

Perennial Filter Strips Reduce Nitrate Levels in Soil and Shallow Groundwater after Grassland-to-Cropland Conversion

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Many croplands planted to perennial grasses under the Conservation Reserve Program are being returned to crop production, and with potential consequences for water quality. The objective of this study was to quantify the impact of grassland-to-cropland conversion on nitrate-nitrogen ($\text{NO}_3\text{-N}$) concentrations in soil and shallow groundwater and to assess the potential for perennial filter strips (PFS) to mitigate increases in $\text{NO}_3\text{-N}$ levels. The study, conducted at the Neal Smith National Wildlife Refuge (NSNWR) in central Iowa, consisted of a balanced incomplete block design with 12 watersheds and four watershed-scale treatments having different proportions and topographic positions of PFS planted in native prairie grasses: 100% rowcrop, 10% PFS (toeslope position), 10% PFS (distributed on toe and as contour strips), and 20% PFS (distributed on toe and as contour strips). All treatments were established in fall 2006 on watersheds that were under brome grass (*Bromus* L.) cover for at least 10 yr. Nonperennial areas were maintained under a no-till 2-yr corn (*Zea mays* L.)–soybean [*Glycine max.* (L.) Merr.] rotation since spring 2007. Suction lysimeter and shallow groundwater wells located at upslope and toeslope positions were sampled monthly during the growing season to determine $\text{NO}_3\text{-N}$ concentration from 2005 to 2008. The results indicated significant increases in $\text{NO}_3\text{-N}$ concentration in soil and groundwater following grassland-to-cropland conversion. Nitrate-nitrogen levels in the vadose zone and groundwater under PFS were lower compared with 100% cropland, with the most significant differences occurring at the toeslope position. During the years following conversion, PFS mitigated increases in subsurface nitrate, but long-term monitoring is needed to observe and understand the full response to land-use conversion.

DURING THE PAST 150 yr, much of the native tallgrass prairie vegetation of the central United States has been converted to intensive production of annual rowcrops, particularly corn (*Zea mays* L.) and soybean [*Glycine max.* (L.) Merr.]. Iowa, located in the heart of the Corn Belt region, currently has <1% of the original extent of its prairie vegetation remaining on the landscape (Samson and Knopf, 1994; Noss et al., 1995). Although government programs such as the Conservation Reserve Program (CRP) have led to increases in perennial vegetation that mitigate the negative effects of agriculture, especially on marginal lands, this trend has been reversed in recent years, as millions of hectares of CRP land have been converted to rowcrop to meet the increasing demand for food and energy (Hart, 2006; Secchi et al., 2008).

Compared with cropland, perennial grassland can have beneficial effects on maintaining ecosystem processes and functions that enhance ecosystem services, including water quality and hydrologic regulation (Wedin and Fales, 2009). For example, by providing a diverse and dense cover of plants with deep roots, grasslands can alleviate peak flows and mitigate flooding by modifying key ecohydrological processes, such as increasing evapotranspiration, promoting greater infiltration rates and soil water storage capacity, and reducing surface and subsurface runoff (Eynard et al., 2005; Gerla, 2007).

Another consequence of grassland-to-cropland conversion is the deterioration of water quality. Nonpoint source (NPS) pollution, particularly nitrate, has led to extensive impairment of water bodies in the U.S. Corn Belt region and has been identified as a significant contributor to hypoxia in the Gulf of Mexico (Alexander et al., 2008). Due to its high mobility, nitrate leaching to groundwater and subsurface drainage has also been a major cause of declining water quality in the midwestern United States, where most streamflow originates from groundwater (Schilling, 2005). One strategy to reduce NPS export from agricultural lands has been the restoration or reconstruction of native tallgrass prairie on relatively large scales. For example, reduction in stream nitrate

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Abbreviations: CRP, Conservation Reserve Program; NPS, nonpoint source; NSNWR, Neal Smith National Wildlife Refuge; PFS, perennial filter strips.

levels was detected at the watershed scale in response to reconstruction of approximately one-third of the watershed to prairie vegetation (Schilling and Spooner, 2006). However, the application of such strategies on large landscape scales is not feasible due to social and economic trade-offs and the increasing societal demand for production of food, feed, fiber, and fuel.

As an alternative to landscape-scale restoration, among the most prominent and promising strategies to mitigate negative effects of rowcrop production on water quality is the incorporation of relatively small amounts of perennial cover in strategic locations within agricultural landscapes (Dosskey et al., 2002; Blanco-Canqui et al., 2006). For example, both perennial filter strips (PFS) and riparian buffers were shown to reduce erosion and loss of nutrients and sediment from agricultural lands into streams by acting as a physical barrier (Barling and Moore, 1994; Helmers et al., 2005). Research has also documented the ability of PFS to reduce $\text{NO}_3\text{-N}$ concentrations in surface runoff and/or groundwater (Lin et al., 2007; Yamada et al., 2007; Ryder and Fares, 2008). Nitrogen can be removed from soil water and groundwater by PFS as a result of either plant uptake or by conversion of nitrate to nitrogen gas via denitrification by microorganisms, a process that is enhanced by addition of organic matter or by incorporation into microbial biomass from PFS (Lowrance and Hubbard, 2001). Multiple factors can affect the degree of reduction in $\text{NO}_3\text{-N}$ concentration, including nitrogen loading, type of vegetation, width of filter strips, and site conditions.

Despite their considerable potential benefits, PFS remain underutilized in the midwestern United States, in part due to a lack of reliable information about their effects on spatial and temporal fluxes in water and nutrients at the watershed scale (Lovell and Sullivan, 2006). Most previous studies were conducted at the plot scale, and the few that have been conducted at the watershed scale lack sufficient replication to draw reliable conclusions (Hickey and Doran, 2004). Further, some observations suggest that a substantial lag time may occur in the response of $\text{NO}_3\text{-N}$ levels in groundwater to agricultural management due to relatively slow groundwater transport in certain landscapes (STAC, 2005; Newbold et al., 2010). For example, groundwater $\text{NO}_3\text{-N}$ levels of a watershed in Iowa's Loess Hills were still influenced by large amounts of fertilizer N experimentally applied more than 30 yr earlier (Tomer and Burkart, 2003). The lag time between changes of N levels in groundwater and changes in a specific agricultural management depends on many factors, including the scale of the monitored area, depth to saturated zone, and meteorological conditions during the monitoring period (Meals et al., 2009).

