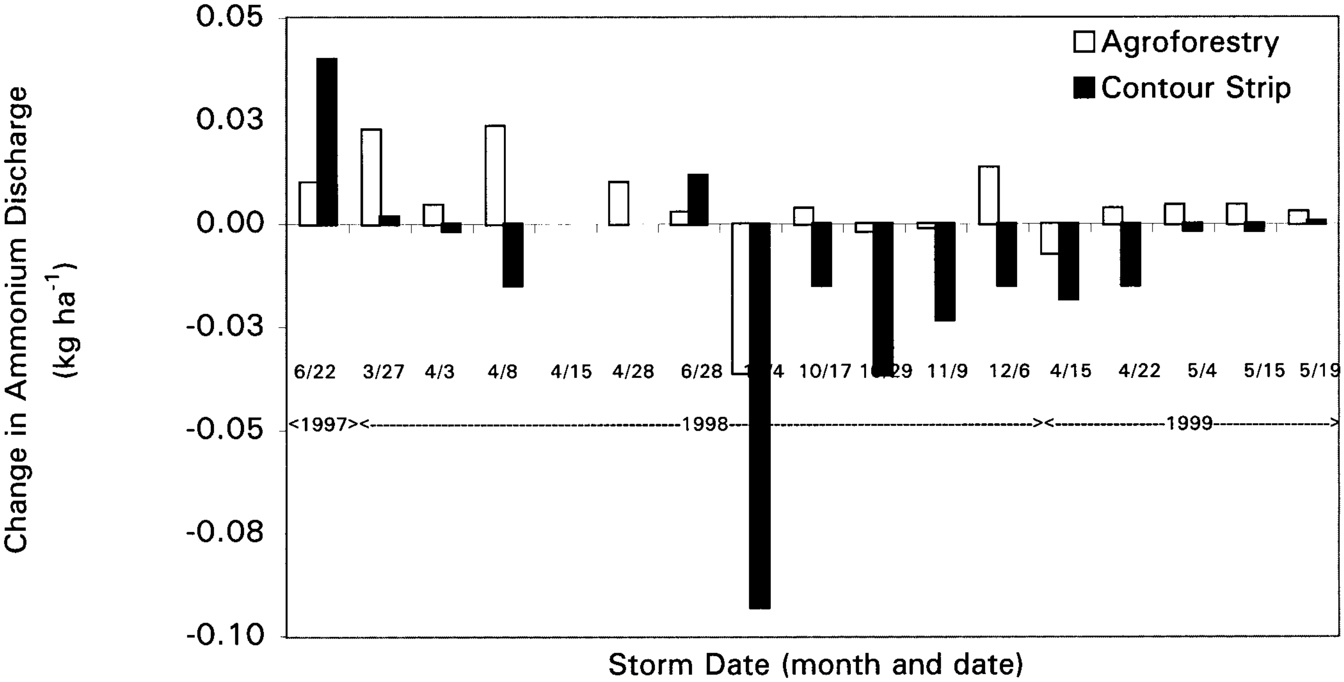

Fig. 7. Observed deviation from predicted (observed minus predicted) ammonium N loss on agroforestry and contour strip watersheds during the treatment period.

kg ha^{-1} ammonium, respectively. Based on calibrations, the contour strip treatment reduced ammonium loss by 32% during the treatment period. Only one runoff event produced samples in 1997 after treatments were established (Fig. 7). During this event, the treatments lost more ammonium than the control. The contour strip treatment had a 40% reduction in ammonium loss in 1998 while the agroforestry lost more than that predicted. In 1999, the contour strip watershed recorded a 33% reduction in ammonium loss. In contrast, the agroforestry treatment consistently lost more than the predicted loss in 1997 and 1998. Out of those 17 sampling events, only three events in 1998 and the first event of 1999 produced less than the predicted loss on the agroforestry treatment. However, it also began to show declining losses in ammonium.

The largest ammonium loss on the control and contour strip occurred on 14 Oct. 1998. The largest loss with agroforestry occurred on 27 Mar. 1998. Although the greatest losses on treatment watersheds occurred on different dates, the greatest reduction in ammonium loss occurred on the same day (14 Oct. 1998). On this day, agroforestry and contour strip treatments reduced ammonium loss by 47 and 57%, respectively.

