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VEGETATED AGRICULTURAL DRAINAGE DITCHES FOR THE MITIGATION OF PYRETHROID-ASSOCIATED RUNOFF

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Abstract—Drainage ditches are indispensable components of the agricultural production landscape. A benefit of these ditches is contaminant mitigation of agricultural storm runoff. This study determined bifenthrin and lambda-cyhalothrin (two pyrethroid insecticides) partitioning and retention in ditch water, sediment, and plant material as well as estimated necessary ditch length required for effective mitigation. A controlled-release runoff simulation was conducted on a 650-m vegetated drainage ditch in the Mississippi Delta, USA. Bifenthrin and lambda-cyhalothrin were released into the ditch in a water–sediment slurry. Samples of water, sediment, and plants were collected and analyzed for pyrethroid concentrations. Three hours following runoff initiation, inlet bifenthrin and lambda-cyhalothrin water concentrations ranged from 666 and 374 μg/L, respectively, to 7.24 and 5.23 μg/L at 200 m downstream. No chemical residues were detected at the 400-m sampling site. A similar trend was observed throughout the first 7 d of the study where water concentrations were elevated at the front end of the ditch (0–25 m) and greatly reduced by the 400-m sampling site. Regression formulas predicted that bifenthrin and lambda-cyhalothrin concentrations in ditch water were reduced to 0.1% of the initial value within 280 m. Mass balance calculations determined that ditch plants were the major sink and/or sorption site responsible for the rapid aqueous pyrethroid dissipation. By incorporating vegetated drainage ditches into a watershed management program, agriculture can continue to decrease potential non–point source threats to downstream aquatic receiving systems. Overall results of this study illustrate that aquatic macrophytes play an important role in the retention and distribution of pyrethroids in vegetated agricultural drainage ditches.

Keywords—Drainage ditches Bifenthrin Lambda-cyhalothrin Aquatic plants Mitigation

INTRODUCTION

Available literature on surface drainage ditch research is limited [1,2]. Other than Moore et al. [3] and Cooper et al. [4], few studies in the United States have focused on the capacity of drainage ditches to decrease the concentration of pesticides entering receiving aquatic ecosystems. More research has been conducted in the Netherlands, where drainage ditches serve as important habitat and potential transports for drinking water [5–8]. Many of these studies focused on ditch maintenance and management practices since the use of ditches is more restrictive in the Netherlands than in the United States. Historically, the value and function of agricultural ditches have been ignored except for their general upkeep, which may include periodic dredging to remove built-up sediment and plants impeding efficient drainage. It has been proposed that these marginal lands colonized with aquatic plants could be used as a simple and inexpensive method for the mitigation of agricultural runoff. Natural or constructed vegetated wetlands are sometimes not an option in agricultural row-crop farming mainly because of space limitations; thus, vegetated drainage ditches are a simple alternative for this currently suggested best management practice.

Recently, research by Moore et al. [3] has shown that vegetation in agricultural ditches aids in the trapping of many commonly used pesticides. Other studies on the role of aquatic plants as a mitigative tool for pesticide reduction and retention have been limited to wetland and mesocosm studies using both spray-drift [9–12] and runoff [13–15] scenarios. Schulz et al. [13] recently showed that the aquatic macrophyte *Typha capensis* facilitated the reduction of the loading of an organophosphate insecticide, azinphos-methyl, into a nearby waterway following a spray-drift event in a South African wetland.

