Ecological Engineering 37 (2011) 1314–1323

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Influence of herbaceous riparian buffers on physical habitat, water chemistry, and stream communities within channelized agricultural headwater streams

Peter C. Smiley Jr*, Kevin W. King, Norman R. Fausey

USDA-ARS, Soil Drainage Research Unit, 590 Woody Hayes Drive, Columbus, OH 43210, USA

ARTICLE INFO

Article history:
Received 29 September 2010
Received in revised form 7 March 2011
Accepted 27 March 2011
Available online 22 April 2011

Keywords:
Filter strip
Agriculture
Headwater streams
Fishes
Amphibians
Conservation practices

ABSTRACT

Herbaceous riparian buffers (CP 21 grass filter strips) are a widely used agricultural conservation practice in the United States for reducing nutrient, pesticide, and sediment loadings to agricultural streams. The ecological impacts of herbaceous riparian buffers on the channelized agricultural headwater streams that are common throughout the midwestern United States have not been evaluated. We sampled riparian habitat, geomorphology, instream habitat, water chemistry, fishes, and amphibians for 4 years from three channelized agricultural headwater streams without herbaceous riparian buffers and three channelized streams with herbaceous riparian buffers in central Ohio. Only seven of 55 response variables exhibited differences between buffer types. Riparian widths were greater in channelized headwater streams with herbaceous riparian buffers than streams without herbaceous riparian buffers. Percent insectivores and minnows were greater in channelized streams without herbaceous riparian buffers than streams with herbaceous riparian buffers. Percent clay, turbidity, specific conductance, and pH differed between buffer types only during one sampling period. No differences in geomorphology and amphibian communities occurred between buffer types. Our results suggest channelized agricultural headwater streams with and without herbaceous riparian buffers are similar physically, chemically, and biologically. Installation of herbaceous riparian buffers alone adjacent to channelized agricultural headwater streams in central Ohio and other parts of the midwestern United States may only provide limited environmental benefits for these stream ecosystems in the first 4–6 years after establishment. Alternative implementation designs combining the use of herbaceous riparian buffers with other practices capable of altering nutrient and pesticide loads, riparian hydrology, and instream habitat are needed.

Published by Elsevier B.V.

1. Introduction

Herbaceous riparian buffers (CP21 grass filter strips) are strips of cool season or warm season grasses planted within sections of agricultural fields immediately adjacent to and running parallel to streams and rivers. The minimum widths of the strips are 6 m from the top of the streambank to the cropland (FSA, 2010a). The recommended maximum width is 37 m and wider strips are allowable based on site-specific water quality needs (FSA, 2010a). This agricultural conservation practice is intended to reduce nutrient, pesticide, and sediment loadings to streams and is a critical component of most non-point source pollution programs in agricultural watersheds in North America (Hickey and Doran, 2004). As of July 2010, 4160 km^2 of herbaceous riparian buffers have been installed adjacent to streams and rivers in the United States through the Conservation Reserve Program (CRP) and the Conservation Reserve Enhancement Program (CREP; FSA, 2010b). The majority (73%) of herbaceous riparian buffers were installed within the midwestern United States (FSA, 2010b). However, herbaceous riparian buffers are a voluntary practice and rarely installed along an entire stream (Barker et al., 2006).

There has been a tremendous interest in the effects of herbaceous riparian buffers and other buffer types as documented by the hundreds of research papers (Shearer and Xiang, 2007) and 17 literature reviews that have been published in peer review journals in the past 60 years (Osborne and Kovacic, 1993; Norris, 1993; Barling and Moore, 1994; Castelle et al., 1994; Hill, 1996; Lyons et al., 2000; Dosskey, 2001; Hickey and Doran, 2004; Krutz et al., 2005; Dorioz et al., 2006; Lovell and Sullivan, 2006; Mayer et al., 2007; Reichenberger et al., 2007; Shearer and Xiang, 2007; Liu et al., 2008; Yuan et al., 2009; Zhang et al., 2010). In general, the reviews indicate that herbaceous riparian buffers have the potential to reduce nutrient and pesticide loadings to streams. However, these reviews also document the wide variability in the efficiency (0–100% efficiency) of herbaceous riparian buffers

* Corresponding author. Tel.: +1 614 292 9806; fax: +1 614 292 9448.
E-mail address: rocky.smiley@ars.usda.gov (P.C. Smiley Jr).

0925-8574/$ – see front matter. Published by Elsevier B.V.
doi:10.1016/j.ecoleng.2011.03.020

to filter agricultural nutrients and pesticides (Hickey and Doran, 2004; Krutz et al., 2005; Mayer et al., 2007; Reichenberger et al., 2007; Liu et al., 2008). Most previous studies were small-scale plot assessments examining the influence of different characteristics of herbaceous riparian buffers (width, vegetation type, slope, etc.) on the removal of nutrients, pesticides, or sediment. Largely lacking from this body of scientific literature are watershed-scale studies examining if implementation of herbaceous riparian buffers reduces nutrient and pesticide concentrations within the streams adjacent to the buffers (Dosskey, 2001; Hickey and Doran, 2004; Krutz et al., 2005; Lovell and Sullivan, 2006; Reichenberger et al., 2007). Nitrate reductions did not occur between headwater agricultural streams without and with planted grass buffers in United Kingdom (Leeds-Harrison et al., 1999). Channelized streams in Indiana exhibited elevated nitrogen, phosphorus, and herbicide concentrations even though 60–77% of their stream length contained riparian zones possessing a combination of herbaceous and woody vegetation (Smith et al., 2008; Pappas et al., 2008).

Only a limited amount of information is available on the ecological effects of herbaceous riparian buffers on stream ecosystems. Agricultural headwater streams with herbaceous riparian buffers in Indiana and North Carolina had greater fish diversity, macroinvertebrate abundance, and macroinvertebrate diversity than streams without herbaceous riparian buffers (Whitworth and Martin, 1990). Coldwater streams in Wisconsin containing greater amounts of CRP grassland created in the watershed and CREP herbaceous buffers adjacent to the streams exhibited increases in biological integrity (Marshall et al., 2008). Conversely, no differences in physical habitat and fish communities occurred between Iowa stream reaches with and without planted mixed species riparian buffers (Fischer et al., 2010).

What is critically missing from our understanding of the effect of herbaceous riparian buffers is a comprehensive appreciation for how riparian habitat, geomorphology, instream habitat, water chemistry, and stream communities respond to the implementation of this conservation practice. This represents a significant knowledge gap as the stream improvements resulting from herbaceous riparian buffers to date have been assumed, not quantified. Others have recommended targeting headwater streams for implementation of herbaceous riparian buffers because: (1) buffer effectiveness is predicted to be greater adjacent to headwater streams; (2) of the need to protect larger downstream tributaries; and (3) of the increased opportunities for implementation due to the large numbers of headwater streams present in all watersheds (Osborne and Kovacic, 1993; Correll, 2005; Teels et al., 2006). Additionally, the headwaters of many agricultural watersheds in the midwestern United States consist of channelized streams that were created or modified for agricultural drainage (Smiley and Gillespie, 2010). Installation of herbaceous riparian buffers adjacent to channelized headwater streams has also been recommended to reduce the need for frequent sediment removal (Lovell and Sullivan, 2006). Therefore, it is also critical to understand how installing herbaceous riparian buffers influences the physical, chemical, and biological characteristics of channelized agricultural headwater streams. We measured riparian habitat, geomorphology, instream habitat, water chemistry, and stream communities for 4 years from three channelized agricultural headwater streams without herbaceous riparian buffers and three channelized agricultural headwater streams with herbaceous riparian buffers. In this manuscript we address the research question: "Is there a difference in physical habitat, water chemistry, and stream communities between channelized agricultural headwater streams with and without herbaceous riparian buffers?"

Table 1
Means (SD) of percent land use and dominant soil types within the upstream watersheds of channelized headwater streams without and with herbaceous riparian buffers in the Upper Big Walnut Creek watershed, Ohio.

