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Nutrient removal by prairie filter strips in agricultural landscapes

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Abstract: Nitrogen (N) and phosphorus (P) from agricultural landscapes have been identified as primary sources of excess nutrients in aquatic systems. The main objective of this study was to evaluate the effectiveness of prairie filter strips (PFS) in removing nutrients from cropland runoff in 12 small watersheds in central Iowa. Four treatments with PFS of different spatial coverage and distribution (No-PFS, 10% PFS, 10% PFS with strips, and 20% PFS with strips) were arranged in a balanced incomplete block design across four blocks in 2007. A no-tillage two-year corn (*Zea mays* L.) –soybean (*Glycine max* [L.] Merr.) rotation was grown in row-cropped areas beginning in 2007. Runoff was monitored by H flumes, and runoff water samples were collected during the growing seasons to determine concentrations of nitrate-nitrogen (NO_3-N), total nitrogen (TN) and total phosphorus (TP) through 2011. Overall, the presence of PFS reduced mean annual NO_3-N, TN, and TP concentrations by 35%, 73%, and 82%, respectively, and reduced annual NO_3-N, TN, and TP losses by 67%, 84%, and 90%, respectively. However, the amount and distribution of PFS had no significant impact on runoff and nutrient yields. The findings suggest that utilization of PFS at the footslope position of annual row crop systems provides an effective approach to reducing nutrient loss in runoff from small agricultural watersheds.

Key words: nutrient loss—reconstructed prairie—row crop—vegetative filter strips

Excess nutrients in fresh water have been identified as contributing to water quality degradation in the Mississippi River Basin and the seasonal occurrence of severe oxygen (O_2) depletion, or hypoxia, in the northern Gulf of Mexico (Conley et al. 2009). Nitrogen (N) and phosphorus (P) from nonpoint agricultural sources, midwestern farms producing corn (*Zea mays L.*) and soybean (*Glycine max* [L.] Merr.) in particular, are considered primary sources of water-borne nutrients (Science Advisory Board 2008). Nine midwestern states accounted for an estimated 75% of the N and P delivery to the Gulf of Mexico (Alexander et al. 2008). During the past decades, various best management practices have been recommended in the upper Mississippi River Basin to reduce the nutrient load to the Gulf. Implementation has reduced the loading of total N from its maximum in 1990, but the loading of total P remains steady (Turner et al. 2007). The 2008 US Environmental Protection Agency Hypoxia Action Plan calls for an additional 45% reduction in both N and P loads down the Mississippi River. To achieve this goal, a variety of conservation practices and nutrient management strategies will need to be implemented across the midwestern United States.

Among other management practices, installing vegetative filter strips (VFS) within crop production systems has proven to be a practical strategy in reducing soil loss and nutrient transport from agricultural landscapes. Vegetative filter strips are bands of perennial vegetation established within crop production systems, typically at the lower portion of the land or upslope along the contour (Dillaha et al. 1989). Numerous studies have demonstrated that VFS can decrease nutrient concentrations in agricultural runoff and reduce nutrient loads by reducing flow velocity, increasing water infiltration, and promoting plant uptake of excess nutrients (Dillaha et al. 1989; Patty et al. 1997; Udawatta et al. 2002; Duchemin and Hogue 2009; Udawatta et al. 2011). According to the meta-analysis by Zhang et al. (2010), the N removal efficacy of VFS has a range of 2.2% to 99.9%, and the P removal efficacy has a range of 22% to 96.3%. Filter strip width plays a significant role in nutrient removal, accounting for about 44% of the variation in N removal efficacy and 35% in P removal efficacy (Zhang et al. 2010). Other key factors affecting nutrient removal efficacy include slope, vegetation type, and flow conditions. Vegetative filter strips generally become less effective under concentrated flow conditions or when water flows over a relatively small effective filter strip area, which is the area of the filter strips that actually contacts runoff water (Dosskey et al. 2002; Helmers et al. 2005).

Most studies assessing the performance of VFS in nutrient removal have been conducted on a plot scale, and assessments at the hillslope and watershed scale are crucial but lacking (Nord and Lyon 2003; Baker et al. 2006). In contrast to homogeneous assumptions in a small plot study, the heterogeneous nature of watersheds in regard to topography and soil type poses challenges for evaluation. Many hydrological processes that are important to water and nutrient transport at the watershed scale may have little influence on plot-scale observations, including subsurface flow, interaction between surface water and groundwater, and water table fluctuation. Deelstra et al. (2005) found that the N concentrations in runoff decreased with an increase in spatial scale. In addition, concentrated flows that commonly develop over long slopes in watersheds are less likely observed on short slopes in plots. Consequently, the results from VFS used in plot studies often cannot be directly applied to assess the effects of filter strips at the watershed scale. This is underscored by findings suggesting that the performance of VFS under on-farm conditions is rarely as effective as that for plot settings (McKergow et al. 2003; Blanco-Canqui et al. 2006; Verstraeten et al. 2006).

Nutrient concentrations and loads in surface runoff water vary among years and

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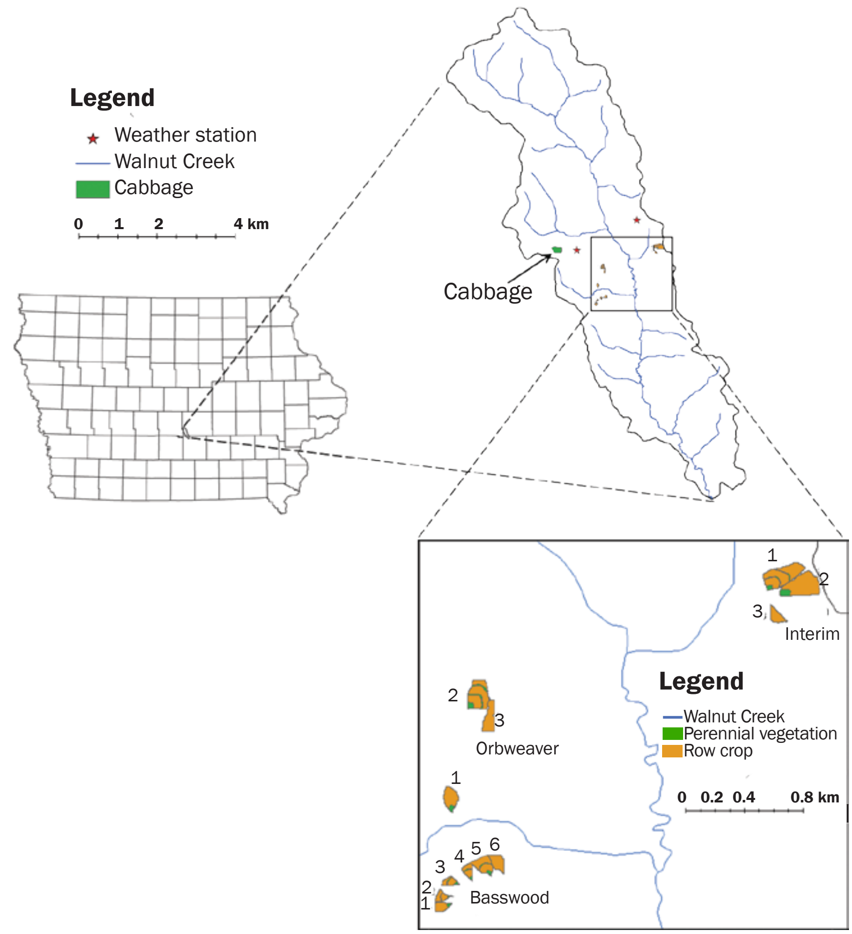
events, due to in-field management, changes in field conditions, and climatic variability. Nutrients can be transported along different hydrological pathways, and both the pathways and loads are further mediated by biogeochemical processes occurring across landscapes and under varying field conditions and precipitation patterns. All these processes, in turn, will affect the nutrient removal efficacy of VFS. The performance of VFS in contaminant removal greatly depends on the establishment of vegetation within the filter strip area and varies under different flow conditions. There is a need for long-term assessments of VFS (Duchemin and Hogue 2009; Helmers et al. 2012). Many VFS consist of single species and/or introduced species, and there may be added hydrologic/nutrient removal benefits by using prairie filter strips (PFS) rather than VFS (Isbell et al. 2011). The objective of the present study was to assess the effectiveness of PFS in reducing N and P loss in runoff from agricultural watersheds through a multiple-year study in central Iowa. Strategic placement of PFS in the landscape was also evaluated in regard to nutrient reduction.