Given the current trends of converting CRP and other perennial vegetation back to rowcrop agriculture in the Midwest combined with growing concern over water quality issues (Hart, 2006; Secchi et al., 2008), there is a critical need for improved understanding of how grassland-to-rowcrop conversion affects nutrient fluxes, particularly $\text{NO}_3\text{-N}$, through the soil water-groundwater system, as well as the potential for using strategically placed perennial vegetation in rowcrop systems to mitigate nutrient losses from agricultural lands. Thus, the main objective of this study was twofold: (i) to quantify changes in $\text{NO}_3\text{-N}$ concentrations in the vadose zone and

shallow groundwater in the period closely following grassland-to-cropland conversion, and (ii) to evaluate the ability of PFS within rowcrop agriculture to reduce $\text{NO}_3\text{-N}$ concentrations in the vadose zone and shallow groundwater in the period following grassland-to-cropland conversion and in response to spatial positioning of PFS. Specifically, we hypothesized that land-use conversion from perennial grassland to rowcrops would result in an increase in $\text{NO}_3\text{-N}$ concentrations in the vadose zone and shallow groundwater followed by a decline with time in areas with the PFS, as they became fully established.

Materials and Methods

Site Description

The study was conducted at the Neal Smith National Wildlife Refuge (NSNWR; 41°33' N; 93°16' W), a 3000-ha area managed by the U.S. National Fish and Wildlife Service, located in the Walnut Creek watershed in Jasper County, Iowa. Created by an act of Congress in 1990, the refuge's mission is to reconstruct the presettlement vegetation on the landscape, particularly native tallgrass prairie. Portions of the refuge awaiting restoration are either leased to area farmers for crop production or maintained in perennial pasture.

The NSNWR comprises part of the southern Iowa drift plain (Major Land Resource Area 108C) (USDA Natural Resources Conservation Service, 2006), which consists of steep rolling hills of Wisconsin-age loess on pre-Illinoian till (Prior, 1991). Walnut Creek is a third-order stream that drains into the Des Moines River at the upper end of the Red Rock Reservoir (Fig. 1). The watershed is well dissected by streams and ephemeral drainage ways, and its terrain is moderately to steeply rolling. Most soils at the research sites are classified as Ladoga (Mollic Hapludalf) or Otley (Oxyaquic Argiudolls) soil series with 5 to 14% slopes and are highly erodible (Nestrud and Worster, 1979; Soil Survey Staff, 2003). The mean annual precipitation over the last 30 yr is 850 mm, with most large storms occurring between May and July. For this site, daily precipitation was obtained from the National Ocean and Atmospheric Administration station at the NSNWR.

Experimental Design

The study was implemented using a balanced incomplete block design with 12 small, zero-order (intermittent in hydrological outflow) watersheds distributed across four blocks. Two blocks are located at Basswood (six watersheds), one block at Interim (three watersheds), and one block at Orbweaver (three watersheds) sites (Fig. 2). The size of the watersheds varied from 0.5 to 3.2 ha, with average slopes ranging from 6.1 to 10.5% (Table 1). Each watershed received one of four treatments (three replicates per treatment): 100% rowcrop (control condition), 10% PFS at the toeslope position, 10% PFS distributed at the toeslope position and in contour strips further up in the watershed, and 20% PFS distributed at the toeslope position and in contour strips further up in the watershed (Table 1). Treatments were randomly assigned to watersheds within each block. Multiple strips were established in the larger watersheds that were treated with upslope strips, leading to 3.3% (Interim-1) or 6.7% (Orbweaver-2) distributed within toeslope, sideslope, and upslope positions. The width of PFS varied

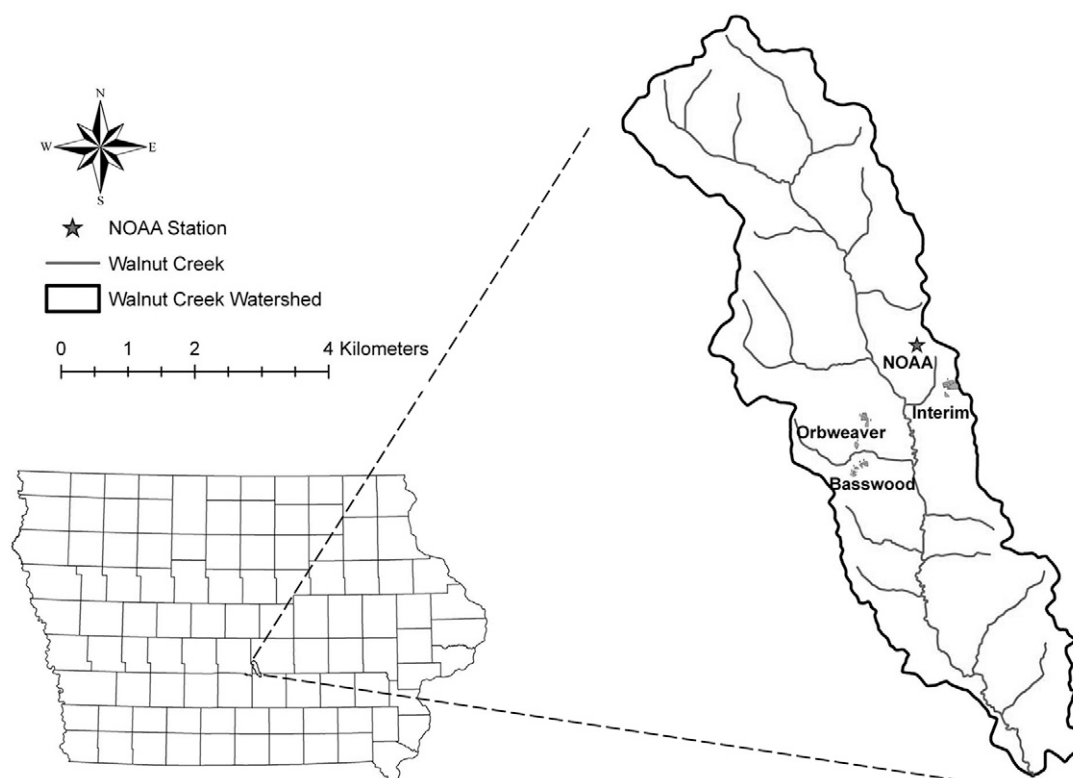


Fig. 1. Location of Walnut Creek Watershed and study watersheds.

