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A B S T R A C T

Decades of farming and fertilization of farm land in the unglaciated/Driftless Area (DA) of southwestern Wisconsin have resulted in the build-up of P and to some extent, N, in soils. This build-up, combined with steep topography and upper and lower elevation farming (tiered farming), exacerbates problems associated with runoff and nutrient transport in these landscapes. Use of an at-grade stabilization structure (AGSS) as an additional conservation practice to contour strip cropping and no-tillage, proved to be successful in reducing organic and sediment bound N and P within an agricultural watershed located in the DA. The research site was designed as a paired watershed study, in which monitoring stations were installed on the perennial streams draining both control and treatment watersheds. Linear mixed effects statistics were used to determine significant changes in nutrient concentrations before and after installation of an AGSS. Results indicate a significant reduction in storm event total P (TP) concentrations ($P = 0.01$) within the agricultural watershed after installation of the AGSS, but not total dissolved P ($P = 0.23$). This indicates that the reduction in P concentration is that of the particulate form. Storm event organic N concentrations were also significantly reduced ($P = 0.03$) after the AGSS was installed. We conclude that AGSS was successful in reducing the organic and sediment bound N and P concentrations in runoff waters thus reducing their delivery to nearby surface waters.

1. Introduction

Agriculturally derived nonpoint source pollution is the primary cause of impairments to rivers and streams in the United States this is largely due to eutrophication (USEPA, 2010). Eutrophication occurs when excessive amounts of nutrients are introduced into surface waters leading to increased plant and algal growth. This increased growth reduces dissolved oxygen levels in waters receiving runoff and sediment, which can negatively impact aquatic ecosystems.

Phosphorus is the nutrient mainly limiting eutrophication in freshwater systems, while N is mainly limiting in most saltwater systems (Sharpley et al., 1994; Correll, 1998). However, given complex nutrient cycling and plant uptake pathways within aquatic systems, as well as the spatial and temporal variability of such pathways, both N and P can be important in impairment in both types of ecosystems (Dodds and Welch, 2000). Nitrate is a form of N that is readily available for plant uptake and can also threaten both human and animal health. For drinking water, the NO_3 form of N has been known to cause methemo-globinemia in infants as well as have toxic effects on livestock (Sandstedt, 1990; Amdur et al., 1991). For this reason, the USEPA has set the drinking water standard for NO_3 (as N) at 10 mg L^{-1} (USEPA, 2009). Nitrate-N levels between 40 and 100 mg L^{-1} in waters being consumed by livestock can also cause adverse effects (Sandstedt, 1990). Ammonium-N concentrations of 0.5 and 2.5 mg L^{-1} have been reported to be harmful to

Abbreviations: DA, unglaciated/Driftless Area; AGSS, at-grade stabilization structure; TP, total phosphorus; DP, dissolved phosphorus; PP, particulate phosphorus; TN, total nitrogen; TKN, total Kjeldahl N; TDP, total dissolved P; NW, north watershed; SW, south watershed; LME, linear mixed effects; CRP, conservation reserve program; CREP, conservation reserve enhancement program.

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both humans and aquatic organisms, respectively (USEPA, 1973; Russo, 1985; Miltner and Rankin, 1998).

Total P (TP), with respect to agricultural runoff, can be separated into dissolved and particulate components. Dissolved P (DP) primarily comprises inorganic PO_4 salts that can readily be taken up and utilized by algae (Walton and Lee, 1972). Particulate P (PP) includes that attached to soil particles and organic matter and is the dominant P component in agricultural runoff. About 75–90% of TP in conventional agricultural runoff is in the PP form (Sharpley et al., 1994). Even though PP is not immediately available for plant uptake, it acts as a long-term source within sediment, although the bioavailability of PP has been shown to be quite variable (e.g., 10–90%; Sharpley et al., 1992). Total P concentrations in agricultural runoff have been shown to decrease with increased conservation and less intensive agricultural practices; however, bioavailable components of phosphorus (DP and bioavailable PP) were shown to represent a much greater proportion of the TP under increased conservation practices (Sharpley et al., 1992). However, it should be noted that the particulate forms of any nutrient in freshwater systems represent a complex continuum of organic particles, microorgansims, and sorbed inorganic ions and that suspended sediment itself can be aggregated (Droppo, 2001).

Particulate N and P tend to be dominant form of these nutrients in runoff from agricultural landscapes. Sharpley et al. (1987) concluded that 75% of the TP and 64% of the TN in runoff from rural areas are in the particulate form. Also, the U.S. Department of Agriculture (USDA, 1989) reported that roughly 80% of the TP and 73% of the TKN in runoff from agricultural areas is attributed to eroded sediment. These particulate transported nutrients are of particular concern for the unglaciated/Driftless Area (DA) of Wisconsin given this area's steep topography and susceptibility to runoff and erosion. This combined with agriculture being the primary land use within the DA, only exacerbates the problems associated with the transport of particulate N and P to surface waters. Total suspended sediment and TP loadings to DA streams were as high as 353.8 Mg km^{-2} yr^{-1} and 693.5 kg km^{-2} yr^{-1}, respectively, and as much as 95% of annual TSS and 87% of the annual TP loadings were attributed to storm runoff events (Corsi et al., 1997).

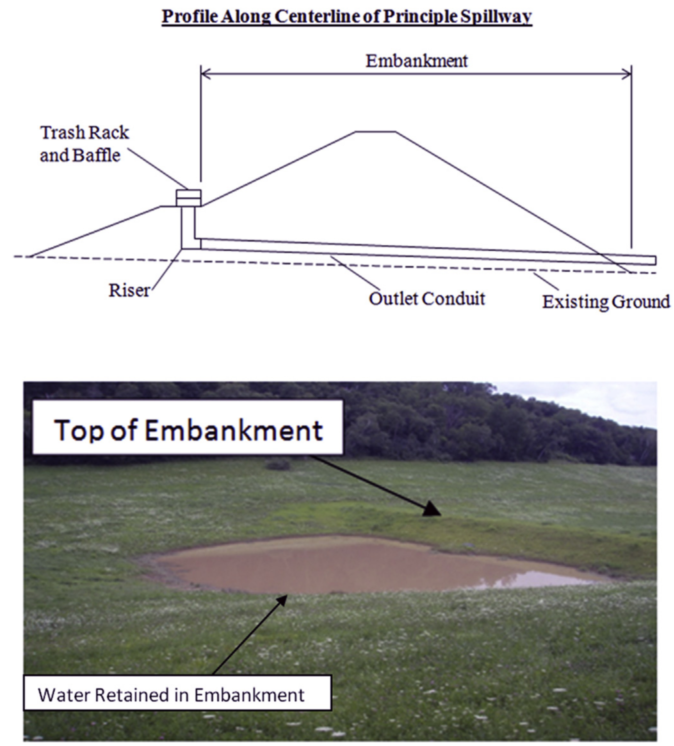

Fig. 2. Cross-sectional view and picture of the at-grade stabilization structure embankment that was installed in the north watershed (USDA NRCS, 2005; Minks et al., 2012).