Response variable	No buffer	Buffer
Land use		
Agriculture	81 (18)	64 (10)
Urban	13 (11)	19 (4)
Woodland	4 (5)	10 (7)
Scrub	1 (2)	6 (9)
Soil type		
Bennington	46 (4)	45 (16)
Pewamo	31 (12)	43 (6)
Centerburg	19 (10)	21 (31)
Other	4 (6)	6 (10)

2. Materials and methods

2.1. Study sites

Upper Big Walnut Creek (UBWC) watershed is located in central Ohio (Fig. 1) and is part of the Scioto River watershed, which is one of the most biologically diverse watersheds in Ohio (Sanders, 2001). Dominant land use in the UBWC watershed is cropland consisting of corn, soybean, or wheat. The majority of headwater streams in the watershed are impaired by nutrient enrichment, pathogens, and habitat degradation stemming from current agricultural management practices (Ohio EPA, 2003, 2004). Six channelized headwater streams (Fig. 1) located on private land were selected based on *a priori* habitat criteria: (1) first or second order headwater streams; (2) dominant land use in the watershed was row-crop agriculture containing systematically tile drained fields; and (3) accessibility that allowed for establishment of two sampling sites spaced at least 150 m apart. Additional *a priori* habitat criteria included the presence or absence of herbaceous riparian buffers (CP21 grass filter strips). Three streams were selected because they lacked herbaceous riparian buffers and possessed riparian zones that exhibited habitat conditions typical of headwater streams within the Upper Big Walnut Creek watershed (Malcom Pirnie, 1999). Typical riparian habitat characteristics included narrow riparian widths (<15 m) and consisting mostly of herbaceous vegetation established through natural colonization and existing seed banks. Three streams were selected because they contained herbaceous riparian buffers (CP21 grass filter strips) on both streambanks that were planted between 2003 and 2005 through the CREP. Post-site selection analyses of the 2006 land use characteristics within the upstream watersheds documented that streams with and without herbaceous riparian buffers were dominated (>60%) by agricultural land use (Table 1). The upstream watersheds of both buffer types contained riparian zones that occupied less than 1% of the watershed area and consisted mostly of herbaceous vegetation. The contributing upstream watersheds for both buffer types were also composed mostly (>80%) of soils formed in undulating glacial till deposits (Table 1). These soils range from moderately well drained to poorly drained soils and possess silt loam to silty clay loam surface textures. The key difference between buffer types is streams without herbaceous riparian buffers possessed riparian zones that were not planted or managed to reduce agricultural impacts, while streams with herbaceous riparian buffers possessed riparian zones containing planted strips of herbaceous vegetation intended to provide water quality benefits. Each channelized stream within a buffer type serves as a replicate in our statistical analyses. Two 125 m-long sites (subsamples) were established in each replicate and were separated by a mean

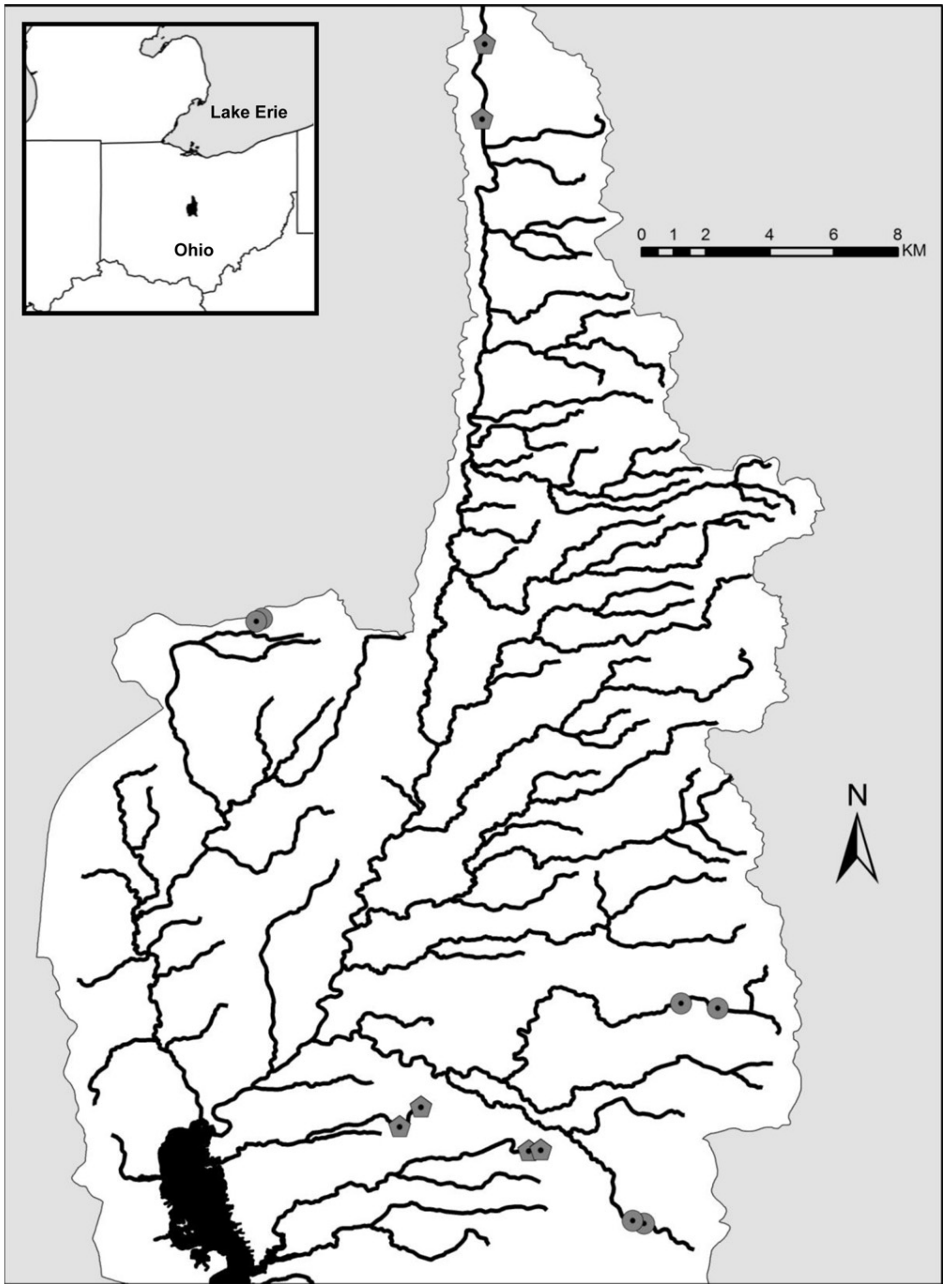

Fig. 1. Location of sampling sites within channelized agricultural headwater streams without and with herbaceous riparian buffers within the Upper Big Walnut Creek watershed, Ohio. Circles are sampling sites within streams without herbaceous riparian buffers and pentagons depict sites within streams with herbaceous riparian buffers. The symbols have been sized to ensure their clarity and do not reflect actual site sizes or distances between sites.

distance of 743 m (range 170–1938 m) to ensure our sampling encompassed a representative range of habitat conditions within each stream.

2.2. Sampling methods

Our sampling protocol was developed following guidelines created for designing sampling protocols to evaluate the influence of agricultural conservation practices on streams (Smiley et al., 2009b). We only provide an overview of our sampling methods as additional details have been published previously (Smiley et al., 2009a,b). We sampled physical habitat, water chemistry, and stream communities within all channelized streams from 2006 to 2009. Our physical habitat sampling protocol involves the use of transect based sampling for the measurement of riparian habitat, geomorphology, and instream habitat. Riparian vegetation, geomorphology, and instream habitat were measured along six permanent transects spaced 25 m apart within each site. Riparian and geomorphology characteristics are measured once a year in the fall (September–November) of each year. Measurements of instream habitat characteristics were obtained in the spring (April–May), summer (July–August), and fall of each year. *In situ* measurements of physicochemical variables were obtained with a multiparameter meter from each site three times a year concurrently with instream habitat, fish, and amphibian sampling. Grab samples for turbidity were collected in conjunction with *in situ* measurements and measured in the lab with a turbidity meter. Water samples for nutrient (nitrate plus nitrite, ammonium, total nitrogen, dissolved reactive phosphorus, total phosphorus, dissolved organic carbon), herbicide (alachlor, atrazine, metolachlor, simazine), and fungicide (chlorothalonil, metalaxyl) measurements were collected weekly from April to November. Nutrient and pesticide measurements follow standard protocols (Menzel and Vaccaro, 1964; Koroleff, 1983; Parsons et al., 1984; U.S. EPA, 1995). Fishes and amphibians were sampled three times a year with a backpack electrofisher and seine concurrently with instream habitat and *in situ* water chemistry measurements. Fishes and amphibians that could be identified in the field were identified, enumerated, and released. Unidentifiable fishes were euthanized with tricaine methanesulfonate, fixed with a 10% formalin solution, and returned to the laboratory for subsequent identification.