In the Mississippi Delta region, USA, a wide range of pesticides are currently used in production farm acreage. Of these, synthetic pyrethroids are one of the main classes of insecticides used, especially in cotton and corn production. For example, bifenthrin [(2-methyl-2-methylbiphenyl-3-ylmethyl (Z)-(1RS,3RS)-3-(2-chloro-3,3,3-trifluoroprop-1-enyl)-2,2-dimethylcyclo-propanecarboxylate] is a fourth-generation pyrethroid insecticide. Sold under such trade names as Capture® and Brigade® (FMC, Philadelphia, PA, USA), approximately 52,000 kg active ingredient bifenthrin were applied to U.S. corn (94%), cotton (3%), and blackberry (3%) crops in 2001 (National Agricultural Statistics Service [NASS], 2003, http://www.nass.usda.gov). Lambda-cyhalothrin, a 1:1 mixture of (S)-α-cyano-3-phenoxybenzyl-(Z)-(1R,3R)-3- (2-chloro-3,3,3-trifluoroprop-1-enyl)–2,2-dimethylcyclo propane carboxylate and (R)-α-cyano-3-phenoxybenzyl-(Z)-(1S,3S)–3-(2-chloro-3,3,3-trifluoroprop-1-enyl)-2,2-dimethylcyclopropanecarboxylate [(RS)-α-cyano-3-phenoxybenzyl3-(2-chloro-3,3,3-trifluorpropenyl)-2,2 dimethylcyclopropanecarboxylate], another fourth-generation pyrethroid, is commonly sold as Karate® (Syngenta, Greensboro, NC, USA). Over 45,000 kg of lambda-

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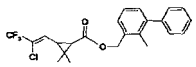
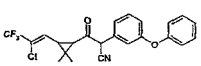
cyhalothrin (as active ingredient) were applied to U.S. cotton (54%), corn (41%), and soybean (5%) crops in 2001 (NASS, 2003, http://www.nass.usda.gov). The basic mode of action for synthetic pyrethroids is the disruption of the central and peripheral nervous system of insects, which causes paralysis [16]. Pyrethroids have been shown to elicit toxic effects at extremely low concentrations in nontarget aquatic organisms, such as mayflies (*Heptageniidae*) and damselflies (*Enellagma* and *Ishnura* spp.) [17]. As a result of these toxic effects, concern has increased about such compounds reaching aquatic environments, especially during agricultural runoff and spray-drift events.

The purpose of this present study was twofold. The first objective was to evaluate the retention and partitioning (water, plant, sediment) of two currently used pyrethroids, bifenthrin and lambda-cyhalothrin, within a vegetated agricultural drainage ditch located in the Mississippi Delta during a simulated, worst-case-scenario runoff event. From these data, the relative importance of aquatic vegetation in facilitating the removal of insecticide from water was evaluated using mass balance calculations and insecticide physicochemical properties. The second objective of this study was to estimate drainage ditch lengths for effective mitigation of bifenthrin and lambda-cyhalothrin using pesticide distribution, given recommended field application rates and other rainfall and runoff variable assumptions.

Table 1. Physical and chemical properties of bifenthrin and lambda-cyhalothrin (adapted from Laskowski [25]); NA = not available; atm = atmospheres

	Bifenthrin	Lambda-cyhalothrin
Structure		
Molecular weight (g/mol)	422.9	449.9
Vapor pressure (mm Hg)	1.8×10^{-7}	1.7×10^{-9}
Water solubility (mg/L)	1.4×10^{-5}	5.0×10^{-3}
Henry's law constant (atm m^3/mol)	7.2×10^{-3}	7.9×10^{-7}
Log K_{ow}	6.4	7.0
Log K_{oc}	5.4	5.5
Hydrolysis half-life (d)		
pH 5	Stable	Stable
pH 7	Stable	Stable
pH 9	Stable	8.7
Photolysis half-life (d)		
Water	408	24.5
Soil	96.9	53.7
Soil half-life (d)		
Aerobic	96.3	42.6
Anaerobic	425	NA

MATERIALS AND METHODS

Ditch exposure

A 650-m segment of an agricultural drainage ditch located within the Mississippi Delta Management Systems Evaluation Area near Indianola, Mississippi, was used for this study. The ditch was approximately 4.5 m wide (top width) and 0.35 m deep (water depth) and had a water width of approximately 2.8 m. Sampling sites were established within the ditch at the simulated runoff inlet 10 m upstream and 25, 50, 75, 100, 200, 400, and 650 m downstream. One week before the simulated storm runoff event, ditch vegetative cover was determined to be approximately 88% by sampling multiple 0.69 m^2 quadrats at each sampling site, and the dominant plant species were determined to be *Ludwiga* followed by *Lemna* and *Polygonum*. In addition to this survey, basic water quality and centerline velocity measurements were made using a handheld YSI-85 m (Yellow Springs Instruments, Yellow Springs, OH, USA) and a Marsh-McBirney electromagnetic current meter (Marsh McBirney, Frederick, MD, USA), respectively. Mean ditch measurements were as follows: velocity, 0.03 ± 0.02 m/s; dissolved oxygen, 3.3 ± 0.09 mg/L; temperature, 30.1 ± 0.4°C; conductivity, 559 ± 5 μmhos; pH, 7.4 ± 0.2.

