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THE INFLUENCE OF TWO-STAGE DITCHES WITH CONSTRUCTED FLOODPLAINS ON WATER COLUMN NUTRIENTS AND SEDIMENTS IN AGRICULTURAL STREAMS

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ABSTRACT: The two-stage ditch is a novel management practice originally implemented to increase bank stability through floodplain restoration in channelized agricultural streams. To determine the effects of two-stage construction on sediment and nutrient loads, we monitored turbidity, and also measured total suspended solids (TSS), dissolved inorganic nitrogen (N) species, and phosphorus (P) after two-stage ditch construction in reference and manipulated reaches of four streams. Turbidity decreased during floodplain inundation at all sites, but TSS and P, soluble reactive phosphorus (SRP) and total phosphorus (TP) decreased only in the two-stage ditches with longer duration of inundation. Both TSS and TP were positively correlated within individual streams, but neither were correlated with turbidity. Phosphorus was elevated in the stream to which manure was applied adjacent to the two-stage reach, but not the reference reach, suggesting that landscape nutrient management plans could restrict nutrient transport to the stream, ultimately determining the efficacy of instream management practices. In addition, ammonium and nitrate decreased in two-stage reaches with lower initial N concentrations. Overall, results suggest that turbidity, TSS, and TP were reduced during floodplain inundation, but the two-stage alone may not be effective for managing high inorganic N loads.

(KEY TERMS: water quality; nutrients; sediment; turbidity; agriculture; streams; two-stage ditch.)

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INTRODUCTION

Agricultural Land Use Impacts Water Quality in the Midwest

Streams in the midwestern United States (U.S.) are associated with row-crop agriculture (Blann *et al.*, 2009); in fact, GIS analyses of land-use patterns show that >90% of streams in the "Corn Belt" of the Midwest (Iowa, Illinois, and Indiana) are within 500 m of a field (Tank *et al.*, 2010). Subsurface tile drainage is essential to ensure high crop yields in this low-gradient landscape (Fausey *et al.*, 1995), but tiles couple agricultural fields to adjacent streams and ditches, making this managed landscape a significant contributor of excess fertilizer nitrogen (N) and phosphorus (P) (Goolsby and Battaglin, 2001; Donner *et al.*, 2004; Foley *et al.*, 2005; Alexander *et al.*, 2008), as well as sediments (Carpenter *et al.*, 1998;

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Foley *et al.*, 2005). Additionally, the majority of nutrients and sediments are exported via intermittent large storm events during <10% of the year. Compositionally, >70% of N (Gentry *et al.*, 1998; Royer *et al.*, 2006), >80% of P (Gentry *et al.*, 2007; Sharpley *et al.*, 2008), and up to 90% of sediments (Withers and Sharpley, 2008; Banner *et al.*, 2009; Blann *et al.*, 2009) are exported during these short time periods. Excess nutrients can stimulate algal production in sensitive recipient ecosystems such as the Gulf of Mexico (Turner and Rabalais, 1991; Donner *et al.*, 2004) and Lake Erie (Ohio Environmental Protection Agency, 2013) where eutrophication may result in periodic hypoxia and fish kills (Turner and Rabalais, 1991; Rabalais *et al.*, 2002). Despite this large impact on downstream ecosystems, agricultural headwater streams remain understudied from an ecological perspective (Martin *et al.*, 2012).

Agricultural Midwest Represents a Mosaic of Farming Practices and Drainage Management

Midwest watersheds are a mosaic of on-field farming practices such as swales and tile drains that contribute to the homogenization of recipient streams and ditches (Fisher *et al.*, 2004). To minimize nutrient and sediment runoff from fields, best management practices (BMPs) on the land (e.g., on-field, edge-of-field, and constructed wetlands) and in drainage ditches and streams have been developed and tested in agricultural watersheds (e.g., Osborne and Kovacic, 1993; Mitsch *et al.*, 2001; Tonitto *et al.*, 2006), but implementation has been patchy and it has been difficult to demonstrate improved water quality (Inamdar *et al.*, 2001; Yates *et al.*, 2007; Lemke *et al.*, 2011). In addition, subsurface tile drainage is pervasive, effectively regulating the water table across the Midwest (Dahl, 1990; Sugg, 2007), but tile drain outlets frequently bypass common edge-of-field practices such as grass buffer strips (Osborne and Kovacic, 1993; Fennessy and Cronk, 1997; Mayer *et al.*, 2007). As a result, intensive agriculture in this tile-drained landscape results in the accumulation of excess fertilizer nutrients within stream networks through loading at discrete locations (e.g., tile drains, swales) (Fisher *et al.*, 2004). To minimize nutrient export from fields to adjacent waterways, a suite of BMPs appropriate to the landscape must be implemented strategically within watersheds.

Integrating BMPs with Drainage Management

Integrating aquatic and terrestrial BMPs at the watershed scale requires collaboration between farmers and regional (e.g., county) drainage management. Economic concerns can be a strong motivator for farmers choosing to implement BMPs, but improving watershed health and land stewardship are also important factors (Baumgart-Getz *et al.*, 2012; Reimer *et al.*, 2012). In either case, BMP implementation is frequently patchy due to varying interest in, and eligibility for, United States Department of Agriculture and state-funded conservation programs. Patchy BMP implementation can lead to variable runoff that can influence both quality and quantity of water delivered to streams (Yates *et al.*, 2007; Lemke *et al.*, 2011). At present, drainage is conventionally managed by dredging streams and maintaining them as trapezoidal channels (i.e., ditches; Figure 1B) that maximize hydraulic capacity, but these channels are often unstable and lead to bank collapses that require frequent dredging (Richards *et al.*, 1996; Powell *et al.*, 2007b). Conventional management of channelized agricultural streams and ditches often compromises ecosystem function in order to optimize drainage capacity, and a critical need exists for innovative stream BMPs that improve ecosystem function while maintaining effective drainage.

Two-Stage Ditch Stabilizes Streambanks

The two-stage ditch is an instream management practice that improves bank stability by constructing floodplains adjacent to incised agricultural streams (Figure 1C; Powell *et al.*, 2007a, b). Floodplain benches are constructed without disturbing or modifying the existing stream channel, and tile drain outflows release water onto floodplains rather than directly into the stream, which mimics the effect of lateral wetlands and can retain water on the floodplains for varying lengths of time depending on discharge (*sensu* Osborne and Kovacic, 1993). The two-stage ditch also removes little or no land from agricultural production as floodplains are often constructed using land formerly in vegetated buffer strips, especially if grass buffer strips are already present (Powell *et al.*, 2007b). The two-stage ditch stabilizes formerly channelized streambanks by increasing wetted channel width, slowing water velocity, and decreasing shear stress on benches during floodplain inundation. The net effect of slower water velocity and decreased shear stress is the reduction in bank slumping and undercutting (Powell *et al.*, 2007a, b). The most effective sizing of the two-stage ditch will result in floodplain benches that inundate several times annually during storm events and snowmelt (Kallio, 2010), and can hold in exceedance of 200-year flood event (Jennifer L. Tank, unpublished data, observed 2007), which may have the added benefit of improving water quality.

