Reducing Herbicides and Veterinary Antibiotics Losses from Agroecosystems Using Vegetative Buffers

Chung-Ho Lin,* Robert N. Lerch, Keith W. Goyne, and Harold E. Garrett

Multiple species vegetative buffer strips (VBSs) have been recommended as a cost-effective approach to mitigate agrochemical transport in surface runoff derived from agronomic operations, while at the same time offering a broader range of long-term ecological and environmental benefits. However, the effect of VBS designs and species composition on reducing herbicide and veterinary antibiotic transport has not been well documented. An experiment consisting of three VBS designs and one continuous cultivated fallow control replicated in triplicate was conducted to assess effectiveness in reducing herbicide and antibiotic transport for claypan soils. The three VBS designs include (i) tall fescue, (ii) tall fescue with a switchgrass hedge barrier, and (iii) native vegetation (largely eastern gamagrass). Rainfall simulation was used to create uniform antecedent soil moisture content in the plots and to generate runoff. Our results suggested that all VBS significantly reduced the transport of dissolved and sediment-bound atrazine, metolachlor, and glyphosate in surface runoff by 58 to 72%. Four to 8 m of any tested VBS reduced dissolved sulfamethazine transport in the surface runoff by more than 70%. The tall fescue VBS was overall most effective at reducing dissolved tylosin and enrofloxacin transport in the runoff (>75%). The developed exponential regression models can be used to predict expected field-scale results and provide design criteria for effective field implementation of grass buffers. Our study has demonstrated that an optimized VBS design may achieve desired agrochemical reductions and minimize acreage removed from crop production.

The presence of organic agrichemicals in water resources and drinking water supplies continues to be a substantial environmental issue. Herbicides have been documented to have adverse impacts on aquatic ecosystems (Hayes et al., 2002), and herbicide concentrations in drinking water have been regulated for many years in the United States because of potential human health effects (Hayes et al., 2002; Swan et al., 2003; USEPA, 1996, 2007). Veterinary antibiotics (VAs) are a class of emerging contaminants receiving increased attention due to potential adverse effects on environmental quality. Veterinary antibiotics have been detected in surface waters (Kim and Carlson, 2007; Lindsey et al., 2001), although the long-term human health effects associated with consuming trace quantities of antibiotics in drinking water are unknown. Of equal or greater concern is the development of antibiotic-resistant bacteria near confined animal feed operations (Chee-Sanford et al., 2001; Koike et al., 2007; Sapkota et al., 2007) and the spread of VA-resistant bacteria to other organisms (Blanco et al., 2009).

A well designed vegetative buffer strip (VBS) can provide many benefits to the ecosystem (De Groot et al., 2002; Lovell and Sullivan, 2006). In particular, VBSs are recognized as one of the most effective approaches to mitigate surface water runoff and contaminant loss from agroecosystems (Blanco-Canqui et al., 2004; Krutz et al., 2003), and we hypothesize that VBSs may also have utility for reducing VA loss. With respect to organic contaminants, VBSs mitigate transport by the following processes: (i) enhancing infiltration and, subsequently, increasing solute–soil interaction; (ii) decreasing surface water runoff velocity, thus promoting the deposition of sediment-bound pollutants; (iii) promoting diverse microbial communities capable of degrading organic agrichemicals; and (iv) improving the capacity of soil to sorb and retain organic contaminants (Krutz et al., 2005; Liu et al., 2008; Reichenberger et al., 2007).

There are several factors that affect VBS efficacy for mitigating surface transport of organic contaminants, including plant species composition, soil properties, buffer width and placement, source to buffer area ratio, runoff flow type (i.e., sheet vs. concentrated flow), landscape, weather and climate,

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Abbreviations: ATR, atrazine; ENF, enrofloxacin; GLY, glyphosate; MET, s-metolachlor; SLF, sulfamethazine; SW, switchgrass; TF, tall fescue; TY, tylosin; VA, veterinary antibiotic; VBS, vegetative buffer strip.

and contaminant chemical properties (Liu et al., 2008; Reichenberger et al., 2007; Sabbagh et al., 2009). The variation in these factors among studies leads to a broad range of reported herbicide trapping efficiencies (Lin et al., 2007; Mersie et al., 1999; Seybold et al., 2001). With respect to VBS design, the factors that can be managed are vegetation type, buffer width, buffer placement, and some soil properties (e.g., pH).

Although the efficacy of VBS for removing organic agrochemicals has been proven in many previous studies, the VBS conservation practices have had marginal success in promoting the use of VBS in many parts of the country (Skelton et al., 2005). These programs often fail due to inappropriate design or complicated guidance (Skelton et al., 2005; Rodriguez et al., 2009). Simple and practical guidelines related to choosing appropriate plant species and VBS widths for specific contaminants are often needed for decision-making, education, training, and technology transfer purposes. There have been many detailed mechanistic, hydrologic models successfully developed to simulate and predict the effectiveness of VBS in reducing the transport of the agrochemicals (Sabbagh et al., 2009). However, a set of site-specific hydrological parameters needs to be determined before any recommendation can be prescribed. Therefore, the goal of this study is to use an empirical modeling approach to deliver a simple but practical tool to action agencies, land managers, and agroforestry practitioners for implementation of VBS near Midwest claypan region. The specific objectives of this study were (i) to compare the effectiveness of three grass buffer treatments in reducing transport of herbicides and VAs in surface runoff and (ii) to establish design criteria, relative to grass buffer widths, for estimating compound-specific load reductions using developed empirical models.