An at-grade stabilization structure (AGSS) can be effective in reducing the amount of suspended sediment being transported to surface waters of the DA (Minks et al., 2012). These structures consist of a large embankment that is designed to retain storm runoff long enough for transported sediments to settle (Fig. 2). In addition, they also serve as a sink in which these transported sediments can be stored for a specified period. In this fashion, it is very similar in function to sedimentation basins and detention

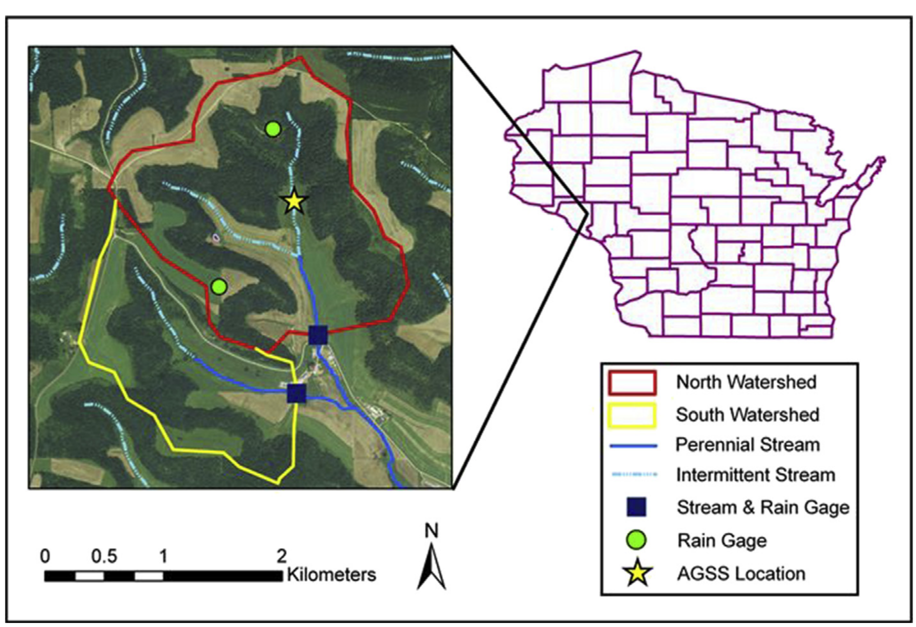

Fig. 1. Location of north and south Travers Valley Creek Watersheds as well as stream and rain gages (Minks et al., 2012).

ponds commonly used in both the mining and construction industries, but these are not one-in-the same. The main difference between a sediment basin and an AGSS is that there is no below-ground excavation for AGSS, and long-term ponding of water does not occur. However, we can look to sediment basin literature for direction on how an AGSS might function and impact water quality.

Past sediment basin research suggests that AGSS might reduce sediment and nutrient loads. Sedimentation basins have been extensively researched since the 1970s and have proven successful in reducing surface water nonpoint source pollution in urban and construction settings (Whipple and Hunter, 1981; Walker, 1987; Fennessey and Jarrett, 1994). Edwards et al. (1999) reported sediment basin TN and TP trapping efficiencies of 72–81% and 32–66%, respectively. Other researchers have also indicated that sedimentation basins are effective in reducing the amount of sediment and nutrients entering surface waters of agricultural watersheds (Edwards et al., 1999; Czapar et al., 2005; Fiener et al., 2005). Runoff with total suspended solid concentrations of about 200 mg L^{-1} was also reported to have decreased to levels between 5 and 20 mg L^{-1} with the installation of a sedimentation basin (Barrett, 2008).

The overall goal of this project was to determine the water quality benefits of an AGSS, with a main objective to determine changes in N and P concentrations and loads before and after installation of an AGSS. As previously noted, it has been shown by Minks et al. (2012) that AGSS can be used to reduce suspended sediment transport to surface waters, but the impact of this structure on nutrient loading has not been evaluated. Thus, we hypothesized that the same AGSS installed by Minks et al. (2012) will reduce the concentration of P and N movement into nearby surface waters. To achieve this objective, a collaborative project was established between the University of Wisconsin Discovery Farms Program and the US Geological Survey (USGS) Wisconsin Water Science Center (WWSC). Storm event nutrient concentrations collected by the USGS WWSC were compared for the two watersheds before and after installation of the AGSS. The two watersheds are adjacent to each other, with one being actively farmed with row crops and the other being used primarily for hay/pasture or enrolled in the Conservation Reserve Program (CRP) (Minks, 2010). Minks et al. (2012) reported data on basin paring and showed that the two basins were similar. They noted that USGS-WWSC used data from 12 storm events occurring from 1 October 2001 to 4 October 2002 to show that the two basins had similar hydrologic behavior. The USGS-WWSC used linear regressions of runoff volumes, suspended solids, TKN, TP, and total dissolved P loadings where one watershed was the dependent variable (south watershed) and the other was the independent variable (north watershed). They reported the r^2 values as ranging from 0.64 to 0.96, and intercepts and slopes as being significant ($P < 0.05$) (Minks et al., 2012). Thus, it was assumed that the two watersheds are similar with respect to nutrient and runoff.

