




TECHNICAL REPORT

Surface Water Quality

Vegetative buffer strips show limited effectiveness for reducing antibiotic transport in surface runoff

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Assigned to Associate Editor Tucker Burch.

Funding information

U.S. Department of Agriculture - Agricultural Research Service

Abstract

Vegetative buffer strips (VBS) have been demonstrated to effectively reduce loads of sediment, nutrients, and herbicides in surface runoff, but their effectiveness for reducing veterinary antibiotic (VA) loads in runoff has not been well documented. The objective of this study was to determine the effectiveness of VBS vegetation and width on surface runoff loads of the VAs sulfamethazine (SMZ) and lincomycin (LIN). Experimental design of the plots (1.5 × 25 m) was a two-way factorial with four vegetation treatments (tall fescue [*Festuca arunifera* Schreb.]; tall fescue with switchgrass [*Panicum virgatum* L.] hedge; warm-season native grass mix; and continuous fallow control), and four buffer widths (0, 2, 5, and 9 m). Turkey litter spiked with SMZ and LIN was applied to the source area (upper 7 m) of each plot, and runoff was collected at each width. Runoff was generated with a rotating boom simulator. Results showed VA loads in runoff at the 0-m sampler ranged from 3.8 to 5.9% of applied, and overall VA transport in runoff was predominately in the dissolved phase (90% for SMZ and 99% for LIN). Among vegetation treatments, only tall fescue significantly reduced loads of SMZ and LIN compared with the control, with load reductions of ~30% for both VAs. Estimated field-scale reductions in VA loads showed that source-to-buffer area ratios (SBARs) of 10:1 to 20:1 reduced VA loads by only 7 to 16%. Overall, the grass VBS tested here were less effective at reducing SMZ and LIN loads in surface runoff than has been previously demonstrated for sediment, nutrients, and herbicides.

Abbreviations: DOC, dissolved organic carbon; INL, input normalized load; LIN, lincomycin; SBAR, source-to-buffer area ratio; SMZ, sulfamethazine; TF, tall fescue; VA, veterinary antibiotics; VBS, vegetative buffer strips.

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1 | INTRODUCTION

The presence of veterinary antibiotics (VA) in water resources and drinking water supplies continues to be a substantial environmental issue affecting human and aquatic ecosystem health (Barrios et al., 2020; Brown et al., 2015). Surface water runoff from manured agroecosystems is considered a primary mechanism for VA transport to surface waters (Hall et al., 2020; Iverson et al., 2022; Lin et al., 2011), and surface runoff events are more problematic for soils with water-restrictive subsoil horizons such as claypans (Anderson et al., 2009; Liu et al., 2020). The presence of such water-restrictive subsoil horizons results in more frequent surface runoff events, greatly increasing the risk of pesticides and VA transport to surface water resources (Lerch and Blanchard, 2003; Udawatta et al., 2006). The global distribution of soils with water restrictive subsoil horizons comprises an area of approximately 2.9 million km² (USDA-NRCS, 1998), which may exacerbate VA transport and presence in streams, rivers, and lakes. Recent studies indicate that vegetative buffer strips (VBS) may have potential for mitigating VA loss from soils prone to surface water runoff events (Chu et al., 2013; Lin et al., 2010, 2011; Unger et al., 2013).

Vegetative buffer strips are land areas of either planted or indigenous vegetation situated between a pollutant source area and a surface water body that receives runoff. These systems are recognized as an effective conservation practice to mitigate the loss of sediment, nutrients, and organic pollutants in surface runoff from agroecosystems (Blanco-Canqui et al., 2004a, 2004b; Lerch et al., 2017; Lin et al., 2011; Reichenberger et al., 2007; Udawatta et al., 2006, 2021). The effectiveness of VBS to reduce surface transport of organic contaminants is affected by numerous factors, including soil properties, buffer width, source-to-buffer area ratio (SBAR), buffer placement, runoff regime (e.g., sheet vs. concentrated flow), slope, rainfall intensity, antecedent soil water content, buffer vegetation, and the chemical properties of the contaminant (Lin et al., 2011; Reichenberger et al., 2007). Because these factors vary considerably between studies, a wide range of mitigation efficiencies have been reported in the literature. However, of the various factors involved, those that can be managed include vegetation type, buffer width, and buffer placement.

A variety of plant species in various VBS designs have shown the capability to enhance the degradation of organic agrichemical contaminants trapped in the rhizosphere because of their ability to stimulate microbial growth and enzyme activities and to produce phytochemical root exudates (Lin et al., 2010; Willett et al., 2013). Forage grasses are commonly used in VBS as they are well adapted to temperate environments, provide wildlife habitat, can be well established in 1–2 yr, and have low maintenance costs (Lovell and Sullivan,

Core Ideas

- Buffer width affected veterinary antibiotic (VA) load reduction more than vegetation treatment.
- Relative VA loads in runoff were similar to pesticide losses (3.8–5.9% of applied).
- Transport of VAs was mainly in the dissolved phase, resulting in limited load reductions by buffers.
- To achieve meaningful VA load reductions at field-scale requires a source-to-buffer area ratio of <20:1.

2006; Udawatta et al., 2021). Some of the species demonstrated to enhance agrichemical degradation in soil include cool season (C3) grasses such as tall fescue (*Festuca arundinacea* Schreb.) and orchardgrass (*Dactylis glomerata* L.) and warm season (C4) grasses such as switchgrass (*Panicum virgatum* L.) and eastern gamagrass [*Tripsacum dactyloides* (L.) L.] (Lin et al., 2003, 2011). Combinations of C3 and C4 species have also been used in VBS by the placement of stiff-stemmed C4 hedges, such as switchgrass, immediately upslope of a C3 buffer strip (Blanco-Canqui et al., 2004a, 2004b; Lerch et al., 2017; Lin et al., 2011). The C4 hedges have been shown to reduce runoff velocity, increase sediment deposition, and prevent or deter the formation of concentrated flow into the VBS (Dunn and Dabney, 1996).