Materials and Methods

Experimental Design

A field rainfall simulation study was conducted during 2004 and repeated during 2006 to test the ability of different VBS designs to mitigate herbicide and antibiotic transport in surface runoff. The experiment, which consisted of three VBS designs and one continuous cultivated fallow control replicated in triplicate, was conducted to assess effectiveness in reducing herbicide and antibiotic transport for claypan soils. The three 8-m VBS treatments included (i) tall fescue (*Festuca arundinacea*) (TF), (ii) TF with a 1-m wide switchgrass (*Panicum virgatum* L.) (SW) hedge at the upslope end of the VBS, and (iii) native warm-season grasses (mainly eastern gamagrass [*Tripsacum dactyloides*] and SW). The continuous cultivated fallow treatment was used as control treatment. Twelve plots (1.5 m wide, 16 m long) with four treatments replicated three times established at the site in 2001 were arranged in a randomized, complete block design (Fig. 1). The plantation was fully mature in 2003, and the plots were maintained with mechanic weed control and fertilization of N, P, K, and micronutrients (Peters 20:20:20; Micromax Micronutrients, The Scotts Company, Marysville, OH) every spring. Each plot is paired with a 1.5 by 8 m source area located directly upslope of the 8-m treatment area, and the source area was managed under continuous cultivated fallow

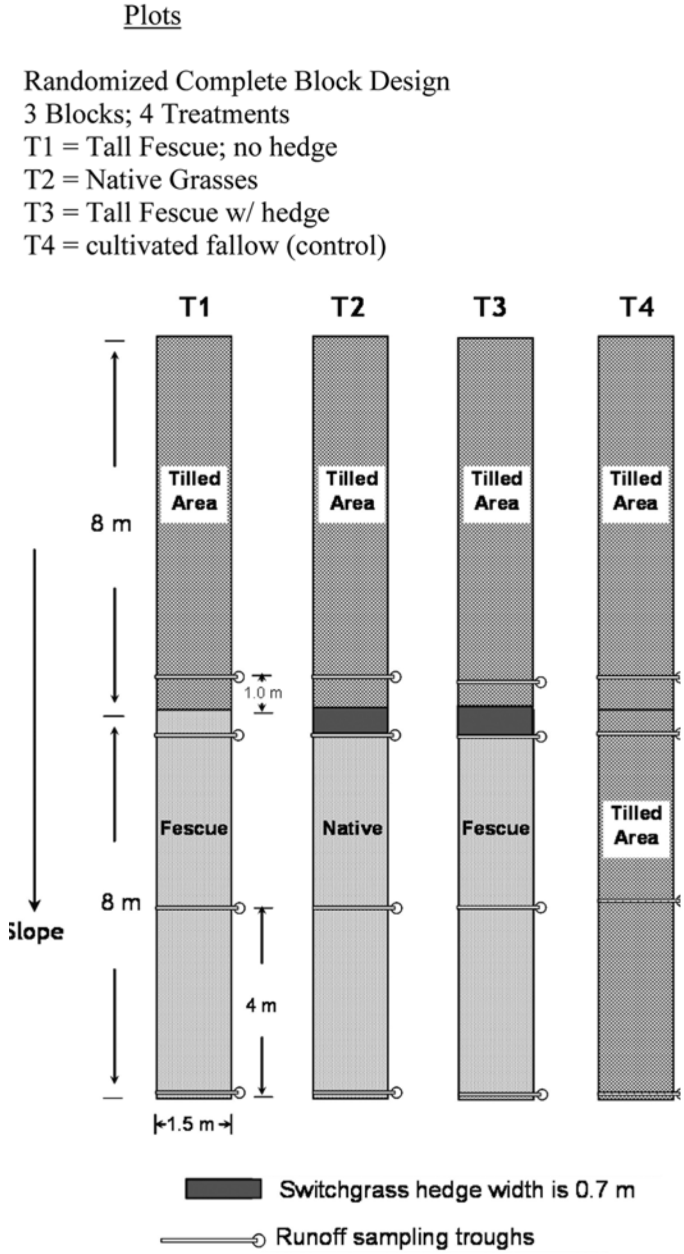


Fig. 1. Schematic diagram showing vegetative buffers and control.

conditions. Runoff collection equipment was installed at locations 1 m above the buffers (−1 m) and within the buffers at 1, 4, and 8 m downslope from the most elevated VBS edge. Thus, this design represents source area to buffer ratios ranging from 1:1 to 8:1. The study was conducted on an eroded Mexico silt loam soil (fine, smectitic, mesic Vertic Epiaqualfs) of 5% slope located at the University of Missouri Bradford Research and Extension Center near Columbia, Missouri. The physical characteristics of the soils collected from each treatment are summarized in Table 1.

A rotating-boom rainfall simulator was used to create uniform antecedent soil moisture content in the plots before herbicide/VA application and to generate runoff after. The rainfall application rate was ~5 cm h^{-1}. Three herbicides, atrazine (ATR) (2-chloro-4-ethylamine-6-isopropylamine-s-triazine), glyphosate (GLY) (N-phosphonomethyl-glycine), and s-metolachlor (MET) (2-chloro-N-2-ethyl-6-methyl-phenyl-

N-1-methoxypropan-2-yl acetamide), and three antibiotics, tylosin (TY) (9-hydroxymethyl-anthracene), sulfamethazine (SLF) (2-*p*-aminobenzenesulfonamido-4,6-dimethylpyrimidine), and enrofloxacin (ENF) (1-cyclopropyl-7–4-ethyl-1-piperazinyl-6-fluoro-1,4-dihydro-4-oxo-3-quinolonecarboxylic acid), were uniformly applied by a well calibrated backpack sprayer to the source areas approximately 24 h before the rainfall application that generated surface water runoff. Application rates of the organic compounds were: ATR, 2.2 kg ha^{-1}; MET, 1.7 kg ha^{-1}; GLY, 1.5 kg ha^{-1}; TY, 4.65 kg ha^{-1}; SLF, 4.29 kg ha^{-1}; and ENF, 4.65 kg ha^{-1}. The selected properties of the herbicides and antibiotics studied are described in Tables 2 and 3.

Sample Collection, Herbicide Analyses, and Load Calculations

Water and suspended sediment samples were collected, starting with the initiation of surface water runoff at the 8-m sampler. At each sampling position within each buffer, samples were collected as a function of time to create composited samples for analysis. Samples were collected for a duration of 5 s at 10-min intervals for 60 min. Individual sample volumes were recorded in the field, and then the samples collected at each location were composited to create a single sample for each of the four samplers within a buffer (i.e., −1, 1, 4, and 8 m). The results of the sampling procedure produced a total of 48 aqueous and 48 sediment samples during each year. The concentrations of herbicides and antibiotics in filtered water samples were quantified using enzyme-linked immunosorbent assays (ELISAs) (Abraxis Kits, Warminster, PA). For herbicide analyses, magnetic particle ELISAs were used, with standard concentrations ranging from 0 to 5 μg L^{-1}. For antibiotic analyses, 96-well plate ELISAs (Abraxis Kits) were used, with standard concentrations ranging from 0 to 50 μg L^{-1}. Analysis of each sample was performed in triplicate, and samples were diluted as needed to achieve concentrations within the range of the calibration standards. The detection limit in water samples was 0.05 μg L^{-1} (ppb).