A storm runoff event was simulated in the drainage ditch in late July 1999. A mixture of bifenthrin (Capture) and lambda-cyhalothrin (Karate) was amended directly into the ditch. Physical and chemical properties of these compounds are presented in Table 1. Pesticide concentrations in simulated runoff (0.89 mg/L bifenthrin and 0.44 mg/L lambda-cyhalothrin) were based on recommended application rates (0.011 and 0.0057 kg/ha, respectively) and an assumed 1% runoff from a 0.64-cm storm event across a 20-ha contributing area [18]. Simulated runoff was pumped through a 2.0-m length of 7.6-cm-diameter polyvinyl chloride pipe with 16 holes (each 1.5 cm) evenly dispersed along the pipe's length for even diffusion. Four water tanks (each 3,800 L) filled with groundwater were connected (one at a time) to the diffuser and used as a water source for the simulated event. The mixture of bifenthrin and lambda-cyhalothrin was added to water in a 110-L mixing chamber and then delivered to the top of the PVC diffuser via Tygon™ (Saint Gobain Performance Plastics Corporation, Valley Forge, PA, USA) tubing using an Attwood™ V450 submersible pump (Attwood Corporation, Lowell, MI, USA) at a rate of 0.02 L/s for 90 min. A 5-cm hose delivered water from the 3,800-L water tanks directly to the diffuser at a rate of approximately 1 L/s for about 250 min.

Collection of water, sediment, and plant samples

Grab samples of water ($n = 1$) were collected in 1-L amber glass bottles one week before the application, at 0 h, at 15-min intervals for 2 h, and at 3, 6, 12, and 24 h and 7, 14, 30, 44, and 99 d postapplication from each site. After samples were collected, they were stored on ice and returned to the laboratory for extraction (within 24 h). Sediment and plant samples were collected one week before application and at 0, 3, 12, and 24 h and 7, 14, 30, 44, and 99 d postapplication from each site, wrapped in solvent washed foil, stored on ice, and returned to the laboratory to be dried. It should be noted that sediment samples were obtained from the top 1 cm using solvent-rinsed stainless-steel spatulas, while plant materials were collected with solvent-rinsed scissors. Only that plant material exposed in the water column (between sediment and water surface) was collected for analysis.

Extraction and analysis of water, sediment, and plant samples

Water, sediment, and plant samples were extracted using previously described methods by Bennett et al. [19] and Moore et al. [3]. Briefly, water samples were extracted by liquid–liquid extraction using ethyl acetate, while both sediment and plant samples were extracted by ultrasonication using ethyl acetate. Sediment and plant extracts were also subject to silica gel cleanup before analysis. Water samples were partitioned

Table 2. Downstream dissipation of bifenthrin and lambda-cyhalothrin concentrations (μg/L) in water at the 3-h, 12-h, and 7-d sampling times (ND = not detected, below detection limits; 1.00 ng/L). E-02 represents 10^{-2} for significant digit purposes

	Bifenthrin			Lambda-cyhalothrin		
Distance (m)	3 h	12 h	7 d	3 h	12 h	7 d
0	666	10.7	0.887	375	5.29	0.250
25	235	25.9	7.76	115	11.8	2.32
50	77.2	6.33	0.178	39.1	3.44	5.50E-02
75	33.8	1.03	0.270	20.6	0.745	7.38E-02
100	27.8	1.32	6.38E-02	16.6	0.899	2.38E-02
200	0.724	0.454	5.07E-02	0.309	0.296	2.00E-02
400	ND	0.471	ND	0.144	9.88E-02	ND
650	ND	ND	ND	ND	ND	ND

with ethyl acetate in the field and transported in coolers back to the laboratory for final extraction.