Buffer width is an important factor to contaminant transport and sediment trapping, and greater buffer widths are required to trap fine-grained particles and moderately sorbed organic pollutants (Lerch et al., 2017; Lin et al., 2011; Liu et al., 2008; Reichenberger et al., 2007). Several studies have shown that the mass of pollutant removed from transport load was an exponential (first-order) function of buffer width (Lerch et al., 2017; Lin et al., 2011). This relationship was observed across a variety of grass and tree species and for three commonly used herbicides transported in the dissolved phase and sediment-bound.

With regard to VBS placement, factors to consider include proximity to the contaminant source, SBAR, drainage patterns in the field, soil wetness, and soil erodibility (Reichenberger et al., 2007). The combination of buffer width and placement will ultimately determine the SBAR. In field settings, this ratio is expected to be 10:1 or greater (Lerch et al., 2017; Lin et al., 2011). In general, VBS effectiveness will increase with decreasing SBAR (Lerch et al., 2017; Lin et al., 2011; Liu et al., 2008). Site-specific factors will determine the optimal width and/or SBAR that strikes a balance between undersized VBS that do not achieve desired reductions in contaminant

load and oversized buffers that unnecessarily remove land from production (Liu et al., 2008).

A significant amount of research has been conducted in Missouri, USA, to investigate the ability of VBS to reduce runoff and agrichemical transport on claypan soils (Blanco-Canqui et al., 2004a, 2004b; Udawatta et al., 2006, 2021; Lin et al., 2011; Lerch et al., 2017). Plot-scale simulated rainfall studies have demonstrated the effectiveness of grass VBS for reducing runoff, sediment, inorganic nutrients, and herbicides when water entered the buffers via concentrated flow or sheet flow (Blanco-Canqui et al., 2004a, 2004b; Lerch et al., 2017; Lin et al., 2011). Studies have shown VBS having improved soil physical properties relative to adjacent croplands and pasture, such as greater saturated hydraulic conductivity, macro and mesoporosity, water storage capacity, aggregate stability, and lower bulk density (Udawatta et al., 2008a, 2008b, 2021). Additionally, soil microbial activities, enzymatic activities, and soil organic carbon content have been observed to be higher in agroforestry and grass buffers relative to cropland soils (Udawatta et al., 2008b, 2021; Unger et al., 2013). These aforementioned studies suggest that soil physical, chemical, and biological properties within VBS should be amenable for reducing VA loss via surface water runoff from agroecosystems.

Sulfamethazine (SMZ) and lincomycin (LIN) are widely used in livestock production as growth promoters and as broad-spectrum antibiotics (FDA, 2020; Wei et al., 2011; Zhao et al., 2010). Both SMZ and LIN have shown environmental behavior indicating their high potential to be mobile in soils with the potential to contaminate water resources (Hall et al., 2020; Kolpin et al., 2002; Kuchta et al., 2009; Pan and Chu, 2017; Wang et al., 2012). The effectiveness of VBS for mitigating the transport of VAs on high runoff potential soils has not been well studied and is fundamental in the design of mitigation strategies to reduce their environmental exposure. Thus, this study evaluated the ability of buffer strips to act as barriers to SMZ and LIN residues in surface runoff. The specific objectives of this study were to assess the effects of buffer width (0, 2, 5, and 9 m) and vegetation (fallow control, tall fescue, tall fescue plus switchgrass hedge, and native grasses) on runoff transport of SMZ and LIN present in surface-applied manure and estimate VBS effectiveness on VA load reduction in surface runoff over a range of field-scale SBARs.

2 | MATERIALS AND METHODS

2.1 | Experimental design

Details of the VBS plot establishment, design, management, and maintenance of the buffers were previously described in detail (Blanco-Canqui et al., 2004a, 2004b; Lerch et al., 2017), with additional details in the Supplemental Material (Plot

Descriptions and Maintenance; Supplemental Figure S1 and Table S1). The VBS plots were established in 2002 and maintained as mature grass stands in this experiment. The plots are located at the University of Missouri Bradford Research and Extension Center (17 km east of Columbia, MO, USA) within the Central Claypan Areas (Major Land Resource Area 113). The experimental design of the plots (1.5 × 16 m) was a two-way factorial design with four vegetation treatments (tall fescue [TF]; TF with a 0.7-m-wide switchgrass hedge at the upslope end of the VBS [Hedge + TF]; native warm-season grasses, mainly comprising Indiangrass [*Sorghastrum nutans* L.], eastern gamagrass, and switchgrass [Native]; and a continuous fallow [control]) and four buffer widths (0, 2, 5, and 9 m). The vegetation treatments were replicated three times and arranged in a completely randomized design (Supplemental Figure S1). All plots had a slope of 5% from top to bottom. Soil within all plots was an eroded Mexico silt loam (fine, smectitic, mesic, Vertic Epiaqualfs) with 2.2% organic matter and pH of 6.7 (Supplemental Table S1). The Mexico silt loam is a typical claypan soil representative of the Central Claypan Areas of northern Missouri and southern Illinois. The high runoff potential of these soils has been well established (Lerch and Blanchard, 2003; Sadler et al., 2015).

2.2 | Manure application

Dry organic turkey litter procured from a farm that does not use antibiotics was spiked with SMZ and LIN, then applied to the surface of the upper 7 m (source area) of the plots approximately 16–20 h before simulated runoff was generated (and after wetting the plots). Turkey litter was spiked with both VAs (see Supplemental Material, Antibiotic Addition to Turkey Litter) to achieve a concentration of 100 mg kg⁻¹ on a dry weight basis. The turkey litter was applied at an equivalent rate of 11 Mg ha⁻¹, a rate commonly used by poultry producers (Heathman et al., 1995). The VA concentrations used in this study were based on VA concentration data reported for manure found in concentrated animal feeding operations (Pan and Chu, 2017; Zhao et al., 2010).

2.3 | Rainfall simulations and sample collection

A rotating-boom rainfall simulator was used to produce uniform antecedent soil water content in the plots before manure application and to generate runoff following application (see Supplemental Material, Runoff Collection and Rainfall Simulation section). The combination of the simulated rainfall rate (~70 mm h⁻¹) with events of ~85-min duration (Supplemental Table S2) translated to a storm recurrence interval of 25–50 yr for the local area (National Weather, Service, 2022);

thus, the simulated events represented a worst-case scenario in which a major runoff event occurred within 24 h of manure application.