from 27 to 41 m at toeslope, and 5 to 10 m at upslope and sideslope. In this study, only the presence of PFS at each watershed position was considered as a treatment, while the amount of PFS was not considered as a treatment since we did not see a significant effect of percentage PFS.

Before treatment, all the watersheds were in brome grass (*Bromus* L.) for at least 10 yr without fertilizer application. Pretreatment data were collected in 2005 and the first half of 2006. In August 2006, all watersheds were uniformly tilled with a mulch tiller. Basswood-1-6 and Orbweaver-1 were tilled again in spring 2007 to further level field residue. Starting in spring 2007, a 2-yr no-till corn-soybean rotation (soybean in 2007) was implemented in areas receiving the rowcrop treatment. Standard herbicide- and fertilizer-based weed and nutrient management practices were applied at each watershed. Anhydrous ammonia was knifed into the field at a rate of 134.4 kg N ha⁻¹ on 24 Apr. 2008, and monoammonium phosphate (MAP 11-52-0) at a rate of 112 kg P₂O₅ ha⁻¹ was applied on 13 May 2008. Areas receiving PFS treatment were seeded with a diverse mixture of native prairie forbs and grasses using a broadcast seeder on 7 July 2007. A total of over 20 species with the four primary species was in the mix, including indiagrass (*Sorghastrum* Nash), little bluestem (*Schizachyrium* Nees), big bluestem (*Andropogon gerardii* Vitman), and aster (*Aster* L.). This method of seeding is consistent with methods used for other prairie reconstruction at the NSNWR. No fertilizer was applied in the PFS areas.

Soil Background Information

Soil core samples were collected in 2004 along two transects in each watershed at the upslope and toeslope positions to

establish the pretreatment baseline conditions ($n = 2$). At each sampling location, a 30-cm soil core was collected using a 6-cm-diameter hand probe and then divided into 0- to 5-, 5- to 15-, and 15- to 30-cm depth increments. Soil bulk density was determined by oven drying at 105°C (Blake and Hartge, 1986). The pipette method was used for the particle size analysis (Gee and Bauder, 1986). The 2-mm sieved samples were used to determine the total C and N by direct combustion with a TruSpec CHN Analyzer (LECO, St. Joseph, MI).

Suction Lysimeters

To measure NO₃-N concentrations in the vadose zone, porous cup suction lysimeters (Model 1920F1L24, Soilmoisture Equipment Corp., Santa Barbara, CA) were installed at the upslope and toeslope positions of each watershed in November 2004. For watersheds receiving treatments with perennial strips, lysimeters were always located within the perennial strip at the toeslope position; however, their position in upslope contour positions varied with respect to the PFS (either within the PFS or outside the PFS under crops). At each sampling location, two lysimeters were installed at a depth of 1 m with a 4-m spacing. To minimize preferential flow of water, a 5-cm-diameter auger was used to drill a hole at a 45° angle into the soil profile via a narrow access trench. Silica flour slurry was poured into the bottom of the cored hole to ensure good soil contact with the porous ceramic cup. Access tubes were attached to each lysimeter for extracting water samples. The trench was then backfilled with native soil. A negative tension (-55 kPa) was applied to each lysimeter by a hand vacuum pump. Water samples were collected monthly between April and October starting in 2005. In 2005–2006, the composite