Herbicide concentrations in sediment were determined after the addition of the flocculent alum [$Al_2(SO_4)_3$] at a concentration of 0.7 g L^{-1} to promote settling of the suspended sediment and to facilitate recovery of sufficient sediment mass for analyses. The addition of $Al_2(SO4)_3$ did not remove the herbicides (USEPA, 2000). Atrazine and MET were extracted twice with 80% MeOH followed by liquid-liquid extraction with chloroform and then solvent exchange into ethyl acetate. Concentrations of extracted ATR and MET were determined by a Varian gas chromatograph with a Saturn 2200 (Varian Inc., Walnut Creek, CA) ion trap mass spectrometer (GC/MS). Details of the GC/MS analysis were described by Lin et al. (2008). Glyphosate was extracted with 1 mol m^{-3} NaOH, and concentrations were determined by the ELISA procedure described above. Suspended sediment concentration was determined by oven-drying 500 mL of the sample in a preweighed beaker at 70°C. To determine the relative load of dissolved herbicide, VAs and sediment transported in surface runoff, herbicide, VAs, and sediment concentrations were multiplied by the total flow volume derived from the hydrographs at each sampling position. The total load of sediment at each sampling position was then multiplied by the concentrations of herbicides in sediment to calculate the sediment-bound herbicide transport in the runoff. The herbicide and VA mass at each sampling position was then normalized to the total mass at −1 m samplers. No sediment analyses were conducted for the antibiotics; thus, only dissolved-phase loads are presented. The biotic and abiotic degradation of the agrochemicals in VBS was considered negligible during the 60-min runoff simulation because the applied pollutants were not in contact with VBS soils until the surface runoff was initiated.

The infiltration potential (reductions in flow volume) was determined by subtracting the volume of outflow at the 8 m sampling position from volume of the inflow at the −1 m sampling position. The flow volume at each sampling position was calculated based on the integration of the hydrographs. Due to the design of the stainless steel berms, presumably no water was lost off the plots, and evapotranspiration should have been minimal for the short duration of the events. Thus, any reductions in volume must have been due to infiltration.

Statistical Analyses

Two-way factorial ANOVA was performed using SYSTAT statistical software (Crane Software International Ltd., Bangalore, India) to compare the effects of treatments (species × buffer width) within blocking factor (year) on load reduction. Differences between the treatments least adjusted means were tested with Fisher's LSD probability test with a significance level of 0.1 ($\alpha = 0.1$). Nonlinear regression analysis using first-order exponential decay model was performed to calculate the predicted reduction rates.

Results and Discussion

Effectiveness of Vegetative Buffer Strips at Reducing Herbicide Transport

Results from 2004 and 2006 (Fig. 2) indicated that all VBS treatments significantly reduced the load of ATR, MET, and

Table 1. Soil physical properties for each treatment.

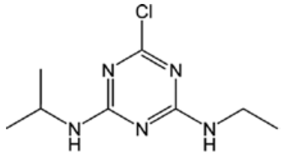
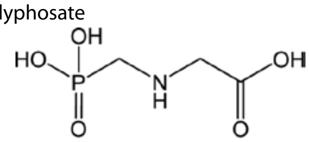
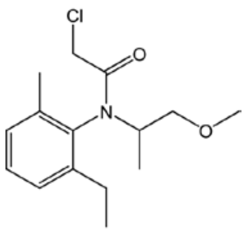
	Clay	Silt	Sand	Total OC†	CEC‡	Bulk density ρb
	%				$cmol_c\ kg^{-1}$	g cm^{-3}
Control	25.6 (±1.3)§	51.8 (±7)	22.6 (±6.1)	1.4 (±0.2)	19.8 (±0.2)	1.21 (±0.07)
Tall fescue	24.6 (±1.5)	53.3 (±3.8)	22.0 (±4.9)	2.2 (±0.2)	21.2 (±0.6)	1.03 (±0.09)
Hedge + fescue	23.7 (±2.7)	52.8 (±4)	23.4 (±2.8)	2.1 (±0.3)	20.5 (±0.8)	1.10 (±0.13)
Native	26.1 (±3.4)	50.6 (±2.4)	23.3 (±3.1)	2.1 (±0.4)	20.4 (±1.7)	1.13 (±0.05)

† OC, organic carbon content.

‡ CEC, cation exchange capacity determined at pH 7 using the ammonium acetate method.

§ Values in parentheses are SD.

Table 2. Selected properties of the herbicides studied.

Chemical	Molecular weight	S_w‡	log K_{ow}§	K_{oc}¶	Field half-life	pK_a#
	g mol^{-1}	mg L^{-1}			d	
Atrazine	215.68	33	2.68	147	173	1.68
		(29.9–52)	(2.34–2.80)	(38–288)	(13–402)	(1.68–1.70)
Glyphosate	169.1	1.2 × 10^{4}	−1.6	2100	37	5.6
				(500–2640)	(2–174)	
Metolachlor	283.8	488	2.6	70	141	–
		(488–550)	(2.6–3.28)	(22–307)	(12–292)	

† Data were obtained from the USDA–ARS Pesticide Properties Database (USDA–ARS, 2007). Individual values represent a recommended value for a select property, and values in parentheses correspond to the range of values reported.

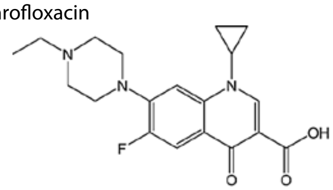
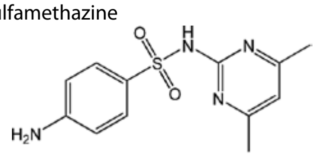
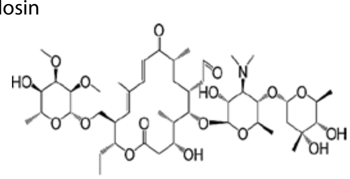
‡ S_W, water solubility.

§ K_{OW}, octanol-water partition coefficient.

¶ K_{OC}, organic carbon partition coefficient.

pK_a, acid dissociation constant.

Table 3. Selected properties of the veterinary antibiotics studied.