The rainfall simulations took place over three consecutive weeks in September 2020, with two simulations (two VBS plots per simulation) taking place each week. To assess the effect of buffer width, runoff sample collectors were installed in each plot at the upslope end of the grass buffers (0 m) and within the buffers at 2, 5, and 9 m, representing SBARs of 7:2 to 7:9. Runoff samples were collected with the initiation of runoff at the 9-m sampler (i.e., runoff was generated over the entire plot area) for a given plot. Seven samples were collected at each sampler location at 10-min intervals from time 0–60 min. Samples were collected for 5–10 s, with the exact sampling time recorded. Sample volume collected was measured with 1,000-ml graduated cylinders and recorded. Samples were composited in 20-L polypropylene carboys for each sampler location within each plot, resulting in 48 composite samples (12 plots \times 4 samplers plot⁻¹) for each dataset.

2.4 | Antibiotic analyses

Composited runoff samples were separated into suspended sediment fractions and water fractions by adding the flocculant $\text{AlK}(\text{SO}_4)_2 \cdot 12\text{H}_2\text{O}$ (Fisher Scientific) to the carboys and allowing complete flocculation to be achieved at ~ 24 h. All fractionated samples were analyzed for suspended sediment concentration, dissolved concentration, and sediment-bound antibiotic concentrations. Details of the sample processing and analysis of all water and sediment samples can be found in the Supplemental Material (Sections 1.1–1.3). Concentrated extracts were analyzed on a Waters Xevo triple quadrupole mass spectrometer (MS/MS) coupled with a Waters Acquity ultra-pressure liquid chromatography (UPLC) H-Class LC system. Results of method validation indicated detection limits ranged between 0.01 and 0.11 $\mu\text{g L}^{-1}$ in water and between 0.05 and 3.26 $\mu\text{g kg}^{-1}$ in sediment. Final concentrations of VAs in water and sediment samples were corrected for matrix effects and surrogate recoveries.

2.5 | Statistical analyses

Data for relative VA loads (i.e., % of applied) at the 0-m sampler, antecedent soil-water content, runoff initiation time, and simulated rainfall rates were analyzed using the general linearized model for one-way analysis of variance with vegetation type as the factor tested. For all VA loads in runoff (dissolved phase, sediment-bound, and totals), normally distributed data were obtained by computing input normalized loads (INLs) relative to the 0-m sampler, expressed as 100%

of the load, and changes in load at other buffer widths were expressed as a percentage of the 0-m load. Because INLs at the 0-m sampler were always 100%, without variance, these data were not included in the ANOVA. The INL data, excluding 0 m, were analyzed as a two-way factorial (vegetation type and buffer width as factors) using the mixed ANOVA procedure (PROC MIXED) in SAS 4.3 (SAS Institute, Inc.) with plots as a random factor (i.e., completely randomized design). All reported treatment means are least square means unless otherwise noted. Differences between treatment means were determined by the PDIF procedure, with an a priori significance level of $\alpha = .10$ due to the highly variable data. Linear and nonlinear regressions were used (SigmaPlot 12.3; Systat Software, Inc.) to correlate INLs to buffer width. Multiple models were explored including linear, quadratic, two parameter first-order decay, and three parameter first-order decay using the Regression Wizard tool in SigmaPlot. Only the highest correlated models with valid parameter estimates are shown, and models were considered significant at $\alpha = .05$ based on the F statistic p -values from the regression ANOVAs performed by SigmaPlot.

3 | RESULTS AND DISCUSSION

3.1 | Hydrologic data

The rainfall simulator performed consistently over the course of the experiment, providing very similar antecedent water content, rainfall rates, and runoff initiation times across the vegetation treatments (Supplemental Table S2). Runoff was significantly affected by vegetation treatment and buffer width ($p = .0003$), and the vegetation \times width interaction was also significant ($p = .029$) (Figure 1). The control had the lowest average runoff depth (57 mm) of any vegetation treatment and significantly lower runoff than the Hedge + TF and Native treatments (Figure 1A). Runoff significantly and exponentially decreased with buffer width (Figure 1B). The vegetation \times buffer width interaction resulted from the varying runoff responses of the grass treatments with buffer width, while the control had consistently lower runoff at every width (Figure 1C). The control was significantly more effective at reducing runoff depth than at least one of the grass treatments at each buffer width beyond 0 m (Supplemental Table S4).

Greater infiltration of the runoff in the control compared with the grass treatments was an unexpected result that was not consistent with earlier studies on these same plots in which VBS decreased surface runoff by enhancing infiltration (Lerch et al., 2017; Lin et al., 2011). Given the conditions of this experiment, tillage of the control plots, which were tilled over their entire length, apparently resulted in greater infiltration than the grass treatments. The end of summer season when the study took place coincided with an annual period