Figure 1
Walnut Creek Watershed and the study watersheds. Prairie filter strips are numbered.



Materials and Methods

The study was conducted at the 3,000 ha (7,410 ac) Neal Smith National Wildlife Refuge (NSNWR) in central Iowa. Located at the central portion of the Walnut Creek Watershed, the NSNWR was established in 1990 to reconstruct native tall-grass prairie on the landscape (Schilling and Spooner 2006), with new prairie seedings added each year. Portions of the NSNWR awaiting restoration are either leased to farmers in the area for crop production or maintained in perennial cover. This study was implemented at three locations slated for future prairie reconstructions. A total of 12 small (zero-order) watersheds in the NSNWR were selected for this study with 6 watersheds at the Basswood site, 3 watersheds at the Interim site, and 3 watersheds at the Orbweaver site (figure 1). The size of the watersheds varied from 0.5 to 3.2 ha (1 to 7 ac), and the average slope of the watersheds ranged from 6.1% to 10.5% (table 1). The predominant soils in the study watersheds were Ladoga silt loam (fine, smectitic, mesic Mollic Hapludalf) and Otley silty clay loam (fine, smectitic, mesic Oxyaquic Argiudolls) (Helmers et al. 2012).

All study watersheds were under bromegrass for at least 10 years without fertilizer application prior to the start of this study in 2006. The watersheds were uniformly tilled with a mulch tiller in preparation for the study in fall of 2006 and spring of 2007. Starting in 2007, a filter strip study in agricultural landscapes was implemented in the 12 watersheds with a balanced incomplete block design across 4 blocks. Each watershed received one of four treatments (three replicates per treatment): 100% row crop (No-PFS), 10% of the watershed area in PFS at the footslope position (10% PFS), 10% of the watershed area in PFS distributed between the footslope position and in upslope contour strips (10% PFS with strips), and 20% of the watershed area in PFS distributed between the footslope position and in upslope contour strips (20% PFS with strips) (figure 2). In July of 2007, areas receiving PFS treatment were seeded with a diverse mixture of native prairie forbs and grasses, dominated by Indian grass (*Sorghastrum nutans* L.), little bluestem (*Schizachyrium scoparium* L.), and big bluestem (*Andropogon gerardii* L.). Prairie filter strip areas were established based on the size of the contributing area for each watershed. Multiple strips were established on contours in the larger watersheds, and the distance between strips was adjusted to accommodate local field equipment. The width of the PFS varied from 37 to 78 m (121 to 256 ft) at the footslope position and from 3 to 10 m (10 to 33 ft) on the contours (table 1). A two-year, no-tillage corn–soybean rotation (soybeans in 2007) was implemented in the areas receiving the row crop treatment. Anhydrous ammonia (NH_3) was applied on corn at a rate of 135 kg ha^{-1} (120 lb ac^{-1}) on April 24, 2008, and 185 kg ha^{-1} (165 lb ac^{-1}) on April 10, 2010. Phosphate (PO_4) was applied at a

Table 1
Site description and experimental design.

Site	Size (ha)	Slope (%)	Maximum slope length (m)	Location and percentage of PFS*	Width of PFS at footslope† (m)	Width of PFS at upslope‡ (m)
Basswood 1	0.53	7.5	120	10% at footslope	38.2	—
Basswood 2	0.48	6.6	113	5% at footslope and 5% at upslope	40.5	3.1
Basswood 3	0.47	6.4	110	10% at footslope and 10% at upslope	37.6	6.0
Basswood 4	0.55	8.2	118	10% at footslope and 10% at upslope	38.1	7.5
Basswood 5	1.24	8.9	144	5% at footslope and 5% at upslope	46.4	7.0
Basswood 6	0.84	10.5	140	All row crops	—	—
Interim 1	3.00	7.7	288	3.3% at footslope, 3.3% at sideslope, and 3.3% at upslope	51.0	6.0
Interim 2	3.19	6.1	284	10% at footslope	78.2	—
Interim 3	0.73	9.3	137	All row crops	—	—
Orbweaver 1	1.18	10.3	187	10% at footslope	57.3	—
Orbweaver 2	2.40	6.7	220	6.7% at footslope, 6.7% at sideslope, and 6.7% at upslope	52.0	9.8
Orbweaver 3	1.24	6.6	230	All row crops	—	—

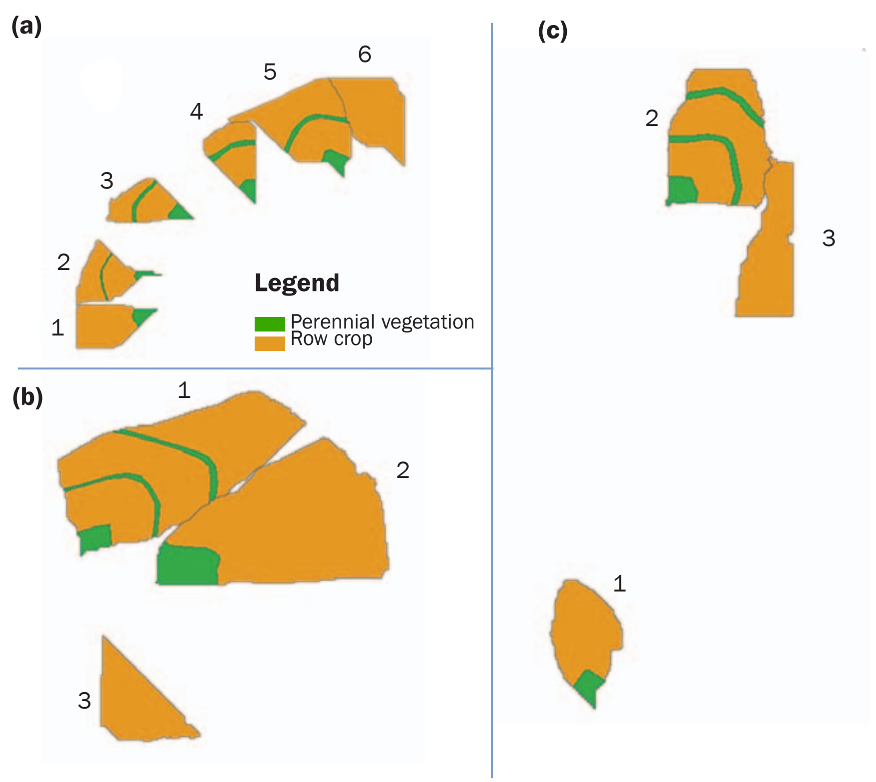
*Percentage of prairie filter strips (PFS) = area of PFS ÷ area of watershed.
†Width of PFS along the primary flow pathway.
‡Average width of PFS if more than one strip at upslope.

rate of 112 kg ha^{-1} (100 lb ac^{-1}) on May 13, 2008 and April 9, 2010. No fertilizer was applied to the PFS areas and during the soybean rotation.