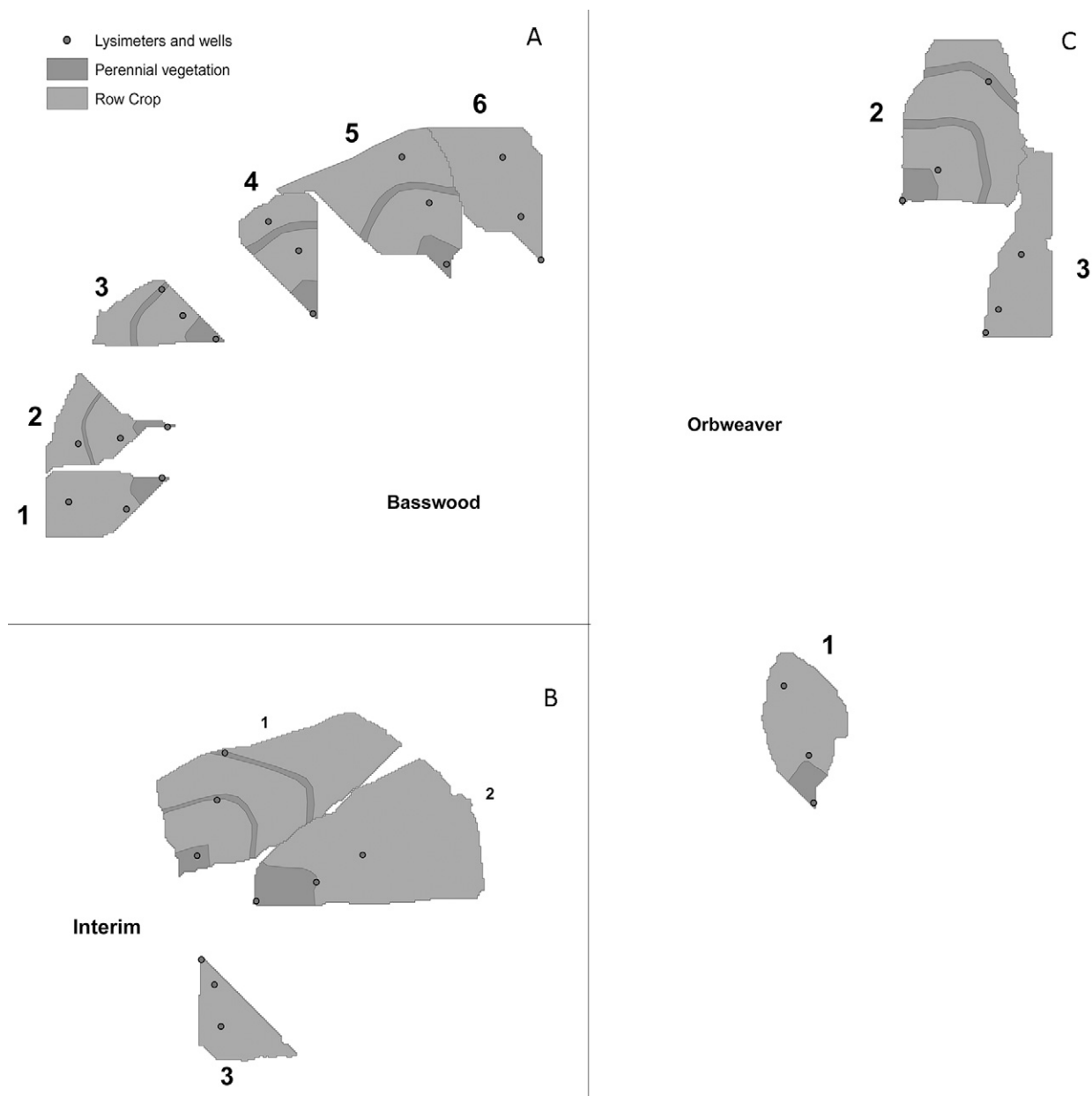


Fig. 2. Experimental design of vegetative filters for the study watersheds at (a) Basswood, (b) Interim, and (c) Orbweaver.

Table 1. Watershed description and experimental design.

Watershed	Size	Slope	Location and percentage of grass filter†
	ha	%	
Basswood-1	0.53	7.5	10% at toeslope
Basswood-2	0.48	6.6	5% at toeslope and 5% at upslope
Basswood-3	0.47	6.4	10% at toeslope and 10% upslope
Basswood-4	0.55	8.2	10% at toeslope and 10% upslope
Basswood-5	1.24	8.9	5% at toeslope and 5% upslope
Basswood-6	0.84	10.5	All rowcrops
Interim-1	3.00	7.7	3.3% at toeslope, 3.3% at sideslope, and 3.3% at upslope
Interim-2	3.19	6.1	10% at toeslope
Interim-3	0.73	9.3	All rowcrops
Orbweaver-1	1.18	10.3	10% at toeslope
Orbweaver-2	2.40	6.7	6.7% at toeslope, 6.7% at sideslope, and 6.7% at upslope
Orbweaver-3	1.24	6.6	All rowcrops

† Percentage of grass filters = area of filters/area of watershed.

samples from each pair of lysimeters were frozen until analyzed. In 2007–2008, samples were filtered through a 0.45- μm cellulose-based filter (DS0210 membrane filter, Nalgene Labware, Rochester, NY) in the laboratory immediately after collection and then refrigerated in the laboratory at 4°C before analysis. Nitrate-nitrogen concentrations in samples were determined on a Quickchem 2000 Automated Ion Analyzer flow injection system with a 0.2 mg L⁻¹ detection limit (Lachat Instruments, Milwaukee, WI).

Shallow Groundwater Wells

At each watershed, shallow groundwater wells were installed at the upslope and toeslope positions in November 2004. Wells were constructed of 50 mm i.d. polyvinyl chloride with 0.6-m screens. The depths of the wells varied between 2.9 and 5.4 m. Bentonite grout was used to seal the holes on the land surface around the wells to prevent surface water from directly entering groundwater. Groundwater samples were extracted from the wells monthly between April and October starting from 2006 because of the snow season from November to March in Iowa. The water samples were filtered and analyzed for NO₃-N concentrations following the same procedures as the lysimeter samples (above). Depth of shallow groundwater from the surface was measured monthly using a submersible level transmitter (Keller America, Inc., Newport News, VA).

Statistical Analyses

Analysis of variance was performed using the General Linear Model (GLM) procedure of SAS (SAS Institute, 2001) to test for significant differences in NO₃-N between experimental treatments (PFS vs. cropland) and watershed position. Because of the similarity in landscape, soil formation (Table 2), and management history among the watersheds, watersheds receiving the same treatment were regarded as randomized replicates. In the statistical analysis, only lysimeters and groundwater wells located completely within or immediately downslope of the PFS strips in contour positions were considered as at the “upslope PFS” position. A total of three watersheds had upslope PFS lysimeters and groundwater wells (Basswood-3, Interim-1, and Orbweaver-2), whereas the upslope lysimeters and groundwater wells in the other nine watersheds were on cropland. For the toeslope position, nine watersheds had lysimeters and groundwater wells installed within PFS, whereas the three 100% rowcrop watersheds had cropland lysimeters and wells (Fig. 2).

Results

Precipitation and Groundwater Levels

Annual precipitation in the NSNWR was highly variable during the study period (Fig. 3), ranging from near the annual mean for the region (850 mm) in 2006 (835 mm), to substantially greater than the annual mean for both 2007 (1053 mm) and 2008 (1169 mm). Spring 2008 was unusually wet, with a total precipitation of 700 mm between April and July, well above the annual mean (425 mm) during this period.

Shallow groundwater level was relatively low in 2006 but much higher during 2007 and 2008 (Fig. 3). In general, groundwater levels were about 1 to 2 m below the surface at toeslope positions and were generally deeper (3–5 m) and exhibited greater temporal variation at the upslope positions. During the period of excessive rainfall in spring 2008, the saturated zone was near the soil surface and was observed within approximately 0.2 m of the surface even at the upslope locations.