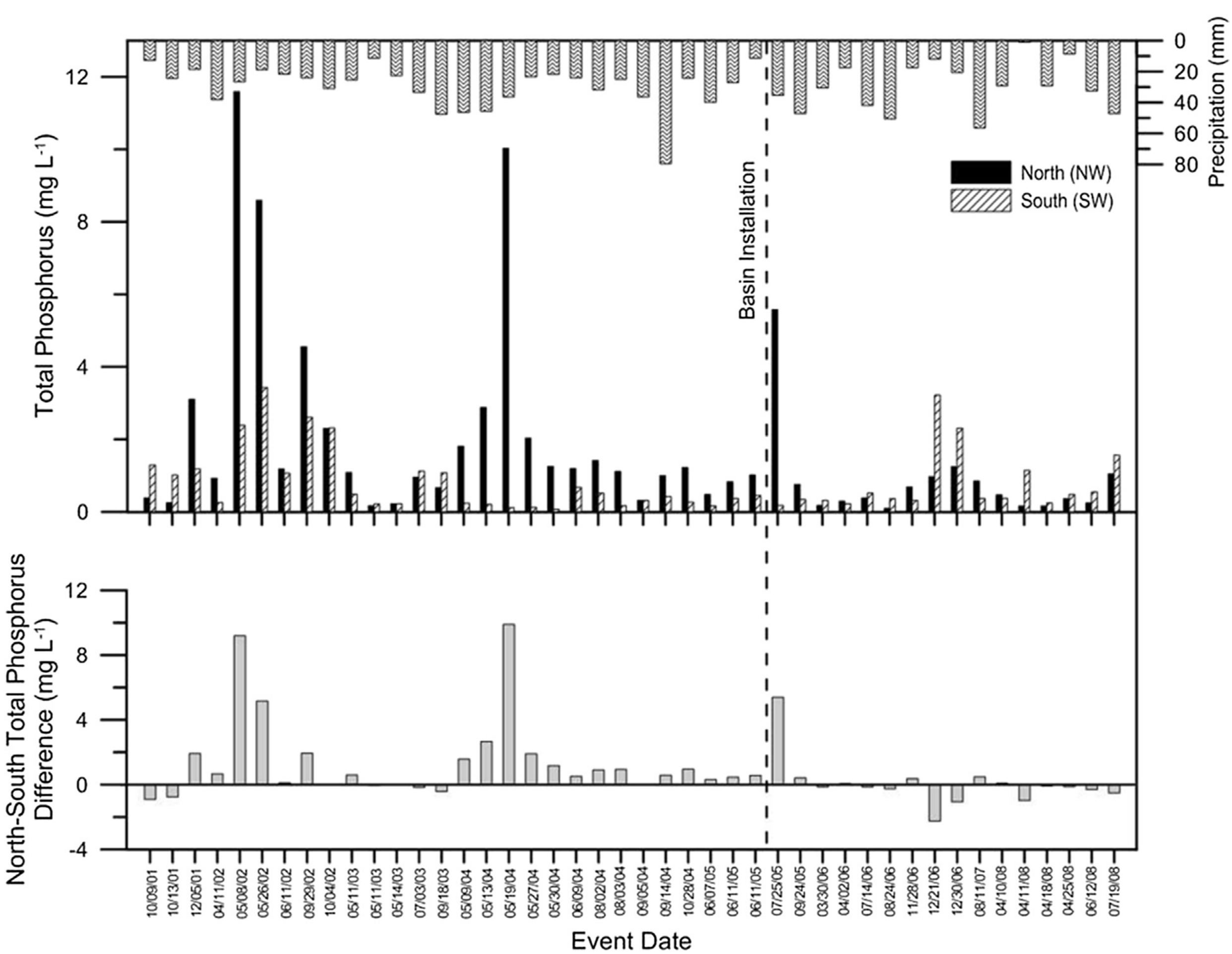

Fig. 3. Total runoff total phosphorus (TP) minus base-flow TP per event (mg L^{-1}) and precipitation (mm) for each of the 44 sampled events from 2001 to 2008 (note: dates are not equally distributed across the x-axis) at both north (NW) and south (SW) watersheds. Difference in total runoff TP minus base-flow TP (north–south) is also presented.

Table 1
Total runoff minus base-flow mean concentrations of total P (TP), total dissolved P (TDP), total N (TN), nitrite+nitrate–N (NO_2+NO_3–N), ammonium–N (NH_4–N), and organic N for 44 paired storm events.[a]

Basin[b]	Before/after[c]	TP	TDP	TN	NO_2+NO_3 –N	NH_4 –N	Organic N
		mg L^{-1}					
Event Non-Ln transformed least squares means							
NW	Before	2.23	0.38	11.63	2.55	0.74	8.39
NW	After	0.84	0.26	5.64	1.99	0.13	3.53
SW	Before	0.81	0.33	4.84	1.47	0.13	3.22
SW	After	0.78	0.60	12.06	8.07	0.27	3.72
Basin before/ after	Basin before/ after	TP	TDP	TN	NO_2+NO_3 –N	NH_4 –N	Organic N
P-values for Event Ln transformed least squares means							
NW After	SW After	0.770	0.007	0.053	0.025*	0.040*	0.460
NW After	NW Before	0.005*	0.232	0.157	0.725	0.860	0.025*
NW After	SW Before	0.925	0.274	0.322	0.078	0.458	0.738
SW After	NW Before	0.010*	0.297	0.668	0.128	0.201	0.091
SW After	SW Before	0.846	0.253	0.006*	0.001*	0.471	0.356
NW Before	SW Before	<0.001*	0.866	<0.001*	0.001*	0.284	<0.001*

* Significant difference at $P < 0.05$.
[a] The *P*-values are also presented using natural log transformations and the linear mixed effects (LME) model.
[b] NW, north watershed (treatment); SW, south watershed (control).
[c] Before the time period before installation of the at-grade stabilization structure August 2001 to June 2005. After, the time period after installation of the at-grade stabilization structure July 2005 to August 2008.

2. Materials and methods

2.1. Study site

The study sites consist of paired watersheds, north (NW) (44° 23′ 55″ latitude, 91° 33′ 05″ longitude) and south (SW) (44° 23′ 44″ latitude, 91° 33′ 13″ longitude), located in the headwaters of Traverse Valley Creek in Buffalo County, Wisconsin (Fig. 1) and were the same watersheds studied in Minks et al. (2012). The NW is approximately 174 ha, of which 58% is woodland (101 ha), 42% is cropland and pasture (73 ha). The SW is approximately 87 ha in size and is comprised of about 16 ha (18%) in cropland and pasture, 43 ha of woodland (50%), with the remainder (32%) enrolled in government programs that require the land to be covered with permanent vegetation for a number of years (CRP and Conservation Reserve Enhancement Program, CREP). Minks (2010) provided detail on the yearly cropping systems, which was mainly a rotation between corn (*Zea mays* L), alfalfa (*Medicago sativa*), and soybean (*Glycine max*) for the cropped areas and grass and woods, with some shrub/brush, for the forest and pasture areas.

This part of the state is known as the DA and is often characterized as having flat top ridges, steep side slopes, and narrow valleys overlying dolostone bedrock. This setting results in upper and lower elevation farming or tiered farming, with the steep side slopes remaining in native forest and not farmed. Soils in both