2.3. Response variables

Physical habitat, water chemistry, and biological data collected from our sampling sites within each stream during each sampling period were either composited or averaged to avoid pseudoreplication. We calculated eight riparian habitat response variables (herbaceous structural diversity, woody structural diversity, riparian structural diversity, Simpson's riparian structural diversity, woody to herbaceous vegetation ratio, percent canopy cover, woody vegetation density, riparian width) for each site from each year. Our metrics of riparian structural diversity were calculated based on the percentages of the presence of woody and herbaceous vegetation in four height stratas within 12 quadrats in a site. Herbaceous and woody structural diversity is the sum of the percent of herbaceous or woody vegetation in each height strata. Riparian structural diversity is the sum of the percent of herbaceous and woody vegetation in each height strata. We also calculated riparian structural diversity using the Simpson's diversity index (Magurran, 1988). Woody to herbaceous vegetation ratio is the sum of the percent of woody vegetation in each height strata divided by the sum of the percent of herbaceous vegetation in each height strata. Woody vegetation density is density of woody vegetation >1 m tall. Riparian width is the width of the riparian zone that extends from the water's edge to the edge of the adjacent agricultural field. Mean values of each riparian response variable for each stream in each year were then calculated.

We calculated the mean of five geomorphology variables (cross-section area, thalweg depth, top bank width, gradient, sinuosity) for each site from each year. Geomorphology variables describing the size of the channel were calculated based on channel size at bankfull capacity. Mean values of each geomorphology response variable for each stream in each year were then calculated.

For instream habitat we calculated the means of four hydrology variables (i.e., water depth, water velocity, wet width, discharge), the percentages of four substrate and cover type variables (i.e., clay, sand, gravel, and wood), and the number of substrate types present in each site during each season. Mean values of each instream habitat response variable for each stream in each season were then calculated.

We calculated the means of 12 nutrient and pesticide variables (nitrate plus nitrite, ammonium, total nitrogen, dissolved reactive phosphorus, total phosphorus, dissolved organic carbon, alachlor, atrazine, metolachlor, simazine, chlorothalonil, metalaxyl) in each site during each season. Mean values of each nutrient and pesticide variable for each stream in each season were then calculated. We also calculated the means of five physicochemical variables (water temperature, pH, turbidity, specific conductance, dissolved oxygen) from each stream during each season.

We calculated 10 fish community response variables (i.e., species richness, abundance, evenness, headwater fish species richness, percent headwater fishes, percent omnivores, percent insectivores, percent minnows (Family Cyprinidae), percent darters (Family Percidae), trophic guild richness, reproductive guild richness) for each stream during each season. Species richness is the number of fish species captured and abundance is the number of fishes captured. Evenness is the reciprocal of the Simpson's diversity index divided by species richness (Smith and Wilson, 1996). Fishes were assigned to habitat (i.e., headwater fish species), feeding, and reproductive guilds based on published literature sources (Pflieger, 1975; Becker, 1983; Robison and Buchanan, 1988; Etnier and Starnes, 1993; Ohio EPA, 2002; Ross, 2002; Smiley et al., 2005). Headwater fish species are those fishes expected to be found in first to third order streams in the midwestern United States, such as creek chub (*Semotilus atromaculatus*), white sucker (*Catostomus commersoni*), and orangethroat darter (*Etheostoma spectabile*) (Ohio EPA, 2002). Omnivores are fishes whose diet consists of plant and animal matter and insectivores are fishes that primarily consume insects and other invertebrates. Trophic guild richness is the number of feeding guilds and reproductive guild richness is the number of reproductive guilds. We also calculated six amphibian community response variables [i.e., taxa richness, abundance, evenness, frog abundance, ratio of adult frogs to tadpoles, Shannon diversity index (Magurran, 1988)] for each stream during each season. Taxa richness is the number of amphibian species and tadpoles captured. Amphibian abundance and evenness were calculated the same way as fish abundance and evenness. The ratio of adult frogs to tadpoles is the number of adult frogs captured divided by the number of tadpoles captured.

2.4. Statistical analyses

We used a two factor repeated measures analysis of variance (ANOVA) coupled with Tukey multiple comparisons test to detect if differences in physical habitat, water chemistry, and stream community variables occurred between channelized streams with and without herbaceous riparian buffers and if the effect of buffer type was influenced by sampling period. The repeated measures ANOVA is specifically designed for the analyses of data obtained by measuring the same experimental subjects (i.e., sites or streams) over time (Littell et al., 1998). We only report results on the effects of buffer type and the interaction of buffer type and sampling period. Selective reporting of our results allows us to focus on the most important results for addressing our research question and enables us to account for the potential influence of sampling period.

The two factors repeated measures ANOVAs were conducted using the Proc Mixed procedure within SAS System for Windows version 8 (SAS Institute, 1999). We used the recommended two step procedure for conducting repeated measures ANOVA with Proc Mixed (Littell et al., 1998). First, we conducted covariance modeling where we performed the analyses with six different covariance structures (first order autoregressive, first order antedependence, toeplitz, heterogeneous first order autoregressive, heterogeneous toeplitz, unstructured) (Littell et al., 1998) to obtain the Akaike's Information Criteria. The unstructured covariance structure is capable of fitting any covariance structure and the remaining covariance structures are intended for use with repeated measures in time (Moser, 2004). For each response variable the covariance structure with the greatest Akaike's Information Criteria was selected (Kincaid, 2005) and used in the second step of the analyses that involved testing of the buffer type and the interaction of buffer type and sampling period effects. This two step approach takes advantage of the inherent variability that occurs when sampling the same sites over time and increases the power and sensitivity

Table 2
P values from two factor repeated measures ANOVA conducted to determine the effect of buffer type (BT) and buffer type by sampling period (BT × SP) on riparian habitat and geomorphology in channelized headwater streams with and without herbaceous riparian buffers in the Upper Big Walnut Creek watershed, Ohio, 2006–2009. Bolded values are significant (P < 0.05). Abbreviations for covariance structures (COVS) used are: ante – first order antedependence; ar – first order autoregressive; toeph – heterogeneous toeplitz; un – unstructured.

Response variable	BT	BT × SP	COVS
Riparian habitat			
Herbaceous riparian structural diversity	0.321	0.109	ar
Woody riparian structural diversity	0.933	0.678	toeph
Riparian structural diversity	0.857	**0.034**	toeph
Simpson's riparian structural diversity	0.770	0.338	ante
Woody to herbaceous vegetation ratio	0.919	0.690	ante
Percent canopy cover	0.783	0.801	un
Woody vegetation density	0.650	**0.012**	ante
Riparian width	**0.013**	0.305	ante
Geomorphology			
Cross-section area	0.830	**0.002**	toeph
Thalweg depth	0.569	0.228	un
Top bank width	0.783	0.509	ar
Gradient	0.988	0.146	ante
Sinuosity	0.730	0.205	toeph

of the two factor repeated measures ANOVA (Littell et al., 1998; Moser, 2004).

The sampling period in our analyses differs among types of response variables. The sampling period in the riparian and geomorphology analyses was year because these response variables were only sampled once a year. We used a combination of season and year (i.e., spring 2006 – sampling period 1; summer 2006 – sampling period 2; etc.) as the sampling period in the instream habitat, water chemistry, fish community, and amphibian community analyses. The season–year combination was selected as the sampling period because only three covariance structures were available to use with a three factor repeated measures ANOVA (SAS Institute, 1999). Additionally, using the same statistical test (two factor repeated measures ANOVA) enables us to compare our results across types of response variables (i.e., riparian habitat, geomorphology, instream habitat, water chemistry, stream community).