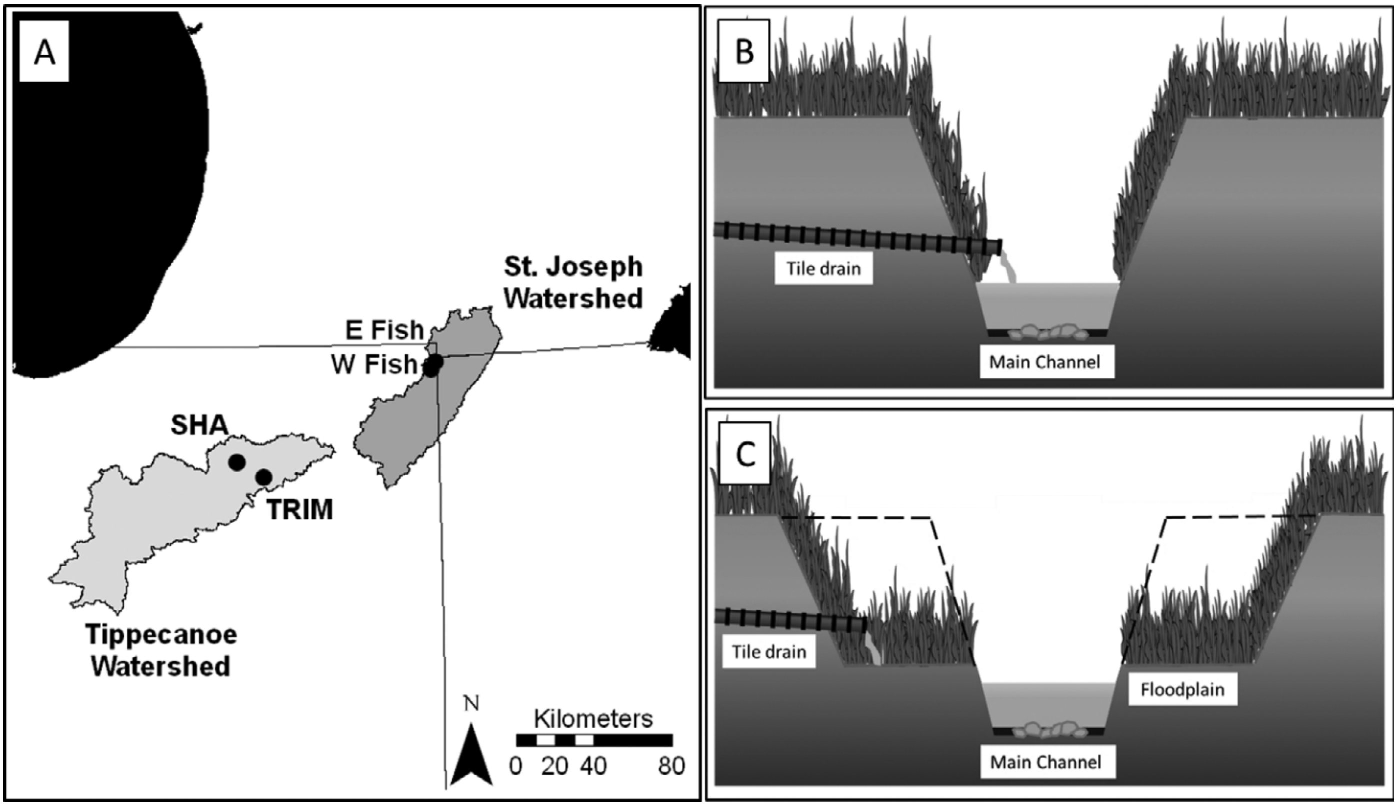

FIGURE 1. (A) Location of Four Study Streams with Two-Stage Ditch Implementation, Denoted by Black Circles, Located in the Tippecanoe and St. Joseph River Watersheds. (B) Cross section of a conventional trapezoidal channel present in upstream reference reaches. (C) Cross section of the two-stage ditch with floodplain bench construction; main channels are not disturbed during construction, and inset channel width to floodplain ratio ranged from 2 to 4:1 among the two-stage reaches in this study.

Two-Stage Has the Potential to Improve Water Quality

In addition to improving channel stability, the two-stage ditch can increase water retention time via floodplain inundation, which can improve water quality via enhanced sediment and nutrient retention. For example, inundated floodplains increase the bioreactive and carbon-rich surface area of the stream channel, which increases the potential for microbially mediated denitrification, a process that permanently removes nitrate (NO_3^--N) from the water column (Knowles, 1982). Recent research has shown that two-stage floodplains enhance denitrification rates, which can remove a significant fraction of NO_3^--N load (Roley *et al.*, 2012a, b; Mahl *et al.*, 2015). Furthermore, the two-stage ditch increases assimilatory uptake of dissolved nutrients into stream biomass resulting in short-term nutrient retention (Roley *et al.*, 2014). On the floodplains, herbaceous vegetation quickly establishes post-construction. Vegetation not only stabilizes streambanks (Hopkinson and Wynn, 2009), but also assimilates nutrients, and further slows water velocity that can increase the deposition of total suspended solids (TSS) and particle-associated total phosphorus (TP) (Powell *et al.*, 2007b). Lastly, the two-stage ditch has been proposed to be "self-cleaning" as increased water velocity in the main channel can flush sediment, decreasing or even eliminating the need for ditch maintenance (Jayakaran *et al.*, 2010; Ward *et al.*, 2007; D'Ambrosio, 2013). However, the potential water quality benefits, as reflected in reductions in water column nutrients and sediments, have yet to be demonstrated at the reach scale.

Study Objectives and Predictions

Our goal was to quantify potential water quality benefits associated with the implementation of the two-stage ditch. We predicted that the two-stage ditch would improve water quality by decreasing water column turbidity and TSS during floodplain inundation by inducing particle settlement as a result of slower water velocities. In addition, we predicted that the two-stage would reduce TP and SRP via increased biomass assimilation or sorption to floodplain sediments. Finally, we predicted that the two-stage would decrease NO_3^--N and ammonium (NH_4^+-N) through increased bioreactive surface area associated with floodplain construction, promoting enhanced assimilatory and dissimilatory uptake (i.e., denitrification).

METHODS

General Site Characteristics

We chose four agricultural streams in northern Indiana, U.S. (Figure 1A, Table 1) and monitored them for two to six years using an upstream-downstream comparative sampling approach to quantify the effect of the two-stage practice on water quality and sediments. The watersheds surrounding each stream are dominated by row-crop agriculture (>70%) and are tile drained with stream channels conventionally managed for effective drainage of surrounding fields, with a trapezoidal channel design, flashy hydrographs, and consistently high dissolved inorganic nutrient concentrations (Table 1). The two-stage reaches in the four streams (Trimble Creek [TRIM], West Fork of Fish Creek [W. Fish], East Fork of Fish Creek [E. Fish], and Shatto Ditch [SHA]) varied in length from 450 to 800 m (Table 1), with floodplain heights ranging from 0.27 to 0.45 m (mean depth above stream bottom; Table 1). Prior to two-stage construction, all four streams had a narrow vegetated buffer strip adjacent to the channel, which ranged in width from 2 to 15 m and was primarily comprised of the invasive reed canary grass (*Phalaris arundinacea*) and rice cutgrass (*Leersia oryzoides*), which dominates the floodplain benches of the two-stage reaches as well. Stream bottom (i.e., benthic) substrate was dominated by two major groups: E. Fish and SHA streams primarily had a mixture of fine benthic organic matter (FBOM), silt, and sand, while TRIM and W. Fish streams were comprised of sand, pebble, and gravel.

Sampling Regime and Continuous Discharge Monitoring

In all four streams, we used a paired upstream-downstream sampling design to quantify any changes in water quality and sediments as a result of two-stage construction. We sampled at the downstream end of each two-stage reach as well as upstream reach immediately above the two-stage reaches as a reference; hereafter we refer to the reaches as "two-stage" and "reference reaches," respectively. We deployed Hydrolab MS5 Minisondes (Hach, Loveland, Colorado) in 4-inch diameter PVC housings at the

TABLE 1. Physical Characteristics of Four Study Streams with Two-Stage Ditch Implementation in Northern Indiana. Included are coordinates of each two-stage reach, watershed area, year of two-stage construction, length of two-stage reach, floodplain height (relative to depth of active channel), median annual stream discharge, flood frequency (events/yr), mean inundation event length (days), D_{50} (mm), and total annual duration of floodplain inundation (day/yr).

Site	Latitude	Longitude	Area (km^2)	Year Constructed	Two-Stage Length (m)	Floodplain Height (cm)	Median Discharge (L/s)	Flood Frequency (events/yr)	Mean Inundation Event Length (days)	Annual Duration of Floodplain Inundation (day/yr)	Richard-Baker Flashiness Index
TRIM	N 41.147442	W 85.880969	20.0	Dec. 2010	500	44	106	6	2.5	17	0.249
W. Fish	N 41.639481	W 84.833039	25.6	Nov. 2009	450	45	157	8	4.8	41	0.320
E. Fish	N 41.669894	W 84.810597	5.3	Nov. 2009	800	38	6	9.3	6.2	59	0.367
SHA	N 41.221528	W 86.044831	10.3	Nov. 2006	600	27	63	10.6	14.5	130	0.400

downstream end of both two-stage and reference reaches. The sondes recorded turbidity (in Nephelometric units), water temperature, conductivity, pH, and dissolved oxygen (DO) at 30-min intervals throughout the study. We downloaded sonde data weekly or bi-weekly. We took grab samples approximately every two weeks at all four streams from July 2010 to July 2012 for dissolved inorganic nutrients (NO_3^--N, NH_4^+-N, and SRP), as well as TP and TSS. At SHA, our sampling for dissolved inorganic nutrients extended from January 2006 to July 2010, with samples taken every two to three weeks. In addition to grab samples, we also conducted sporadic stormflow sampling for TSS and TP using automated water samplers (models 3700 and 6172; ISCO, Lincoln, Nebraska).

At each site, we estimated discharge from stream stage in both the two-stage and reference reaches. We recorded stage at 10-min intervals using capacitance meters (Odyssey, Christchurch, New Zealand) contained within PVC wells placed in the thalweg of each reach. We measured discharge periodically using pulse additions of the conservative tracers NaCl or Rhodamine-WT (Stream Solute Workshop, 1990). Alternatively, on many sampling dates we measured discharge using the partial summation method (Gore, 2007), in which we recorded velocity at 6/10 stream depth using a Marsh-McBirney Flo-Mate 2000 (Flowmate, Grey, Maine) at 0.1 m intervals across a transect. Finally, for all streams, we constructed reach-specific stage-discharge relationships. We also related stream and reach-specific physical parameters (e.g., discharge, mean water velocity, width, and depth) to nearby USGS stream gauge measurements (gauge number 03328000 for SHA and TRIM, number 04177720 for E. Fish and W. Fish) so that we could fill in periodic discontinuities in our stage data (e.g., from storm damage or power failure). To quantify flashiness of the two-stage reach we used the *R-B* index, defined as $R\text{-}B \text{ Index} = \frac{(\sum_{i=1}^{n} |q_i - q_{i-1}|)}{\sum_{i=1}^{n} (q_i)}$ where q was mean daily discharge for day i (Baker *et al.*, 2004).