A fiberglass H flume was installed at the bottom of each watershed in 2005 and early 2006 with the size designed for a 10-year, 24-hour return storm. A total of eight 0.61 m (2.00 ft) H-flumes and four 0.76 m (2.49 ft) H-flumes were installed in the 12 watersheds. Plywood wing walls (5 m [16 ft] at each side of a flume) were constructed at the bottom of the watersheds to guide surface runoff to the flumes. In 2007, ISCO 6,712 automated water samplers (ISCO Inc., Lincoln, Nebraska) equipped with pressure transducers (720 Submerged Probe Module) were installed at each flume to measure flow rate and collect water samples. Flow stage was measured by pressure transducers every five minutes to calculate the flow rate using the stage-discharge rating curve for that specific flume (Walkowiak 2006).

For water sampling during storm events, the ISCO autosampler took a 300 mL (10.6 oz) sample in a 1L (10.6 ft^3) bottle for every 1 mm (0.03 in) runoff. Up to three samples were placed in each bottle in sequential fashion. A maximum of 24 bottles could be filled during a single storm event, and no additional samples could be taken until the bottles were replaced. Typically, a total of 8 bottles could be filled for a 2-year storm and 14 to 15 bottles for a 10-year storm. Water samples were retrieved within 24 hours of a storm event and refrigerated at 4°C (39.2°F) until analysis. ISCO units were removed

Figure 2
Placement of prairie filter strips at (a) Basswood, (b) Interim, and (c) Orbweaver.



from the field during winter (late November to March) to avoid freeze damage.

Concentrations of nitrate-nitrogen (NO_3-N), total nitrogen (TN), and total phosphorus (TP) in runoff water were analyzed in the Agricultural and Biosystems Engineering Water Quality Research Laboratory at Iowa State University, Ames, Iowa. Nitrate-nitrogen concentration was determined with a Lachet Quickchem 2000 Automated Ion Analyzer flow injection system (Lachet Instruments, Milwaukee, Wisconsin) on water samples filtered through a 0.45 mm (0.018 in) cellulose-based filter (DS0210 membrane filter, Nalgene Labware, Rochester, New York). Total N and TP concentrations of unfiltered samples were determined from a HACH DR2800 spectrometer (Hach company, Loveland, Colorado) following a digestion with acid persulfate (Method 10208 for TN and Method 8190 for TP). The detection limit was 0.2, 0.5, and 0.02 mg L^{-1} (0.2, 0.5, and 0.02 ppm) for NO_3-N, TN, and TP, respectively. Nutrient loads (NO_3-N, TN, and TP) were then calculated based on the measured nutrient concentration for each sample and total flow volume for the period during which the sample was collected. Flow-weighted nutrient concentrations were calculated by dividing the total nutrient loss by the total flow volume of the period of interest (daily or annually).

Daily precipitation data during 2007 to 2011 was obtained by averaging the observed precipitation amount from the two nearby weather stations within the NSNWR; a Mesonet weather station operated by the National Weather Service and a weather station operated by the National Oceanic and Atmospheric Administration. Both weather stations are 1.1 to 3.6 km (0.68 to 2.24 mi) from the study watersheds.

The General Linear Model (GLM) procedures for SAS (SAS Institute 2003) were used for statistical analysis. To control variation by date and focus on the treatment effect, daily flow-weighted nutrient concentrations and loads of NO_3-N, TN, and TP were compared among treatments at the 5% significance level for each individual year between 2007 and 2011. Annual flow-weighted nutrient concentrations and loads were compared among treatments for the entire study period. Data from Orbweaver 1 (with 10% PFS at footslope) in 2010 was not included in the analysis due to the frequent failure of the ISCO unit in that year.

For additional analysis and assessment, two adjacent watersheds under 100% prairie reconstruction were also similarly gauged and sampled for runoff and nutrient transport. The two watersheds were within 3 km (1 mi) of the nearest study watersheds and were 4.2 and 5.1 ha (10.4 and 12.6 ac) in size (noted as Cabbage on figure 1). Native prairie was planted in 2004 in the two watersheds as described by Tomer et al. (2010), providing a 100% prairie restoration reference comparison to the 12 agricultural watersheds for the years of 2010 and 2011. These prairie watersheds were not part of the original experimental design and were sampled only from 2010.

Results and Discussion

Precipitation. Precipitation during the growing season (April to October) showed a high year-to-year variability within the study period (2007 to 2011), ranging from 719 to 1,221 mm (28.3 to 40.1 in) (table 2). Exceptionally high precipitation occurred in 2008 (966 mm [38 in]) and 2010 (1,221 mm [40.1 in]), both of which were well above the long-term normal of 713 mm (28.1 in). The precipitation amount in 2011 (719 mm [28.3 in]) was very close to the normal, while the precipitation in 2007 (838 mm [33 in]) and 2009 (811 mm [31.9 in]) were slightly greater than the normal. Precipitation also showed a high seasonal variability within a year. For example, year 2011 had 272 mm (10.7 in) of precipitation in June but only 185 mm (7.28 in) during the following four-month period (July to October). The total precipitation in June and August of 2010 alone was 710 mm (28 in), which was almost equal to the long-term normal for the entire growing season.

Runoff. The mean annual runoff during the growing season ranged from 32.4 mm (1.3 in) in 2007 to 347.6 mm (13.7 in) in 2010 for the study watersheds (table 3). Corresponding to the high precipitation in 2008 and 2010, a large amount of runoff was produced in these two years. A total of 208 mm (8.2 in) of runoff was observed for the storm event during August 8 to 11, 2010, the largest event for the study period. Among the different land use treatments, the No-PFS treatment had the highest runoff with a mean annual runoff of 206 mm (8.1 in). The PFS treatments reduced runoff to varying extents with the treatment of 10% PFS at footslope having the lowest runoff. The average annual runoff during the growing season was 82, 150, and 145 mm (3.2, 5.9, and 5.7 in) for the 10% PFS, 10% PFS with strips, and 20% PFS with strips, respectively, approximately a 60%, 27%, 29% reduction as compared with that for the No-PFS treatment. The greater PFS area at the footslope position for the 10% PFS treatment could lead to the greater runoff reduction than other PFS treatments (Hernandez-Santana et al. 2013). The runoff from the No-PFS treatment was significantly higher than that from most of PFS treatments since the second year of PFS establishment.