NO₃-N Concentration in the Vadose Zone

Before installation of the treatments in the period extending from fall 2006 through spring 2007, NO₃-N concentrations in the vadose zone were very low (close to zero) in all watersheds and topographical positions and remained low during the first year following treatment (Fig. 4). Nitrate-nitrogen concentrations increased at both the upslope and toeslope positions from April 2007 through May 2008 under both cropland and PFS. The NO₃-N concentrations in the vadose zone beneath the PFS then dropped to a low level in subsequent months. In contrast, the NO₃-N concentrations remained relatively high during most of 2008 in the row-cropped areas especially for the toeslope position (Fig. 4).

For the upslope position, NO₃-N concentrations generally decreased in the vadose zone beneath the cropland after the initial peak in April during the growing season of 2008, ranging from a maximum of 8.5 mg L⁻¹ in April and then dropping to 0.4 mg L⁻¹ in September (Fig. 4a). This range was similar to that exhibited under PFS (11.1–0.1 mg L⁻¹), except that the reduction in NO₃-N occurred more quickly and earlier in the season. Conversely, at the toeslope position, NO₃-N concentrations under the cropland remained high (3.1–10.6 mg L⁻¹) during the entire growing season (Fig. 4b), with NO₃-N concentrations at their lowest level during the early growing season (April–June) and then increasing thereafter. Nitrate-nitrogen concentrations under PFS in the toeslope peaked in April at 2.9 mg L⁻¹ and declined to almost zero by July.

Table 2. Soil characteristics (0–30 cm in depth) of the sampling sites.

	Basswood		Interim		Orbweaver	
	Upslope	Toeslope	Upslope	Toeslope	Upslope	Toeslope
Sand (%)	2.54	16.81	3.75	10.52	2.26	12.99
Silt (%)	28.58	25.76	26.38	23.47	30.85	25.79
Clay (%)	68.88	57.43	69.87	66.01	66.89	61.22
C (g kg ⁻¹ soil)	1929.70	1683.23	2080.71	2367.13	1644.45	2060.61
N (g kg ⁻¹ soil)	172.84	145.69	206.41	145.90	142.87	162.24
C:N	11.16	11.55	10.04	16.22	11.52	12.70
Bulk density (g cm ⁻³)	1.37	1.44	1.42	1.37	1.39	1.43

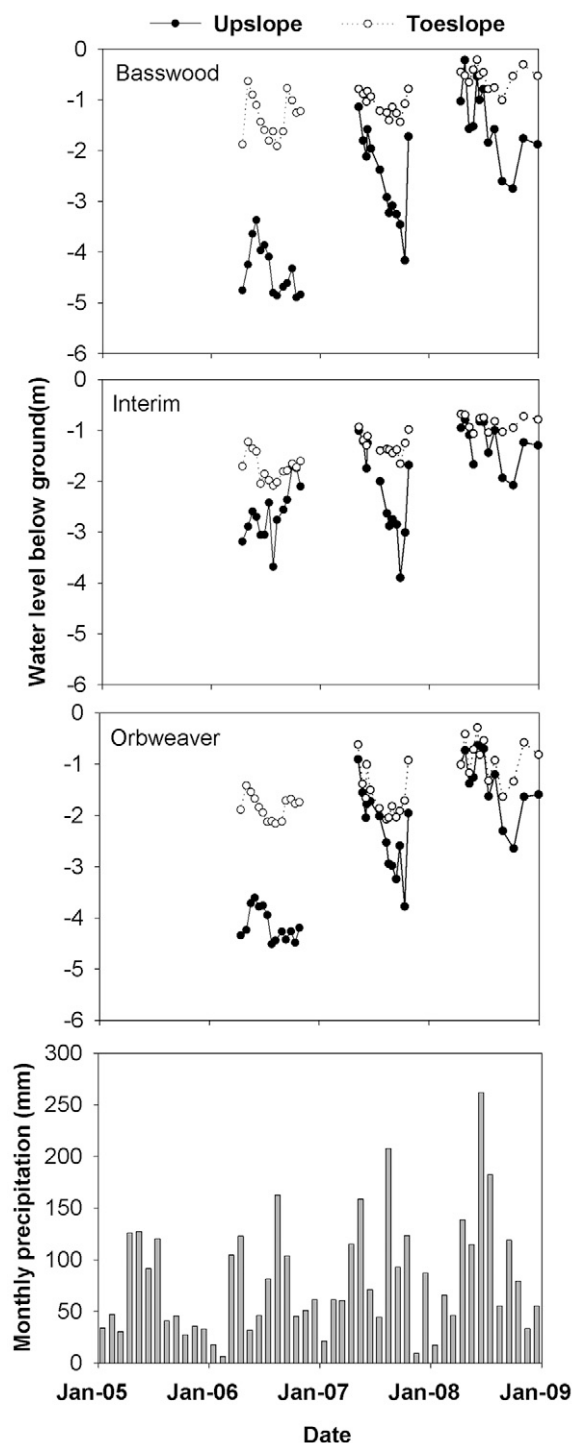


Fig. 3. Depth of shallow groundwater level below ground at the upslope and toeslope positions for three study sites and monthly precipitation in the Walnut Creek Watershed.

Comparing between years, $\text{NO}_3\text{-N}$ concentrations under cropland in the vadose zone in 2008 were significantly higher than during the pretreatment years (2005 and 2006) and the first post-treatment year (2007). Nitrate-nitrogen concentrations under PFS in the fall 2007 and spring 2008 were higher than the previous years but then decreased to a low level. Comparing $\text{NO}_3\text{-N}$ concentrations in the vadose zone between cropland and PFS, for the upslope position, concentrations were only significantly different for the month of July