Field Sampling for Benthic Substrate Composition and Water Chemistry

We characterized benthic substrate composition annually in both the two-stage and reference reaches of each stream using the Wolman pebble count approach (Wolman, 1954; Bunte and Abt, 2001). We established 20 equidistant transects along each reach and identified substrate every 0.1 m across each transect. We identified substrate sizes ranging from 2 to 286 mm using a gravelometer (Forestry Suppliers, Inc., Jackson, Mississippi) and then lumped substrate into five general size classes: FBOM (0.002-0.0625 mm), clay (<0.002 mm), sand (0.01-2 mm), gravel (2-64 mm), and cobble (>64 mm) that best characterized the benthos in these streams (*sensu* Cummins, 1962). We then calculated the median (D_{50}) substrate size class for each site and reach. To compare two-stage to upstream reference reaches, we compared the proportion of substrate coverage in each size class by reach.

For routine water chemistry sampling, at every site visit (at most every three weeks), we collected grab samples for NO_3^--N, NH_4^+-N, SRP, TP, and TSS at the downstream end of the two-stage and reference reaches in each stream. We filtered 60 mL of stream water using syringe-mounted glass fiber filters (nominal 0.45 μm pore size; Pall, Ann Arbor, Michigan) into acid-washed, stream water-rinsed, polyethylene bottles for dissolved inorganic nutrients. In addition, we collected one liter of unfiltered stream water in rinsed, polyethylene bottles for TSS and TP. Stream water samples were stored on ice and transported back to the laboratory; samples for nutrients were then frozen until analysis while TSS samples were refrigerated prior to filtering. Samples frozen for dissolved inorganic nutrients were thawed and analyzed within 24 h.

Laboratory Analyses

We analyzed water samples for inorganic nutrient concentrations using a Lachat QC8500 Flow Injection Autoanalyzer (FIA; Lachat Instruments, Loveland, Colorado). We analyzed NO_3^--N using the cadmium reduction method (APHA, 1995), NH_4^+-N using the phenol-hypochlorite method (Solórzano, 1969), and SRP was quantified using the ascorbic acid method (Murphy and Riley, 1962). For TP analysis, we used a modified persulfate digestion to oxidize all P to SRP (APHA, 1995). Briefly, we re-suspended unfiltered water samples (using stir plate and magnet) prior to taking an 8.0 mL subsample. Samples were then amended with 3.2 mL of 0.185 M potassium persulfate followed by 160 μL of 3.75 N NaOH within one minute, sealed with polyethylene septa (Qorpack, Bridgeville, Pennsylvania), and shaken to mix reagents. Digestions for TP were performed in 22-mL acid-washed, glass scintillation vials (Qorpack) and we began the digestions within 30 min of persulfate and NaOH addition, autoclaving samples at 121°C, 17 psi for 45 min. We allowed samples to cool, then filtered to remove particulates (glass fiber filter, nominal 0.45 μm pore size; Pall) into a borosilicate vial for analysis using the ascorbic acid method for SRP as described above.

We measured TSS by filtering samples through pre-ashed glass fiber filters (nominal 0.45 μm pore size; Pall) until filters clogged, or the full volume of collected sample (~900 mL) was filtered (APHA, 1995). We rinsed the filters with reagent-grade reverse osmosis (RO) water to remove sorbed constituents not part of the TSS fraction. We then dried filters at 70°C for a minimum of 72 h prior to weighing; samples were reweighed twice with a minimum of 24 h drying between measurements. To determine ash-free dry mass (AFDM), we combusted samples at 450° for four hours, wetted with reagent-grade water, dried, and weighed. We calculated AFDM and as TSS dry weight minus combusted TSS dry weight.

Statistical Analyses

We performed statistical analyses using R (R Development Core Team, 2012) to analyze differences in nutrients and sediments between two-stage and reference reaches, as well as among streams, and we deemed $p < 0.05$ to represent statistically significant differences. To meet the assumptions of parametric statistics, we transformed all nutrient data using log base-10 or square-root transformations to ensure normality (Shapiro-Wilk test; $p > 0.05$). We used paired one-tailed *t*-tests to identify significant differences between NO_3^--N, NH_4^+-N, TSS, TP, substrate, and turbidity between the two-stage and upstream reference trapezoidal reaches. We considered whether outliers would significantly impact the mean (average) of a given dataset in a way that would skew results, which was especially important to consider given that we are trying to draw conclusions about the efficacy of the two-stage practice to influence water chemistry based on those means. Therefore, we removed points that were statistical outliers when their distance from the median exceeded three times the interquartile range. These data points were generally infrequent, and influenced <10% of the dataset. We used simple linear regression (SLR) to examine relationships between floodplain bench height and inundation frequency, turbidity and TSS, turbidity and TP, and TSS and TP. We used an ANCOVA to examine the interaction of TSS and TP, using both stream and reach as the covariates.

RESULTS

Two-Stage Floodplains Are Inundated due to Flashy Stream Discharge

The four streams exhibited typical hydrology for agriculturally impacted midwestern systems, as discharge varied widely within and among sites. Discharge ranged from <1 to >2,000 L/s and spanned three orders of magnitude in each stream, while the median discharge ranged from 6 L/s in E. Fish to 157 L/s in W. Fish (Table 1). In general, highest discharges occurred during spring snowmelt and storms, although infrequent storms with high rainfall in mid- to late-summer increased discharges similarly between reaches and watersheds receiving a similar amount of precipitation. We characterized precipitation in the region using data from local weather stations within 15 km from each site. Additionally, the range in discharge was highly variable among calendar years, largely as a function of precipitation. All study streams experienced drought conditions in 2012, characterized by lower discharge, verified using long-term USGS gauging records; unusually low stream discharges in 2012 reflected decreased regional snowpack and precipitation throughout the year. We also used the R-B index as a relative metric of flashiness (Baker *et al.*, 2004) with values closer to 0 denoting predominantly base-flow conditions and higher values denoting large daily variation in discharge. Based on the R-B index, all four agricultural streams were flashy, with the index ranging from 0.240 to 0.400; the R-B index was consistent across years despite considerable variation in mean discharge.

Given these flashy discharge regimes, we also quantified the frequency and duration of floodplain inundation in the two-stage reaches in the four streams as we hypothesized that improvement in water quality would occur primarily during inundation events. We found that floodplains were inundated an average of 6-11 times per year among the four sites, primarily during spring and early summer. We estimated floodplain wetting using stream-specific rating curves, and confirmed these with capacitance meters placed directly on floodplains. Despite relatively little variation in the number of inundation events across sites and years, the total number of days that floodplains were inundated varied significantly across sites and years. The duration of individual inundation events ranged from 1 to 60 days, and annual inundation duration ranged from 17 day/yr in TRIM to 130 day/yr in SHA (Table 1). As predicted, total annual inundation duration was inversely related to floodplain height (SLR, $p < 0.05$, $r^2 = 0.863$, df = 2) with lower floodplains being inundated longer. In addition, we found that the mean flood frequency (number of events per year) was positively correlated with R-B Flashiness Index rankings (SLR, $p < 0.05$, $r^2 = 0.984$, df = 2), meaning that flashier systems inundated more frequently. Thus, the number of high-flow events (quantified by R-B index) constrained the frequency of floodplain inundation events, but bench height most strongly affected the duration of floodplain inundation.

Benthic Substrate Composition Varied by Stream, but Not Influenced by the Two-Stage

We examined whether the two-stage ditch altered the substrate composition in the main channel. Ward *et al.* (2010) hypothesized that water velocity would increase in the main channel of the two-stage ditch at the beginning of an inundation event, which could flush fine particles from the stream bottom. Using annual substrate surveys, we found that floodplain construction did not alter substrate composition in the main channel. The predominance of finer substrates, FBOM and clay, was not statistically different between the reference and two-stage reaches at any of the four sites (Figure 2; paired *t*-test, $p > 0.05$), but sand did decrease in the two-stage reaches relative to upstream reference reaches (Figure 2; paired *t*-test, $p < 0.05$). In contrast, the proportion of coarser substrates, including gravel and cobble, decreased in the two-stage reach at all sites except TRIM (Figure 2; paired *t*-test, $p < 0.05$). In the two streams with lower discharge, SHA and E. Fish, the streams were dominated by FBOM ($D_{50} < 0.0625$ mm). In contrast, we found that the two streams with higher discharge, TRIM and W. Fish, were dominated by coarser substrate (i.e., $D_{50} = 2$ mm) with the thalweg containing very little FBOM. Across all four sites, FBOM content was significantly lower during normal water years compared to 2012, when there were fewer storms and resultant stream discharge was lower (2010 *vs.* 2012 comparison; paired *t*-test, $p < 0.05$), and FBOM content was negatively correlated with mean annual discharge (SLR, $p < 0.05$).

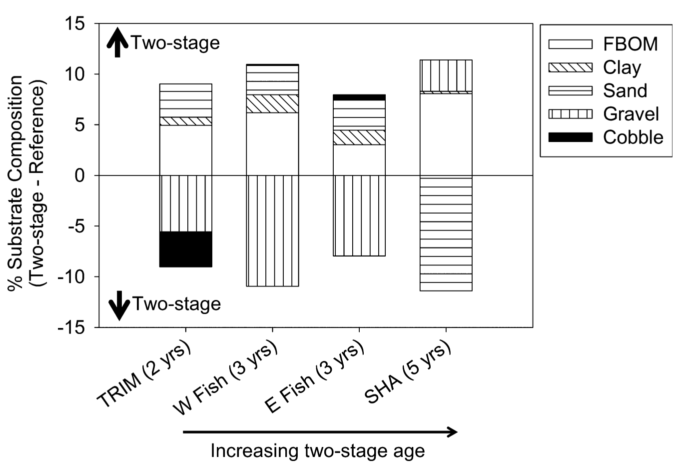

FIGURE 2. Stream Substrate Composition, Shown as the Difference between the Two-Stage and Reference Reaches during Summer 2012 for Five Substrate Sizes (fine benthic organic matter — FBOM; clay; sand, <2 mm; gravel, 2-32 mm; cobble, >32 mm). Coarse substrate did not generally increase in the two-stage reach relative to the upstream reference reach (paired *t*-test, $p > 0.05$) and substrate size varied interannually.