Pyrethroids were analyzed by gas chromatography/microelectron capture detection using a Hewlett Packard 6890 gas chromatograph (Hewlett Packard, New Haven, CT, USA) equipped with a 30-m HP-1MS column. The following oven temperature program was used: 75°C (held for 1 min) to 225°C at a rate of 40°C/min. The injector and detector temperatures were set to 250 and 325°C, respectively. The carrier gas, ultra-high-purity helium (nexAir, Memphis, TN, USA), was set to a constant flow of 1 ml/min, and makeup gas, ultra-high-purity nitrogen (nexAir), was set at a constant makeup flow of 60.0 ml/min. A multilevel calibration procedure was used with bifenthrin and lambda-cyhalothrin standards (AccuStandard, New Haven, CT, USA) and was updated every ninth sample. Procedural blanks and solvent blanks were analyzed with each batch of standards. Limits of detection for bifenthrin and lambda-cyhalothrin in water, sediment, and plants were 1 to 10 ng/L (ppt). Mean extraction efficiencies, based on fortified samples, were >90% for water, sediment, and plants.

Data analysis

Ordinary least-squares linear regression analyses [20] were used to fit curves to log-transformed bifenthrin and lambda-cyhalothrin water concentrations (y) versus the log of the distance down ditch from the inlet (x). For simplicity, only the maximum concentrations observed at each sampling site during the first 3 h following injection were used in the analyses. Concentrations at the most distant sampling site (x = 650 m) were below detection limits, so regressions were run either without this data point or with the assumption that the concentration at this site equaled the detection limit. The latter set of regressions produced results similar to those without the x = 650 m data, so they will not be presented here. Concentrations at the injection point were also omitted from the regression in order to improve curve fits at greater distances.

Mass balances were performed using data on water, plant, and sediments collected along transects of the 650-m ditch length for each sample time point (3, 12, and 24 h and 7, 24, 30, and 44 d). This enabled quantitative evaluation of chemical partitioning and losses that occurred over the study duration. The mass balance at a given time point was determined as

$$m_{\text{total}(t)} = m_{\text{w}(0\text{–}650)} + m_{\text{p}(0\text{–}650)} + m_{\text{s}(0\text{–}650)} \quad (1)$$

where $m_{\text{w}(0\text{–}650)}$, $m_{\text{p}(0\text{–}650)}$, and $m_{\text{s}(0\text{–}650)}$ reflect the total chemical mass (g) in water, plants, and sediments over the 650-m ditch length. Integration of chemical masses in water was performed according to the trapezoidal rule:

$$m_{\text{w}(0\text{–}650)} = \sum_{i=0}^{n-1} \left\{ [V_{\text{w}(i+1)} - V_{\text{w}(i)}] \cdot \left[\frac{C_{\text{w}(i+1)} + C_{\text{w}(i)}}{2} \right] \right\} \quad (2)$$

where the term ($V_{\text{w}(i+1)} - V_{\text{w}(i)}$) represents the volume of water (L) bounded by transects i and i + 1 (i.e., 0–25, 25–50, 200–400 m, and so on) and $C_{\text{w}(i)}$ is the water concentration measured at transect i. Water volumes between transects were estimated as the product of mean water depth, mean water width, and the distance between transects less the water displacement by plants (assuming that an estimated 20% of water was displaced by plant biomass in each transect). The mass calculations for plants and sediments were similar to Equation 2, except that concentrations were measured in units of mg/kg and the volume terms were replaced by bulk sediment mass (kg) or plant biomass (kg). In the former case, the ditch bed surface area (m^2) within a given interval was multiplied by a sediment depth of 0.01 m and converted to sediment mass by assuming a bulk sediment density of 1,200 kg/m^3. Plant biomass was calculated at 0-, 25-, 50-, 75-, 100-, 200-, 400-, and 600-m transects and ranged from 0 to 4.22 kg/m^2 (dry wt). The average area plant biomass within a transect interval was multiplied by the ditch width and interval distance to arrive at a plant biomass (kg) estimate.