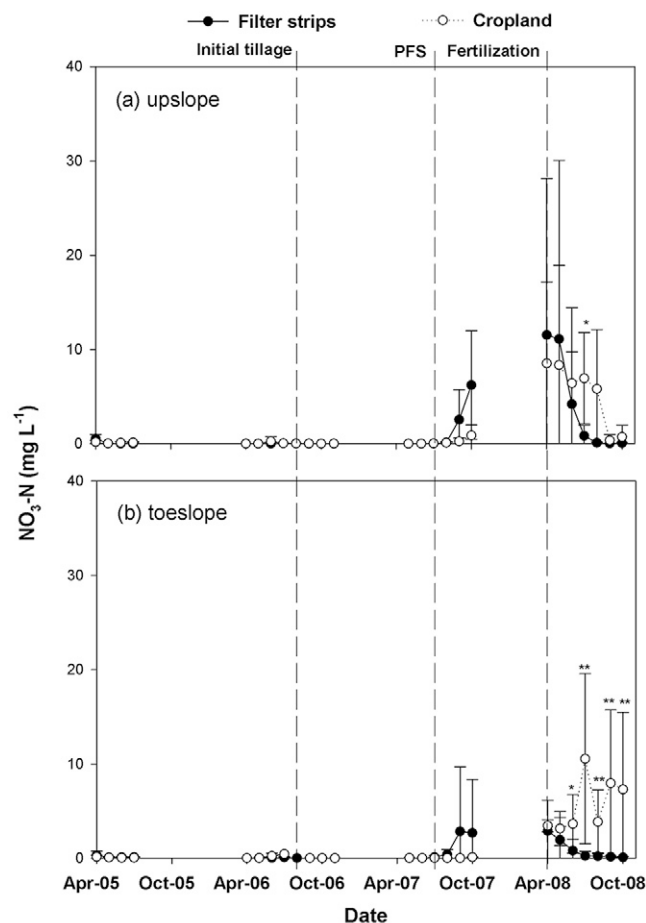


Fig. 4. $\text{NO}_3\text{-N}$ concentrations in lysimeters at the (a) upslope and (b) toeslope positions. Error bars denote the standard deviation of the replicates. Statistical difference of mean nitrate concentration between treatments (grass filters vs. cropland) was indicated for each monitoring period using two significant levels (** $p < 0.05$, * $p < 0.1$). PFS, perennial filter strips.

2008 (Fig. 4a). In contrast, for the toeslope position, the cropland had consistently and significantly higher $\text{NO}_3\text{-N}$ concentrations compared with PFS from June through October in 2008 (Fig. 4b).

$\text{NO}_3\text{-N}$ Concentration in Shallow Groundwater

Before treatment implementation, $\text{NO}_3\text{-N}$ concentrations in the shallow groundwater were very low ($< 2 \text{ mg L}^{-1}$) across all study watersheds and vegetative covers (Fig. 5). Interestingly, for the upslope position, $\text{NO}_3\text{-N}$ concentrations remained at relatively low levels during the two growing seasons following treatment implementation under both cropland and PFS (Fig. 5a). In contrast, at the toeslope position, groundwater $\text{NO}_3\text{-N}$ concentrations under cropland increased significantly in 2008 (Fig. 5b), reaching an average of 11 mg L^{-1} in June. This increase in $\text{NO}_3\text{-N}$ concentration beginning in early 2008 occurred in the period before nitrogen application (anhydrous ammonia) on 24 Apr. 2008. However, $\text{NO}_3\text{-N}$ concentrations under PFS in the toeslope position increased slightly in April 2008 (1.3 mg L^{-1}) and then declined gradually through the remainder of the growing season.

Discussion

The lack of a significant response in $\text{NO}_3\text{-N}$ concentrations across all treatments and landscape positions the first year after treatment establishment was somewhat surprising, given that tillage of the bromegrass sod and subsequent exposure of the organic rich soil would likely have accelerated decomposition processes. Other studies have documented sharp increases in soil $\text{NO}_3\text{-N}$ immediately following conversion from perennial cover to annual crops. For example, Huggins et al. (2001) reported a 125% increase in residual soil $\text{NO}_3\text{-N}$ the first year following conversion of CRP bromegrass to corn. However, this same study also reported that $\text{NO}_3\text{-N}$ losses in drainage water remained low during the first year following bromegrass-to-cropland conversion, but then increased to levels similar to continuous rowcrop systems during subsequent years (Huggins et al., 2001). The 1-yr time lag observed in our study for both the vadose zone and shallow groundwater may in part be attributed to a combination of the use of no-till practices for crop production, which potentially minimized organic matter losses by decomposition (Follett et al., 2009) and resulted in greater soil nitrogen conservation (Spargo et al., 2008; Purakayastha et al., 2009), as well as immobilization of nitrogen through microbial nitrification or assimilation processes (Booth et al., 2005; Yang et al., 2008). In addition, the high clay content (Table 2) of the soil in the study watersheds may result in a low water permeability, which could lead to a longer time before elevated nitrate shows up in the deep soil and groundwater.

The initial increase in $\text{NO}_3\text{-N}$ concentrations in the vadose zone under both the cropland and PFS in the upslope and toeslope positions in April 2008, which occurred before nitrogen fertilizer application, may have resulted from enhanced soil microbial N mineralization in response to disturbance from the initial tillage (Elliott, 1986; Grandy and Robertson, 2006). Further, mineralization rates were probably stimulated by the exceptionally high rainfall that year (Stanford and Epstein, 1974; Borken and Matzner 2009).

The subsequent decrease in $\text{NO}_3\text{-N}$ concentrations from early to late growing season in the upslope vadose zone for both cropland and PFS in 2008 was probably due to a combination of factors, including $\text{NO}_3\text{-N}$ leaching, plant uptake, and denitrification. The wet soil condition and large storms during 2008 likely enhanced the leaching of $\text{NO}_3\text{-N}$ in soil. In addition, large amount of recharge may also have diluted nitrate and contributed to decline in late spring and early summer. A similar trend was observed for a continuous corn system in central Iowa by Kalita and Kanwar (1993) and for continuously cropped hillslopes during a year-round crop rotation with winter wheat and summer maize in central China (Zhu et al., 2009). It is also probable that both the crops and PFS actively sequestered $\text{NO}_3\text{-N}$ from the soil (Dawson et al., 2008). Whereas $\text{NO}_3\text{-N}$ concentrations remained relatively high under cropland for several months before eventually decreasing, probably due to the fertilizer application in late April, the trend of rapidly declining $\text{NO}_3\text{-N}$ concentrations under PFS suggests effective uptake of nitrogen by perennial plants, which would be consistent with the findings of Sainju et al. (2007) and Schilling and Jacobson (2010). Denitrification, although