Two-Stage Construction Decreased Water Column Turbidity

We hypothesized that two-stage construction would reduce water column turbidity because water velocity is slower over inundated floodplains and could promote sediment deposition and increase water clarity. Using Minisonde data, we found that turbidity was significantly lower in three of the four two-stage reaches during floodplain inundation (E. Fish, SHA, and W. Fish) compared to their respective upstream reference reaches (Figure 3; paired *t*-test, $p < 0.05$). Turbidity was variable both across and within streams, and ranged from 0 to >3,000 NTU (above detection), but highest turbidity occurred during high discharge events (e.g., spring snowmelt, large thunderstorms). The two-stage reach reduced water column turbidity by 29 and 47% in both SHA and E. Fish, which were the streams dominated by finer substrates. In contrast, in the streams with coarser sediments, turbidity was reduced by just 15% in W. Fish, and not at all in TRIM, which also had the highest floodplain bench height and shortest inundation duration. In general, the shorter the duration of floodplain inundation, the lower the potential for sediment retention, and there is no potential for sediment retention on the two-stage floodplains during base-flow conditions. In contrast to turbidity measured during inundation events, we also found that base-flow turbidity differed among sites, and was statistically lower in the two-stage reaches at E. Fish

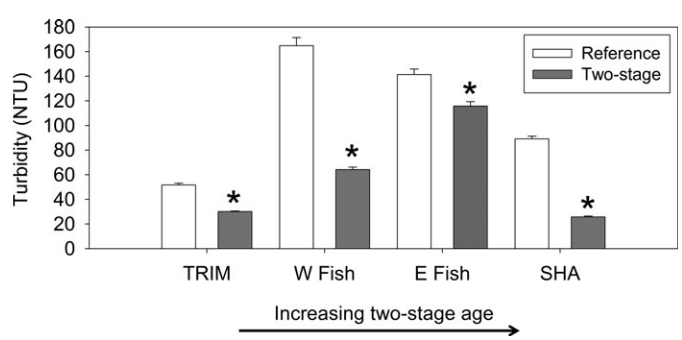

FIGURE 3. Mean Turbidity (± SE) of Paired Sampling Events during Floodplain Inundation, Measured at 0.5 h Intervals. Turbidity was significantly lower in two-stage reaches relative to the upstream reference reach (paired *t*-test, $p < 0.05$) and * denotes statistical significance.

and W. Fish (paired t-test, $p < 0.05$), but higher in TRIM and SHA.

The Influence of Two-Stage Construction on Sediments and Water Chemistry

Because we found that turbidity was significantly lower in three of the four two-stage reaches during floodplain inundation, we predicted that TSS would also decrease as suspended solids are the major contributors to water column turbidity. However, we found no correlation between turbidity and TSS either within (i.e., two-stage *vs*. reference reaches) or among streams (SLR and ANCOVA, $p > 0.05$, data not shown). In examining streams individually, the two-stage reach significantly decreased TSS only in SHA (paired t-test, $p < 0.05$, data not shown), but did not reduce TSS when all data were pooled. Nevertheless, we also examined turbidity as a potential proxy measurement for predicting TSS and TP, which cannot currently be continuously monitored. We found that turbidity did not correlate with TSS or TP at any site (SLR, $p > 0.05$, data not shown). However, TSS was positively correlated with TP at each site (Figure 4; SLR, $p < 0.05$, n = 31-69), consistent in both the reference and two-stage reaches.

Given that the two-stage decreased turbidity, and TSS was related to TP, we predicted that two-stage construction would also influence P concentrations (as TP and SRP). Using routine grab samples to compare reference and two-stage reaches, we found that TP and SRP concentrations were extremely variable among streams and over time. Mean TP was 105 μg/L and ranged from 0 to 1,528 μg/L while mean SRP was 40 μg/L and ranged from 0 to 762 μg/L (Table 2). We found that both SRP and TP decreased in the two-stage reach at SHA (Figures 5A and 5B; paired t-test, $p < 0.05$), but no P reduction was observed in the two-stage reach at the other three streams. Overall, SRP concentrations were significantly lower in the two-stage reaches when data were pooled for all four streams (paired t-test, $p = 0.004$) but TP was not different between reference and two-stage reaches (paired t-test, $p > 0.05$).

We also examined the influence of two-stage construction on concentrations of NO_3^--N and NH_4^+-N during floodplain inundation as increased bioreactive surface area associated with floodplain construction has been shown to increase N removal (Roley *et al.*, 2012a, b). In general, NH_4^+-N concentrations were much lower than NO_3^--N; NH_4^+-N ranged from 0 to 2,846 μg/L with a mean of 83 μg/L, and NO_3^--N ranged from 0.01 to 13.0 mg/L with a mean of 3.6 mg/L (Table 2). In comparing the two-stage to upstream reference reaches, we found that NO_3^--N concentrations were lower in the two-stage reach when data were pooled among streams (paired t-test, $p = 0.04$); this pattern was driven by reductions at W. Fish. In contrast, NH_4^+-N concentrations did not change in response to two-stage implementation (Figure 5D; paired t-test, $p > 0.05$), although NH_4^+-N concentration was also lower in the two-stage at W. Fish (paired t-test, $p = 0.002$).

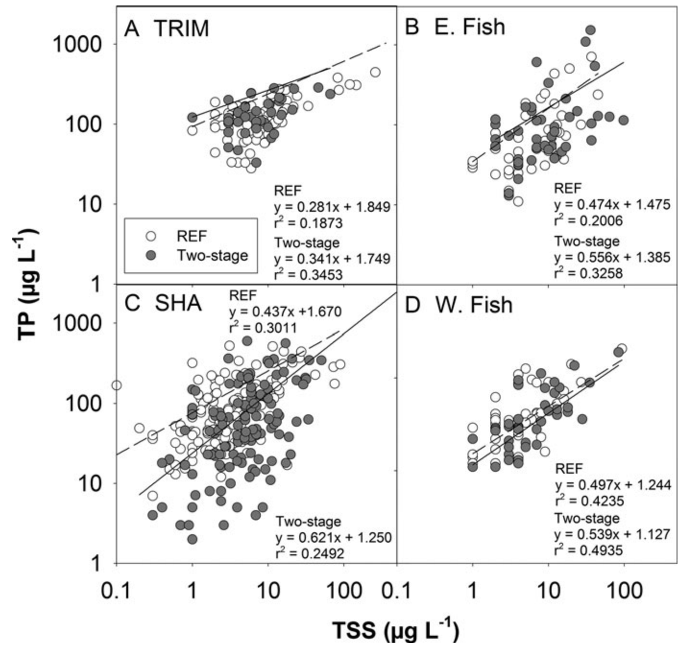

FIGURE 4. Correlation between Total Suspended Solids (TSS) and Total Phosphorus (TP) in the Four Agricultural Streams for Hydrological Conditions Ranging from Base Flow to Floodplain Inundation; TSS Was Positively Correlated with TP within Reach Types (reference = dashed line, two-stage = solid line; SLR; $p < 0.05$), and Correlations Did Not Differ among Reaches (ANCOVA; TP * reach, $p > 0.05$) Except at SHA (ANCOVA; TP * reach, $p < 0.05$). The correlations between TSS and TP were stream-specific (ANCOVA, TP * stream; $p < 0.05$).

DISCUSSION

Potential for Two-Stage to Improve Water Quality in Agricultural Streams

Few studies have documented post-restoration monitoring to quantify the effect of stream restoration (Bernhardt *et al.*, 2005; Bernhardt and Palmer, 2011), but completing these studies are critical in managing freshwater in human-impacted landscapes. The effects of restoration and management can vary across the landscape and can change over time. Our objective was to determine whether the two-stage ditch improves water quality in agricultural streams.

TABLE 2. Chemical Characteristics, Reported in μg/l, for Reference and Two-Stage Reaches in Four Study Streams Reported as Mean ± Standard Error (SE). Dissolved nutrients, total phosphorus (TP), and total suspended solids (TSS) were measured over a 1.5-6 year period, with sample number ranging from 35 to 106 samples per stream for dissolved nutrients and 31-69 samples per stream for TP and TSS.