Temporal Change of Nutrient Export. Nutrient concentrations exhibited a wide range of year-to-year variation. Although 2008 had less precipitation than 2010, it had the highest nutrient concentrations for the five-year study period (table 4). The high concentrations might be attributed to the disturbance from the initial tillage of the bromegrass sod that occurred in 2006 and 2007 and the limited PFS cover in 2008. As a result, severe soil erosion occurred in 2008 with high sediment concentration in runoff water (Helmers et al. 2012).

Overall, the annual flow-weighted NO_3-N concentrations in the surface runoff were low during the study period, ranging from 0.13 to 2.43 mg L^{-1} (0.13 to 2.43 ppm). NO_3-N concentrations were relatively low in the years with low precipitation and were higher in the years with high precipitation. While NO_3-N concentrations were generally low and comparable during the study period, high NO_3-N concentrations occurred in the early growing season in the corn years (2008 and 2010) with concentrations greater than or close to 10 mg L^{-1} (figure 3). This same trend was not observed in the 2007, 2009, and 2011. It appears that the high mean annual NO_3-N concentration in 2008 and 2010 might not primarily be directly associated with the exceptionally high precipitation but is likely be attributed to the high NO_3-N concentration in the spring. High storm flows generally resulted in low NO_3-N concentrations (figure 3), which could be due to the dilution effect by precipitation (Mendez et al. 1999). The application of N fertilizers and wet field conditions likely contributed to the high NO_3-N concentration in the spring of the corn years (2008 and 2010). High groundwater table level during the spring could also facilitate the interaction between surface water and groundwater, which usually has higher NO_3-N concentrations. Nitrate-nitrogen concentrations in shallow groundwater at footslope were around 5 mg L^{-1} (5 ppm) in the spring in the No-PFS watersheds, significantly higher than NO_3-N concentrations in the PFS watersheds (Zhou et al. 2010). Return flows of shallow groundwater to the surface can dominate the late recession of a runoff hydrograph. Increased NO_3 concentrations were observed during late runoff hydrograph in an agricultural watershed in Iowa (Tomer et al. 2010).

The flow-weighted annual TN concentrations were highest in 2008 with about 23 mg L^{-1} (23 ppm) and similar to 2009 through 2011 with concentrations between 7.25 and 7.91 mg L^{-1} (7.25 and 7.91 ppm) (table 4). Similarly, total P concentrations were highest in 2008 and then decreased in the following years. The seasonal change of TN and TP concentrations was closely associated with the precipitation pattern, particularly during the early growth stage (figures 4 and 5). Phosphorus and N in surface runoff are often attached and transported with sediment, which is readily produced by soil erosion during storm events. Soils are more susceptible to runoff-induced erosion during the early growth stage of crop due to the poor ground cover. Approximately 80% of the total sediment export in 2009 was contributed from the first large storm event on April 25 to 27, 2009 (Helmers et al. 2012). It was unexpected that 2010 would have the highest precipitation but the lowest TP concentration. This could be in part because a great proportion of precipitation in 2010 occurred in August when the corn canopy was well developed thereby reducing water-induced erosion. The increased perennial cover in the PFS watersheds may also have attributed to the relatively low nutrient concentrations in 2010. Approximately 30%, 59%, and 94% of ground cover was established in PFS in the summer of 2008, 2009, and 2010, respectively (Hirsh and Liebman, *unpublished data*). Schultz et al. (1995) found that the amount of vegetative biomass produced in a grass buffer had a positive impact on nutrient removal.

Most N in soils is in the organic N form. The availability of soluble N in the upper soil profile, mostly as NO_3-N, depends on many factors including soil moisture, precip-

Table 2
Monthly precipitation during April through October in 2007 to 2011 and the long-term normal at the Neal Smith National Wildlife Refuge, Iowa.

Month	2007 (mm)	2008 (mm)	2009 (mm)	2010 (mm)	2011 (mm)	Normal (mm)
April	123.2	115.2	125.2	124.4	113.4	91.2
May	148.5	122.9	75.3	117.2	149.1	117.9
June	87.0	265.8	147.9	337.0	272.0	120.6
July	45.8	205.9	83.9	155.1	94.7	115.6
August	212.5	56.5	157.2	372.7	58.8	109.0
September	94.8	119.1	56.4	102.3	22.7	90.2
October	126.4	81.0	165.3	12.4	8.5	69.1
Total	838.2	966.2	811.1	1220.9	719.2	713.6

Table 3
Annual surface runoff during the growing season (April to October) for the treatments of without prairie filter strips (No-PFS) and with PFS. Letters after numbers indicate the significance test of mean difference among four treatments within each year at p <0.05.

Year	No-PFS (mm)	10% PFS (mm)	10% PFS with strips (mm)	20% PFS with strips (mm)	Mean (mm)
2007	39.6a	16.2a	32.6a	41.2a	32.4
2008	254.2a	61.1b	178.1ab	178.0ab	167.8
2009	129.0a	53.5b	96.5b	74.1b	88.3
2010	477.6a	224.8c	329.4bc	358.4b	347.6
2011	128.9a	53.9b	112.6b	74.1b	92.4
Average	205.9a	81.9b	149.8ab	145.2ab	

Table 4
Annual flow-weighted concentrations of nitrate-nitrogen (NO_3-N), total nitrogen (TN), and total phosphorus (TP) in surface runoff during the growing season (April to October) for the treatments of without prairie filter strips (No-PFS) and with PFS. Letters after numbers indicate the significance test of mean difference among four treatments within each year at p <0.05.

Year	No-PFS (mg L^{-1})	10% PFS (mg L^{-1})	10% PFS with strips (mg L^{-1})	20% PFS with strips (mg L^{-1})	Mean (mg L^{-1})
NO_3-N					
2007	0.51a	1.10a	1.34a	0.32a	0.82
2008	3.70a	3.15ab	2.04ab	0.82b	2.43
2009	0.16a	0.18a	0.16a	0.04a	0.13
2010	2.16a	1.76ab	1.05ab	0.95b	1.48
2011	0.80a	0.64a	0.48a	0.28a	0.55
Average	1.47a	1.37a	1.01a	0.48a	—
TN					
2007	3.87a	5.85a	3.99a	4.23a	4.48
2008	68.81a	10.64b	7.99b	4.71b	23.04
2009	17.46a	4.25b	5.05ab	2.24b	7.25
2010	12.87a	6.29b	6.36b	6.13b	7.91
2011	9.17a	8.05a	7.83a	6.31a	7.84
Average	22.44a	7.01b	6.25b	4.72b	—
TP					
2007	0.58a	0.64a	0.35a	0.40a	0.49
2008	15.59a	2.49b	2.08b	1.22b	5.35
2009	5.74a	1.00b	0.78b	0.53b	2.01
2010	1.58a	0.89b	0.50bc	0.43c	0.85
2011	2.36a	1.42a	1.02a	0.50a	1.33
Average	5.17a	1.29b	0.95b	0.62b	–

Figure 3
Daily runoff flow and nitrate-nitrogen (NO_3-N) concentration of water samples during the growing season (April to October) for the treatments of (a) without prairie filter strips (PFS) and (b) 20% PFS.