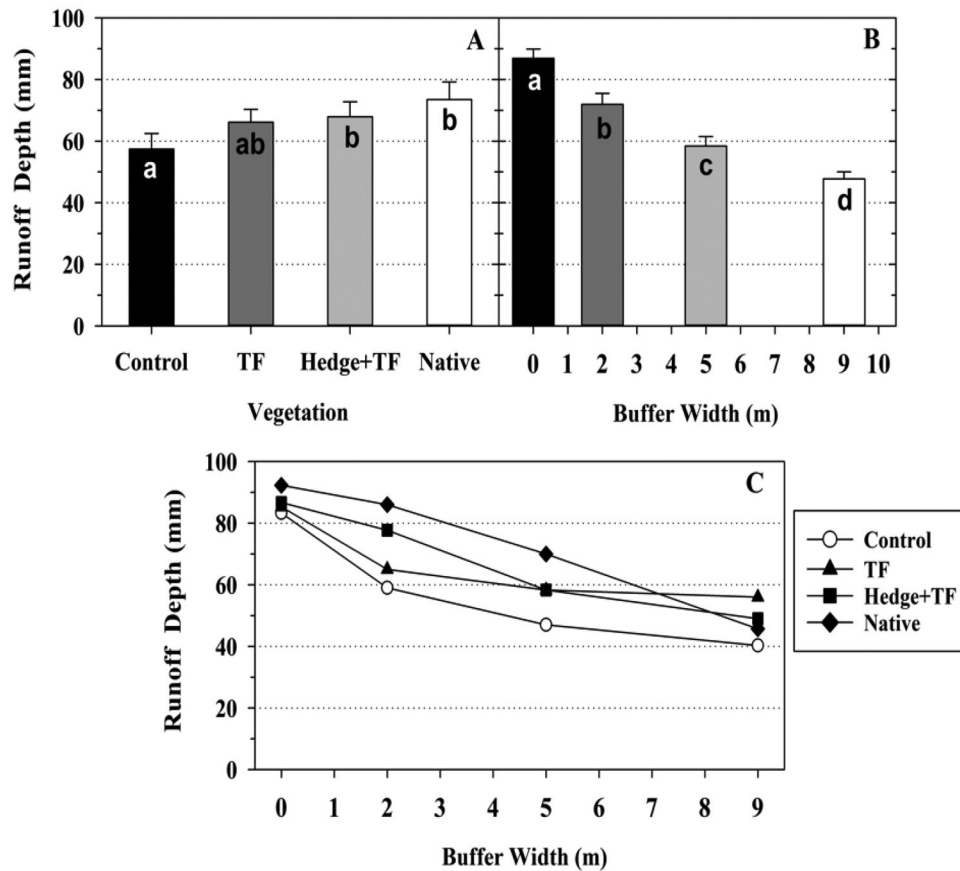


FIGURE 1 Effect of (A) vegetation type and (B, C) buffer width on runoff. Bars with different letters were significantly different at $\alpha = .10$. Vegetation \times width interaction was significant ($p = .029$). Error bars represent the 95% confidence intervals. TF, tall fescue

where water infiltration into soil under grass vegetation is reduced due to extremely hot and dry conditions that facilitated soil crusting. Tilling can break surface soil crusting and increase surface roughness, leading to enhanced infiltration compared with the untilled grass buffers. While grass roots may increase porosity and improve soil structure, resulting in enhanced infiltration and percolation via channeling of water through macropores (Rachman et al., 2004), the observed increases in infiltration in the control plots were greater than the grass treatments, suggesting that tillage resulted in the observed decreases in surface runoff.

3.2 | Relative veterinary antibiotic loads

Relative VA loads at the 0-m sampler were consistent over the vegetation treatments (Supplemental Figure S2) with no significant differences between treatments for either target compound. This was an important result as it showed that the VA loads at the 0-m sampler were comparable across all treatments. Mean relative loads entering the VBS for SMZ varied from 3.8% of applied VA for the Hedge + TF to 5.8% for the Native treatment. The LIN results were similar,

with relative loads ranging from 4.4% for the Hedge + TF to 5.9% for the control treatment. These results were similar to reported relative loads for these VAs from plots, fields, and incubation assays (Albero et al., 2018; Barrios et al., 2020; Kuchta et al., 2009). Thus, the relative load results indicated that the intense simulated storms represented robust scenarios for testing the ability of VBS to reduce VA transport.

3.3 | Dissolved phase and sediment-bound VA transport

Dissolved phase transport as a proportion of total VA load at the 0-m sampler was $90 \pm 2.6\%$ (95% CI) for SMZ and $99 \pm 0.3\%$ for LIN. These results are consistent with previous runoff studies (Albero et al., 2018; Barrios et al., 2020; Kuchta et al., 2009). The results in the current study showed SMZ to have a greater affinity for soil sorption compared with LIN and the importance of both dissolved phase and sediment-bound transport to SMZ losses in runoff. The sorption distribution coefficient (K_d) value range for SMZ (0.6–3.1 L kg⁻¹; Boxall et al., 2003; Tolls, 2001) is reported

to be lower than the K_d value range for LIN (29–72 L kg⁻¹; Wang et al., 2012) so SMZ was expected to have a higher proportion of total VA load in the dissolved phase when compared to LIN, but this was not observed in this study. Previous studies reported increases of SMZ sorption in soils after the addition of manure as a soil amendment, following the hypothetical rule that highly polar compounds, such as SMZ, can have an increased sorption affinity after manure amendment (Wang et al., 2015). Conversely, Albero et al. (2018) reported that compared with unamended soil, manure-amended soil had decreased LIN sorption and a higher proportion of LIN transport in runoff was in the dissolved phase. These observations were attributed to the high affinity of LIN for dissolved organic carbon (DOC) causing more LIN to partition into the dissolved phase and decreasing its interaction with soil colloids (Albero et al., 2018). Although DOC might enhance the sorptive behavior of some antibiotics through cumulative sorption or co-sorption, other processes such as competition or co-transport with DOC in runoff may facilitate their mobility in soil (Chu et al., 2013; Haham et al., 2012). In the present study, DOC content was not measured in water samples but DOC likely influenced dissolved phase LIN transport by preventing sorption of LIN to the soil or plant detritus, as inferred from the literature.