		NO_3^--N		NH_4^+-N		SRP		TP		TSS	
Site	**Reach**	**Mean ± SE**	**Range**	**Mean ± SE**	**Range**	**Mean ± SE**	**Range**	**Mean ± SE**	**Range**	**Mean ± SE**	**Range**
E. Fish	REF	1,072 (131)	2-4,512	103 (56)	6-2,846	28 (7)	3-475	73 (14)	11-711	8 (1)	1-37
	TRT	1,037 (134)	1-6,029	107 (56)	3-2,740	30 (6)	3-762	79 (15)	13-1,528	9 (1)	2-36
SHA	REF	5,760 (285)	549-10,447	49 (6)	7-1,733	22 (2)	6-236	95 (10)	19-324	5 (1)	0-21
	TRT	5,762 (291)	553-10,985	58 (4)	0-1,493	16 (1)	0-313	83 (10)	20-364	6 (1)	1-14
TRIM	REF	5,039 (353)	612-8,467	51 (15)	3-509	52 (8)	10-189	119 (10)	28-379	6 (1)	1-17
	TRT	5,052 (362)	571-8,549	49 (14)	3-489	50 (7)	10-318	121 (10)	33-286	8 (1)	1-22
W. Fish	REF	482 (74)	6-3,148	34 (8)	7-424	15 (2)	2-89	49 (5)	12-207	8 (2)	1-93
	TRT	472 (79)	9-3,539	31 (8)	2-421	14 (2)	3-88	45 (4)	11-190	10 (2)	1-84
All streams	REF	2,618 (197)	2-10,447	57.5 (13.3)	3-2,846	26 (2.4)	2-475	80.8 (5.3)	11-711	6.8 (0.7)	0-93
	TRT	2,610 (199)	1-10,985	60.8 (13.2)	0-2,740	24.5 (2.3)	0-762	77.9 (5.4)	11-1,528	8.2 (0.7)	1-98

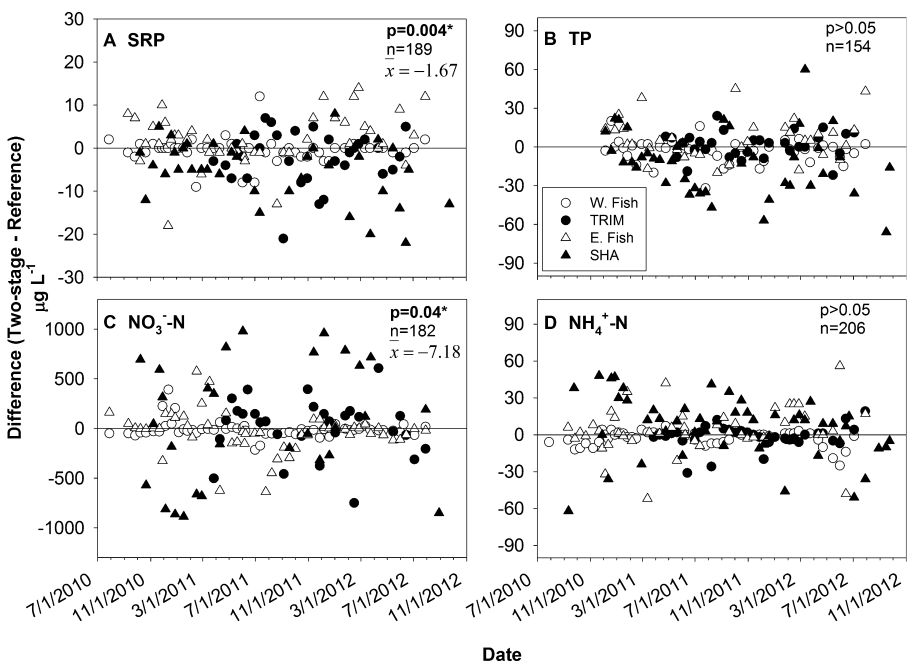

FIGURE 5. Differences in Concentration between the Two-Stage Reach and the Reference Reach for (A) Soluble Reactive Phosphorus (SRP), (B) Total Phosphorus (TP), (C) NO_3^--N, and (D) NH_4^+-N over the Course of the Monitoring Period for the Four Agricultural Streams. Sampling dates include base-flow conditions, storm events, and floodplain inundation.

We found that restoration of floodplains through two-stage ditch construction had the potential to reduce sediment and P loads but the effectiveness of the restoration varied over the two years and differed among sites based on variation in bench design and management of adjacent fields. However, while NO_3^--N concentration was reduced overall when data were pooled among all the two-stage reaches, this pattern was driven by the significant reduction in W. Fish. Nitrate concentrations in other streams were likely too high to be significantly reduced by the relatively short two-stage reaches examined in this study (450-800 m), or due to continued NO_3^--N gains within a reach, despite increased sediment denitrification rates previously measured on constructed floodplains (Roley *et al.*, 2012a, b; Mahl *et al.*, 2015). Our two-year study conducted in four agricultural streams with the two-stage implementation allowed us to quantify potential benefits of this new practice, highlighting the positive impact of instream BMPs on water quality in agricultural watersheds.

Two-Stage Improves Water Column Turbidity, but Does Not Change Benthic Substrate Composition

We found that the two-stage ditch decreased turbidity in the two-stage reach under both base-flow and storm-flow conditions in three of the four streams (Figure 3). We also examined whether turbidity could be used as a high-frequency surrogate for estimating TSS (e.g., Gippel, 1995; Prestigiacomo *et al.*, 2007; Jones *et al.*, 2011) in order to examine the ability of the two-stage ditch to "self-clean" the main channel of excess fine sediments. Previous research has shown that the two-stage ditch could serve this function by (1) slowing water velocity over floodplains which would allow sediments to accumulate on floodplains and (2) by increasing water velocity in the main channel thalweg (Figure 1C) thereby flushing sediments when floodplains are inundated (Powell *et al.*, 2007a, b; Ward *et al.*, 2007). We found that turbidity was not correlated linearly with TSS, but rather exhibited a pattern of hysteresis (data not shown) as also previously shown (Krueger *et al.*, 2009). This complex pattern of rise-and-fall of suspended solids with storm events makes the use of turbidity as a metric to estimate TSS more complicated, and site-specific relationships would need to be developed to conduct such monitoring. Nevertheless, our use of high-frequency turbidity measurements in these dynamic agricultural streams allowed us to isolate and quantify the benefits of the two-stage ditch in reducing stream water turbidity associated with agricultural land use. For example, increased light penetration could lead to higher assimilatory N and P uptake by primary producers (Roley *et al.*, 2014).

Using annual benthic substrate surveys in the four study streams, we found that implementation of the two-stage ditch did not increase coarse substrate availability in the main channels. Upon close inspection of survey data, we found coarse substrate actually decreased in three of the four 2-stage reaches which was opposite to our prediction that the two-stage would be "self-cleaning" and flush fine sediments from the main channel during floodplain inundation, leaving coarser substrates behind (Powell *et al.*, 2007a, b). The lack of improved substrate is most likely because midwestern agricultural streams generally have low substrate variability (Fischer *et al.*, 2010; Smiley *et al.*, 2011) and coarse substrate may not be present and available for exposure. We identified a longitudinal pattern in the two streams most dominated by fine substrates (SHA and E. Fish). In these streams, the occurrence of sand increased in the two-stage reach, and these two streams also showed the largest decline in turbidity at the reach scale. Data from the sub-reach scale suggest that two-stage streams dominated by fine substrate may indeed be self-cleaning, but sediment mobilized from the upstream trapezoidal reach may be delivered into the two-stage reach, masking reach-scale benefits. If implemented at a larger scale (i.e., to the headwaters), floodplain construction using the two-stage may increase coarse substrate in main stream channels, if coarse substrate is present and could be exposed.

Suspended Sediment Reductions due to the Two-Stage May Also Reduce TP Loads

Total suspended solids were positively correlated with TP in all four streams (Figure 4) but the relationship was stream specific, although in three of the four streams, there were no differences between the slope of the relationship in the upstream reference and two-stage reaches. The consistency of the TSS-TP relationship suggests that P is likely sorbed to suspended sediments, and sediment-bound P is not changing over short distances (<800 m). Previous research has shown that TP concentrations are strongly associated with TSS across a range of land-use types within a watershed (e.g., Grayson *et al.*, 1996; Krueger *et al.*, 2009). Our results further suggest the need for stream-specific TSS-TP relationships in order to accurately estimate stream-specific P fluxes and export when using TSS as a proxy measurement for TP.

Using periodic grab samples to compare the reference to two-stage reaches in four streams, we found that two-stage implementation decreased TSS and TP concentrations only in SHA, but not in the other streams (Figure 5A). It is notable that floodplain height was the lowest in SHA, and thus this stream had the highest floodplain inundation frequency and longest inundation duration (Table 1), which optimized the opportunity for floodplains to induce sediment settling. In contrast to SHA, E. Fish had higher TP concentrations in the two-stage reach, likely associated with manure application in the field directly adjacent to the two-stage reach, but not applied upstream, which has been shown to increase P runoff during precipitation events (Sharpley *et al.*, 1994; Sharpley and Moyer, 2000; Borda *et al.*, 2011). The remaining two streams (TRIM and W. Fish) also did not have TSS or TP reductions in the two-stage reaches; these were deeper streams with more flow, higher floodplain benches with less frequent inundation, and shorter inundation duration. These floodplains were less likely to promote sediment deposition, consistent with what has previously been seen in deeper, increased streams with lower likelihood of sediment and P retention (Olli *et al.*, 2009). These trends in stream-specific TSS and TP retention suggest the widths of the two-stage floodplain must be the appropriate size for the stream or else efficacy may decline. All four two-stage ditches in this study met the minimum dimensions as identified by Powell *et al.* (2007b), suggesting that floodplain height may be the best predictor of constructed floodplains to retain nutrients and sediments.