Chemical	Molecular weight	S_w‡	log K_{ow}§	K_{oc}¶	Field half-life	pK_a#
	g mol^{-1}	mg L^{-1}			d	
Enrofloxacin	359.4	1.3 × 10^{5}	1.1	16,510–76,8740	–	6.27
						~8.3
Sulfamethazine	278.3	1.5 × 10^{3}	0.89	80–139	–	2.65
						7.65
Tylosin	917.1	500	3.5	553–7990	–	7.1
						

† Data were obtained from Tolls (2001) and Thiele-Bruhn (2003).

‡ S_W, water solubility.

§ K_{OW}, octanol-water partition coefficient

¶ K_{OC}, organic carbon partition coefficient.

pK_a, acid dissociation constants.

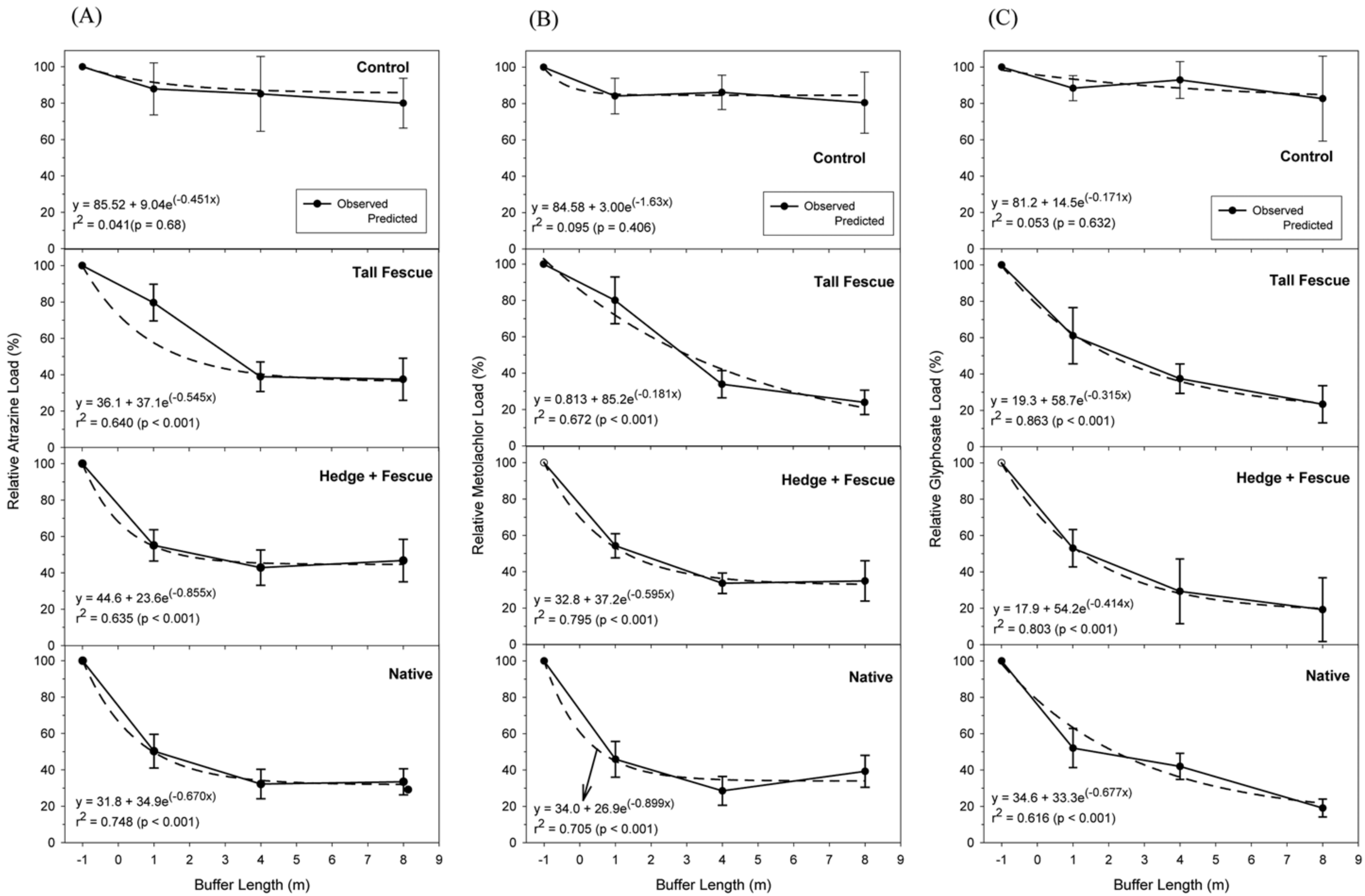


Fig. 2. Relationship between relative total herbicide load reduction (*y*) and buffer length (*x*) for (A) atrazine, (B) s-metolachlor, and (C) glyphosate. The total herbicide load reduction was determined by dissolved and sediment-bound transport of herbicides. Error bars represent 95% confidence interval. *y*-intercept, load reduction at the interface between source and buffer; *x*-intercept, buffer length required to completely remove the pollutants.

GLY in surface runoff, including dissolved-phase and sediment-bound transport ($p < 0.1$) (Fig. 2A, 2B, 2C). All the buffer treatments were similarly effective in reducing ATR and MET transport at 4 and 8 m into the buffers ($p > 0.1$), but TF was overall less effective at 1 m into the buffer ($p < 0.1$). Four meters of VBS removed about 58 to 72% of ATR and MET and 60 to 71% of GLY in surface runoff, and additional buffer width improved the GLY trapping efficiency. In contrast to the VBS treatments, the relative herbicide loads of the control treatments were not affected by the distance. Overall, the herbicide trapping efficiency of VBS was determined by the buffer width, species composition, and chemical properties of the herbicides.