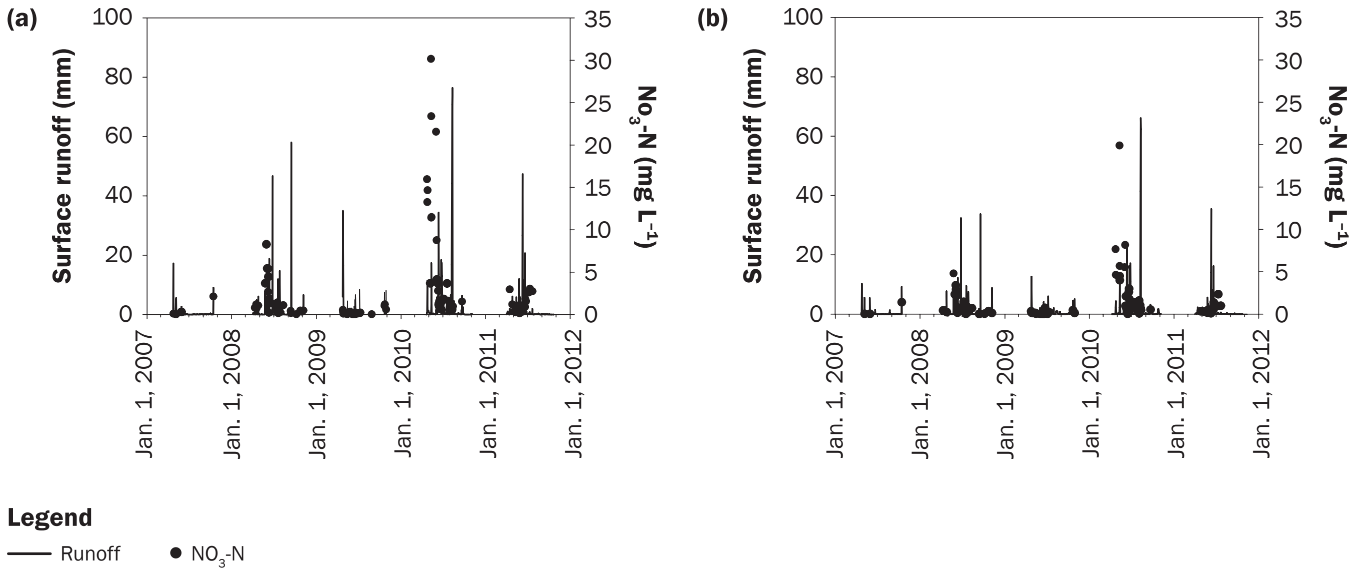
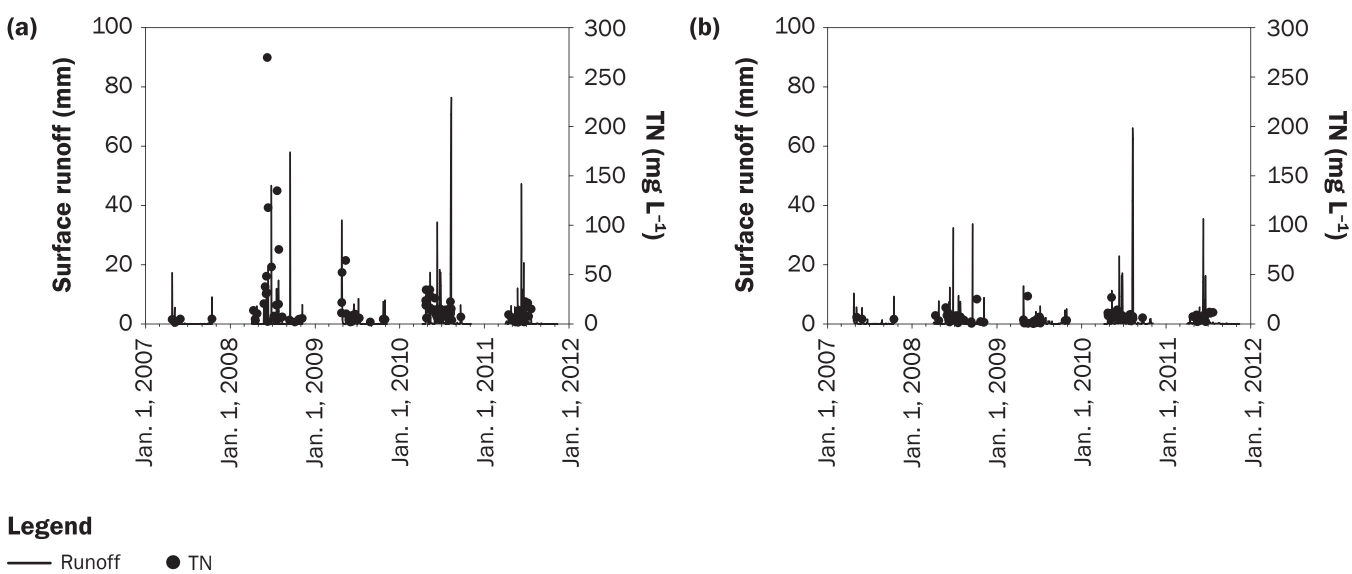


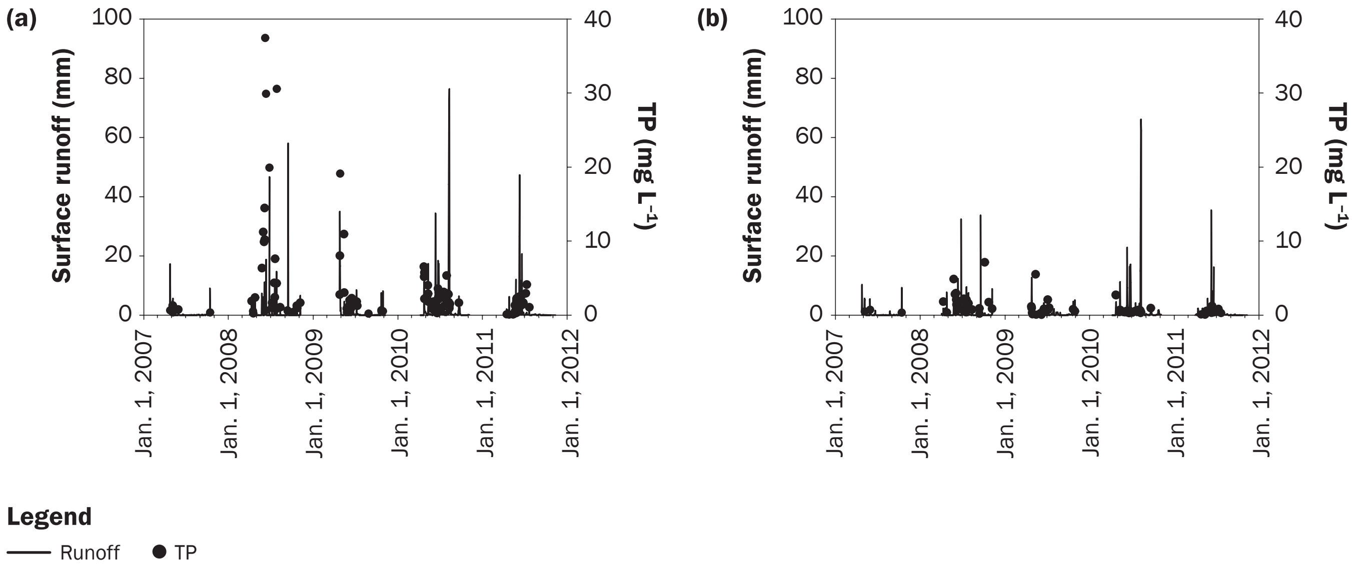
Figure 4
Daily runoff flow and total nitrogen (TN) concentration of water samples during the growing season (April to October) for the treatments of (a) without prairie filter strips (PFS) and (b) 20% PFS.