Treatments have reduced ammonium loss on row-cropped watersheds, especially during large catastrophic events. To date, our results have not shown that trees in a grass strip are more effective in reducing ammonium loss compared with grass strips alone. However, Fig. 7 indicates that reductions in ammonium loss occur on both watersheds with time. During the first three years of treatment, the grass strip treatment appeared to be more effective in controlling ammonium loss from row-cropped watersheds. However, we anticipate greater reductions from the agroforestry watershed as trees occupy greater soil volumes and grow larger.

SUMMARY AND CONCLUSIONS

Decreases in runoff and nutrient yields resulting from filter strips, forested buffers, and single trees have been reported by others (Gilliam, 1994; O'Neill and Gordon, 1994; Schmitt et al., 1999). However, a paired watershed research approach to examine runoff, sediment, and nutrient reductions as influenced by agroforestry and contour grass strips has been used infrequently. The larger scale of watersheds is likely to introduce heterogeneity not found in plots but the paired approach statistically controls climatic and hydrological differences and allows water quality differences to be attributed to treatment.

In this study, we examined the agroforestry and contour strip effects on runoff, sediment, and nutrient loss reductions on corn–soybean rotations. The agroforestry treatment, after only three years, reduced runoff and total phosphorus losses by 1 and 17% based on calibration relationships. The contour grass strip treatment reduced runoff and total phosphorus losses by 10 and 8%, respectively. Most reductions occurred in second and third years after treatment establishment, as the vegetation cover increased and roots of the vegetation occupied more soil volume.

Extreme precipitation events were found to contribute significantly to the export of nutrients and runoff. The largest three runoff events accounted for 36% of the total runoff during the treatment period. The contour strip and agroforestry treatments reduced the runoff of the largest three events by 10 and 11%, respectively. The largest single TP and TN losses accounted for 19 to 22% and 25 to 41% of the total on the three watersheds. Our results clearly indicate that agroforestry and contour strip practices, when incorporated directly into corn–soybean watersheds in the Midwest, can be used to effectively reduce runoff volume and sediment and nutrient loss. These decreases in runoff, sediment, and nutrients following treatment application are especially significant given the relatively small number of runoff events and the short time that treatments have been in place. In particular, the contribution made in reducing N and P loss should increase with tree growth on the agroforestry watershed.

Buffer strips can be used to control degradation of stream water quality from agricultural nonpoint source pollution. In particular, site-specific slope and precipitation factors should be factored in when determining the appropriate width and tree density of buffer strips. Until such design factors become widely studied, however, landowners and policymakers can effectively use buffers similar to those of our study to reduce NPS pollution significantly.

ACKNOWLEDGMENTS

We gratefully acknowledge the Soil and Water Program of the Missouri Department of Natural Resources, the Missouri Agricultural Experiment Station, and the USEPA for financial assistance. The results presented are the sole responsibility of the investigators and/or the University of Missouri and may not represent the policies or positions of the funding agencies. We thank Randall Smoot at the Greenley Center and Dan Ramsey for help with field work and laboratory analyses.

REFERENCES

American Public Health Association, American Water Works Association, and Water Environment Federation. 1992. Standard methods for the examination of water and wastewater. APHA, Washington, DC.

Burwell, R.E., G.E. Schuman, K.E. Saxton, and H.G. Heinemann. 1976. Nitrogen in subsurface discharge from agricultural watersheds. J. Environ. Qual. 5:325–329.

Cooper, C.M., and W.M. Lipe. 1992. Water quality and agriculture: Mississippi experiences. J. Soil Water Conserv. 47:220–223.

Cooper, J.R., and J.W. Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. Soil Sci. Soc. Am. J. 51:1600–1604.

DeWalle, D.R., B.R. Swistock, and W.E. Sharpe. 1988. Three component tracer model for stormflow on a small Appalachian forested catchment. J. Hydrol. (Amsterdam) 104:301–310.

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

Edwards, W.M., and L.B. Owens. 1991. Large storm effects on total soil erosion. J. Soil Water Conserv. 46:75–78.

Garrett, H.E., and L.E. Buck. 1997. Agroforestry practice and policy in the United States of America. For. Ecol. Manage. 91:5–15.

Gburek, W.J., and A.N. Sharpley. 1998. Hydrologic controls on phosphorus loss from upland agricultural watersheds. J. Environ. Qual. 27:267–277.

Gilliam, J.W. 1994. Riparian wetlands and water quality. J. Environ. Qual. 23:896–900.

Gold, M.A., and J.W. Hanover. 1987. Agroforestry systems for temperate zones. Agrofor. Syst. 5:109–121.