3.4 | Total INLs for veterinary antibiotics

Both vegetation type and buffer width significantly affected total (dissolved + sediment-bound) INLs for each VA ($p = .6309$ for SMZ, $p = .0351$ for LIN). Compared with the control, the TF and Native treatments significantly reduced SMZ total INLs by 29 and 22% respectively (Figure 2). The Hedge+TF treatment did not significantly affect total SMZ loads, with only a 17% reduction in SMZ INLs compared with the control. Within vegetation treatment differences showed that at 2 m none of the grass treatments significantly reduced total SMZ and LIN INLs compared with the control, and only TF significantly reduced total SMZ loads, at 5 m by 40% (Supplemental Table S5). At 9 m, the TF and native treatments significantly reduced total SMZ INLs compared with the control, but the Hedge+TF treatment did not. The effect of buffer width (averaged over vegetation treatments) on total SMZ INLs showed that compared with 0 m, the 2-, 5-, and 9-m treatments significantly reduced INLs, but there were no significant differences among these three widths (Figure 2). Buffer width effects within each vegetation treatment showed the same pattern except TF, which significantly reduced total SMZ INLs at 9 m compared with 2 and 5 m (Supplemental Table S5). For LIN, the TF treatment significantly reduced total LIN INLs compared with the control by 34%, but the Hedge+TF and Native treatments did not significantly reduce total LIN loads. Within vegetation treat-

ment effects showed that the grass treatments significantly reduced total LIN INLs compared with the control only at 9 m, by 47–61% (Supplemental Table S5). The effect of buffer width on total LIN INLs showed that loads significantly decreased with increasing buffer width. Within the individual treatments, all three treatments showed significant reduction in total LIN INLs among the >0-m buffer widths, with a 47% load reduction in the Native grass treatment, a 48% load reduction in the Hedge+TF treatment, and a 61% load reduction in the TF treatment (Supplemental Table S5). Overall, the reduction of total INLs showed that the TF treatment was the only treatment to significantly reduce INL for both VAs.

3.5 | Dissolved phase INLs

Vegetation type and buffer width significantly affected dissolved phase INLs for both VAs ($p = .5233$ for SMZ, $p = .0345$ for LIN, Figure 3), but because both VAs were primarily transported in the dissolved phase the dissolved phase INL results strongly correlated to the total INL results (Figure 2). Averaged over buffer widths, only the TF treatment significantly reduced dissolved phase INLs compared with the control for both VAs (Figure 3). Within vegetation treatment differences showed that none of the grass treatments significantly reduced dissolved SMZ INLs compared with the control at any buffer width (Supplemental Table S6). For LIN, all three treatments showed significant reduction in dissolved INLs at 9 m compared with the control (between 47–61% load reduction), but no treatment showed significant load reductions ($\leq 42\%$) at shorter widths (Supplemental Table S6). Similar to the total INLs, buffer width effects on dissolved phase SMZ INLs showed a large reduction in load from 0 to 2 m and no or small reductions at 5 and 9 m, while LIN showed an increasingly significant reduction in load at each buffer width (Figure 3). Buffer width effects on dissolved phase INLs within individual treatments reflected these same patterns for both VAs (Supplemental Table S6).

3.6 | Sediment-bound INLs

Buffer width was the only factor that significantly affected sediment-bound INLs for both SMZ and LIN; there was no vegetation effect on sediment-bound INLs for either VA ($p = .6417$ for SMZ, $p = .9845$ for LIN, Figure 4). For both SMZ and LIN, within vegetation treatment differences showed that none of the grass treatments significantly reduced sediment-bound INLs compared with the control at any buffer width (Supplemental Table S7). These results were similar to the dissolved phase INLs for SMZ but contrasted with the dissolved phase INLs for LIN and with the total

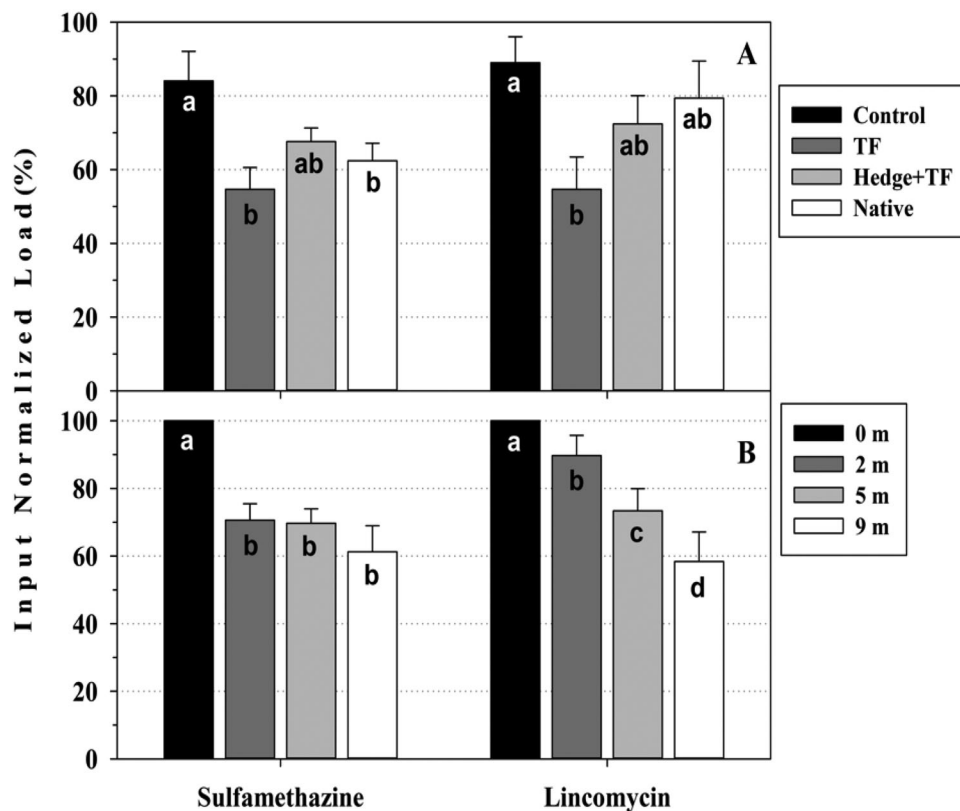


FIGURE 2 Least square means of total input normalized loads (dissolved + sediment-bound) of sulfamethazine and lincomycin by (A) vegetation type and (B) buffer width. Bars with different letters were significantly different at $\alpha = .10$. Vegetation \times width interaction was significant for lincomycin ($p = .035$). Error bars represent the 95% confidence intervals. TF, tall fescue

(dissolved + sediment-bound) INLs for both SMZ and LIN. Sediment-bound INLs by buffer width showed an increasingly significant reduction in load at each buffer width for both SMZ and LIN (Figure 4). Within the individual treatments, all three grass treatments showed significant reductions in sediment-bound SMZ INLs at 2 m (between 44 and 60%), 5 m (between 57 and 90%), and 9 m (between 90 and 98.6%) compared with 0 m (Supplemental Table S7). Only the Hedge+TF treatment showed significant reduction in sediment-bound INLs for LIN among the >0 -m buffer width (reduction up to 96.4%), with the 9-m width being significantly lower than the 2-m width (Supplemental Table S7).