Buffer width has been shown to be an important factor to the herbicide trapping efficiency of VBS. In this study, the mitigation effectiveness of VBS for reducing herbicide loads in surface runoff followed a first-order exponential decay relationship as a function of buffer width. Regression models developed for VBS from this study can be successfully used to predict the load reduction rates in differing VBS designs implemented on claypan soils ($r^2 > 0.61$; $p < 0.001$). Our results suggested that the source to buffer area ratios of 8:1 or greater can still effectively reduce herbicide transport in surface runoff. Because field implementation of buffers will likely result in source to buffer area ratios >20:1, this relationship can provide an estimate of anticipated field-scale effects. For example, at a source to buffer area ratio of 20:1 (equivalent to 0.4 m of buffer width), the expected reduction of ATR loads for the treatments are as follows: (i) TF, 34%; (ii) TF+SW, 38%; and (iii) native plant species, 42%. Four meters of the buffer treatments were equally effective at reducing ATR and MET loads, as was 8 m for all treatments in both years. Thus, source to buffer area ratios less than 2:1 provide no additional mitigation for ATR and MET. On the other hand, our results indicated that 8 m of VBS provided additional benefits by reducing GLY loads by an additional 10 to 21%, compared with 4 m of buffer width ($p = 0.015$). Similar results were obtained from previous studies, which reported that buffer width is an important factor in reducing contaminant and sediment transport and that greater buffer widths are required to trap fine-grained particles and moderately sorbed pesticides (Reichenberger et al., 2007; Liu et al., 2008; Krutz et al., 2005). In contrast, Sabbagh et al. (2009) developed an empirical model based on the partitioning of pesticides between solution and sorbed phases in which buffer width was not a statistically significant parameter.

The results from regression simulation and ANOVA analysis suggested that the effects of species on ATR and GLY load reduction between TF, native, and TF+SW were not significantly different ($p > 0.1$). Native VBS had the greatest effectiveness at reducing the transport of MET ($p = 0.079$; $\alpha = 0.1$). The addition of 1 m of SW hedge to TF enhanced the reduction of ATR, MET, and GLY load by approximately 25, 30, and 10%, respectively, compared with TF alone, but it did not affect the overall reduction at 4 and 8 m.

Differences in the chemical properties of the herbicides also greatly affected the extent of transport and the mitigation mechanism of the VBS. Our study showed that ATR and MET were primarily transported in the dissolved phase (97% or more), whereas GLY showed a high proportion of sediment-bound transport (41–69%). The total loss of total applied herbicides via surface runoff was <4%, with averages of 2.7, 3.7, and 2.4% for ATR, MET, and GLY, respectively.

The mobility of the herbicides studied decreases in the following order: ATR > Met > GLY (Bowman, 1990; Sanchez-Marten et al., 1994). Much stronger adsorption of the ionic GLY to sediments leads to the much higher proportion of sediment-bound transport relative to ATR and MET. Native and TF+SW VBS were more effective in sediment trapping than the TF VBS ($p = 0.019$). Eight meters of native or TF+SW VBS reduced the sediment load by 80%, whereas 8 m of TF VBS trapped 47% of the sediment load. At a source to buffer area ratio of 20:1 (equivalent to 0.4 m of buffer width), the expected reductions in total GLY loads for the treatments are (i) TF, 14.1%; (ii) TF+SW, 36.3%; and (iii) native, 29.2%. The strongly sorbing GLY was largely removed by sediment deposition and trapping within the first 4 m of VBS. Therefore, sediment trapping and erosion control by VBS were identified as the primary mitigation mechanisms reducing the transport of GLY. On the other hand, the primary mechanisms by which VBS reduced transport of dissolved-phase ATR and MET were enhanced infiltration and reduced runoff volume resulting from plant evapotranspiration and greater water holding capacity in the surface soil. In addition, the accumulation of organic matter in VBS (Table 1) may facilitate the sorption and retention of these moderately hydrophobic herbicides (Krutz et al., 2006). Therefore, buffers can reduce herbicide transport via enhanced infiltration as well as through sediment trapping, depending on the chemical properties of the herbicide. This study also showed that the VBSs were capable of increasing the infiltration of surface runoff for a high-runoff-potential claypan soil ($p = 0.01$). As compared with control treatment, our results have shown that 8 m of TF, TF+SW hedge, and native VBS reduced approximately 992, 743, and 1039 L of the runoff volumes, respectively, during the 60-min rainfall simulation period.

Effectiveness of Vegetative Buffer Strips at Reducing Veterinary Antibiotic Transport

With 4 to 8 m of VBS, SLF loads in the surface runoff were significantly reduced by more than 70% ($p = 0.002$), but the effectiveness between plants species was not significantly different ($p = 1$) (Fig. 3A). One meter of VBS reduced 32 to 40% of dissolved SLF. For TY and ENF, the TF VBS had the greatest impact on reducing transport in runoff ($p = 0.05$ and 0.001 for TY and ENF, respectively) (Fig. 3B and 3C). Four meters of TF VBS removed more than 75% of dissolved TY and ENF in the surface runoff. Similar to our findings for ATR and MET, 4 m of the buffer treatment was equally effective at reducing SLF and TY loads, as was 8 m of VBS for all treatments (SLF, $p = 0.48$; TY, $p = 0.26$). The native and TF+SW VBS did not significantly affect TY transport. The average total applied antibiotic losses via the dissolved phase were 11.7% for SLF, 0.18% for TY, and 0.076% for ENF, demonstrating that SLF is more mobile than TY and ENF. This is in agreement with Davis et al. (2006), who documented much higher runoff losses of sulfonamides than the other classes of VAs on application to cropland.