We predicted that P would be retained in vegetation or associated with floodplain soils either adsorbed to particles or incorporated into microbial biomass. Rewetting after previously dry periods frequently mobilize P from wetland and floodplain soils (Krueger *et al.*, 2009; Tanner and Sukias, 2011; Knowles *et al.*, 2012; Schönbrunner *et al.*, 2012) but increased inundation frequency could counteract drying effects by maintaining low redox conditions that prevent P release. In addition to the influence of P mobility, P equilibrium concentration and the ability of soils to retain P could affect net retention by floodplain soils. For example, Liu *et al.* (2013) found that a two-stage ditch had lower equilibrium P concentrations, and lower potential P sorption capacity. Where P reduction is an objective of two-stage implementation, P sorption capacity should be measured because floodplains are usually constructed in C and D soil horizons, which may have low P sorption capacity (Anderson, 1988; Jayakaran *et al.*, 2010), that may limit the potential for recently excavated floodplains to retain P.

Two-Stage Ditch Can Reduce SRP and NH_4^+-N Concentrations

We found that the two-stage ditch decreased SRP and NH_4^+-N concentrations in one of the four streams. Previous research has shown that both SRP and NH_4^+-N sorb preferentially to fine substrates (Meyer, 1979; Newbold *et al.*, 1983a, b); sorption could serve as one retention mechanism for these species both on fine sediments in main channels at base flow and on floodplain soils during inundation. The potential is high that sorption is retaining SRP and NH_4^+-N in the two-stage, but the relative role of abiotic retention remains unclear.

An alternative mechanism for the decrease in SRP and NH_4^+-N in the two-stage reaches is assimilatory uptake into biomass such as microbes, algae, and plants. Both SRP and NH_4^+-N are highly bioavailable dissolved species and thus are readily assimilated, and strongly correlated with ecosystem functional metrics such as nutrient uptake and stream metabolism (Mulholland *et al.*, 2001; Webster *et al.*, 2003). Despite relatively high concentrations, saturation of SRP and NH_4^+-N uptake appears to be uncommon in agricultural streams (Bernot *et al.*, 2006), and high primary producer biomass (e.g., filamentous algae or submerged macrophytes) is common in agricultural streams and may be a significant sink for nutrients in agricultural streams (Bernot *et al.*, 2006; Roley *et al.*, 2014). Several potential mechanisms exist for NH_4^+-N and SRP retention in agricultural streams; we would expect to see reductions where nutrient loads are moderate but not in streams with high N or P loads.

When Loading Is High, the Two-Stage Ditch Cannot Decrease NO_3^--N

We were able to document a reduction in NO_3^--N concentration among all the streams (Figure 5C), and this trend was driven by a significant reduction at W. Fish and a marginally significant reduction at E. Fish, and both of these streams were characterized by generally lower NO_3^--N concentrations. Previous research has demonstrated that two-stage floodplains significantly enhance NO_3^--N removal via denitrification, with potential rates being high, as well as increased bioreactive surface area being added as part of floodplain construction (Roley *et al.*, 2012a, b; Mahl *et al.*, 2015). Despite higher reach-scale N removal via denitrification, our research suggests that the two-stage ditch practice implemented at the reach scale is insufficient as a stand-alone BMP to reduce NO_3^--N when concentrations are high (e.g., >1 mg/L). However, in a previous study, Roley *et al.* (2012a) found that up to 47% of NO_3^--N loads were removed via denitrification in SHA, and yet, we observed no difference in NO_3^--N concentration or flux in this stream. In order to make the two-stage ditch effective at NO_3^--N reduction in such watersheds, additional management practices that reduce N inputs to streams from the surrounding watershed would need to be implemented as well.

Wetland and stream BMPs that are designed to remove NO_3^--N rely on a large bioreactive surface area and extended water residence times to maximize dissimilatory NO_3^--N removal (i.e., denitrification). However, NO_3^--N concentrations in runoff from conventional row-crops are high (commonly 2-10 mg/L) and when coupled with tile drainage that reduces water residence times and shunts drainage water directly to stream channels, results in insufficient NO_3^--N removal to prevent downstream eutrophication. Decreasing the NO_3^--N concentration in runoff from agricultural fields through the use of on-field BMPs (e.g., cover crops, nutrient management plans) is needed in order to optimize the mitigation potential of aquatic BMPs such as the two-stage ditch.

The agricultural landscape is heterogeneous and thus requires coordinated and targeted implementation of multiple BMPs, rather than relying on any single BMP as a "silver bullet." We recognize that while agricultural streams are ubiquitous within the landscape, the two-stage is not suitable for mitigating excess agricultural nutrients in every case and farm-specific management practices (e.g., timing of manure and fertilizer application) can impact the efficacy of any BMP. Balancing landscape practices and the costs of alternative BMPs such as wetlands and cover crops will be essential to effectively implement a stacked suite of BMPs to optimize water quality in adjacent and downstream ecosystems (Christianson *et al.*, 2013; Tomer *et al.*, 2013). Watershed-scale BMP implementation remains relatively rare, but where it has been achieved has shown some tremendous successes (Cook *et al.*, 1996; Yates *et al.*, 2007), as well as negligible impact (Lemke *et al.*, 2011). Finally, it is critical that managers identify hot spots of nutrient and sediment mobilization within agricultural watersheds in order to effectively target BMP implementation (e.g., Heathwaite *et al.*, 2000; Mayer *et al.*, 2007; Tomer *et al.*, 2010).

Best Practices to Optimize Water Quality Improvements from Two-Stage Ditch Implementation

The results from our two-year study across four streams demonstrate that the two-stage ditch has the potential to decrease N, P, and sediment export from agricultural streams, but proper floodplain bench height is critical to optimize floodplain inundation frequency and duration, which can lead to maximal water quality benefits. Early two-stage ditches were sized to target a specific flood recurrence interval to most effectively stabilize banks (Powell *et al.*, 2007a); however, this flood recurrence interval is less frequent than what would be best to optimize nutrient and sediment retention. Prior to two-stage implementation, a compromise on inundation frequency must be struck between hydrologists who use two-stage sizing tools based upon regional curves (Mecklenburg and Ward, 2008) and water quality managers who may desire nutrient mitigation benefits. Active collaboration between hydrologists and biogeochemists can result in the construction of stable floodplains that inundate frequently (i.e., SHA), which creates optimal conditions to mitigate runoff in agricultural streams.

For example, in studying four streams with different histories leading to two-stage implementation, we found consistent reductions in TSS, TP, and SRP only in SHA where floodplain height was designed to maximize floodplain inundation frequency and duration, a result achieved by a collaboration between hydrologists and biogeochemists. The other three two-stage ditches studied were constructed solely using the two-stage sizing tool (Mecklenburg and Ward, 2008) and flooded much less frequently and for shorter periods. These results, while anecdotal, provide evidence that the efficacy of the two-stage ditch practice for improving water quality depends on optimal two-stage sizing to maximize floodplain inundation.

CONCLUSION

Targeted implementation of new conservation practices (e.g., two-stage ditch), paired with other landscape BMPs, has the potential to improve water quality in agricultural watersheds. A watershed-scale management approach has the greatest likelihood of success (Yates *et al.*, 2007; Delgado *et al.*, 2011; Tomer *et al.*, 2013), but this systems approach will require the identification of nutrient-specific reduction criteria and appropriate "stacking" of BMPs. While the two-stage ditch was initially designed to stabilize streambanks, this practice may reduce nutrient and sediment export from watersheds, and could be helpful in treating agricultural runoff, which frequently bypasses riparian buffer strips via subsurface tile drains (Osborne and Kovacic, 1993; Mayer *et al.*, 2007). The two-stage ditch can augment ecosystem services provided by additional instream habitat, and could maximize the underutilized riparian zone of agricultural streams, thus improving the "kidneys" of the agricultural landscape (Fennessy, 1993). The two-stage ditch can also be effectively combined with other landscape conservation practices that reduce nutrient and sediment export to waterways, maximizing the role agricultural streams can play in improving water quality within agricultural watersheds.

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LITERATURE CITED

Alexander, R.B., R.A. Smith, G.E. Schwarz, E.W. Boyer, J.V. Nolan, and J.W. Brakebill, 2008. Differences in Phosphorus and Nitrogen Delivery to the Gulf of Mexico from the Mississippi River Basin. Environmental Science & Technology 42:822-830.

Anderson, D.W., 1988. The Effect of Parent Material and Soil Development on Nutrient Cycling in Temperate Ecosystems. Biogeochemistry 5:71-97.

APHA, 1995. Standard Methods for the Examination of Water and Wastewater. American Public Health Association, Washington, D.C.

Baker, D.B., R.P. Richards, T.T. Loftus, and J.W. Kramer, 2004. A New Flashiness Index: Characteristics and Applications to Midwestern Rivers and Streams. Journal of the American Water Resources Association 40:503-522.

Banner, E.B.K., A.J. Stahl, and W.K. Dodds, 2009. Stream Discharge and Riparian Land Use Influence In-Stream Concentrations and Loads of Phosphorus from Central Plains Watersheds. Environmental Management 44:552-565.

Baumgart-Getz, A., L.S. Prokopy, and K. Floress, 2012. Why Farmers Adopt Best Management Practice in the United States: A Meta-Analysis of the Adoption Literature. Journal of Environmental Management 96:17-25.

Bernhardt, E.S. and M.A. Palmer, 2011. River Restoration: The Fuzzy Logic of Repairing Reaches to Reverse Catchment Scale Degradation. Ecological Applications 21:1926-1931.

Bernhardt, E.S., M.A. Palmer, J.D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G.M. Kondolf, P.S. Lake, R. Lave, J.L. Meyer, T.K.O. O'Donnell, L. Pagano, B. Powell, and E. Sudduth, 2005. Synthesizing U.S. River Restoration Efforts. Science 308:636-637.