A ditch chemical depuration rate constant (k_2), representative of all measured compartments, was determined by plotting ln m_{total} as a function of time and determining the slope using linear regression analysis. The ditch chemical depuration rate constant represents the summation of individual clearance rate constants occurring within each phase and includes losses to advection (e.g., outflow from the ditch via water), volatilization, and abiotic/biotic reactions. Chemical half-lives ($t_{½}$) were subsequently estimated as ln(2)/k_2. Chemical depuration rate constants and half-lives were also derived using mass balances for individual media (water, plants, and sediments).

RESULTS

Concentrations in water

Both bifenthrin and lambda-cyhalothrin concentrations measured in water decreased rapidly with distance (Table 2). For example, 3 h following the initiation of the storm runoff simulation, bifenthrin water concentrations ranged from 666 μg/L at the inlet to 7.24 μg/L at 200 m downstream, while concentrations were below detection limits at the 400-m sampling site. A similar trend was observed throughout the first 14 d of the study where water concentrations were elevated at the front end of the ditch (0–25 m) and greatly reduced by the 400-m sampling site. By day 30, the distribution of bifenthrin water concentrations showed slightly greater dispersion,

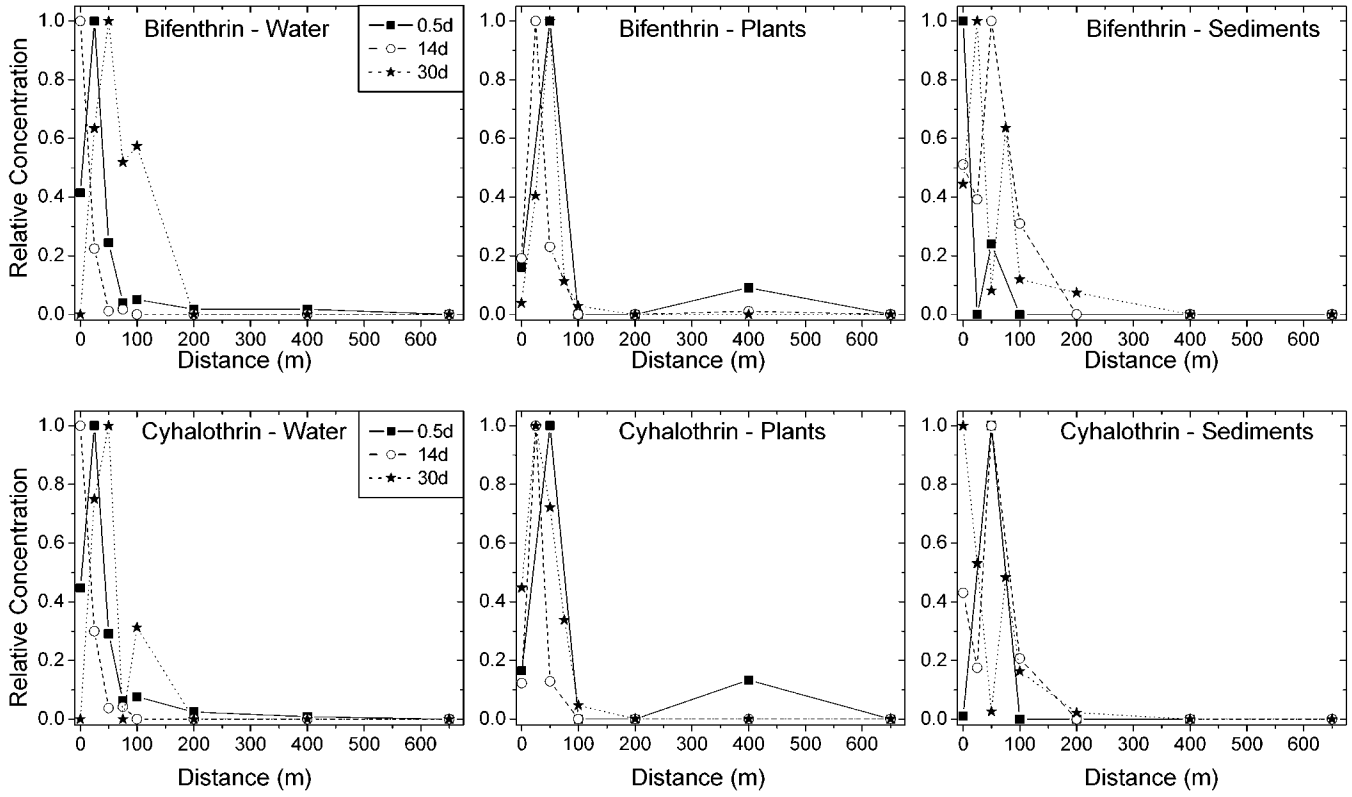


Fig. 1. Comparison of relative bifenthrin and lambda-cyhalothrin concentrations measured at 0.5, 14, and 30 d for each compartment (water, plants, sediment) to sampling sites downstream of the injection point.