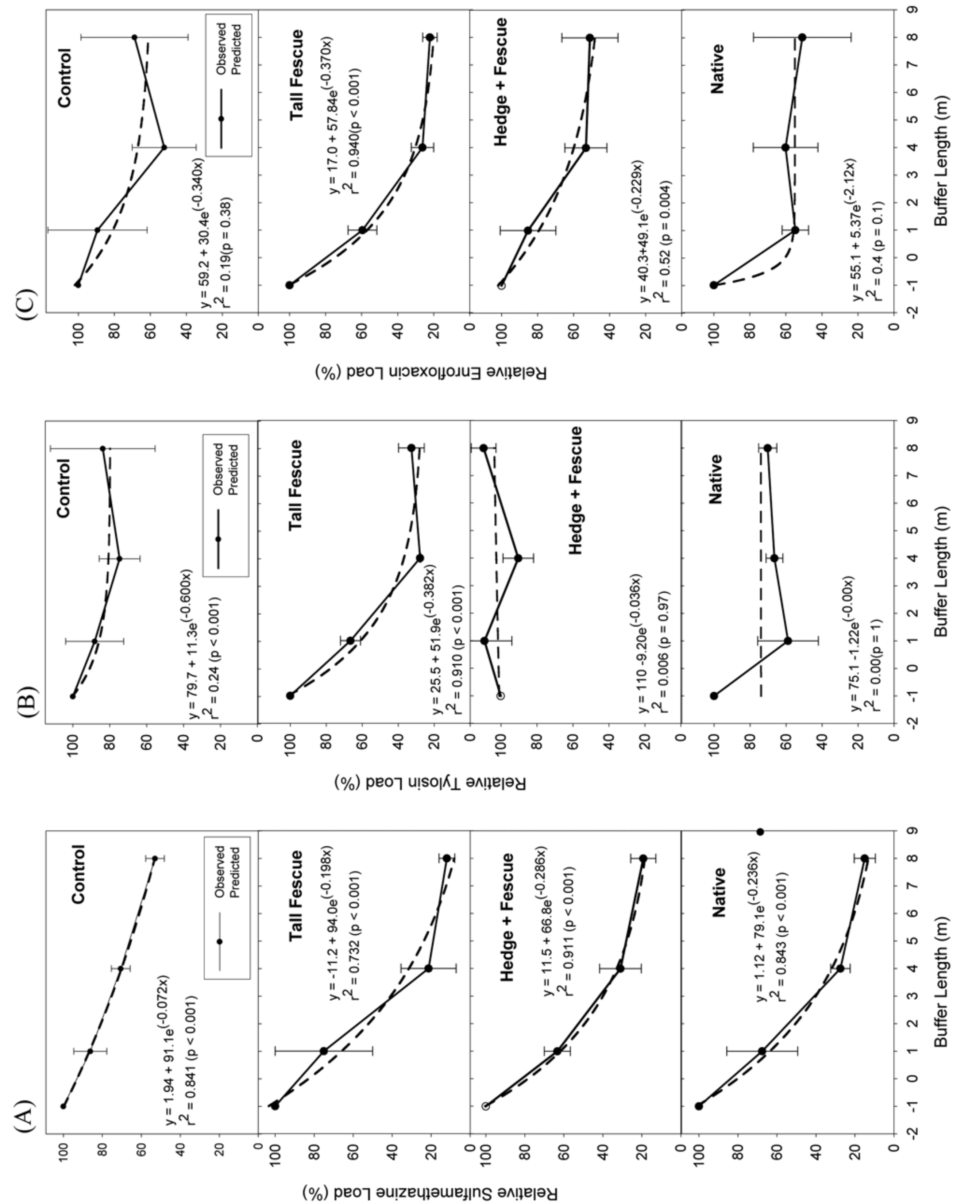


Fig. 3. Relationship between relative dissolved antibiotics load reduction (*y*) and buffer length (*x*) for (A) sulfamethazine, (B) tylosin, and (C) enrofloxacin. Error bars represent 95% confidence interval. *y*-intercept, load reduction at the interface between source and buffer; *x*-intercept, buffer length required to completely remove the pollutants.

The processes governing herbicide fate also apply to VAs. Similar to the results from the herbicide runoff study, grass VBS treatments generally showed that the mitigation effectiveness for SLF and ENF load followed a first-order exponential decay relationship with buffer width. The nonlinear regression models developed in this study allowed us to estimate and predict the load reduction of SLF ($r^2 > 0.73$; $p < 0.001$) and ENF ($r^2 = 0.4$–0.94; $p < 0.1$) by VBS as a function of the buffer width. These equations can be used to calculate the buffer width required to achieve the desired reduction rates with a given source to buffer area ratios. However, the effect of VBS width on TY trapping was not uniform across the VBS treatments. Except for TF VBS ($r^2 = 0.91$; $p < 0.001$), the exponential decay model did not adequately fit TY load reduction in the VBS studied.

Relative concentrations and runoff loss of the VAs measured in surface runoff in the present study were also very similar to those reported from previous studies (Davis et al., 2006). It has been documented that the mobilities of the studied VAs decrease in the following order: SLF > TY > ENF. The ranges of sorption coefficients (K_d, L kg^{-1}) for these compounds were 0.9 to 3.5, 62 to 128, and 427 to 5612, respectively (Thiele-Bruhn, 2003). The SLF concentrations in runoff were two orders of magnitude greater than TY and ENF. Due to the low K_d and high water solubility of sulfonamides, we anticipated the SLF loss to occur predominantly in the aqueous phase and that losses via suspended particulates would be low (<1%), which has previously been reported to be the case (Kay et al., 2005; Kreuzig and Höltge, 2005). Therefore, the predominant mitigation mechanism for SLF was enhanced infiltration resulting from the greater water-holding capacity of the soil in the VBS treatments. Moreover, the root exudates and root decomposition products in the VBS, including specific functional groups such as phenolic and carboxylic groups, N-heterocyclic compounds, and lignin decomposition products, are believed to serve as preferred binding sites that may further limit SLF mobility in the VBS (Cheng and Kuzyakov, 2005; Thiele-Bruhn et al., 2004). Recent work in our laboratory has demonstrated enhanced sulfonamide sorption to VBS soils relative to cropland soils (Chu et al., 2010), and we have also documented enhanced sulfonamide dissipation in the root zone of particular plant species commonly used in VBS (Lin et al., 2010).

For TY and ENF, which are the VAs with high soil/water partition coefficients, the relative load reduction by VBS would be anticipated to be much higher if the sediment-bound transport is taken into account. Davis et al. (2006) reported that more than 50% of TY loss is associated with the sediment-bound transport. Enrofloxacin is expected to have had the highest relative losses associated with sediment among the VAs studied because of its extremely high partition coefficients (Table 3).