The assumptions of normality and equal variance were only met for five response variables (cross-section area, gradient, dissolved organic carbon, number of substrate types, percent sand). Therefore, the two factors repeated measures ANOVAs were conducted with rank transformed values for 50 response variables that did not meet the normality and equal variance assumptions. Rank transformation is commonly recommended in these situations and its use with a parametric test is the equivalent of a nonparametric two factor repeated measures ANOVA (Conover, 1999). A significance level of P < 0.05 was used for all statistical tests.

3. Results

3.1. Physical habitat

Riparian widths were greater (P < 0.05) in channelized streams with herbaceous riparian buffers than streams without herbaceous riparian buffers (Table 2, Fig. 2). Herbaceous structural diversity, woody structural diversity, Simpson's riparian structural diversity, woody to herbaceous vegetation ratio, and canopy cover did not differ (P > 0.05) between buffer types (Table 2). Woody vegetation density and riparian structural diversity exhibited a significant (P < 0.05) buffer type by sampling period effect (Table 2). Subsequent analyses of Tukey multiple comparison tests did not reveal any significant differences (P > 0.05) in these two riparian response variables between buffer types during any sampling period.

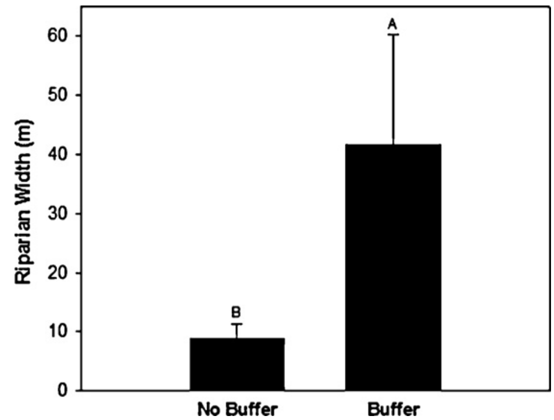

Fig. 2. Mean riparian width in channelized agricultural headwater streams without and with herbaceous riparian buffers in the Upper Big Walnut Creek watershed, Ohio, 2006–2009.

Thalweg depth, top bank width, gradient, and sinuosity did not differ (P > 0.05) between buffer types (Table 2). Channel cross-section area exhibited a significant (P < 0.05) buffer type by sampling period effect (Table 2). Subsequent analyses of Tukey multiple comparison tests did not reveal any significant differences (P > 0.05) in cross-section area between buffer types during any sampling period.

Mean water velocity, wet width, number of substrate types, percent sand, percent gravel, and percent wood did not differ (P > 0.05) between buffer types (Table 3). Mean water depths, percent clay, and mean discharge exhibited a significant (P < 0.05) buffer type by sampling period effect (Table 3). The Tukey multiple comparison tests did not reveal any significant differences (P > 0.05) in mean water depths and discharge between buffer types during any sampling period. Percent clay was greater (P < 0.05) in channelized streams with herbaceous riparian buffers than streams without buffers only in the summer 2007 and no differences occurred between buffer types for any other sampling period (Fig. 3).

3.2. Water chemistry

Ammonium, dissolved reactive phosphorus, total nitrogen, dissolved organic carbon, alachlor, atrazine, metolachlor, simazine, chlorothalonil, and metalaxyl did not differ (P > 0.05) between buffer types (Table 4). Nitrate plus nitrite and total phosphorus exhibited a significant (P < 0.05) buffer type by sampling period

Table 3
P values from two factor repeated measures ANOVA conducted to determine the effect of buffer type (BT) and buffer type by sampling period (BT × SP) on instream habitat in channelized headwater streams with and without herbaceous riparian buffers in the Upper Big Walnut Creek watershed, Ohio, 2006–2009. Bolded values are significant (P < 0.05). Abbreviations for covariance structures (COVS) used are: ante – first order antedependence; ar – first order autoregressive; arh – heterogeneous first order autoregressive; toep – toeplitz.

Response variable	BT	BT × SP	COVS
Water depth	0.559	**0.026**	ante
Water velocity	0.206	0.072	arh
Wet width	0.963	0.188	ante
Discharge	0.534	**<0.001**	toep
Number of substrate types	0.371	0.319	arh
Percent clay	**0.017**	**<0.001**	ante
Percent sand	0.100	0.175	ar
Percent gravel	0.088	0.074	ante
Percent wood	0.289	0.804	ante

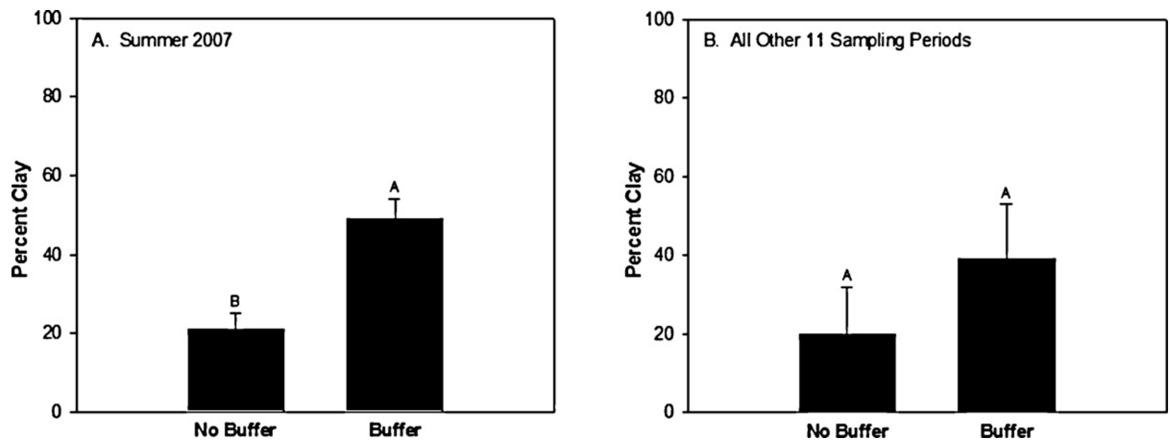

Fig. 3. Mean percent clay within channelized agricultural headwater streams without and with herbaceous riparian buffers during the summer 2007 (A) and all other sampling periods (B) in the Upper Big Walnut Creek watershed, Ohio.

effect (Table 4). No significant differences ($P > 0.05$) in mean nitrate plus nitrite and total phosphorus between buffer types were observed with the Tukey multiple comparison test during any sampling period.

Percent dissolved oxygen did not differ ($P > 0.05$) between buffer types (Table 4). Water temperature, pH, turbidity, and specific conductance exhibited a significant ($P < 0.05$) buffer type by sampling period effect (Table 4). No significant differences ($P > 0.05$) in mean water temperature between buffer types were documented with the Tukey multiple comparison test during any sampling period. Mean turbidity and specific conductance only differed ($P < 0.05$) between buffer types during fall 2007 (Fig. 4). Mean turbidity and specific conductance were greater in the channelized streams without herbaceous riparian buffers than streams with buffers during this one sampling period (Fig. 4). In the summer 2008 mean pH was greater ($P < 0.05$) in channelized streams without herbaceous riparian buffers than streams with buffers and no differences occurred between buffer types for any other sampling period (Fig. 4).

Table 4
P values from two factor repeated measures ANOVA conducted to determine the effect of buffer type (BT) and buffer type by sampling period (BT × SP) on water chemistry in channelized headwater streams with and without herbaceous riparian buffers in the Upper Big Walnut Creek watershed, Ohio, 2006–2009. Bolded values are significant ($P < 0.05$). Abbreviations for covariance structures (COVS) used are: ante – first order antedependence; ar – first order autoregressive; arh – heterogeneous first order autoregressive; toep – toeplitz.