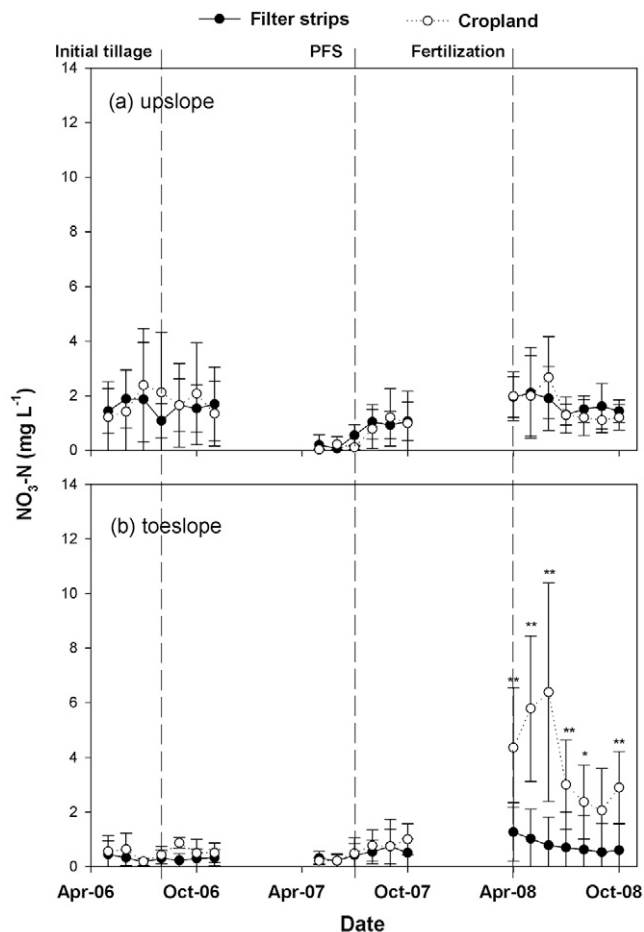


Fig. 5. $\text{NO}_3\text{-N}$ concentrations in shallow groundwater at the (a) upslope and (b) toeslope positions. Error bars denote the standard deviation of the replicates. Statistical difference of mean nitrate concentration between treatments (grass filters vs. cropland) was indicated for each monitoring period using two significant levels (** $p < 0.05$, * $p < 0.1$). PFS, perennial filter strips.

generally more pronounced under saturated conditions (see details below), may also have contributed to declining $\text{NO}_3\text{-N}$ concentrations in the upslope, especially during intermittent periods of high soil moisture.

In the toeslope vadose zone, the continued increase in $\text{NO}_3\text{-N}$ concentrations under cropland may reflect both the initial response to the fertilizer application and the following transport of $\text{NO}_3\text{-N}$ by lateral subsurface flows from the upper to lower regions of the watersheds as the 2008 growing season progressed (Bishop et al., 2004). The relatively steep slope (6–10%) of the study watersheds would probably promote rapid lateral preferential flow under wet soil conditions (Lin and Zhou, 2008). Even when the saturated zone is close to the soil surface during intense storms, water may still primarily flow laterally since horizontal hydraulic conductivity is usually much greater than vertical hydraulic conductivity on hillslopes (Lin, 2006). The “oxyaquic” subgrouping of the Otley soil series refers to the tendency to perch shallow water for brief periods that can flow laterally within hillslopes. Another recent study within the Walnut Creek watershed on groundwater recharge revealed that a considerable portion of precipitation recharges the groundwater in the riparian area as downslope runoff rather than as baseflow; the study concluded

that lateral flow dominates the three-dimensional flow system along hillslopes (Schilling, 2009). In this study, the observation that the amount of PFS area on watershed hillslopes did not affect soil and groundwater nitrate could also support the potential lateral flow in the study watersheds.

In contrast to cropland, under PFS in the toeslope vadose zone, $\text{NO}_3\text{-N}$ concentrations consistently declined throughout the growing season. In addition to plant uptake and leaching (discussed above), denitrification probably played a more dominant role in explaining this trend as denitrification is generally enhanced under saturated conditions when abundant organic carbon is present (Young and Briggs 2005; Schilling et al., 2007). During the 2008 season, groundwater levels in the toeslope were consistently higher (<1 m) than levels for the upslope position (3–5 m) and were also generally higher under the PFS toeslope compared with the cropland toeslope (Fig. 6). Consequently, the vadose zone and groundwater at the toeslope under PFS could be connected for much of the time,

while organic carbon was probably high due to postdisturbance mineralization of the grassland soils, thereby supporting conditions favorable to denitrification. Another possible contributing factor that cannot be overlooked is that dilution effects may have been greater in PFS areas that are not nitrate sources relative to cropland during wet periods (Maitre et al., 2003).

Although some of the differences in $\text{NO}_3\text{-N}$ concentrations between the PFS and cropland were arguably related to the fertilizer application, the observation that $\text{NO}_3\text{-N}$ concentrations under PFS and cropland reached the same early spring peak in response to mineralization, but then declined rapidly under PFS, underscores the functional capacity of the PFS strips to remove $\text{NO}_3\text{-N}$. Although we cannot determine to what extent fertilizer $\text{NO}_3\text{-N}$ was transported from the cropland to the PFS area during the study period, this will most certainly occur with time. More long-term monitoring is needed to assess the capacity of PFS to take up and store this additional $\text{NO}_3\text{-N}$ and prevent its movement from the watershed.