Bernot, M.J., J.L. Tank, T.V. Royer, and M.B. David, 2006. Nutrient Uptake in Streams Draining Agricultural Catchments of the Midwestern United States. Freshwater Biology 51:499-509.

Blann, K.L., J.L. Anderson, G.R. Sands, and B. Vondracek, 2009. Effects of Agricultural Drainage on Aquatic Ecosystems: A Review. Critical Reviews in Environmental Science and Technology 39:909-1001.

Borda, T., L. Celi, L. Zavattaro, D. Sacco, and E. Barberis, 2011. Effect of Agronomic Management on Risk of Suspended Solids and Phosphorus Losses from Soil to Waters. Journal of Soils Sediments 11:440-451.

Bunte, K. and S.R. Abt, 2001. Sampling Frame for Improving Pebble Count Accuracy in Coarse Gravel-Bed Streams. Journal of the American Water Resources Association 37:1001-1014.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith, 1998. Nonpoint Pollution of Surface Waters with Phosphorus and Nitrogen. Ecological Applications 8:559-568.

Christianson, L., J. Tyndall, and M. Helmers, 2013. Financial Comparison of Seven Nitrate Reduction Strategies for Midwestern Agricultural Drainage. Water Resources and Economics 2-3:30-56.

Cook, M.G., P.G. Hunt, K.C. Stone, and J.H. Canterberry, 1996. Reducing Diffuse Pollution Through Implementation of Agricultural Best Management Practices: A Case Study. Water Science and Technology 33:191-196.

Cummins, K.W., 1962. An Evaluation of Some Techniques for the Collection and Analysis of Benthic Samples with Special Emphasis on Lotic Waters. American Midland Naturalist 67:477-504.

Dahl, T.E., 1990. Wetlands: Losses in the United States 1780s to 1980s. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C., 13 pp.

D'Ambrosio, J.L., 2013. Perspectives on the Geomorphic Evolution and Ecology of Modified Channels and Two-Stage Ditches in the Agriculturally-Dominated Midwestern United States. The Ohio State University, Columbus, Ohio.

Delgado, J.A., R. Khosla, and T. Mueller, 2011. Recent Advances in Precision (Target) Conservation. Journal of Soil and Water Conservation 66:167A-170A.

Donner, S.D., C.J. Kucharik, and J.A. Foley, 2004. Impact of Changing Land Use Practices on Nitrate Export by the Mississippi River. Global Biogeochemical Cycles 18:GB1028.

Fausey, N.R., L.C. Brown, H.W. Belcher, and R.S. Kanwar, 1995. Drainage and Water Quality in Great Lakes and Cornbelt States. Journal of Irrigation and Drainage Engineering 121:283-288.

Fennessy, M.S. and J.K. Cronk, 1997. The Effectiveness and Restoration Potential of Riparian Ecotones for the Management of Nonpoint Source Pollution, Particularly Nitrate. Critical Reviews in Environmental Science and Technology 27:285-317.

Fennessy, S., 1993. Riparian Buffer Strips: Their Effectiveness for the Control of Agricultural Pollution. World Wide Fund for Nature Project, London.

Fischer, J.R., M.C. Quist, S.L. Wigen, A.J. Schaefer, T.W. Stewart, and T.M. Isenhart, 2010. Assemblage and Population-Level Responses of Stream Fish to Riparian Buffers at Multiple Spatial Scales. Transactions of the American Fisheries Society 139:185-200.

Fisher, S.G., R.A. Sponseller, and J.B. Heffernan, 2004. Horizons in Stream Biogeochemistry: Flowpaths to Progress. Ecology 85:2369-2379.

Foley, J.A., R. DeFries, G.P. Asner, C. Barford, G. Bonan, S.R. Carpenter, F.S. Chapin, M.T. Coe, G.C. Daily, H.K. Gibbs, J.H. Helkowski, T. Holloway, E.A. Howard, C.J. Kucharik, C. Monfreda, J.A. Patz, I.C. Prentice, N. Ramankutty, and P.K. Snyder, 2005. Global Consequences of Land Use. Science 309:570-574.

Gentry, L.E., M.B. David, T.V. Royer, C.A. Mitchell, and K.M. Starks, 2007. Phosphorus Transport Pathways to Streams in Tile-Drained Agricultural Watersheds. Journal of Environmental Quality 36:408-415.

Gentry, L.E., M.B. David, K.M. Smith, and D.A. Kovacic, 1998. Nitrogen Cycling and Tile Drainage Nitrate Loss in a Corn/Soybean Watershed. Agriculture, Ecosystems and Environment 68:85-97.

Gippel, C.J., 1995. Potential of Turbidity Monitoring for Measuring the Transport of Suspended Solids in Streams. Hydrological Processes 9:83-97.

Goolsby, D.A. and W.A. Battaglin, 2001. Long-Term Changes in Concentrations and Flux of Nitrogen in the Mississippi River Basin, USA. Hydrological Processes 15:1209-1226.

Gore, J.A., 2007. Discharge Measurements and Streamflow Analysis. *In*: Methods in Stream Ecology, R.F. Hauer and G.A. Lamberti (Editors). Elsevier Ltd., Burlington, Massachusetts, pp. 51-78.

Grayson, R.B., B.L. Finlayson, C.J. Gippel, and B.T. Hart, 1996. The Potential of Field Turbidity Measurements for the Computation of Total Phosphorus and Suspended Solids Loads. Journal of Environmental Management 47:257-267.

Heathwaite, L., A. Sharpley, and W. Gburek, 2000. A Conceptual Approach for Integrating Phosporus and Nitrogen Management at Watershed Scales. Journal of Environment Quality 29:158-166.

Hopkinson, L. and T. Wynn, 2009. Vegetation Impacts on Near Bank Flow. Ecohydrology 2:404-418.

Inamdar, S.P., S. Mostaghimi, P.W. McClellan, and K.M. Brannan, 2001. BMP Impacts on Sediment and Nutrient Yields from an Agricultural Watershed in the Coastal Plain Region. Transactions of the ASAE 44:1191-1200.

Jayakaran, A.D., D.E. Mecklenburg, J.D. Witter, A.D. Ward, and G.E. Powell, 2010. Fluvial Processes in Agricultural Ditches in the North Central Region of the United States and Implications for Their Management. *In*: Agricultural Drainage Ditches: Mitigation Wetlands for the 21st Century, M.T. Moore and R. Kröger (Editors). Research Signpost, Kerala, India, pp. 1-27.

Jones, A.S., D.K. Stevens, J.S. Horsburgh, and N.O. Mesner, 2011. Surrogate Measures for Providing High Frequency Estimates of

Total Suspended Solids and Total Phosphorus Concentrations. Journal of the American Water Resources Association 47:239-253.

Kallio, S.E., 2010. Determining the Bankfull Discharge Exceedance Potential of Agricultural Ditches in Ohio. The Ohio State University, Columbus, Ohio.

Knowles, L., J. Iles, Y. Lu, T. Kobayashi, and L. Wen, 2012. Phosphorus Dynamics in an Ephemeral Wetland Ecosystem after Re-Flooding. Environmental Modelling & Software 35:31-37.

Knowles, R., 1982. Denitrification. Microbiological Reviews 46: 43-70.

Krueger, T., J.N. Quinton, J. Freer, C.J.A. Macleod, G.S. Bilotta, R.E. Brazier, P. Butler, and P.M. Haygarth, 2009. Uncertainties in Data and Models to Describe Event Dynamics of Agricultural Sediment and Phosphorus Transfer. Journal of Environment Quality 38:1137-1148.

Lemke, A.M., K.G. Kirkham, T.T. Lindenbaum, M.E. Herbert, T.H. Tear, W.L. Perry, and J.R. Herkert, 2011. Evaluating Agricultural Best Management Practices in Tile-Drained Subwatersheds of the Mackinaw River, Illinois. Journal of Environmental Quality 40:1215-1228.

Liu, F., R. Xiao, Y. Wang, Y. Li, S. Zhang, Q. Luo, and J. Wu, 2013. Effect of a Novel Constructed Drainage Ditch on the Phosphorus Sorption Capacity of Ditch Soils in An Agricultural Headwater Catchment in Subtropical Central China. Ecological Engineering 58:69-76.

Mahl, U.H., J.L. Tank, S.S. Roley, and R.T. Davis, 2015. Two-Stage Ditch Floodplains Enhance N-Removal Capacity and Reduce Turbidity and Dissolved P in Agricultural Streams. Journal of the American Water Resources Association, doi: 10.1111/1752-1688.12340.

Martin, L.J., B. Blossey, and E. Ellis, 2012. Mapping Where Ecologists Work: Biases in the Global Distribution of Terrestrial Ecological Observations. Frontiers in Ecology and the Environment 10:195-201.

Mayer, P.M., S.K. Reynolds, M.D. McCutchen, and T.J. Canfield, 2007. Meta-Analysis of Nitrogen Removal in Riparian Buffers. Journal of Environmental Quality 36:1172-1180.

Mecklenburg, D. and A. Ward, 2008. Stream Restoration & Management Program — STREAM Modules. Ohio Department of Natural Resources. http://www.dnr.state.oh.us/tabid/24137/Default.aspx, *accessed* August 2010.

Meyer, J.L., 1979. The Role of Sediments and Bryophytes in Phosphorus Dynamics in a Headwater Stream Ecosystem. Limnology and Oceanography 24:365-375.