Response variable	BT	BT × SP	COVS
Nutrients			
Nitrate plus nitrite	0.490	**0.007**	ante
Ammonium	0.620	0.617	ante
Total nitrogen	0.363	0.170	ante
Dissolved reactive phosphorus	0.698	0.115	ante
Total phosphorus	0.825	**<0.001**	ante
Dissolved organic carbon	0.369	0.263	ar
Pesticides			
Alachlor	0.985	0.391	toep
Atrazine	0.058	0.429	ante
Metolachlor	0.178	0.877	ar
Simazine	0.066	0.074	ante
Chlorothalonil	0.855	0.550	ar
Metalaxyl	0.668	0.172	toep
Physicochemical			
Water temperature	0.246	**0.019**	toep
pH	**0.020**	**<0.001**	ante
Turbidity	0.142	**0.002**	arh
Specific conductance	**0.002**	**0.002**	ante
Dissolved oxygen	0.207	0.849	ante

3.3. Stream communities

Mean percent insectivores and percent minnows were greater in channelized streams without buffers than streams with buffers (Table 5, Fig. 5). Mean fish abundance, fish evenness, percent headwater fishes, percent darters, reproductive guild richness, and trophic richness did not differ ($P > 0.05$) between buffer types (Table 5). Mean fish species richness and percent omnivores exhibited a significant ($P < 0.05$) buffer type by sampling period effect (Table 5), but the Tukey multiple comparison test indicated that no differences between buffer types occurred during any sampling period. No differences ($P > 0.05$) in mean amphibian species richness, abundance, evenness, ratio of adult frogs to tadpoles, and diversity occurred between buffer types (Table 5). Frog abundance exhibited a significant ($P < 0.05$) buffer type by sampling period effect (Table 5). No significant differences ($P > 0.05$) in mean frog abundance between buffer types were documented with the Tukey multiple comparison test during any sampling period.

Table 5
P values from two factor repeated measures ANOVA conducted to determine the effect of buffer type (BT) and buffer type by sampling period (BT × SP) on fish and amphibian communities in channelized headwater streams with and without herbaceous riparian buffers in the Upper Big Walnut Creek watershed, Ohio, 2006–2009. Bolded values are significant ($P < 0.05$). Abbreviations for covariance structures (COVS) used are: ante – first order antedependence; arh – heterogeneous first order autoregressive.

Response variable	BT	BT × SP	COVS
Fish			
Species richness	0.863	**0.022**	ante
Abundance	0.536	0.351	ante
Evenness	0.355	0.055	ante
Percent headwater fishes	0.103	0.155	ante
Percent omnivores	0.782	**0.033**	ante
Percent insectivores	**0.034**	0.709	arh
Percent minnows	**0.038**	0.329	ante
Percent darters	0.813	0.429	ante
Reproductive guild richness	0.595	0.053	ante
Trophic guild richness	0.413	0.248	arh
Amphibian			
Taxa richness	0.448	0.348	ante
Abundance	0.279	0.293	ante
Evenness	0.536	0.324	ante
Frog abundance	0.958	**0.038**	ante
Ratio of adult frogs to tadpoles	0.378	0.366	arh
Shannon diversity index	0.626	0.158	ante

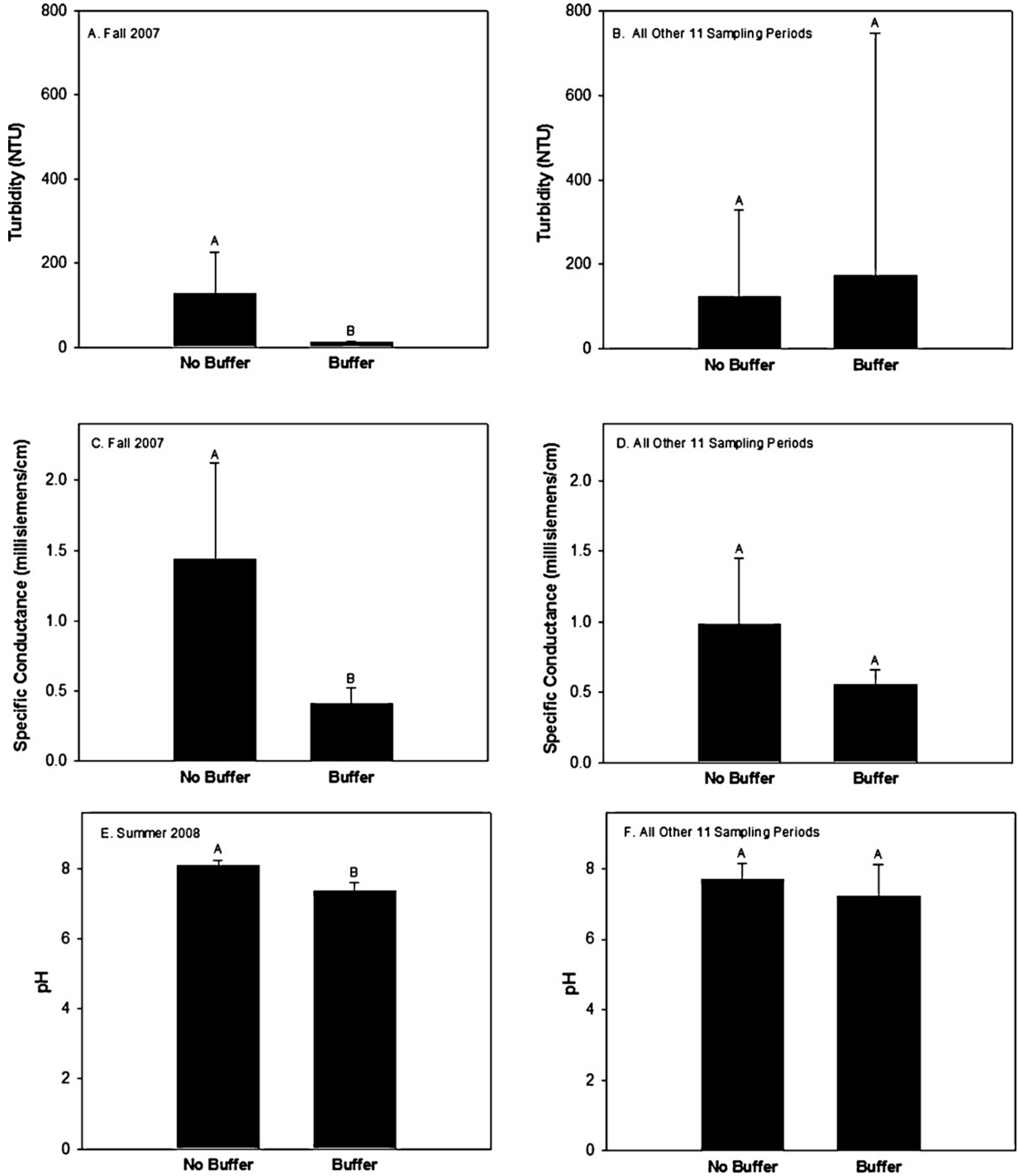

Fig. 4. Mean turbidity, specific conductance, and pH within channelized agricultural headwater streams without and with herbaceous riparian buffers during the fall 2007 (A, C), summer 2008 (E), and all other sampling periods (B, D, F) in the Upper Big Walnut Creek watershed, Ohio.

4. Discussion

Our results suggest that channelized agricultural headwater streams with and without herbaceous riparian buffers are similar physically, chemically, and have similar stream communities. Only seven out of 55 physical, chemical, and biological response variables exhibited a difference between buffer types and four of these response variables (percent clay, turbidity, specific conductance, pH) only exhibited differences between buffer types for one sampling period during our 4 year study. Therefore, our results suggest the effect of installing herbaceous riparian buffers adjacent to channelized agricultural headwater streams is simply widening of the riparian zones. Our results also represent the potential effects of herbaceous riparian buffers during the first 4–6 years after establishment, and may be representative of the effects of this conservation practice under a best-case scenario. The herbaceous riparian buffers evaluated in this study are much wider than the minimum recommended widths. Also, herbaceous buffer installation occurred on both sides of the stream and represents atypical installation conditions within the Upper Big Walnut Creek watershed.