Design and Limitation

This work illustrates the additional ecosystem services that VBS systems may provide to improve environmental quality. We have demonstrated the effectiveness of VBS at reducing agrochemical transport. Furthermore, load reductions as a function of buffer width generally showed a first-order exponential decay relationship, demonstrating that a relatively short VBS can be highly effective at reducing transport of herbicides and VAs. This has important practical implications for the development of design criteria that can achieve meaningful reductions in contaminant transport while minimizing the amount of land taken out of production. In contrast to the empirical model developed by Sabbagh et al. (2009), our results suggested that the buffer width is one of the predominant factors dictating agrochemical load reduction by VBS. These predictive regression models will be useful to landowners, regulatory agencies, and consultants for optimizing VBS designs based on the pollutant, source to buffer ratios, and VBS species composition for claypan soils and perhaps other soils with restrictive subsoil horizons. However, many factors may affect the successful long-term effectiveness of VBS to mitigate herbicide and VA losses from agroecosystems. For example, these native warm-season species can be difficult to maintain and establish, and they are not well suited for growth in colder climates or shaded environments (Lin et al., 2004). Thus, cool-season C3 species may be effective alternatives for grass buffers when the use of a C4 species is impractical. In addition, C4 and C3 species have different tolerances to herbicide exposure, which may also dictate the choice of grass species for use in a VBS (Lin et al., 2004). Field implementation of VBS that use C3 and C4 grasses could enhance season-long effectiveness for reducing agrochemical pollutants transport because of their complementary growth patterns and detoxification mechanisms.

The studies presented here show that VBS can be an effective strategy for reducing organic chemical transport. More importantly, the knowledge generated from this study provides a simple but practical tool to conservation agencies, land managers, and agroforestry practitioners for the implementation of VBS.

Conclusions

This field rainfall simulation study demonstrated that all grass VBS treatments significantly reduced the transport of dissolved and sediment-bound herbicides and dissolved SLF in surface runoff. The TF treatment was overall most effective at reducing VA transport in runoff. The TF+SW treatment did not significantly affect TL transport. Buffers can reduce the transport of these agrochemical pollutants via enhanced infiltration of runoff as well as through sediment trapping. The mitigation effectiveness of VBS for the herbicides, SLF, and ENF followed a first-order decay relationship as function of buffer width. The developed exponential regression models can be used to predict expected field-scale results and provide design criteria for effective field implementation of grass buffers. The implementation of the tested grass VBS could provide desired reductions in herbicide and VA transport in surface runoff from cropland. Additionally, an optimized VBS design may achieve desired agrochemical reductions and remove less land from production.

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References

Blanco, G., J.A. Lemus, and J. Grande. 2009. Microbial pollution in wildlife: Linking agricultural manuring and bacterial antibiotic resistance in red-billed choughs. Environ. Res. 109:405–412. doi:10.1016/j.envres.2009.01.007

Blanco-Canqui, H., C.J. Gantzer, S.H. Anderson, E.E. Alberts, and A.L. Thompson. 2004. Grass barrier and vegetative filter strip effectiveness in reducing runoff, sediment, nitrogen, and phosphorus loss. Soil Sci. Soc. Am. J. 68:1670–1678. doi:10.2136/sssaj2004.1670

Bowman, B.T. 1990. Mobility and persistence of alachlor, atrazine and metolachlor in plainfield sand, and atrazine and isazofos in honeywood silt loam, using field lysimeters. Environ. Toxicol. Chem. 9:453–461. doi:10.1002/etc.5620090406

Chee-Sanford, J.C., R.I. Aminov, I.J. Krapac, N. Garrigues-Jeanjean, and R.I. Mackie. 2001. Occurrence and diversity of tetracycline resistance genes in lagoons and groundwater underlying two swine production facilities. Appl. Environ. Microbiol. 67:1494–1502. doi:10.1128/AEM.67.4.1494-1502.2001

Cheng, W., and Y. Kuzyakov. 2005. Root effects on soil organic matter decomposition. *In* R. Zobel and S. Wright (ed.) Roots and soil management: Interactions between roots and the soil. Agron. Monogr. 48. ASA, CSSA, SSSA, Madison, WI.

Chu, B., K.W. Goyne, S.H. Anderson, C.-H. Lin, and R.P. Udawatta. 2010. Veterinary antibiotic sorption to agroforestry buffer, grass buffer and cropland soils. Agrofor. Syst. 79:67–80. doi:10.1007/s10457-009-9273-3

Davis, J.G., C.C. Truman, S.C. Kim, J.C. Ascough II, and K. Carlson. 2006. Antibiotic transport via runoff and soil loss. J. Environ. Qual. 35:2250–2260. doi:10.2134/jeq2005.0348

De Groot, R.S., M.A. Wilson, and R.M.J. Boumans. 2002. A typology for the classification, description, and valuation of ecosystem functions, goods, and services. Ecol. Econ. 41:393–408. doi:10.1016/S0921-8009(02)00089-7

Hayes, T., K. Haston, M. Tsui, A. Hoang, C. Haeffele, and A. Vonk. 2002. Herbicides: Feminization of male frogs in the wild. Nature 419:895–896. doi:10.1038/419895a

Kay, P., P.A. Blackwell, and A.B.A. Boxall. 2005. Transport of veterinary antibiotics in overland flow following the application of slurry to arable land. Chemosphere 59:951–959. doi:10.1016/j.chemosphere.2004.11.055

Kim, S.-C., and K. Carlson. 2007. Temporal and spatial trends in the occurrence of human and veterinary antibiotics in aqueous and river sediment matrices. Environ. Sci. Technol. 41:50–57. doi:10.1021/es060737+

Koike, S., I.G. Krapac, H.D. Oliver, A.C. Yannarell, J.C. Chee-Sanford, R.I. Aminov, and R.I. Mackie. 2007. Monitoring and source tracking of tetracycline resistance genes in lagoons and groundwater adjacent to swine production facilities over a 3-year period. Appl. Environ. Microbiol. 73:4813–4823. doi:10.1128/AEM.00665-07

Kreuzig, R., and S. Höltge. 2005. Investigations on the fate of sulfadiazine in manured soils: Laboratory experiments and test plot studies. Environ. Toxicol. Chem. 24:771–776. doi:10.1897/03-582R.1

Krutz, L.J., S.A. Senseman, R.M. Zablotowicz, and M.A. Matocha. 2005. Reducing herbicide runoff from agricultural fields with vegetative filter strips: A review. Weed Sci. 53:353–367. doi:10.1614/WS-03-079R2

Krutz, L.J., S.A. Senseman, M.C. Dozier, D.W. Hoffman, and D.P. Tierney. 2003. Infiltration and adsorption of dissolved atrazine and atrazine metabolites in buffalograss filter strips. J. Environ. Qual. 32:2319–2324. doi:10.2134/jeq2003.2319