exhibiting a peak concentration at 50 m, elevated levels at 100 m, and nondetectable levels after 200 m (Fig. 1).

This pattern was also observed for lambda-cyhalothrin (Table 2 and Fig. 1). During and after day 14 of sampling, bifenthrin and lambda-cyhalothrin water concentrations were approximately 0.05 and 0.02 μg/L, respectively, when detected. By day 99, all water samples collected were below detection limits. When these data are expressed on a percentage basis, pesticide concentrations at the 200-m sampling site were >99% lower than concentrations detected at the 25-m sampling throughout the study.

Concentrations in/on plants

Ditch plant samples collected one week before experiment initiation indicated bifenthrin and lambda-cyhalothrin concentrations that were below detection. The highest concentrations for both bifenthrin and lambda-cyhalothrin in/on plants were detected in the ditch within 50 m of the inlet throughout the study (Fig. 1). Maximum measured concentrations for bifenthrin and lambda-cyhalothrin were 10.8 (dry wt) and 8.79 mg/kg (dry wt), respectively. By 12 h after runoff initiation, both pesticides were detected at all the sampling sites between the inlet and the 400-m sampling site. Unlike water, bifenthrin and lambda-cyhalothrin showed little evidence of dispersion from the initial distribution along the ditch length over time (Fig. 1).

Pesticide concentrations associated with plants stabilized after 12 h throughout the ditch for approximately 30 d postapplication. Bifenthrin concentrations were between 7.26 and 10.8 mg/kg at the 50-m sampling site within the first day of the study with concentrations stabilizing (~3 mg/kg) after 7 d until they dropped considerably after 30 d (Fig. 2). A similar trend was observed for lambda-cyhalothrin, with the exception that no considerable decline in concentration was observed after 30 d.

Concentrations in sediment

Background sediment samples collected one week before experiment initiation were below detection limits for concentrations of the two insecticides. Bifenthrin and lambda-cyhalothrin in sediment reached maximum concentrations of 0.0917 and 0.0538 mg/kg, respectively, within the first 25 m of the

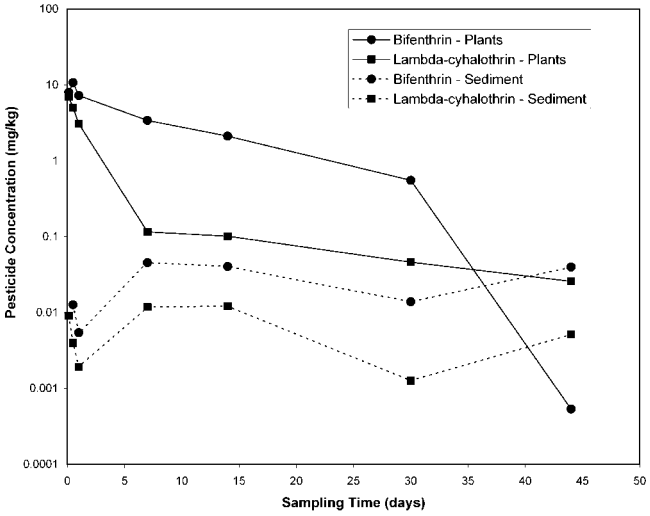


Fig. 2. Measured bifenthrin and lambda-cyhalothrin concentrations (mg/kg) in plants and sediments collected from the 50-m sampling site at the 3-, 12-, and 24-h and 7-, 14-, 30-, and 44-d sampling periods.