itation intensity and amount, depth of water table, temperature, and carbon (C) content (Mendez et al. 1999). Overall, the proportion of TN lost as NO_3-N tended to increase over the growing season in the soybean years but decrease over the season in the corn years (figure 6). During the early growing season of the corn years, most N in runoff was transported as NO_3 as a result of fertilization or surface/groundwater interaction. Approximately 90% of N in runoff was in NO_3-N form in the spring of 2010 (figure 6). The proportion of NO_3 in runoff dropped to a lower level at the late growth stage due to the lower soil NO_3 level resulting from leaching and plant uptake over the growing season and little surface/groundwater interaction. The mean proportion of NO_3-N to TN was 15% and 9% in corn and soybean years, respectively. In an N loss study from agricultural watersheds in northeastern Missouri, the mean proportion of NO_3-N to TN in runoff was found to be 63% and 43% in corn years and soybean years, respectively

Figure 5

Daily runoff flow and total phosphorus (TP) concentration of water samples during the growing season (April to October) for the treatments of (a) without prairie filter strips (PFS) and (b) 20% PFS.



(Udawatta et al. 2006). The large losses in the NO_3-N form were attributed to above normal precipitation conditions.

The annual nutrient losses for the study period were 0.13 to 4.38 kg ha^{-1} (0.12 to 3.90 lb ac^{-1}) for NO_3-N, 0.87 to 33.68 kg ha^{-1} (0.77 to 30.00 lb ac^{-1}) for TN, and 0.1 to 7.85 kg ha^{-1} (0.09 to 6.99 lb ac^{-1}) for TP (table 5). Figure 7 shows the annual cumulative nutrient loss for each treatment. As expected, high nutrient loss occurred in 2008 and 2010 because of the high precipitation and runoff amounts. Only a small amount of NO3 was transported with runoff in 2007, 2009, and 2011. The NO_3-N and TN losses dramatically increased in 2010 as compared with in 2009 due to the much greater runoff volume in 2010. Water pathway differences between years could be contributing to the difference in NO_3-N too. Saturated shallow conditions in 2010 can accentuate return flows that often have higher NO_3-N concentration (Tomer et al. 2010). The TP loss had only a slight increase in 2010. The largest NO_3-N loss occurred from the storm event of May 2010 (117 mm [4.6 in] precipitation) while the largest TN and TP occurred from the storm event of June 2008 (266 mm [10.5 in] precipitation) (figure 7), suggesting that NO_3-N loss was more dependent on the timing between storm event and antecedent subsurface moisture conditions but the loss of TN and TP was more associated with the water-induced erosion and sediment transport.

Figure 6

Proportion of nitrate-nitrogen (NO_3-N) to total nitrogen (TN) concentrations for each treatment.

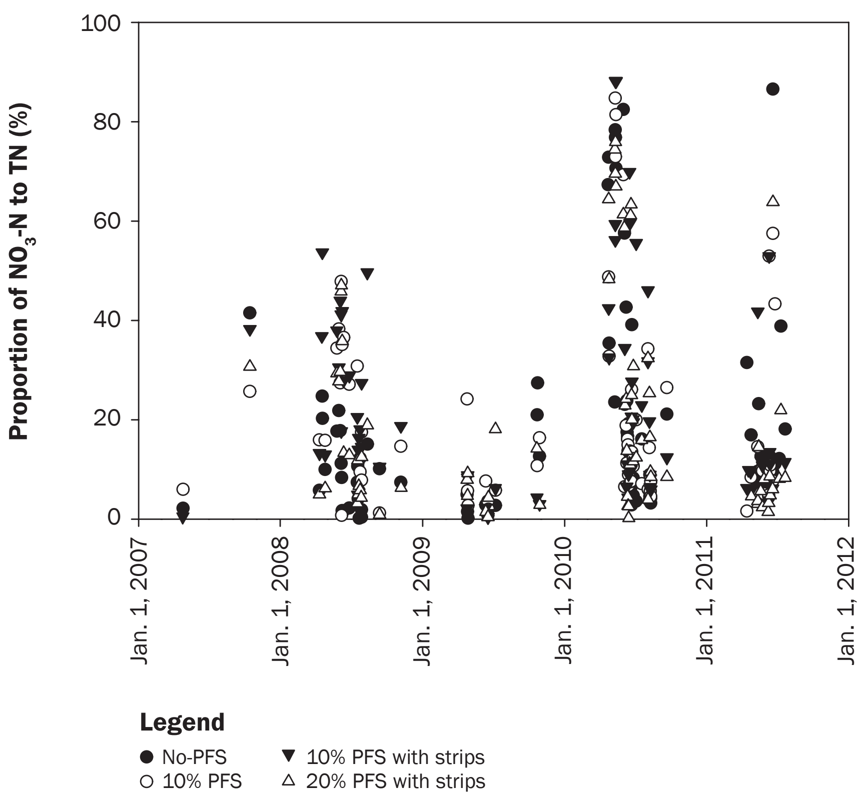


Table 5

Annual losses of nitrate-nitrogen (NO_3-N), total nitrogen (TN), and total phosphorus (TP) in surface runoff during the growing season (April to October) for the treatments of without prairie filter strips (No-PFS) and with PFS. Letters after numbers indicate the significance test of mean difference among four treatments within each year at $p < 0.05$.