3.7 | INLs for sediment loss

Both vegetation type and buffer width significantly affected INLs for sediment transport in runoff in all treatments ($p = .0041$). Compared with the control, each grass treatment significantly reduced sediment INL, with reductions of $>60\%$ (Figure 5). Within vegetation treatments, reductions in sediment INL for all grass treatments were significant compared with the control at 5 and 9 m, but not at 2 m (Supplemental Table S8). At 5 and 9 m, reductions in sediment INLs averaged 71.3 and 86.3%, respectively (Supplemental Table

S8). The effect of buffer width on sediment INLs was consistent across the three grass treatments, with significant and exponential reductions in load with increasing buffer width (Figure 5). By contrast, sediment INLs for the control were significantly lower at 2 m (72%), then returned to $\sim 100\%$ at 5 and 9 m (Figure 5). This finding indicated that for the control treatment significant, sediment deposition of eroded soil from the source area (i.e., the upper 7 m of the plot where manure was applied) occurred between 0 and 2 m, and the observed sediment increases at 5 and 9 m were derived from eroded sediment below the 2-m sampler, within the lower portion of the plots. Additionally, significant deposition of sediment-bound VAs also occurred from 0 to 2 m in the control treatment, and the observed decreases in sediment-bound INLs at 5 and 9 m resulted from a dilution of the sediment-bound VAs with the uncontaminated sediment in the lower portions of the plot. While this pattern of sediment deposition and suspension may take place in the grass buffer treatments, these treatments also significantly reduced sediment INLs compared with the control, yet they did not significantly reduce sediment-bound VA INLs compared with the control. These results highlight how sediment deposition and suspension dynamics controlled sediment-bound VA transport in runoff. The importance of buffer width on reducing sediment loads has been previously reported (Blanco-Canqui et al., 2004a, 2004b; Lerch et al.,

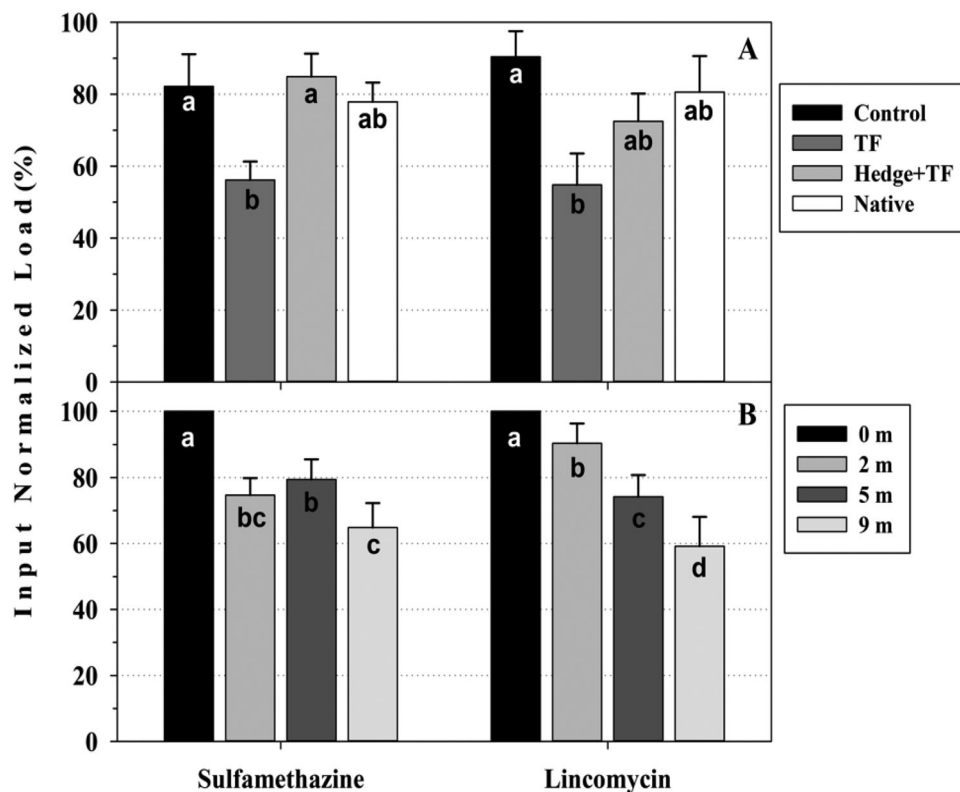


FIGURE 3 Least square means of input normalized dissolved phase loads of sulfamethazine and lincomycin by (A) vegetation type and (B) buffer width. Bars with different letters were significantly different at $\alpha = .10$. Vegetation \times width interaction was significant ($p = .035$) for lincomycin. Error bars represent the 95% confidence intervals. TF, tall fescue

2017; Liu et al., 2008), and the results presented here are consistent with these studies. However, the results for the sediment-bound VA loads did not correspond with those of the sediment loads (Figures 4 and 5).

3.8 | Predicting input normalized VA loads and expected field-scale reductions

Regression equations to predict total INLs as a function of buffer width resulted in significant models for all grass treatments, but no models were significant for the control (Figure 6). The linear models shown for the control had the highest correlation coefficients of any model examined, but the slopes were not significantly different from zero. For SMZ, the TF and Native treatments were best fit with a three-parameter exponential model, while the Hedge+TF treatment was best fit with a quadratic model. For LIN, the TF treatment was also best fit with the three-parameter exponential model, while the Hedge+TF and Native treatments were best fit with linear models.

The linear model fits for reducing LIN INLs in two of the grass treatments implies that grass buffers were generally less effective at reducing LIN loads compared with SMZ loads. A linear model implies lower load reductions per meter of buffer

when compared to exponential models, resulting in greater vegetation buffer width, or lower SBARs to achieve LIN load reductions. For both VAs, the TF treatment resulted in significant exponential models for total INLs as a function of buffer width. These results indicated that for each incremental increase in buffer width, estimated VA load reductions would be greater for TF compared with the other grass treatments.