Hewlett, J.D., and L. Pienaar. 1973. Design and analysis of catchment experiment. p. 88–106. *In* E.H. White (ed.) Proc. Symp. The use of small watersheds in determining effects of forest land use on water quality. College of Agric., Univ. of Kentucky, Lexington.

Jordan, T.E., D.L. Correll, and D.E. Weller. 1997. Effects of agriculture on discharge of nutrients from coastal plain watersheds of Chesapeake Bay. J. Environ. Qual. 26:836–848.

Kang, B.T., G.F. Wilson, and T.L. Lawson. 1984. Alley cropping: A stable alternative to shifting cultivation. Int. Inst. Tropical Agric., Ibadan, Nigeria.

Krstansky, J.J., G.S. Henderson, and H.E. Garrett. 1997. An evaluation of the use of agroforestry practices to reduce soil erosion on Missouri soils. Final Report MDNR 90-3. School of Natural Resour., Univ. of Missouri, Columbia.

Lowrance, R., R. Todd, J. Fail, O. Hendrickson, R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34:374–377.

Lyons, J.B., J.H. Gorres, and J.A. Amador. 1998. Spatial and temporal variability of phosphorus retention in a riparian forest soil. J. Environ. Qual. 27:895–903.

O'Neill, G.J., and A.M. Gordon. 1994. The nitrogen filtering capability of Carolina poplars in an artificial riparian zone. J. Environ. Qual. 23:1218–1223.

Owenby, J.R., and D.S. Ezell. 1992. Monthly station normals of temperature, precipitation, and heating and cooling degree days 1961–90. Climatography of the United States no. 81. Natl. Climatic Data Center, Ashville, NC.

Patty, L., B. Real, and J.J. Gril. 1997. The use of grassland buffer strip to remove pesticides, nitrate, and soluble phosphorus compounds from runoff water. Pestic. Sci. 49:243–251.

Peterjohn, W.T., and D.L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observation on the role of riparian forest. Ecology 65:1466–1475.

Quinton, J.N., J.A. Catt, and T.M. Hess. 2001. The selective removal of phosphorus: Is event size important? J. Environ. Qual. 30:538–545.

Robinson, C.A., M. Ghaffarzadeh, and R.M. Cruse. 1996. Vegetative filter strip effects on sediment concentration in cropland runoff. J. Soil Water Conserv. 50:227–230.

Sanchez, P.A. 1995. Science in agroforestry. Agrofor. Syst. 30:5–55.

SAS Institute. 1999. SAS user's guide. Statistics. SAS Inst., Cary, NC.

Schmitt, S.J. 1999. Application of a flow source mixing model and remote sensing to the hydrology and water quality of two small watersheds in northern Missouri. M.S. thesis. Univ. of Missouri, Columbia.

Schmitt, T.J., M.G. Dosskey, and K.D. Hoagland. 1999. Filter strip performance and processes for different vegetation, widths, and contaminants. J. Environ. Qual. 28:1479–1489.

Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Smith, T.C. Daniel, and K.R. Reddy. 1994. Managing agricultural phosphorus for the protection of surface water: Issues and options. J. Environ. Qual. 23:437–451.

Sharpley, A.N., S.J. Smith, O.R. Jones, W.A. Berg, and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30–35.

Sims, J.T., R.R. Simard, and B.C. Joern. 1998. Phosphorus loss in agricultural drainage: Historical perspective and current research. J. Environ. Qual. 27:277–293.

USEPA. 1993. Paired watershed study design. USEPA Rep. 841-F-93-009. Office of Water, Washington, DC.

Van Noordwijk, M., G. Lawson, A. Soumare, F.F.R. Groot, and K. Hairiah. 1996. Root distribution of trees and crops: Competition and/or complementarity. p. 319–364. *In* C.K. Org and P. Huxley (ed.) Tree–crop interaction: A physiological approach. University Press, Cambridge, England.

Verchot, L.V., E.C. Franklin, and J.W. Gilliam. 1998. Effects of agricultural runoff dispersion on nitrate reduction in forested filter zone soils. Soil Sci. Soc. Am. J. 62:1719–1724.

Wall, G.J., A.W. Bos, and A.H. Marshall. 1996. The relationship between phosphorus and suspended sediment loads in Ontario watersheds. J. Soil Water Conserv. 51:504–507.

Watson, F.C. 1979. Soil survey of Knox, Monroe, and Shelby Counties, Missouri. USDA Soil Conservation Serv., Missouri Agric. Exp. Stn., U.S. Govt. Print. Office, Washington, DC.

Young, A. 1997. Agroforestry for soil management. 2nd ed. CAB Int., Oxon, UK.