Krutz, L.J., T.J. Gentry, S.A. Senseman, I.L. Pepper, and D.P. Tierney. 2006. Mineralisation of atrazine, metolachlor and their respective metabolites in vegetated filter strip and cultivated soil. Pest Manag. Sci. 62:505–514. doi:10.1002/ps.1193

Lin, C.H., R.N. Lerch, H.E. Garrett, and M.F. George. 2004. Incorporating forage grasses in riparian buffers for bioremediation of atrazine, isoxaflutole and nitrate in Missouri. Agrofor. Syst. 63:91–99. doi:10.1023/B:AGFO.0000049437.70313.ef

Lin, C.H., R.N. Lerch, H.E. Garrett, and M.F. George. 2008. Bioremediation of atrazine-contaminated soil by forage grasses: Transformation, uptake, and detoxification. J. Environ. Qual. 37:196–206. doi:10.2134/jeq2006.0503

Lin, C.H., R.N. Lerch, H.E. Garrett, C.J. Gantzer, and S.H. Anderson. 2007. Utilizing vegetative buffer strips to remove dissolved and sediment-bound atrazine, metolachlor and glyphosate from surface water runoff. p. 113–121. *In* Proceedings of the 10th North American Agroforestry Conference, 9–14 June 2007, Quebec, Canada.

Lin, C.-H., K.W. Goyne, R.J. Kremer, R.N. Lerch, and H.E. Garrett. 2010. Dissipation of sulfamethazine and tetracycline in the root zone of grass and tree species. J. Environ. Qual. 39:1269–1278. doi:10.2134/jeq2009.0346

Lindsey, M.E., M. Meyer, and E.M. Thurman. 2001. Analysis of trace levels of sulfonamide and tetracycline antimicrobials in groundwater and surface water using solid-phase extraction and liquid chromatography/mass spectrometry. Anal. Chem. 73:4640–4646. doi:10.1021/ac010514w

Liu, X., X. Zhang, and M. Zhang. 2008. Major factors influencing the efficacy of vegetated buffers on sediment trapping: A review and analysis. J. Environ. Qual. 37:1667–1674. doi:10.2134/jeq2007.0437

Lovell, S.T., and W.C. Sullivan. 2006. Environmental benefits of conservation buffers in the United States: Evidence, promise, and open questions. Agric. Ecosyst. Environ. 112:249–260. doi:10.1016/j.agee.2005.08.002

Mersie, W., C.A. Seybold, C. McNamee, and J. Huang. 1999. Effectiveness of switchgrass filter strips in removing dissolved atrazine and metolachlor from runoff. J. Environ. Qual. 28:816–821. doi:10.2134/jeq1999.00472425002800030012x

Reichenberger, S., M. Bach, A. Skitschak, and H.-G. Frede. 2007. Mitigation strategies to reduce pesticide inputs into ground- and surface water and their effectiveness: A review. Sci. Total Environ. 384:1–35. doi:10.1016/j.scitotenv.2007.04.046

Rodriguez, J.M., J.J. Molnar, R.A. Fazio, E. Sydnor, and M.J. Lowe. 2009. Barriers to adoption of sustainable agriculture practices: Change agent perspectives. Renewable Agric. Food Syst. 24:60–71. doi:10.1017/S1742170508002421

Sanchez-Martin, M.J., T. Crisanto, M. Arienzo, and M. Sanchez-Camazano. 1994. Evaluation of the mobility of C14-labelled pesticides in soils by thin layer chromatography using a linear analyser. J. Environ. Sci. Health B 29:473–484. doi:10.1080/03601239409372891

Sabbagh, G.J., G.A. Fox, A. Kamanzi, B. Roepke, and J.-Z. Tang. 2009. Effectiveness of vegetative filter strips in reducing pesticide loading: Quantifying pesticide trapping efficiency. J. Environ. Qual. 38:762–771. doi:10.2134/jeq2008.0266

Sapkota, A.R., F.C. Curriero, K.E. Gibson, and K.J. Schwab. 2007. Antibiotic-resistant enterococci and fecal indicators in surface water and groundwater impacted by a concentrated swine feeding operation. Environ. Health Perspect. 115:1040–1045. doi:10.1289/ehp.9770

Seybold, C., W. Mersie, and D. Delorem. 2001. Removal and degradation of atrazine and metolachlor by vegetative filter strips on clay loam soil. Commun. Soil Sci. Plant Anal. 32:723–737. doi:10.1081/CSS-100103904

Skelton, P., S.J. Josiah, J.W. King, J.R. Brandle, G.A. Helmers, and C.A. Francis. 2005. Adoption of riparian forest buffers on private lands in Nebraska, USA. Small-scale For. Econ. Manag. Policy 4:185–204.

Swan, S.H., C. Brazil, E.Z. Drobnis, F. Liu, R.L. Kruse, M. Hatch, J.B. Redmon, C. Wang, and J.W. Overstreet. 2003. Geographic differences in semen quality of fertile U.S. males. Environ. Health Perspect. 111:414–420. doi:10.1289/ehp.5927

Thiele-Bruhn, S. 2003. Pharmaceutical antibiotic compounds in soils: A review. J. Plant Nutr. Soil Sci. 166:147–167.

Thiele-Bruhn, S., T. Seibicke, H.-R. Schulten, and P. Leinweber. 2004. Sorption of sulfonamide pharmaceutical antibiotics on whole soils and particle-size fractions. J. Environ. Qual. 33:1331–1342. doi:10.2134/jeq2004.1331

Tolls, J. 2001. Sorption of veterinary pharmaceuticals in soils: A review. Environ. Sci. Technol. 35:3397–3406. doi:10.1021/es0003021

USDA–ARS. 2010. The ARS pesticide properties database. Available at http://www.ars.usda.gov/Services/docs.htm?docid=14199 (verified 2 Mar. 2011).

USEPA. 1996. Drinking water regulations and health advisories. U.S. Gov. Print. Office, Washington, DC.

USEPA. 2000. Summary of pesticide removal/transformation efficiencies from various drinking water treatment processes. Prepared for the Committee to Advise on Reassessment and Transition (CARAT), 3 Oct. 2000. USEPA, Alexandria, VA.

USEPA. 2007. Preliminary interpretation of the ecological significance of atrazine stream-water concentrations using a statistically-designed monitoring program. Office of Prevention, Pesticides, and Toxic Substances. Alexandria, VA.