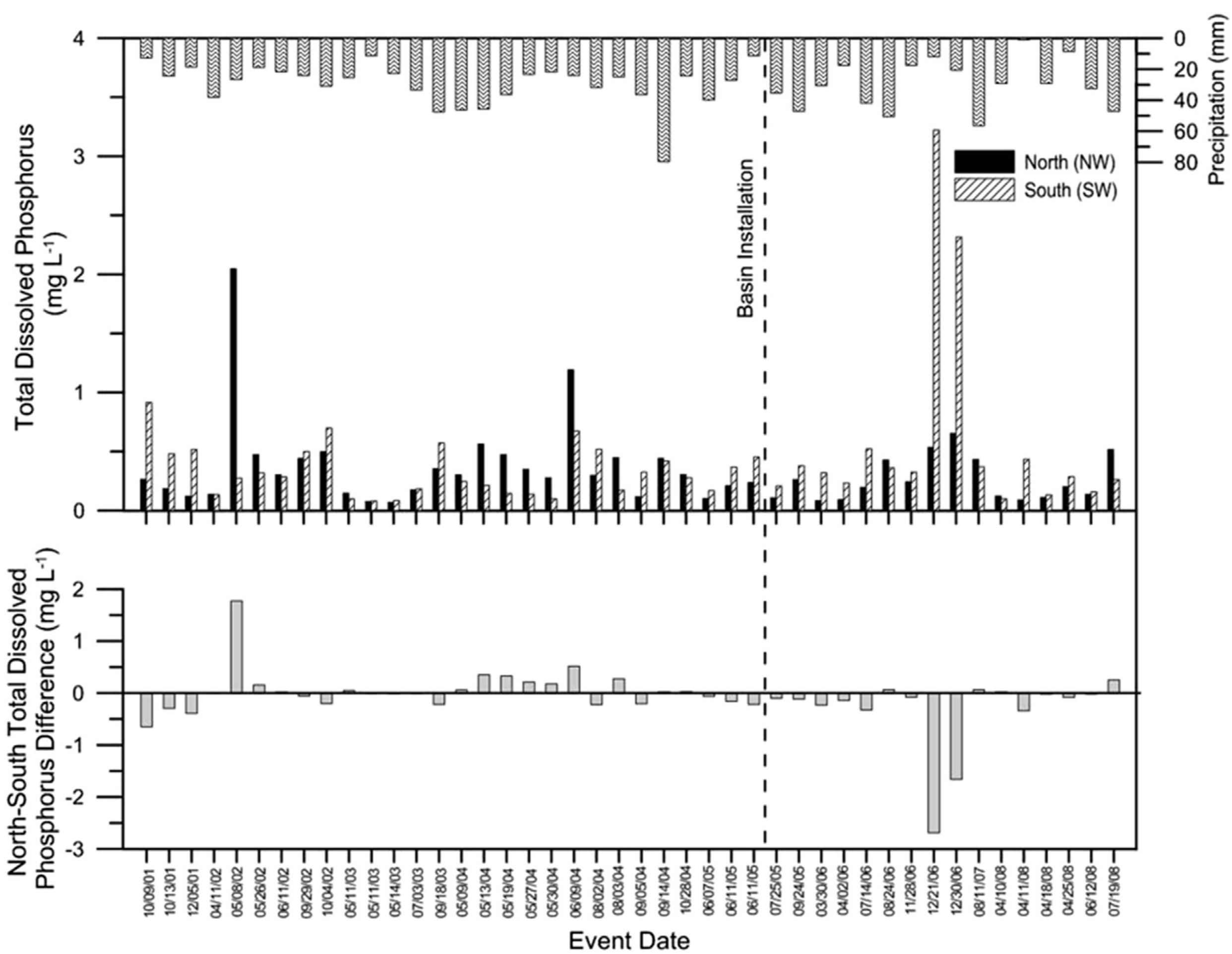

Fig. 4. Total runoff total dissolved phosphorus (TDP) minus base-flow total TDP per event (mg L^{-1}) and precipitation (mm) for each of the 44 sampled events from 2001 to 2008 at both north (NW) and south (SW) watersheds. Difference in total runoff total TDP minus base-flow total TDP (north–south) is also presented.

watersheds consisted of Dubuque silt loams (fine-silty, mixed, superactive, mesic Typic Hapludalfs) on the ridge tops, stony and rocky debris occupying much of the steep side slopes, along with Fayette (fine-silty, mixed, superactive, mesic Typic Hapludalfs) and Norden (fine-silty, mixed, superactive, mesic Typic Hapludalfs) silt loams comprising the majority of soils in the valleys (Soil Survey Staff, 2013). The area receives an average 846 mm of snowfall and 1151 mm of total precipitation annually, and has an average monthly temperature of −8.4 °C in winter and 20.3 °C in summer (Wisconsin State Climatology Office, 2007).

2.2. Monitoring stations, sampling, and analysis

For these watersheds, the USGS, in cooperation with the University of Wisconsin Discovery Farms program, installed two stream gages on private farms. Monitoring stations were installed, maintained, and operated according to detailed methods described in Stuntebeck et al. (2008) and Minks et al. (2012) provided an overview, thus only a brief summary is provided here. Streamflow volumes were quantified with H-flumes. The stage reading of the H-flumes (depth of water flowing) was measured with non-submersible pressure transducers (Sutron Accubar Model 5600-0125, Sutron Corporation, Sterling, Virginia) coupled with a nitrogen bubble system, and pressure reading were recorded with a datalogger (CR10 datalogger, Campbell Scientific, Inc. Logan, Utah). An automated and refrigerated sampler (ISCO refrigerated sampler R3700 with 1L polyethylene bottles, Teledyne ISCO, Inc., Lincoln, Nebraska) collected discrete volume-based samples during rainfall- and snowmelt-induced runoff events. Samples were generally retrieved (>95% of the time) within 24 h of the end of a runoff event and were placed on ice and transported to the Water and Environmental Analysis Laboratory, a state- and USGS-certified facility at the University of Wisconsin–Stevens Point for analysis. Filtration of samples occurred in the laboratory. Tipping-bucket rain gages were used to determine rainfall amount and intensity of unfrozen precipitation. Volumetric soil water content was measured with soil moisture probes installed at 10-, 20-, 30-, and 50-cm depth (EasyAg, Sentek Sensor Technologies, Stepney, SA, Australia) and 0–30 cm with CR615 and 616 units from Campbell Scientific. A standard Campbell Scientific weather station was installed at the site with air and soil temperature measurements, and the soil temperature was measured at 2, 5, 10, 20, 40, and 80 cm with Campbell Scientific 107 temperature probes.

Flow volumes were measured for all runoff events, and water samples were collected and analyzed for most of the period from October 2001 through September 2008; however, only runoff events occurring when the ground was not frozen (mean daily soil temperature above freezing) were included in this analysis. Monthly base-flow samples were taken to characterize the base-flow concentrations of constituents.