Mitsch, W.J., J.W. Day, J. Wendell Gilliam, P.M. Groffman, D.L. Hey, G.W. Randall, and N. Wang, 2001. Reducing Nitrogen Loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to Counter a Persistent Ecological Problem. BioScience 51:373-388.

Mulholland, P.J., C.S. Fellows, J.L. Tank, N.B. Grimm, J.R. Webster, S.K. Hamilton, E. Marti, L. Ashkenas, W.B. Bowden, W.K. Dodds, W.H. McDowell, M.J. Paul, and B.J. Peterson, 2001. Inter-Biome Comparison of Factors Controlling Stream Metabolism. Freshwater Biology 46:1503-1517.

Murphy, J. and J.P. Riley, 1962. A Modified Single Solution Method for the Determination of Phosphate in Natural Waters. Analytica Chimica Acta 27:31-36.

Newbold, J.D., J.W. Elwood, R.V. O'Neill, and A.L. Sheldon, 1983a. Phosphorus Dynamics in a Woodland Stream Ecosystem: A Study of Nutrient Spiralling. Ecology 64:1249-1265.

Newbold, J.D., J.W. Elwood, M.S. Schulze, R.W. Stark, and J.C. Barmeier, 1983b. Continuous Ammonium Enrichment of a Woodland Stream: Uptake Kinetics, Leaf Decomposition, and Nitrification. Freshwater Biology 13:193-204.

Ohio Environmental Protection Agency, 2013. Ohio Lake Erie Phosphorus Task Force II Final Report. Cleveland, Ohio.

Olli, G., A. Darracq, and G. Destouni, 2009. Field Study of Phosphorous Transport and Retention in Drainage Reaches. Journal of Hydrology 365:46-55.

Osborne, L.L. and D.A. Kovacic, 1993. Riparian Vegetated Buffer Strips in Water-Quality Restoration and Stream Management. Freshwater Biology 29:243-258.

Powell, G.E., A.D. Ward, D.E. Mecklenburg, J. Draper, and W. Word, 2007a. Two-Stage Channel Systems: Part 2, Case Studies. Journal of Soil and Water Conservation 62:286-296.

Powell, G.E., A.D. Ward, D.E. Mecklenburg, and A.D. Jayakaran, 2007b. Two-Stage Channel Systems: Part 1, A Practical Approach for Sizing Agricultural Ditches. Journal of Soil and Water Conservation 62:277-286.

Prestigiacomo, A.R., S.W. Effler, D. O'Donnell, J.M. Hassett, E.M. Michalenko, Z. Lee, and A. Weidemann, 2007. Turbidity and Suspended Solids Levels and Loads in a Sediment Enriched Stream: Implications for Impacted Lotic and Lentic Ecosystems. Lake and Reservoir Management 23:231-244.

Rabalais, N.N., R.E. Turner, and W.J.J. Wiseman, 2002. Gulf of Mexico Hypoxia, A.K.A. "The Dead Zone". Annual Review of Ecology and Systematics 33:235-263.

R Development Core Team, 2012. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria, ISBN-3-900051-07-0. http://www.r-project.org/, *accessed* October 2012.

Reimer, A.P., A.W. Thompson, and L.S. Prokopy, 2012. The Multi-Dimensional Nature of Environmental Attitudes among Farmers in Indiana: Implications for Conservation Adoption. Agriculture and Human Values 29:29-40.

Richards, C., L.B. Johnson, and G.E. Host, 1996. Landscape-Scale Influences on Stream Habitats and Biota. Canadian Journal of Fisheries and Aquatic Sciences 53:295-311.

Roley, S.S., J.L. Tank, N.A. Griffiths, R.O.J. Hall, and R.T. Davis, 2014. The Influence of Floodplain Restoration on Whole-Stream Metabolism in an Agricultural Stream: Insights from a 5-Year Continuous Data Set. Freshwater Science 33:1043-1059.

Roley, S.S., J.L. Tank, M.L. Stephen, L.T. Johnson, J.J. Beaulieu, and J.D. Witter, 2012a. Floodplain Restoration Enhances Denitrification and Reach-Scale Nitrogen Removal in an Agricultural Stream. Ecological Applications 22:281-297.

Roley, S.S., J.L. Tank, and M.A. Williams, 2012b. Hydrologic Connectivity Increases Denitrification in the Hyporheic Zone and Restored Floodplains of An Agricultural Stream. Journal of Geophysical Research 117:G00N04.

Royer, T.V., M.B. David, and L.E. Gentry, 2006. Timing of Riverine Export of Nitrate and Phosphorus from Agricultural Watersheds in Illinois: Implications for Reducing Nutrient Loading to the Mississippi River. Environmental Science & Technology 40:4126-4131.

Schönbrunner, I.M., S. Preiner, and T. Hein, 2012. Impact of Drying and Re-Flooding of Sediment on Phosphorus Dynamics of River-Floodplain Systems. Science of the Total Environment 432:329-337.

Sharpley, A. and B. Moyer, 2000. Phosphorus Forms in Manure and Compost and Their Release during Simulated Rainfall. Journal of Environment Quality 29:1462-1469.

Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel, and K.R. Reddy, 1994. Managing Agricultural Phosphorus for Protection of Surface Waters: Issues and Options. Journal of Environmental Quality 23:437-451.

Sharpley, A.N., P.J.A. Kleinman, A.L. Heathwaite, W.J. Gburek, G.J. Folmar, and J.P. Schmidt, 2008. Phosphorus Loss from An Agricultural Watershed as a Function of Storm Size. Journal of Environmental Quality 37:362-368.

Smiley, P.C., K.W. King, and N.R. Fausey, 2011. Influence of Herbaceous Riparian Buffers on Physical Habitat, Water Chemistry, and Stream Communities within Channelized Agricultural Headwater Streams. Ecological Engineering 37:1314-1323.

Solórzano, L., 1969. Determination of Ammonia in Natural Waters by the Phenolhypochlorite Method. Limnology and Oceanography 14:799-801.

Stream Solute Workshop, 1990. Concepts and Methods for Assessing Solute Dynamics in Stream Ecosystems. Journal of the North American Benthological Society 9:95-119.

Sugg, Z., 2007. Assessing U.S. Farm Drainage: Can GIS Lead to Better Estimates of Subsurface Drainage Extent? World Resources Institute, Washington, D.C.

Tank, J.L., E.J. Rosi-Marshall, T.V. Royer, M.R. Whiles, N.A. Griffiths, T.C. Frauendorf, and D.J. Treering, 2010. Occurrence of Maize Detritus and a Transgenic Insecticidal Protein (Cry1Ab) within the Stream Network of an Agricultural Landscape. Proceedings of the National Academy of Sciences USA 107:17645-17650.

Tanner, C.C. and J.P.S. Sukias, 2011. Multiyear Nutrient Removal Performance of Three Constructed Wetlands Intercepting Tile Drain Flows from Grazed Pastures. Journal of Environment Quality 40:620-633.

Tomer, M.D., S.E. Porter, D.E. James, K.M.B. Boomer, J.A. Kostel, and E. McLellan, 2013. Combining Precision Conservation Technologies into a Flexible Framework to Facilitate Agricultural Watershed Planning. Journal of Soil and Water Conservation 68:113A-120A.

Tomer, M.D., C.G. Wilson, T.B. Moorman, K.J. Cole, D. Heer, and T.M. Isenhart, 2010. Source-Pathway Separation of Multiple Contaminants During a Rainfall-Runoff Event in an Artificially Drained Agricultural Watershed. Journal of Environmental Quality 39:882-895.

Tonitto, C., M.B. David, and L.E. Drinkwater, 2006. Replacing Bare Fallows with Cover Crops in Fertilizer-Intensive Cropping Systems: A Meta-Analysis of Crop Yield and N Dynamics. Agriculture, Ecosystems & Environment 112:58-72.

Turner, R.E. and N.N. Rabalais, 1991. Changes in Mississippi River Water Quality This Century. BioScience 41:140-147.

Ward, A.D., J. Witter, A.D. Jayakaran, D.E. Mecklenburg, and G.E. Powell, 2007. Two-Stage Channel Geometry: Active Floodplain Requirements. *In*: Encyclopedia of Water Science, S.W. Trimble, B.A. Stewart, and T.A. Howell (Editors). CRC Press, New York, New York, pp. 1253-1260.

Webster, J.R., P.J. Mulholland, J.L. Tank, H.M. Valett, W.K. Dodds, B.J. Peterson, W.B. Bowden, C.N. Dahm, S. Findlay, S.V. Gregory, N.B. Grimm, S.K. Hamilton, S.L. Johnson, E. Marti, W.H. McDowell, J.L. Meyer, D.D. Morrall, S.A. Thomas, and W.M. Wollheim, 2003. Factors Affecting Ammonium Uptake in Streams — An Inter-Biome Perspective. Freshwater Biology 48:1329-1352.

Withers, P.J.A. and A.N. Sharpley, 2008. Characterization and Apportionment of Nutrient and Sediment Sources in Catchments. Journal of Hydrology 350:127-130.

Wolman, M.G., 1954. A Method of Sampling Coarse River-Bed Material. Transactions, American Geophysical Union 35:951-956.

Yates, A.G., R.C. Bailey, and J.A. Schwindt, 2007. Effectiveness of Best Management Practices in Improving Stream Ecosystem Quality. Hydrobiologia 583:331-344.