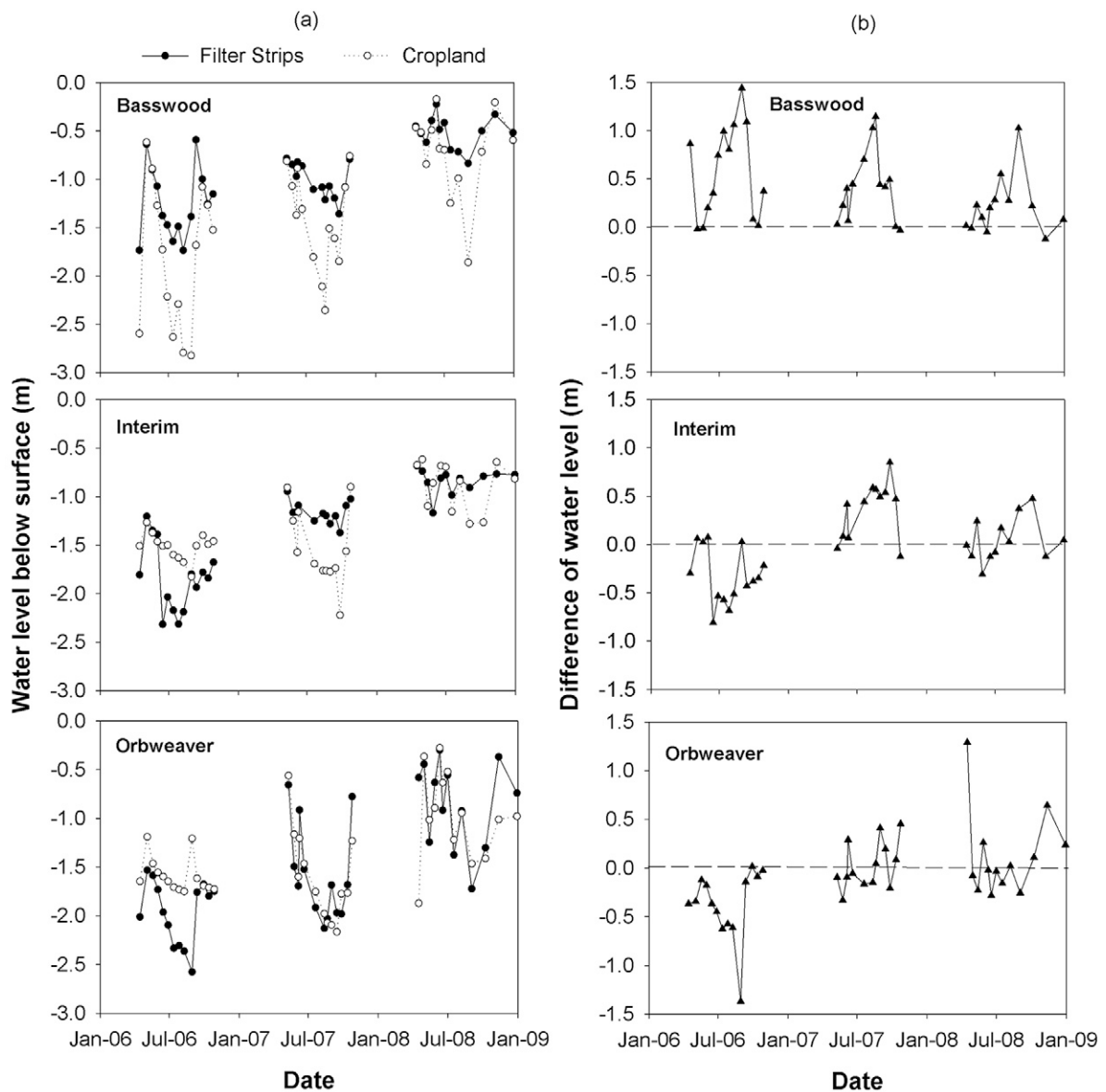


Fig. 6. (a) Depth of shallow groundwater level belowground in the perennial filter strip (PFS) and cropped areas at the toeslope positions, and (b) the difference of water level (PFS– Cropland) between the PFS and cropped areas. Positive values indicate that the water tables under PFS were higher than those under cropland. (Table 1).

During the 2 yr following grassland-to-cropland conversion, $\text{NO}_3\text{-N}$ concentrations in the shallow groundwater remained relatively low in the upslope position, and significantly lower than the toeslope groundwater concentration. In addition to the processes of leaching, plant uptake, and denitrification, it is also possible that $\text{NO}_3\text{-N}$ was transported from the upper hillslope to lower portions of the watershed by either overland runoff or subsurface lateral flow in the vadose zone before it reaches the shallow groundwater, as discussed above. Because the water table wells in the upslope position are generally between 3 and 5 m deep, the relatively low $\text{NO}_3\text{-N}$ concentrations at the upslope position in the shallow groundwater may also have resulted from a lag time in movement of the nitrate contamination into the groundwater at the upslope position where groundwater was deeper than relatively shallow toeslope areas after the land treatment change. This study's results suggest that incorporation of PFS in the toeslope position following grassland-to-cropland conversion was effective at reducing the $\text{NO}_3\text{-N}$ concentrations in shallow groundwater. A variety of N transport and transformation processes operating at the landscape scale may have contributed to this finding.

Conclusions

Nitrate-nitrogen concentrations in the vadose zone and shallow groundwater of the cropland areas exhibited sharp and in some cases sustained increases the second year after grassland-to-cropland conversion in the study watersheds, whereas increases in $\text{NO}_3\text{-N}$ concentrations only occurred for brief time periods in the early spring under the perennial strips. The response of $\text{NO}_3\text{-N}$ concentration to the land treatment change was statistically significant at the toeslope positions, which may be attributed to lateral transport of upslope $\text{NO}_3\text{-N}$ to toeslope positions.

The use of vegetative filters at the toeslopes within cropland was effective at reducing $\text{NO}_3\text{-N}$ concentrations in the shallow groundwater compared with toeslopes under crops in the first 2 yr after land-use conversion. Nitrogen uptake or physically withholding by vegetation, microbial denitrification and immobilization, and leaching are possible mechanisms for the nitrogen removal in vadose zone and shallow groundwater. The results suggest that converting perennial vegetation, such as brome grass typically used in CRP practices, to rowcrop production can contaminate groundwater within two growing seasons after grassland-to-cropland conversion and that PFS have the potential to mitigate increases in subsurface nitrate in early years following the conversion.

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