A flow-weighted composite sample was produced for each runoff event at each station by combining the volume-based discrete samples to represent each runoff event (Stuntebeck et al., 2008). The composite sample was analyzed for TP, NO_2+NO_3–N,

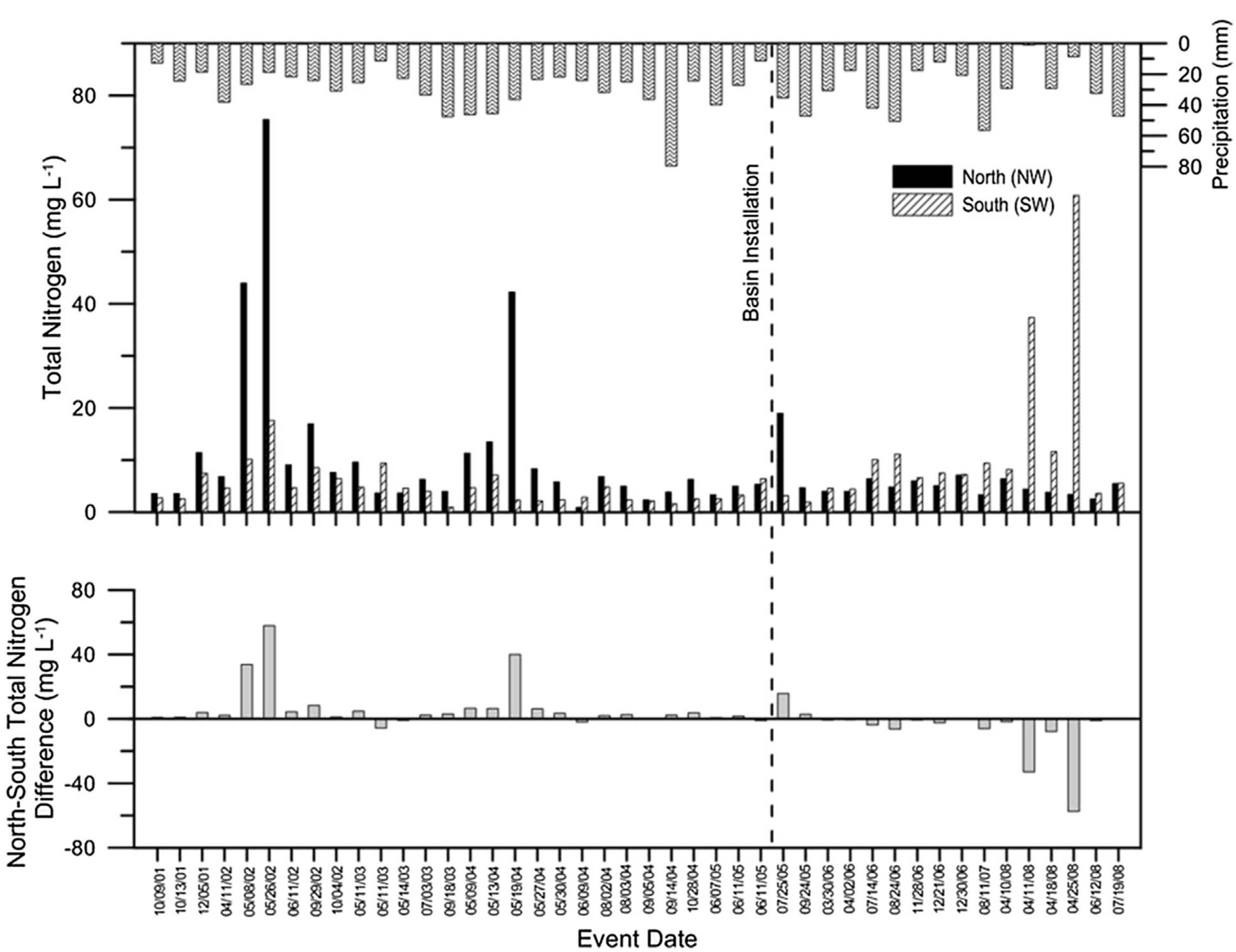

Fig. 5. Total runoff total N (TN) minus base-flow TN per event (mg L^{-1}) and precipitation (mm) for each of the 44 sampled events from 2001 to 2008 at both north (NW) and south (SW) watersheds. Difference in total runoff TN N minus base-flow TN (north–south) is also presented.

NH_4–N, total Kjeldahl N (TKN), and total dissolved P (TDP). Analyses were performed by automated colorimetery on Lachat 8500 (for NO_2+NO_3–N) (Hach, Loveland, Colorado) or Lachat 8000 instruments (for all others). Analyses for NO_2+NO_3–N, NH_4–N, and TDP were on filtered (0.45 μm) samples, while the analyses for TKN and TP were not filtered. The TDP, TKN, and TP were block-digested prior to analysis. Analyses were done by method 4500-NO_3–F (all methods are according to APHA, 2012) for NO_2+NO_3–N, 4500-NH_3–H for NH_4–N, 4500-NH_3–G for TKN, 4500-P–F for TDP, and 4500-P–J for TP. Total N was estimated as the sum of NO_2+NO_3–N and TKN, and organic N was estimated as the difference between TKN and NH_4–N.

2.3. Basin pairing

As previously noted, Minks et al. (2012) reported that during the first year of monitoring (1 October 2001 to 4 October 2002), the USGS-WWSC conducted statistical analysis on 12 storms to determine if the nutrient transport in each basin was similar. Thus, the treatment for this study was the installation of an AGSS in the NW. The first 4 years of the study (1 October 2001 to 11 June 2005) was a calibration period that consisted of collecting runoff water samples and recording background measurements in both watersheds. The AGSS was then installed in June 2005 at the base of the wooded hill slope in the NW. Data continued to be collected from 25 July 2006 to 30 September 2008 from both watersheds. The AGSS was designed according to NRCS specifications (Fig. 2) (USDA NRCS, 2005). The structure consists of 100-m long embankment that bisects the major flow path of the NW. It stands roughly 8.33 m tall from the toe end of the embankment, and contains a 76.2-cm diameter 30-m long flow pipe positioned at a 4.8% grade. The purpose of the structure is to reduce the velocity of the water flowing down the forested hill slope, allowing suspended debris time to settle out.

2.4. Data preparation

Over the course of the 8-yr study, 44 non-frozen ground paired storm events were sampled. Computations were done to remove the base-flow component of each storm event so that only the overland flow component remained. Monthly base-flow concentrations were multiplied by base-flow volumes to estimate base-flow loadings. Base-flow volumes were estimated by multiplying storm durations by the pre-storm event steady state flow rates. Base-flow loadings were then subtracted from total runoff loadings (event-mean runoff concentration multiplied by storm volumes) to obtain the loading of each constituent as the result of overland flow only. The total overland flow loading was then divided by the difference between the storm volume and base-flow volume to obtain the flow-weighted average concentration of each constituent within the overland flow component of the total runoff (Stuntebeck et al., 2008; Minks et al., 2012).