Planted grass buffers (5–50 m wide) did not reduce nutrient concentrations in agricultural headwater streams in the United Kingdom (Leeds-Harrison et al., 1999). The lack of a buffer effect

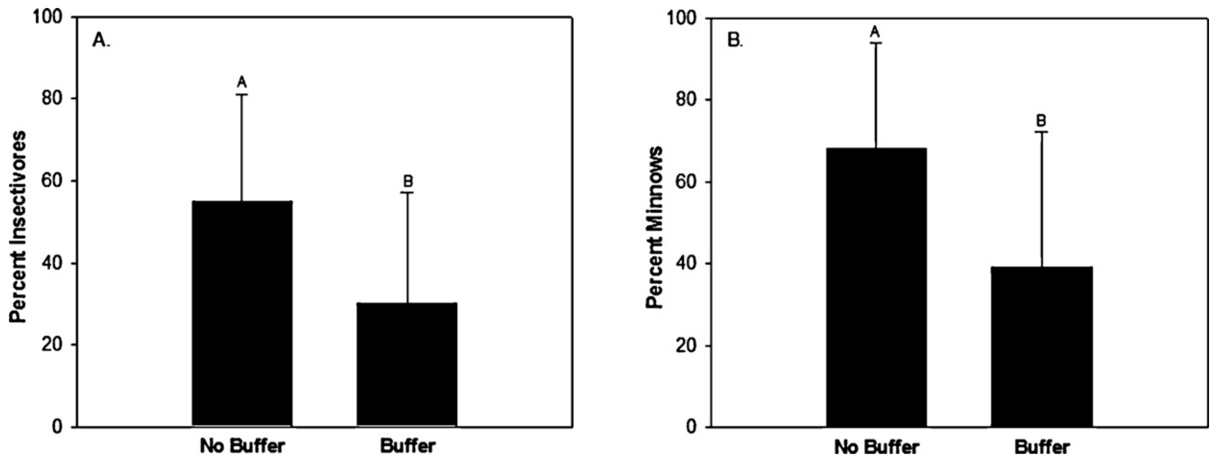

Fig. 5. Mean percent insectivores (A) and percent minnows (B) in channelized agricultural headwater streams without and with herbaceous riparian buffers in the Upper Big Walnut Creek watershed, Ohio, 2006–2009.

was attributed to the tile drains in the watershed that may have enabled the runoff to bypass the buffers. Our nutrient results are consistent with these results (Leeds-Harrison et al., 1999), as we did not observe concentration differences between buffer types. Given that the watersheds of our streams also contain tile drains this is also a likely explanation for our water chemistry results. The overall effectiveness of any buffer will be determined by its ability to intercept and uptake agricultural pollutants from surface and subsurface runoff (Mayer et al., 2007). Previous research has also suggested agricultural runoff may bypass the buffers if tile drains within the agricultural fields reduce surface runoff or if groundwater levels adjacent to the entrenched channelized streams are below the root zone of the buffers (Osborne and Kovacic, 1993; Leeds-Harrison et al., 1999; Hickey and Doran, 2004; Mayer et al., 2007).

The influence of herbaceous riparian buffers on riparian habitat, geomorphology, and instream habitat has not been examined previously. However, comparisons of riparian and instream habitat characteristics between stream reaches in Iowa with remnant herbaceous riparian zones and reaches adjacent to planted mixed species riparian buffers have been conducted (Fischer et al., 2010). Although woody and herbaceous vegetation were planted, the riparian buffers during the Iowa study (Fischer et al., 2010) consisted of mostly herbaceous vegetation since the buffers were less than 20 years old. No differences in percent canopy cover, substrate types, cover types, percent microhabitat types (i.e., pools, riffles, runs), water depth, and wet width between buffer types were observed (Fischer et al., 2010). Our results are consistent with Fischer et al. (2010) as we did not observe differences in riparian habitat or instream habitat variables between buffer types.

Information regarding the influence of herbaceous riparian buffers on amphibian communities is lacking, but information on fish community responses is available. Fish communities in headwater agricultural streams with herbaceous riparian buffers exhibited greater species richness, diversity, and index of biological integrity (IBI) scores than streams without herbaceous riparian buffers in Indiana and North Carolina (Whitworth and Martin, 1990). Increasing the amount of grassland habitat within the watershed and riparian zones adjacent to Wisconsin coldwater streams led to decreased livestock, percent crop, and percent pasture in the watershed and increased coldwater IBI scores and decreased fish species richness (decreasing species richness in coldwater streams is considered a positive response) (Marshall et al., 2008). Conversely, comparisons of 16 fish community response variables between Iowa stream reaches with and without planted mixed species riparian buffers found that only species composition differed between buffer types (Fischer et al., 2010). Stream reaches with planted mixed species buffers contained more black bullheads (*Ameirus melas*) and black crappie (*Pomoxis nigromaculatus*) than reaches without buffers (Fischer et al., 2010). Our results are consistent with Fischer et al. (2010) in that fish communities responded weakly to buffer types as only two (percent insectivores and percent minnows) of 10 fish community response variables differed between buffer types. Our results and Fischer et al. (2010) are representative of the responses of warmwater fish communities to agricultural conservation practices as coldwater fish communities respond more readily than warmwater fish communities (Westra et al., 2005; Wang et al., 2006). Differences between the results of Whitworth and Martin (1990) with our results and those of Fischer et al. (2010) are difficult to explain. Fischer et al. (2010) attributed the lack of fish community responses to planted mixed species buffers due to: (1) the tolerance of prairie fish communities to increased sediment, water temperatures, and variable discharge; (2) a reduction of the available species pools due to the widespread impact of agriculture; and (3) presence of tile drains. However, our study and Fischer et al. (2010) did not observe differences in riparian habitat (excluding riparian width) or instream habitat between buffer types. We feel the lack of difference in physical habitat characteristics, particularly instream habitat, explains why fish communities in our study and Fischer et al. (2010) and the amphibian communities in our study did not respond to herbaceous riparian buffers. Instream habitat is an important determinant of fish community structure in channelized headwater agricultural streams in the midwestern United States (Smiley and Gillespie, 2010). Our previous research findings from channelized headwater agricultural streams in Indiana and Ohio documented that fish communities are more strongly correlated with instream habitat than riparian habitat or water chemistry (Smiley et al., 2008).

Both woody and herbaceous riparian buffers should not be expected to resolve all environmental problems that headwater streams experience as a result of agriculture (Teels et al., 2006). Our results suggest that herbaceous riparian buffers planted adjacent to channelized headwater streams with tile drained agricultural watersheds have a minimal impact on these stream ecosystems. We are not implying that our results are contradictory with the concept (Lyons et al., 2000) that herbaceous riparian zones are important components of stream ecosystems. Instead our results suggest that planting herbaceous riparian buffers (CP 21) following FSA guidelines adjacent to channelized headwater stream simply widen the riparian zones of these streams. Yet, despite our confidence in our results we do not recommend discontinuing the use of this conservation practice adjacent to channelized agricultural headwater streams. First, there is not enough empirical information available on the watershed-scale influence of herbaceous riparian buffers on agricultural streams. Our study is one of five watershed-scale evaluations (Whitworth and Martin, 1990; Leeds-Harrison et al., 1999; Marshall et al., 2008; Fischer et al., 2010) of this conservation practice and the results are not consistent among studies. Secondly, agricultural land use has resulted in the narrowing and alteration of riparian zones within agricultural watersheds in the United States. Many farmers are not willing to voluntarily establish forested riparian zones even with the availability of cost-share funds (Lyons et al., 2000). The removal of agriculture, planting of herbaceous vegetation, and the widening of these riparian zones represents a first step towards riparian restoration that is likely to be more widely adopted by farmers than forested riparian zones (FSA, 2010b). Thirdly, herbaceous riparian buffers are one of the most evaluated practices with regard to terrestrial wildlife benefits (Clark and Reeder, 2007). Herbaceous riparian buffers have been documented to provide habitat for terrestrial insects, small mammals, and birds, although wildlife diversity and reproductive success are not always as great as those in natural grassland habitats (Brady, 2007; Clark and Reeder, 2007). Fourthly, our results only represent the potential effects of this practice 4–6 years after planting. It has been suggested that conservation practices that involve planting vegetation require long time periods (≥10 years) to become fully effective (Frimpong et al., 2006; Munro et al., 2009; Meals et al., 2010). Long term evaluations of the effects of herbaceous riparian buffers on channelized agricultural headwater streams are lacking. However, we predict that even allowing longer time periods for the herbaceous riparian buffers in our study to more fully develop will not result in a difference between buffer types because the tile drains enable agricultural runoff to bypass the buffers.