For expected field-scale reductions, three different SBAR values were applied: 10:1 equivalent to 0.7-m-wide buffer at plot-scale; 20:1 (0.35-m-wide buffer); and 50:1 (0.14-m-wide buffer). Applying these SBARs to the equivalent plot-scale buffer width, the regression equations provided load reductions estimates for SMZ and LIN. For LIN, only the TF treatment was estimated to provide meaningful load reductions at field scale with a 12.6% reduction at 10:1 SBAR and 7.2% reduction at 20:1 SBAR. For SMZ, each grass treatment at 10:1 and 20:1 SBARs achieved load reductions in the range of 7.5 to 16.3% (for 10:1 and 20:1 SBARs, respectively—12.4 and 7.5% for TF; 12.2 and 8.1% for Hedge + TF; 16.3 and 9.2% for Native). Overall, these grass treatments are expected to provide only modest reductions in VA loads at expected field-scale implementation, and the grass treatments studied here were more effective for reducing loads of SMZ in runoff than LIN.

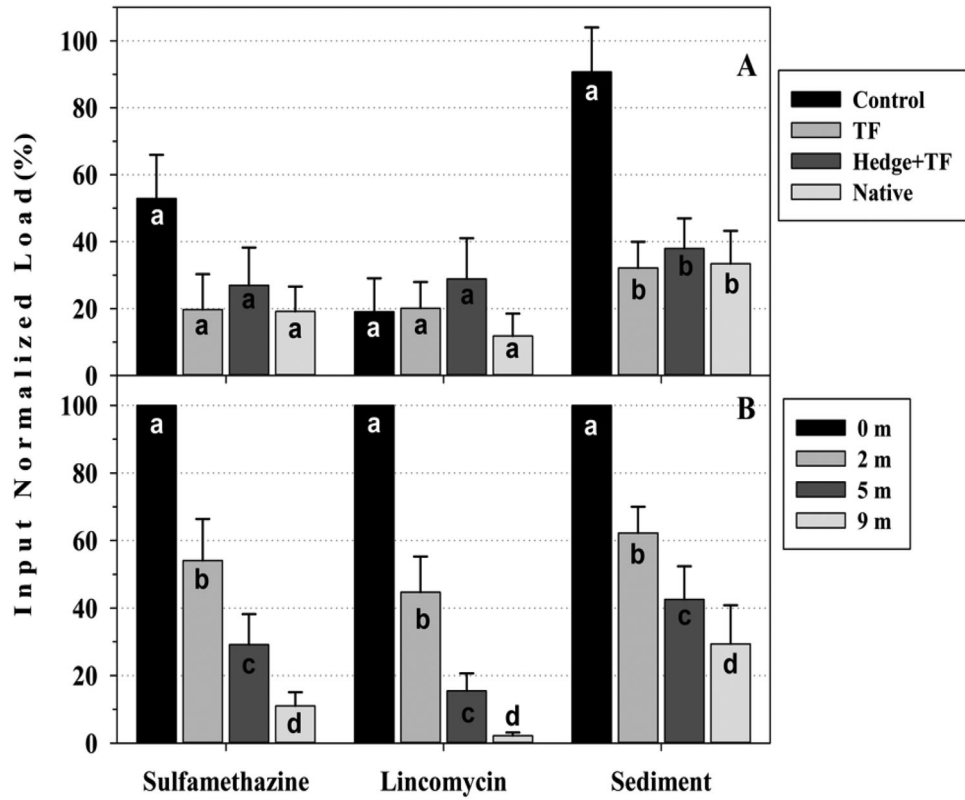
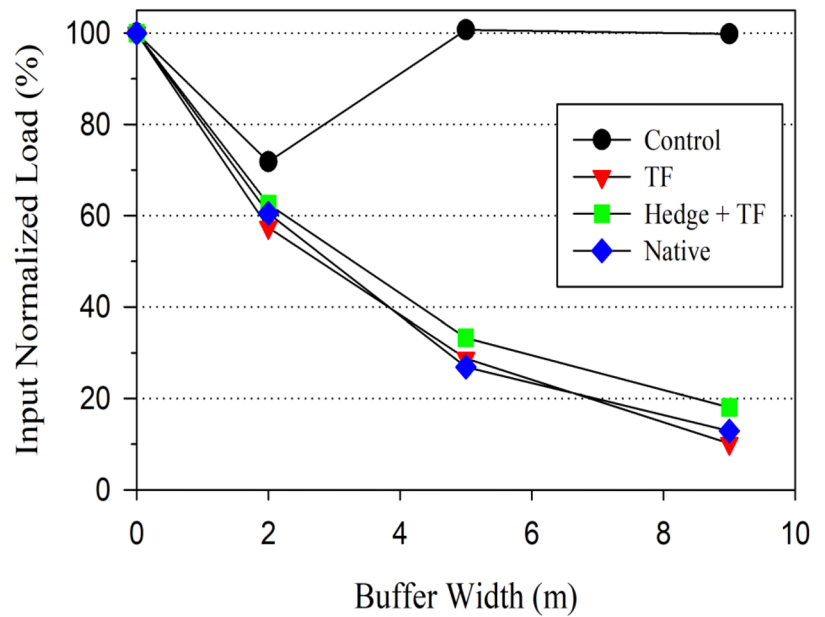


FIGURE 4 Least square means of input normalized sediment-bound loads of sulfamethazine and lincomycin by (A) vegetation type and (B) buffer width. Bars with different letters were significantly different at $\alpha = .10$. Vegetation \times buffer width interaction was significant for sediment ($p = .004$). Error bars represent the 95% confidence intervals. TF, tall fescue

FIGURE 5 Input normalized sediment loads of each vegetation type \times buffer width. TF, tall fescue