On 9 October 2001, the NO_2+NO_3–N constituent in the NW had a negative value once base-flow was removed. In order to allow for

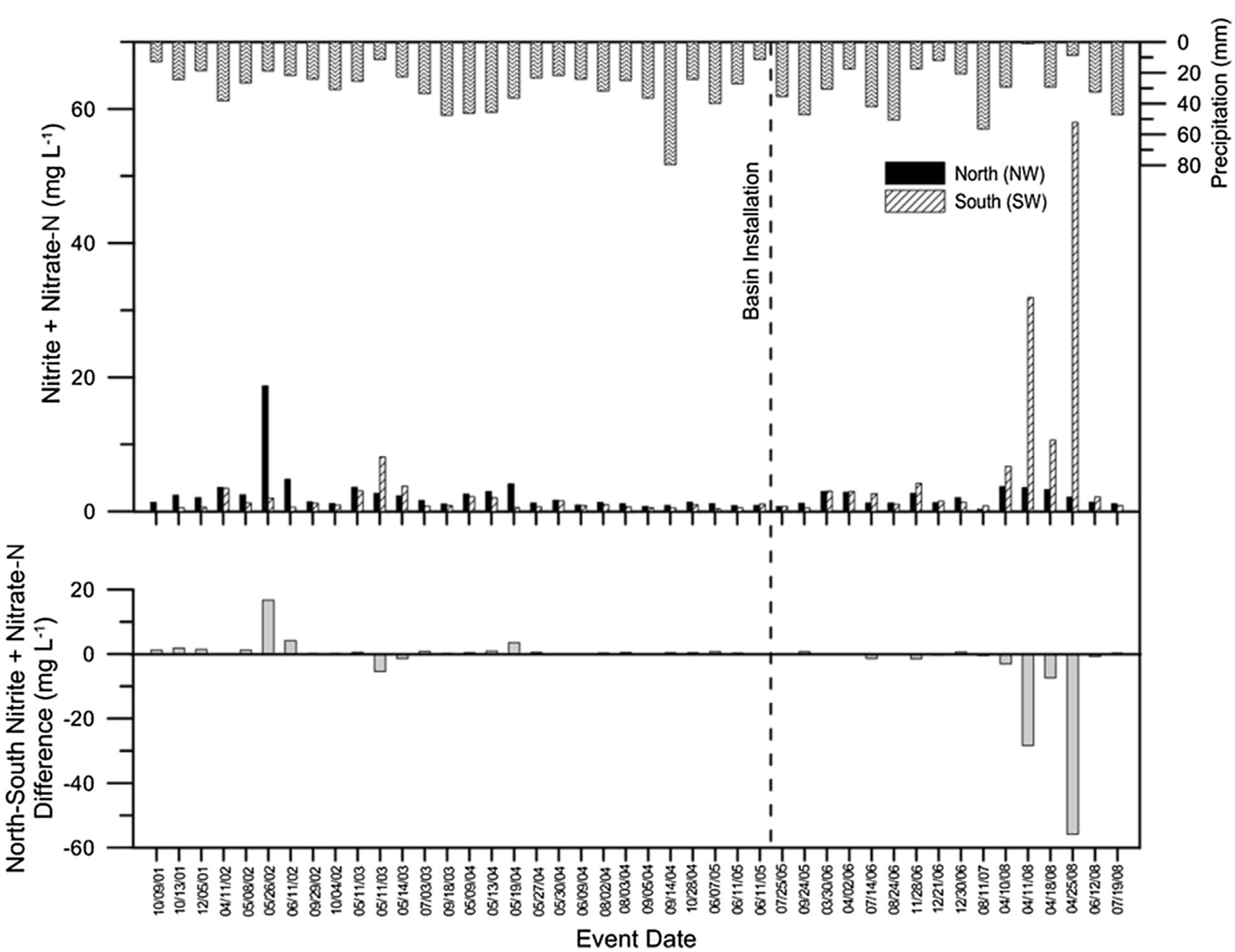

Fig. 6. Total runoff nitrite + nitrate–N (NO_2+NO_3) minus base-flow NO_2+NO_3–N per event (mg L^{-1}) and precipitation (mm) for each of the 44 sampled events from 2001 to 2008 at both north (NW) and south (SW) watersheds. Difference in total runoff NO_2+NO_3–N minus base-flow NO_2+NO_3–N (north–south) is also presented.

the natural log (Ln) transformations (for statistical analysis) of this constituent, a constant value (0.02) slightly greater than the minimum value of all the storms, was added to each storm event. Original values were still used to determine non-Ln transformed event mean values.

2.5. Statistics

Statistically significant changes in event mean nutrient concentrations were determined using a linear mixed effects (LME) model. The LME model is:

$$Y = (ns_i)ba_j + ev_k + e_{ijk} \tag{1}$$

with Y representing the variable of interest (Ln TP, Ln TDP, Ln TN, Ln NO_2+NO_3–N, Ln NH_4–N, and Ln organic N); ns_i is either the NW or SW; ba_j indicates when the measurement was taken, before or after installation of the AGSS, and is represented by b or a, respectively; ev_k is the random event and illustrates the storm event that has taken place in both watersheds; and e_{ijk} is the error associated with the above three parameters. The Ln transformations were implemented in order to hold normality assumptions within modeling. Analysis was conducted using Proc Mixed SAS 9.2 (SAS, 2008) with ns and ba as fixed effects and ev as a random effect. Statistical significance was set at P = 0.05.

Storm events (ev_k) represent a specific individual storm with a varying probability of intensity and duration that occurred within both watersheds. As a result, this variable was held as a random factor within Equation (1). In addition, stream water samples were taken in the same location before and after installation of the AGSS within both the NW and SW. Thus, the parentheses in Equation (1) represent a nested design in which the ba_j factor is nested within the ns_i factor.

3. Results and discussion

3.1. Phosphorus

Forty-four storm events were analyzed, the same amount analyzed for runoff and sediment loss by Minks et al. (2012). The AGSS had a noticeably reduced TP concentration during storm events in the NW (Fig. 3). Statistical analysis using the LME model supports this observation. Application of the model revealed a significant (P = 0.01) decrease in average event TP concentration before (2.23 mg L^{-1}) and after (0.84 mg L^{-1}, a 62% reduction) installation of the AGSS within the NW (Table 1). However, no significant (P = 0.85) change was evident in the SW with pre- and post-installation TP values of 0.81 and 0.78 mg L^{-1}, respectively. Since a significant change was evident in the NW and not the SW, it can be concluded that the installation of the AGSS resulted in the observed decreases. The AGSS was also successful in reducing TP concentrations in the NW (0.84 mg L^{-1}) to similar levels as those seen in the SW (0.78 mg L^{-1}) (Table 1).