Year	No-PFS (kg ha^{-1})	10% PFS (kg ha^{-1})	10% PFS with strips (kg ha^{-1})	20% PFS with strips (kg ha^{-1})	Mean (kg ha^{-1})
NO_3-N					
2007	0.21a	0.06a	0.15a	0.11a	0.13
2008	6.02a	2.16b	2.29ab	1.09b	2.89
2009	0.18a	0.05a	0.03a	0.04a	0.07
2010	8.72a	3.15b	2.66b	3.00b	4.38
2011	1.04a	0.42b	0.36b	0.21b	0.51
Average	3.24a	1.17ab	1.10b	0.89b	
TN					
2007	1.38a	0.39a	0.82a	0.91a	0.87
2008	112.58a	7.29b	8.70b	6.15b	33.68
2009	21.15a	1.39b	1.76b	1.39b	6.42
2010	48.47a	11.17b	17.79b	16.74b	23.54
2011	11.64a	5.06b	6.68b	5.19b	7.14
Average	39.04a	5.05b	7.04b	6.08b	
TP					
2007	0.20a	0.04a	0.08a	0.08a	0.10
2008	25.73a	1.72b	2.26b	1.69b	7.85
2009	7.06a	0.32b	0.31b	0.34b	2.01
2010	5.82a	1.58b	1.31b	1.26b	2.49
2011	2.43a	0.92ab	0.73ab	0.40b	1.12
Average	8.25a	0.92b	0.92b	0.75b	

Impact of Prairie Filter Strips on Nutrient Concentrations. The mean annual flow-weighted NO_3-N concentration was 1.47, 1.37, 1.01, and 0.48 mg L^{-1} for the No-PFS, 10% PFS, 10% PFS with strips, and 20% PFS with strips treatments, respectively (table 4). The NO_3-N concentrations were not significantly different among the treatments in most years and the entire study period, except the No-PFS treatment had significantly higher NO_3-N concentrations than the 20% PFS with strips treatment in 2008 and 2010, both of which had above normal precipitation. Overall, the annual TN concentration of the No-PFS treatment was significantly higher than that the PFS treatments, except no significant difference existed in 2007 and 2011. The mean annual TN concentration over the study period was also significantly higher in the No-PFS watersheds than in the PFS watersheds. The mean annual TN concentration was 22.44, 7.01, 6.25, and 4.72 mg L^{-1} (22.44, 7.01, 6.25, and 4.72 ppm) for the No-PFS, 10% PFS, 10% PFS with strips, and 20% PFS with strips, respectively (table 4). Similarly, No-PFS treatment had a significantly higher TP concentration in individual years of 2008 to 2010 than the PFS watersheds. The mean annual TP concentration was 5.17, 1.29, 0.95, and 0.62 mg L^{-1} (5.17, 1.29, 0.95, and 0.62 ppm) for the No-PFS, 10% PFS, 10% PFS with strips, and 20% PFS with strips treatments, respectively (table 4), with TP concentrations of the No-PFS treatment significantly higher than the other treatments.

Overall, the implementation of PFS in row crop fields reduced mean annual TN and TP concentration by 73% and 82%, respectively. The infiltration of runoff in PFS can facilitate the reduction of both soluble nutrients and sediment-bound nutrients. Previous work has shown that TN and TP transport from runoff is highly correlated with sediment detachment and transport (Lee et al. 2003). A 96% sediment load trapping efficiency was reported for the PFS of the study watersheds for a four-year (2007 to 2010) study period (Helmers et al. 2012). In contrast, the removal of the NO_3-N is primarily through infiltration and leaching (Parn et al. 2012), and as such, the removal efficiency of PFS is relatively less evident in surface runoff.

The daily nutrient concentrations of the No-PFS treatment were compared to those of the 20% PFS with strips treatment for the sampling events from 2007 to 2011, as shown in figures 3 through 5. The reduction of nutrient concentration was more evident during large storm events, especially in the spring. The effect of PFS on the reduction of TN and TP was more notable in 2008 and 2009 when high nutrient concentrations were observed. Compared to TN and TP, the reduction of NO_3-N in the watersheds with PFS was less obvious, except for the storm events in spring of 2010.

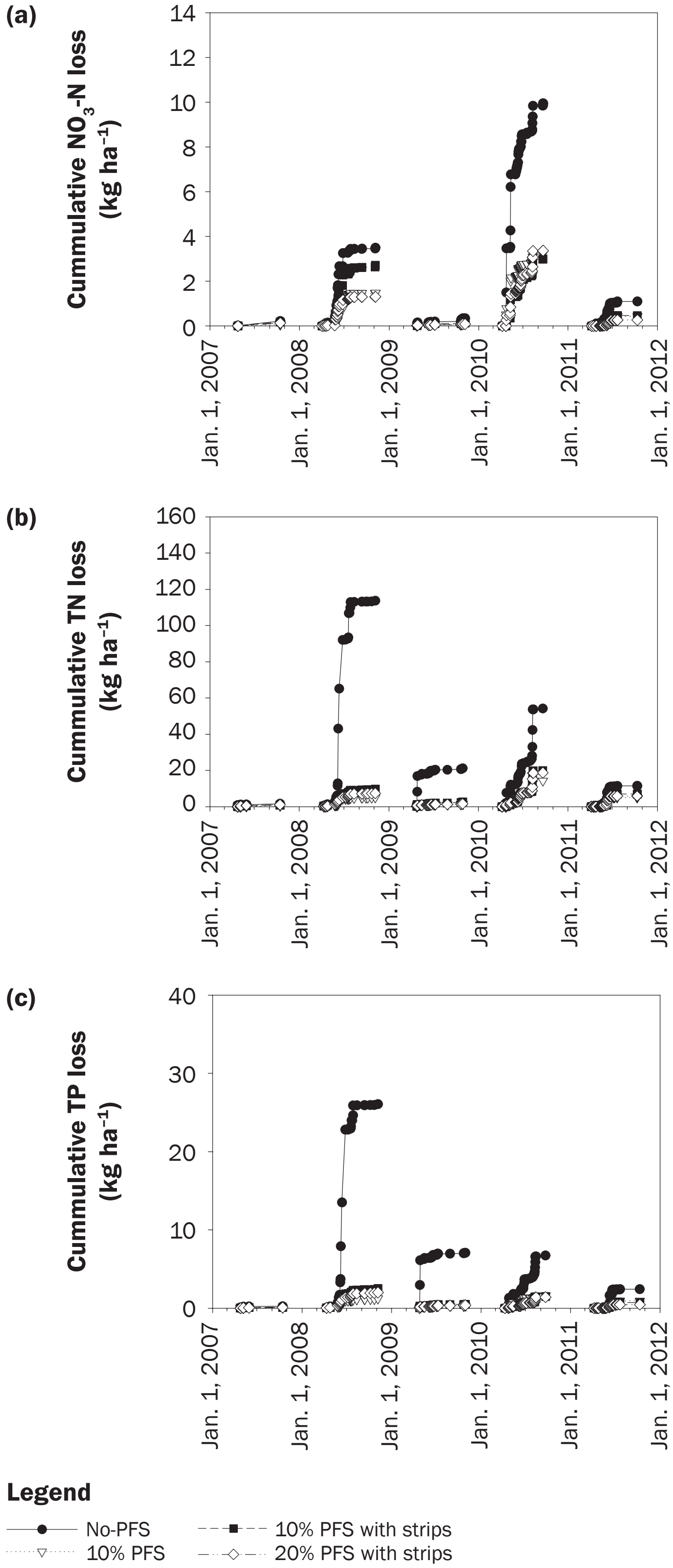
Impact of Prairie Filter Strips on Nutrient Losses. Prairie filter strips significantly reduced annual NO_3-N loss in 2008, 2010, and 2011 as well as mean annual loss for the study period. The mean annual NO_3-N loss was 3.24, 1.17, 1.10, and 0.89 kg ha^{-1} (2.89, 1.04, 0.98, 0.79 lb ac^{-1}) for the No-PFS, 10% PFS, 10% PFS with strips, and 20% PFS with strips treatments, respectively (table 5).