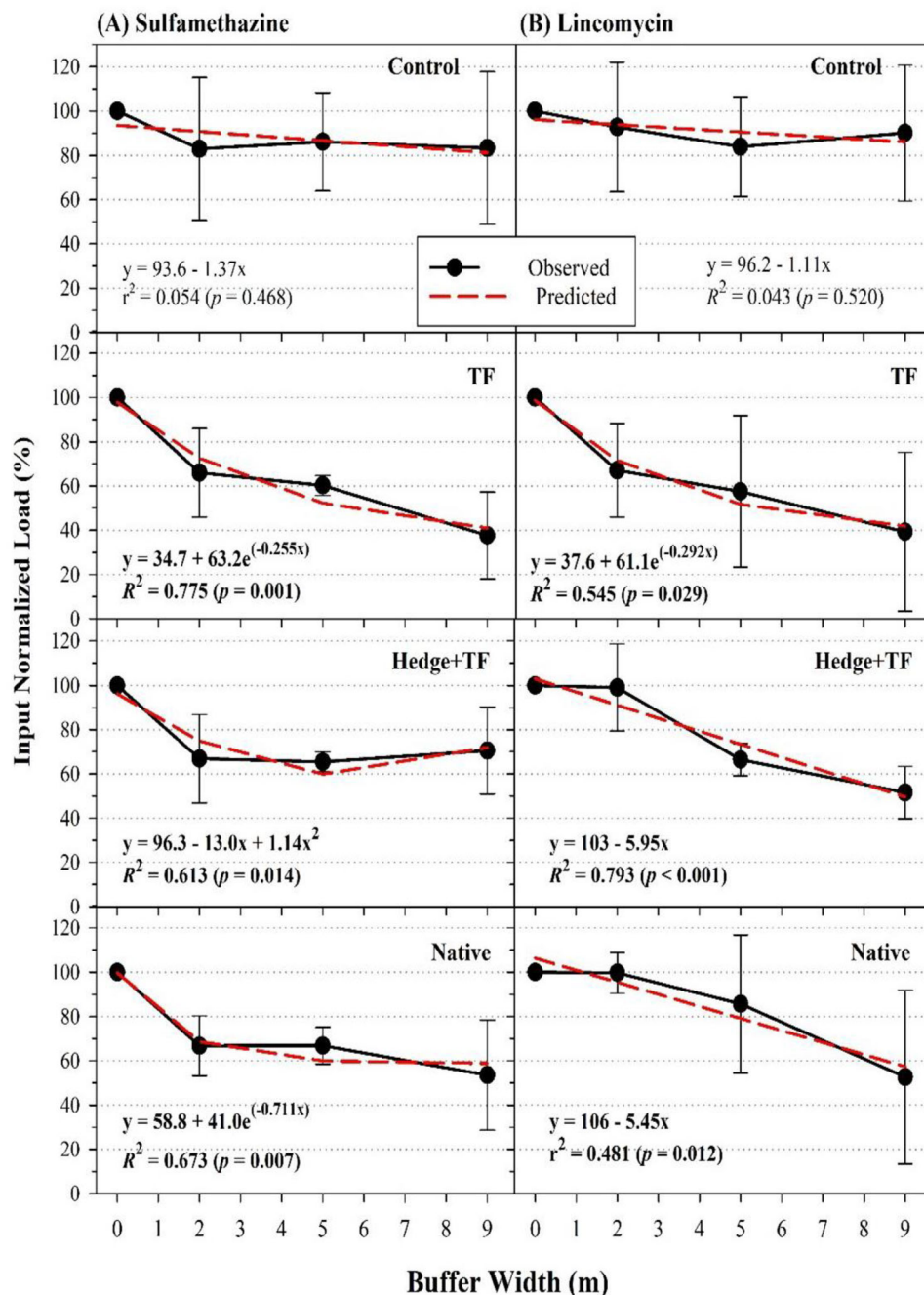


FIGURE 6 Regression equations correlating relative load reduction as a function of buffer width for (A) sulfamethazine and (B) lincomycin. Error bars represent 95% confidence interval for observed data. Model equations in bold were significant at $\alpha = .05$

4 | CONCLUSIONS

Under the conditions studied, grass VBS had limited ability to reduce SMZ and LIN loads in runoff. Although there were statistically significant load reductions, particularly for the TF treatment, expected field-scale load reductions were estimated to be $\leq 16\%$ for all grass treatments. Key factors affecting the results included the high proportion of dissolved phase transport for both VAs and the lack of increased runoff

infiltration in the grass treatments compared with the control. Further, grass VBS did not significantly reduce sediment-bound VA loads compared with the control despite achieving significant reductions in sediment loads. Partitioning of VA loads in runoff was apparently affected by the presence of DOC, especially for LIN, which was dominantly transported in the dissolved phase despite reports of much greater soil sorption than SMZ. Compared with reported reductions for sediment, nutrients, and herbicides on these same plots, these grass treatments were less effective for reducing SMZ and

LIN loads. This emphasizes the importance of developing other management strategies to mitigate VA contamination of surface waters, such as manure composting prior to land application, subsurface manure application techniques, the development of novel VAs with limited persistence in soils, and efforts to reduce the large-scale use of VAs in livestock animal production.

ACKNOWLEDGMENTS

Funding for this research was provided by the USDA-ARS. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

AUTHOR CONTRIBUTIONS

Adam H. Moody: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Resources; Software; Validation; Visualization; Writing – original draft; Writing – review & editing. Robert N. Lerch: Conceptualization; Funding acquisition; Methodology; Resources; Software; Supervision; Validation; Writing – review & editing. Keith W. Goynes: Validation; Writing – review & editing. Stephen H. Anderson: Validation; Writing – review & editing. David G. Mendoza-Cózatl: Validation; Writing – review & editing. David A. Alvarez: Funding acquisition; Project administration; Resources; Validation; Writing – review & editing.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

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How to cite this article: Moody, A. H., Lerch, R. N., Goyne, K. W., Anderson, S. H., Mendoza-Cózatl, D. G., & Alvarez, D. A. (2023). Vegetative buffer strips show limited effectiveness for reducing antibiotic transport in surface runoff. *Journal of Environmental Quality*, 52, 137–148. <https://doi.org/10.1002/jeq2.20441>