There was no observable TDP reduction within the NW after the

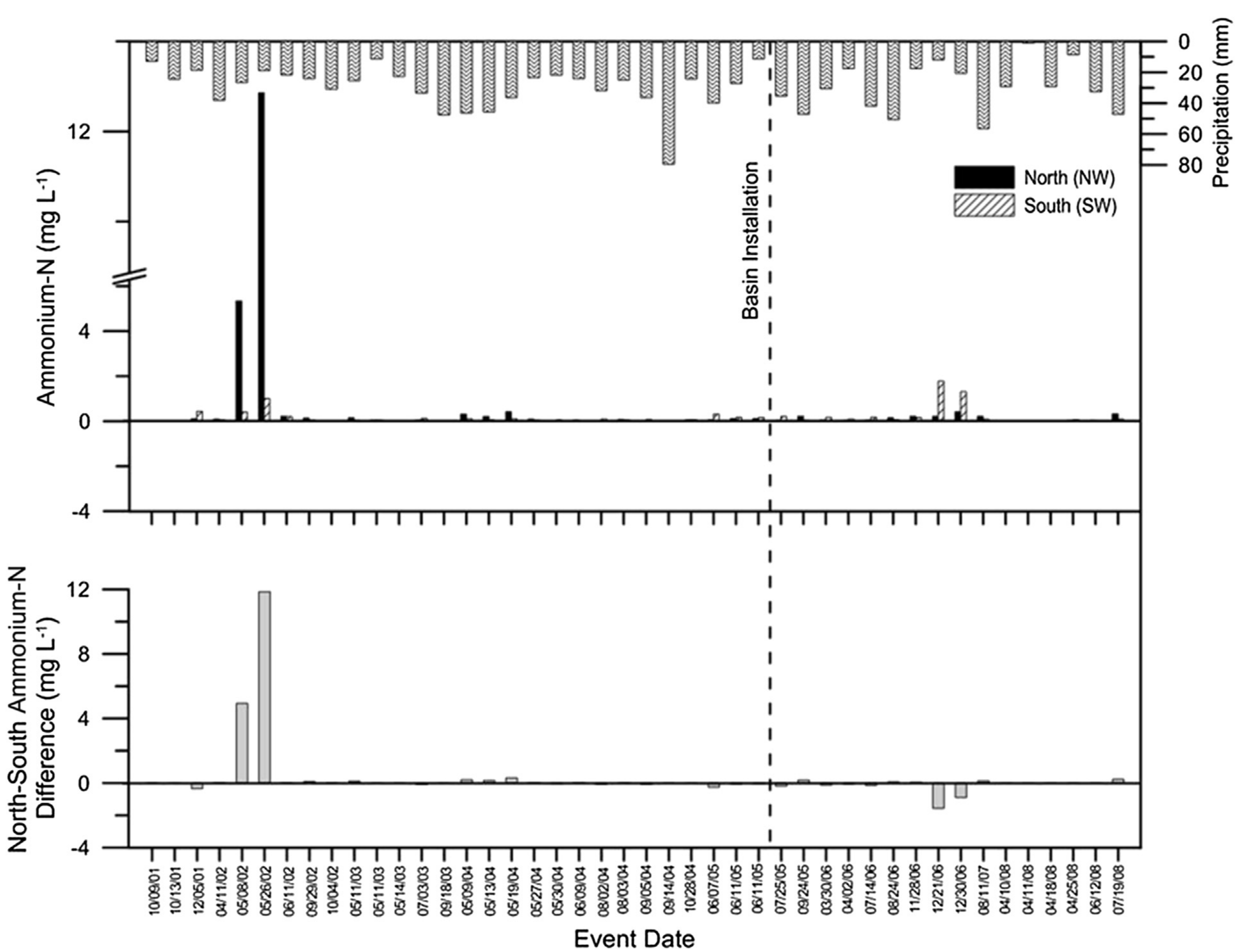

Fig. 7. Total runoff ammonium–N (NH_4–N) minus base-flow NH_4–N per event (mg L^{-1}) and precipitation (mm) for each of the 44 sampled events from 2001 to 2008 at both north (NW) and south (SW) watersheds. Difference in total runoff NH_4–N minus base-flow NH_4–N (north–south) is also presented.

AGSS was installed (Fig. 4). The average pre-installation TDP was 0.38 mg L^{-1} compared to 0.26 mg L^{-1} after (Table 1). Average before and after values in the SW were 0.33 and 0.60 mg L^{-1}, respectively (Table 1). No significant differences were evident in average event TDP concentrations before and after installation in either the NW ($P = 0.23$) or SW ($P = 0.25$) (Table 1).

The TP reduction in the NW is supported and consistent with research of Edwards et al. (1999), which determined TP trapping efficiencies using a sedimentation basin to be between 32 and 66%. The significant reduction in TP, but not TDP, is consistent with a trapping of PP. The 63% reduction in mean event TP concentration based on the before and after installation of the AGSS can thus be attributed to the trapping of the PP component.

Previous research conducted on this watershed supports the result that AGSS significantly reduced PP in the NW. Minks et al. (2012) evaluated the same storm events and watersheds presented in this research and concluded that installation of the AGSS in the NW significantly reduced ($P = 0.02$) average suspended sediment concentrations from 972 mg L^{-1} before installation to 263 mg L^{-1} after (73% reduction). Given that PP includes P that is attached to both soil particles and organic matter, significantly reducing the amount of suspended sediment reduces the amount of PP.

Although the AGSS is not a sediment basin, the reduction in average event TP concentration of 62% (2.23 mg L^{-1} before and 0.84 mg L^{-1} after) with the AGSS installation in the NW compares well with other studies of engineered structures like sediment basins. Edwards et al. (1999) reported retention efficiencies in mean TP with a sediment basin of 32–66%. Brown et al. (1981) found a sediment pond reduced TP in runoff between 25 and 33%. Bjorneberg and Lentz (2005) found a combination of polyacrylamide (PAM) and sediment basin reduced average TP 66% for a 3-year irrigation runoff study, but did not reduce dissolved reactive P. Thus, AGSS should be considered as efficient as sediment basins with respect to reducing TP losses and similar to sediment basins in their inability to reduced DP losses.

3.2. Nitrogen

Event TN concentrations within the NW exhibited an apparent decrease after installation of the AGSS (Fig. 5). Prior to AGSS installation, the average event TN concentration in the NW was 11.63 mg L^{-1} (Table 1) compared to 5.64 mg L^{-1} after (Table 1). However, this reduction was not a significant ($P = 0.16$). Average event TN in the SW increased significantly ($P = 0.08$) from 4.84 mg L^{-1} before to 12.06 mg L^{-1} after the AGSS was installed (Table 1). Given the non-significant reduction of TN within the NW and the significant increase of TN in the SW, it is difficult to conclude that the AGSS reduces TN concentrations. This can be further explained with additional analysis of the individual components that make up TN.