Similarly, the PFS treatments effectively reduced mean annual TN and TP losses, and annual losses during all years of the study period except for 2007. The mean annual TN loss was 39.04, 5.05, 7.04, and 6.08 kg ha^{-1} (34.77, 4.50, 6.27, and 5.42 lb ac^{-1}) for the No-PFS, 10% PFS, 10% PFS with strips, and 20% PFS with strips treatments, respectively; and the mean annual TP loss was 8.25, 0.92, 0.92, and 0.75 kg ha^{-1} (7.35, 0.82, 0.82, and 0.67 lb ac^{-1}) for the No-PFS, 10% PFS, 10% PFS with strips, and 20% PFS with strips treatments, respectively (table 5). The PFS treatments reduced NO_3-N loss by 69%, TN loss by 93%, and TP by 93% in the second year of the treatment period in this study. In a grass strip and agroforestry study, treatments did not reduce TN loss until the third year of treatments and slightly reduced NO_3-N and TP loss during the first two years of treatments (Udawatta et al. 2002).

Overall, the annual NO_3-N, TN, and TP loss in cropland runoff was reduced by 67%, 84%, and 90% in the PFS watersheds, respectively, as compared with those in the No-PFS treatment. The reduction of nutrient loss was more evident in years with greater flow associated with a number of large storms accounting for most of the total nutrient loss (table 5 and figure 7). The heavy nutrient losses in spring were likely accentuated by open and wet field conditions before and during early crop

Figure 7

Annually cumulative (a) nitrate-nitrogen (NO_3-N), (b) total nitrogen (TN), and (c) total phosphorus (TP) losses during the growing season (April to October) for the treatments of cropland and prairie filter strips (PFS).



development following N fertilizer application in April. It is encouraging that PFS can effectively reduce nutrient transport and loss from cropland during large storms even though the effectiveness of PFS might be reduced when concentrated flow occurs (Dosskey et al. 2002). During the storm event of June 12, 2008, (29.8 mm [1.2 in] precipitation and 13.3 mm [0.5 in] runoff), as an example, the PFS treatments reduced NO_3-N, TN, and TP losses by 43%, 96%, and 95%, respectively, as compared with the No-PFS treatment.

No significant difference was found in nutrient concentration or loss among the PFS treatments, which could be due to the wide PFS at all footslope positions. The width of PFS along the primary flow path at the footslope ranged between 37 to 78 m (121 to 256 ft) regardless of PFS treatment. Previous research revealed that much of particulate contaminants were trapped upslope or within the first meters of filter strips, resulting in almost the same effectiveness in trapping efficiency for filters with different widths (Mendez et al. 1999; Liu et al. 2008). The greater filter strip area might also facilitate greater uptake of nutrients by plants (Dosskey et al. 2010).

Nutrient Loss From Reconstructed Prairie Watersheds. Two adjacent reconstructed prairie watersheds provide a reference comparison in nutrient loss to the 12 agricultural watersheds. The annual runoff during the growing season for the reconstructed prairie watersheds was 152.5 and 46.9 mm (6.0 and 1.8 in) in 2010 and 2011, respectively (table 6), approximately 44% and 51% of the average runoff for the agricultural watersheds during the two years. Similarly, the annual loss of NO_3-N, TN, and TP was reduced by 98%, 85%, and 83% in 2010, respectively, and 87%, 90%, and 95% in 2011, respectively, compared to the No-PFS treatment. The low nutrient loss in the reconstructed prairie watersheds is likely because no fertilizers were applied in those prairie watersheds. Only a very small amount of NO_3-N was lost from surface runoff, being 0.14 and 0.13 kg ha^{-1} (0.12 and 0.12 lb ac^{-1}) in 2010 and 2011, respectively, due to the much greater total area of the prairie watersheds covered by perennial vegetation. Nitrogen in the form of NO_3-N would be readily taken up by the prairie plants that became well established during 2007 in the reconstructed prairie watersheds. In another

Table 6

Annual surface runoff and losses of nitrate-nitrogen (NO_3-N), total nitrogen (TN), total phosphorus (TP) and in surface runoff during the growing season (April to October) for the native prairie watersheds.

Year	Runoff (mm)	NO_3-N (kg ha^{-1})	TN (kg ha^{-1})	TP (kg ha^{-1})
2010	156.46	0.14	7.39	1.00
2011	46.99	0.13	1.21	0.13

study in these reconstructed prairie watersheds, nondetectable groundwater NO_3-N concentrations were observed along drainage ways after 2007 (Tomer et al. 2010). Compared to N, the reduction of P was less evident, which might be because P in runoff is often attached and transported with sediment particles. It has been found that the sediment export from the two native prairie watersheds was similar to that from the watersheds with PFS (Helmers et al. 2012). Controlled burns in April each year reduced vegetation cover before reemergence of the perennial plants. The lack of vegetation cover thereby contributes to appreciable P losses from the reconstructed prairie during storm events following prairie burns. From a groundwater nutrient study in these prairie watersheds, Tomer et al. (2010) found that groundwater P showed little temporal trend following prairie reconstruction.

Summary and Conclusions

The effectiveness of PFS in reducing nutrient concentration and export from cropland runoff was investigated in 12 small agricultural watersheds in central Iowa. The findings suggest that utilization of PFS at the footslope position of annual row crop systems provides an effective approach to reducing nutrient loss in runoff from agricultural watersheds. During the five-year study period, the annual flow-weighted concentrations of TN and TP in the watersheds with PFS were significantly reduced with mean reductions of 73% and 82%, respectively. The PFS treatments also reduced the NO_3-N, TN, and TP losses by 67%, 84%, and 90%, respectively. The results are encouraging considering the mean measured monthly precipitation during the study period exceeded normal monthly precipitation by over 25%. The effectiveness of PFS under high precipitation is meaningful under global climate change during which extreme events are expected to occur more frequently (Core Writing Team 2007). Utilization of PFS in agricultural landscapes can be a part of integrated solutions to the hypoxia problem.

Generally, heavy NO_3-N loss tended to occur in the spring while the export of TN and TP was closely associated with the sediment transport during large storms. That transport was significant despite the fact that all 12 watersheds were managed in no-tillage and that several had been under cool season grass cover prior to establishment of the experiment. Under large precipitation events, no-tillage alone may not provide enough protection from off-site nutrient losses. For the same study watersheds, sediment under no-tillage alone was reduced by 96% with PFS. Much of the nutrient loss was attributed to a few large events, and the PFS were found to be effective in removing nutrients from cropland runoff during the large events. The different PFS treatments (amount and distribution of PFS) showed no significant differences in their effectiveness for nutrient removal. From the practical point of view, converting 10% of agricultural cropping system to PFS at the bottom of a watershed would be more convenient for field operations while still effectively controlling off-site runoff and nutrient losses from the cropped area. However, the cropped soils in upslope areas are susceptible to soil and nutrient loss; their long term productivity and the ecological dynamics/biological conservation benefits are likely to be adversely affected. The distribution of contour strips could reduce the spatial scale of net effects of erosion in upslope areas and be beneficial for long term site productivity as well as provide other essential ecosystem services such as increasing biodiversity, wildlife habitat and a refuge for beneficial insects.

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