Our initial results indicate a need for changing how herbaceous riparian buffers are implemented adjacent to channelized streams possessing tile drained agricultural watersheds in the midwestern United States. Specifically, we recommend that herbaceous riparian buffers should not be installed alone, but their use adjacent to channelized headwater streams needs to occur in conjunction with other practices capable of improving the physical, chemical, and biological characteristics of these small streams. This recommendation represents a change in how herbaceous riparian buffers are implemented as many conservation programs focus on soliciting landowner adoption of single conservation practices. Our recommendation would ideally be implemented as part of an adaptive management strategy (Zedler, 2003) that involves implementing the above recommendation, evaluating its effectiveness, and then using the new information as the basis for future decisions regarding the use of herbaceous riparian buffers.

Future research is needed to identify which practices should be used in conjunction with herbaceous riparian buffers. Wang et al. (2006) found a combination of upland and riparian habitat management improved water chemistry and physical habitat within agricultural watersheds in Wisconsin. Thus, perhaps future conservation plans should seek to establish upland management practices, such as precision nutrient management, pesticide management, and/or no-till tillage, in conjunction with herbaceous riparian buffers. Additionally, created wetlands designed to capture and treat surface and subsurface runoff have been recommended as a cost-effective solution capable of reducing excess nutrient inputs from agricultural streams in the midwestern United States (Osborne and Kovacic, 1993; Mitsch and Day, 2006). Increases in riparian width as a result of installing herbaceous riparian buffers would provide greater area for creating wetlands and would increase habitat diversity within riparian zones currently lacking wetlands. Drainage water management (Cooke et al., 2008) has the potential of altering the hydrology of agricultural runoff through the riparian zones and increasing the filtration efficiency of herbaceous riparian buffers during certain times of the year. However, current drainage water management guidelines only involve manipulating tile hydrology during the winter and the summer. Water quality benefits would not be expected to occur during free drainage that is implemented in the spring and fall. However, these practices (upland management, riparian wetland creation, drainage water management) will not address the instream physical habitat degradation caused by channelization. Thus, the use of instream habitat structures and other practices that can directly alter the instream habitat characteristics (i.e., water depth, velocity, substrate types, cover types) in combination with herbaceous riparian buffers will most likely be needed to produce changes in the stream communities.

5. Conclusions

Our comparisons of 55 physical, chemical, and biological response variables between channelized agricultural headwater streams with and without herbaceous riparian buffers in central Ohio suggest that these streams are similar physically, chemically, and have similar stream communities. Our results suggest herbaceous riparian buffers planted adjacent to channelized headwater streams possessing tile drained agricultural watersheds in the midwestern United States have a minimal impact on these stream ecosystems in the first 4–6 years after planting. Our results are indicative of a need for changing how herbaceous riparian buffers are implemented adjacent to channelized headwater streams. Specifically, we recommend that future use of herbaceous riparian buffers be paired with upland management practices, riparian wetland creation, and/or instream habitat practices that are capable of addressing the chemical and physical habitat degradation exhibited by channelized agricultural headwater streams.

Acknowledgements

A. Casteneda, E. Dibble, and D. Shields Jr. reviewed an earlier draft of the manuscript and provided helpful comments. D. Gamble, S. Hess, A. Kemble, P. Levison, A. Rapp, G. Roberts, J. Risley, K. Seger, and R. Shaw assisted with field work. We also thank the other numerous past and current USDA-ARS Soil Drainage Research Unit personnel for their assistance with laboratory work and logistical support. D. Shields Jr., S. Knight, and L. Williams provided constructive feedback on the initial sampling design. R. Wilson provided information on design and implementation of herbaceous riparian buffers. R. Gates provided assistance with the statistical analyses. B. Bacon provided information on watershed characteristics. Landowner and site information were provided by Soil and Water Conservation and NRCS districts in Delaware and Morrow counties. We also are grateful to the landowners who provided access to the sites.

References

Barker, L.S., Felton, G.K., Russek-Cohen, E., 2006. Use of Maryland biological stream survey data to determine the effects of agricultural riparian buffers on measures of biological stream health. Environ. Monit. Assess. 117, 1–19.

Barling, R.D., Moore, I.D., 1994. Role of buffer strips in management of waterway pollution: a review. Environ. Manag. 18, 543–558.

Becker, G.C., 1983. Fishes of Wisconsin. University of Wisconsin Press, Madison, WI.

Brady, S.J., 2007. Effects of cropland conservation practices on fish and wildlife habitat. In: Haufler, J.B. (Ed.), Fish and Wildlife Response to Farmland Conservation Practices. Technical Review 07-1. the Wildlife Society, Bethesda, MD, pp. 9–23.

Castelle, A.J., Johnson, A.W., Connolly, C., 1994. Wetland and stream buffer size requirements – a review. J. Environ. Qual. 23, 878–882.

Clark, W.R., Reeder, K.F., 2007. Agricultural buffers and wildlife conservation: a summary about linear practices. In: Haufler, J.B. (Ed.), Fish and Wildlife Response to Farmland Conservation Practices. Technical Review 07-1. the Wildlife Society, Bethesda, MD, pp. 45–55.

Conover, W.J., 1999. Practical Nonparametric Statistics. Wiley, New York, NY.

Cooke, R.A., Sands, G.R., Brown, L.C., 2008. Drainage water management: a practice for reducing nitrate loads from subsurface drainage systems. In: Upper Mississippi River Sub-basin Hypoxia Nutrient Committee. Final Report: Gulf Hypoxia and Local Water Quality Concerns Workshop. American Society of Agricultural and Biological Engineers, St. Joseph, MI, pp. 19–28.

Correll, D.L., 2005. Principles of planning and establishment of buffer zones. Ecol. Eng. 24, 433–439.

Dorioz, J.M., Wang, D., Poulenard, J., Trevisan, D., 2006. The effect of grass buffer strips on phosphorus dynamics – a critical review and synthesis as a basis for application in agricultural landscapes in France. Agric. Ecosyst. Environ. 117, 4–21.

Dosskey, M.G., 2001. Toward quantifying water pollution abatement in response to installing buffers on crop land. Environ. Manag. 38, 577–598.

Etnier, D.A., Starnes, W.C., 1993. The Fishes of Tennessee. University of Tennessee, Knoxville, TN.

Fischer, J.R., Quist, M.C., Wigen, S.L., Schaefer, A.J., Stewart, T.W., Isenhart, T.M., 2010. Assemblage and population-level responses of stream fish to riparian buffers at multiple spatial scales. Trans. Am. Fish. Soc. 139, 185–200.

Frimpong, E.A., Lee, J.G., Sutton, T.M., 2006. Cost effectiveness of vegetative filter strips and instream half logs for ecological restoration. J. Am. Water Resour. Assoc. 42, 1349–1361.

FSA, 2010. FSA Agricultural Resource Conservation Program Handbook for State and County Offices. Revision 5. USDA Farm Service Agency, Washington, DC.

FSA, 2010. Monthly Conservation Reserve Program Summary – July 2010. USDA Farm Service Agency, Washington, DC.

Hickey, M.B.C., Doran, B., 2004. A review of the efficiency of buffer strips for the maintenance and enhancement of riparian ecosystems. Water Qual. Res. J. Can. 39, 311–317.

Hill, A.R., 1996. Nitrate removal in stream riparian zones. J. Environ. Qual. 25, 743–755.

Kincaid, C., 2005. Guidelines for selecting the covariance structure in mixed model analysis. Paper 198-30. In: Proceedings of the Thirtieth Annual SAS Users Group International Conference. SAS Institute Inc., Cary, NC.

Koroleff, J., 1983. Determination of total phosphorus by alkaline persulphate oxidation. In: Grasshoff, K., Ehrhardt, M., Kremling, K. (Eds.), Methods of Seawater Analysis. Verlag Chemie, Wienheim, pp. 136–138.

Krutz, L.J., Senseman, S.A., Zablotowicz, R.M., Matocha, M.A., 2005. Reducing herbicide runoff from agricultural fields with vegetative filter strips: a review. Weed Sci. 53, 353–367.