Nitrite + nitrate–N event concentrations were not affected by the installation of the AGSS (Fig. 6). Statistical evaluation of the data reveals no significant change ($P = 0.73$) in the NW post installation

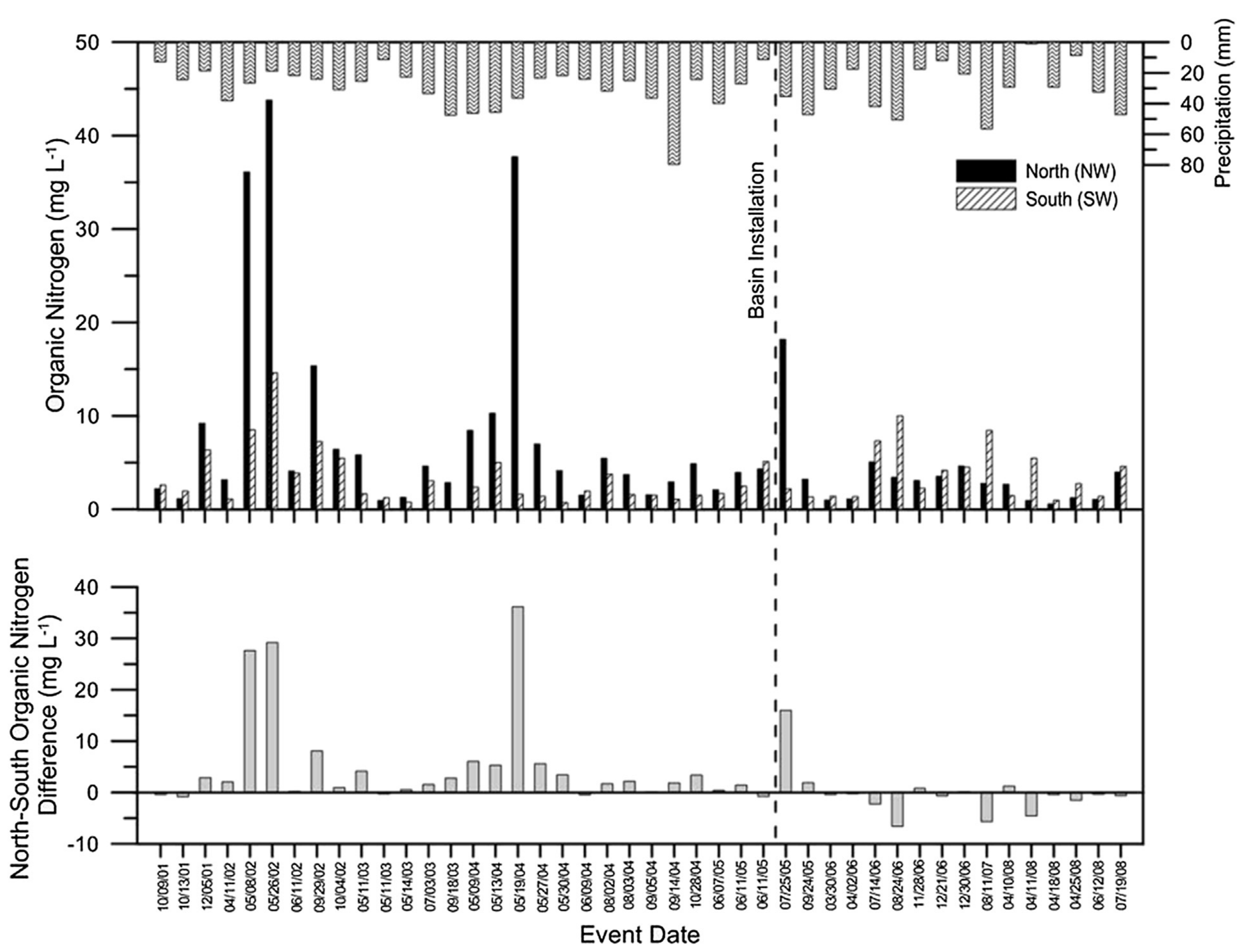

Fig. 8. Total runoff organic N minus base-flow organic N per event (mg L^{-1}) and precipitation (mm) for each of the 44 sampled events from 2001 to 2008 at both north (NW) and south (SW) watersheds. Difference in total runoff organic N minus base-flow organic N (north–south) is also presented.

of the AGSS. However, significant increase ($P = 0.01$) was evident in the SW with a mean event concentration of 1.47 mg L^{-1} before and 8.07 mg L^{-1} after (Table 1). Given these results, it is unlikely that the AGSS reduced the concentration of NO_2+NO_3-N. This can be explained by the fact that NO_2+NO_3-N concentrations are likely primarily controlled by groundwater in both watersheds (USGS-WWSC, personal communication, 2010). In addition, the significant increase in $NO_2 + NO_3-N$ in the SW are likely a result of the high concentrations occurring during two events in April 2008 (Fig. 6), which had low flow volumes (Minks et al., 2012). Thus, the large difference in post AGSS installation between NW and SW is a result of the enrichment effect from two events rather than the AGSS in the NW watershed preventing large NO_2+NO_3-N concentrations at those times.

Further evaluation of the data revealed no change in NH_4-N in either the NW or SW after installation of the AGSS with a P-value of 0.86 and 0.47 before and after installation, respectively (Fig. 7, Table 1). Comparing the TN average values per event pre- and post-installation of the AGSS to NH_4-N concentrations reveals that NH_4-N accounts for less than 7% of the TN. This relatively small proportion can be explained by the rapid conversion of NH_4-N to NO_2+NO_3-N under most environmental conditions.

Of the three components comprising TN (NO_2+NO_3-N, NH_4-N, and organic N), the organic form appears to be the most affected by the AGSS. A noticeable reduction in the organic N event concentration within the NW after installation of the AGSS was observed (Fig. 8). Statistical analysis supports this observation, with a significant ($P = 0.03$) reduction in the average event organic N concentration before AGSS installation of 8.39 to 3.53 mg L^{-1} after (Table 1). Evaluation of the SW reveals no such significant ($P = 0.36$) change in average concentrations before and after AGSS installation, with values of 3.22 and 3.72 mg L^{-1}, respectively (Table 1). In addition, data analyzed by the USGS-WWSC (personal communication, 2010) indicated that the major N component during storm runoff events is the organic form. These data indicate that the AGSS was effective in reducing the organic N component of TN, within the NW. It has been suggested by van Kessel et al. (2009) that dissolved organic N is often overlooked when determining dissolved N losses from agricultural fields, which can comprise between 1 and 83% of the total N leaching losses depending on the agricultural system. Our results suggest that organic N can also be overlooked in surface runoff as they represented between 31 and 72% of the TN exported from fields in this study. The organic N in this study includes both particulate and dissolved organic N, but since sediment and particulate P were reduced after installation of the AGSS, it is likely the sediment associated organic N is the fraction that is reduced as well.

4. Conclusions

The AGSS was effective in decreasing P and N delivery to nearby surface waters within the agricultural watershed in the unglaciated/Driftless Area of Wisconsin. Total P concentrations declined from an average 2.23 to 0.84 mg L^{-1} during runoff events, or 62%. Given the lack of a significant decrease in TDP, most of the TP decline is attributed to removal of PP. Organic N declined from average 8.39 mg L^{-1} prior to AGSS to 3.53 mg L^{-1} after AGSS installation. This fraction of the TN was then responsible for reducing TN concentrations as well. The results from this research, the installation of multiple AGSSs within agricultural watersheds of the DA could have a profound impact on improving the water quality of this area.

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