Leeds-Harrison, P.B., Quinton, J.N., Walker, M.J., Sanders, C.L., Harrod, T., 1999. Grassed buffer strips for the control of nitrate leaching to surface waters in headwater catchments. Ecol. Eng. 12, 299–313.

Littell, R.C., Henry, P.R., Ammerman, C.B., 1998. Statistical analysis of repeated measures data using SAS procedures. J. Anim. Sci. 76, 1216–1231.

Liu, X., Zhang, X., Zhang, M., 2008. Major factors influencing the efficacy of vegetated buffers on sediment trapping: a review and analysis. J. Environ. Qual. 37, 1667–1674.

Lovell, S.T., Sullivan, W.C., 2006. Environmental benefits of conservation buffers in the United States: evidence, promise, and open questions. Agric. Ecosyst. Environ. 112, 249–260.

Lyons, J., Trimble, S.W., Paine, L.K., 2000. Grass versus trees: managing riparian areas to benefit streams of central North America. J. Am. Water Resour. Assoc. 36, 919–930.

Magurran, A.E., 1988. Ecological Diversity and its Measurement. Croom Helm, London.

Malcom Pirnie, Inc., 1999. Upper Big Walnut Creek Watershed Water Quality Management Plan. Columbus, OH.

Marshall, D.W., Fayram, A.H., Panuska, J.C., Baumann, J., Hennessy, J., 2008. Positive effects of agricultural land use changes on coldwater fish communities in southwest Wisconsin streams. North Am. J. Fish. Manag. 28, 944–953.

Mayer, P.M., Reynolds Jr., S.K., McCutchen, M.D., Canfield, T.J., 2007. Meta-analysis of nitrogen removal in riparian buffers. J. Environ. Qual. 36, 1172–1180.

Meals, D.W., Dressing, S.A., Davenport, T.E., 2010. Lag time in water quality response to best management practices: a review. J. Environ. Qual. 39, 85–96.

Menzel, D.W., Vaccaro, R.F., 1964. The measurement of dissolved organic and particulate carbon in seawater. Limnol. Oceanogr. 9, 138–142.

Mitsch, W.J., Day Jr., J.W., 2006. Restoration of wetlands in the Mississippi-Ohio-Missouri (MOM) river basin: experience and needed research. Ecol. Eng. 26, 55–69.

Moser, E.B., 2004. Repeated measures modeling with proc mixed. Paper 188–29. In: Proceedings of the Twenty-ninth Annual SAS Users Group International Conference. SAS Institute Inc., Cary, NC.

Munro, N.T., Fischer, J., Wood, J., Lindenmayer, D.B., 2009. Revegetation in agricultural areas: the development of structural complexity and floristic diversity. Ecol. Appl. 19, 1197–1210.

Norris, V., 1993. The use of buffer zones to protect water quality: a review. Water Resour. Manag. 7, 257–272.

Ohio EPA, 2002. Field Evaluation Manual for Ohio's Primary Headwater Habitat Streams. Ohio Environmental Protection Agency, Division of Surface Water, Columbus, OH.

Ohio EPA, 2003. Biological and Water Quality Study of the Big Walnut Creek Basin 2000. Delaware, Fairfield, Franklin, Licking, Morrow, and Pickaway Counties, Ohio. OEPA Technical Report DSW/EAS 2003-11-10, Ohio Environmental Protection Agency, Division of Surface Water, Columbus, OH.

Ohio EPA, 2004. Total Maximum Daily Loads for the Big Walnut Creek Watershed. Draft Report for Public Comment, December 8, 2004. State of Ohio Environmental Protection Agency, Division of Surface Water, Columbus, OH.

Osborne, L.L., Kovacic, D.A., 1993. Riparian vegetated buffer strips in water quality restoration and stream management. Freshw. Biol. 29, 243–258.

Pappas, E.A., Huang, C., Bucholtz, D.L., 2008. Implications of sampling frequency to herbicide conservation effects assessment. J. Soil Water Conserv. 63, 410–419.

Parsons, T.R., Maita, Y., Lalli, C.M., 1984. A Manual of Chemical and Biological Methods for Seawater Analysis. Pergamon Press, Oxford.

Pflieger, W.L., 1975. The Fishes of Missouri. Missouri Department of Conservation, Jefferson City, MO.

Reichenberger, S., Bach, M., Skitschak, A., Freds, H., 2007. Mitigation strategies to reduce pesticide inputs into ground- and surface water and their effectiveness: a review. Sci. Total Environ. 284, 1–35.

Robison, H.W., Buchanan, T.M., 1988. Fishes of Arkansas. The University of Arkansas Press, Fayetteville, AR.

Ross, S.T., 2002. The Inland Fishes of Mississippi. University Press of Mississippi, Jackson, MS.

Sanders, R.E., 2001. A Guide to Ohio Streams. Watkins Printing, Columbus, OH.

SAS Institute, 1999. Statistical Analysis System for Windows, version 8. Cary, NC.

Shearer, K.S., Xiang, W.N., 2007. The characteristics of riparian buffer studies. J. Environ. Inf. 9, 41–55.

Smiley Jr., P.C., Dibble, E.D., Schoenholtz, S.H., 2005. Fishes of first-order streams in north-central Mississippi. Southeast Nat. 4, 219–236.

Smiley Jr., P.C., Gillespie, R.B., 2010. Influence of physical habitat and agricultural contaminants on fishes within agricultural drainage ditches. In: Moore, M.T., Kroger, R. (Eds.), Agricultural Drainage Ditches: Mitigation Wetlands for the 21st Century. Research Signpost, Kerula, India, pp. 37–73.

Smiley Jr., P.C., Gillespie, R.B., King, K.W., Huang, C., 2008. Relative contributions of habitat and water quality to the integrity of fish communities in agricultural drainage ditches. J. Soil Water Conserv. 63, 218A–219A.

Smiley Jr., P.C., Gillespie, R.B., King, K.W., Huang, C., 2009a. Management implications of the relationships between water chemistry and fishes within channelized headwater streams in the midwestern United States. Ecohydrology 2, 294–302.

Smiley Jr., P.C., Shields Jr., F.D., Knight, S.S., 2009b. Designing impact assessments for evaluating the ecological effects of conservation practices on streams in agricultural landscapes. J. Am. Water Resour. Assoc. 45, 867–878.

Smith, D.R., Livingston, S.J., Zuercher, B.W., Larose, M., Heathman, G.C., Huang, C., 2008. Nutrient losses from row crop agriculture in Indiana. J. Soil Water Conserv. 63, 396–409.

Smith, B., Wilson, J.B., 1996. A consumer's guide to evenness indices. Oikos 76, 70–82.

Teels, B.M., Rewa, C.A., Myers, J., 2006. Aquatic condition response to riparian buffer establishment. Wildl. Soc. Bull. 34, 927–935.

U.S. EPA, 1995. Methods for the Determination of Organic Compounds in Drinking Water Supplement III. EPA-600/R-95/131 National Exposure Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati, OH.

Wang, L., Lyons, J., Kanehl, P., 2006. Habitat and fish responses to multiple agricultural best management practices in a warm water stream. J. Am. Water Resour. Assoc. 42, 1047–1062.

Westra, J.V., Zimmerman, J.K.H., Vondracek, B., 2005. Bioeconomic analysis of selected conservation practices on soil erosion and freshwater fisheries. J. Am. Water Resour. Assoc. 41, 309–322.

Whitworth, M.R., Martin, D.C., 1990. Instream benefits of CRP filter strips. In: McCabe, R.E. (Ed.), Transactions of the 55th North American Wildlife and Natural Resources Conference. Wildlife Management Institute, Washington, DC, pp. 40–45.

Yuan, Y., Bingner, R.L., Locke, M.A., 2009. A review of effectiveness of vegetative buffers on sediment trapping in agricultural areas. Ecohydrology 2, 321–336.

Zedler, J.B., 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Front. Ecol. Environ. 2003, 65–72.

Zhang, X., Liu, X., Zhang, M., Dahlgren, R.A., Eitzel, M., 2010. A review of vegetated buffers and a meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. J. Environ. Qual. 39, 76–84.