Table 3. Summary of linear regression formulas produced using maximum pesticide concentrations versus distance downstream from inlet. Data were log transformed. Relations are power functions of the form $y = ax^b$, where y is the maximum concentration in mg/L (water) observed during the first 3 h following injection and x is the distance in meters. Alternatively, $y' = a'x^b$, where y' is concentration in mg/L (water) divided by the maximum observed concentration at the inlet. Data for the injection point ($x = 0$) were not used in the regression in order to improve curve fits for larger x values (Fig. 3)

Compound	No. of observations	a	a'	b	r^2	p
Bifenthrin	6	20,460	30,770	−3.19	0.937	0.0015
Lambda-cyhalothrin	6	1,537	4,099	−2.69	0.902	0.0037

ditch system. Generally, sediment pesticide concentrations were two orders of magnitude lower at sites beyond the 50-m sampling site. However, after 14 d, both bifenthrin and lambda-cyhalothrin showed some evidence of downstream dispersion, with low to moderate concentrations being measured at 100 and 200 m (bifenthrin only; Fig. 1).

Overall, bifenthrin and lambda-cyhalothrin sediment concentrations stabilized in the ditch system after 24 h. Bifenthrin sediment concentrations at the 50-m sampling site stabilized to approximately 0.0400 mg/kg 24 h postapplication, and this concentration was maintained for the remainder of the study (Fig. 2). A similar trend was also observed for lambda-cyhalothrin.

Ditch length predictions

Maximum observed pesticide concentrations (determined during 0–3 h) were inversely proportional to distance downstream from the inlet, producing regression coefficients that were significant at $p < 0.004$ (Table 3 and Fig. 3). The fit standard errors were about ±0.01, but 95% confidence intervals were rather broad because of the relatively small number of data points. The formulas predict that bifenthrin and lambda-cyhalothrin concentrations in ditch water were reduced to 1% of the initial value within the first 120 m of ditch downstream from the inlet and fall to only 0.1% of the initial value within 280 m.

Mass balance and half-lives

The total mass of bifenthrin and lambda-cyhalothrin at 3 h ($m_{total(3h)}$) was consistent with the nominal mass of chemical added to the system (Table 3). The $m_{total(3h)}$ for bifenthrin was 12.1 g, or 106% of the nominal mass added to the system, while the $m_{total(3h)}$ for lambda-cyhalothrin was 163% of the nominal mass. Mass balance calculations for different sampling times showed that changes occurred in phase distribution occurring over the study duration. At the 3-h sampling time point, the majority of mass for both bifenthrin and lambda-cyhalothrin was concentrated in the water and plant compartments (Table 4). By 12 h into the study, almost all the mass of both pesticides had shifted to the plant compartment. This trend continued throughout the study. For both water and plant compartments, a steady reduction of pesticide mass was observed, while enrichment and/or retention of lambda-cyhalothrin in the sediment compartment occurred over time. However, the total mass in sediment was relatively low.

Combined changes in $m_{w(0–650)}$, $m_{s(0–650)}$, and $m_{p(0–150)}$ with time were used to estimate ditch chemical half-lives. Bifenthrin and lambda-cyhalothrin half-lives were calculated to be 6.12 and 1.35 d, respectively. Bifenthrin data used in this calculation followed first-order kinetics, while initial lambda-cyhalothrin half-life calculations were based on assumed first-order kinetics. In fact, lambda-cyhalothrin data followed a biphasic pattern of elimination that was occurring with an initial rapid drop in mass during the first 7 d followed by a much slower elimination rate over the remaining 36 d (Table 4). Alternately, half-lives for lambda-cyhalothrin were calculated to be 1.24 d within the 7-d sampling period and >100 d after the 7-d sampling period.

DISCUSSION

This study was designed to determine the effectiveness of a vegetated drainage ditch on the retention and partitioning of pyrethroids during a simulated runoff event. Results showed that vegetated drainage ditches are effective tools for lowering/removing pyrethroid-associated runoff from the ditch water column, consequently reducing pesticide loadings into receiving water bodies. In this study, 33.9 and 36.3% of the original dose of bifenthrin and lambda-cyhalothrin, respectively, remained in the water 3 h postapplication and were further reduced to 3.09 and 1.24% after 1 d. Similar results by Leistra

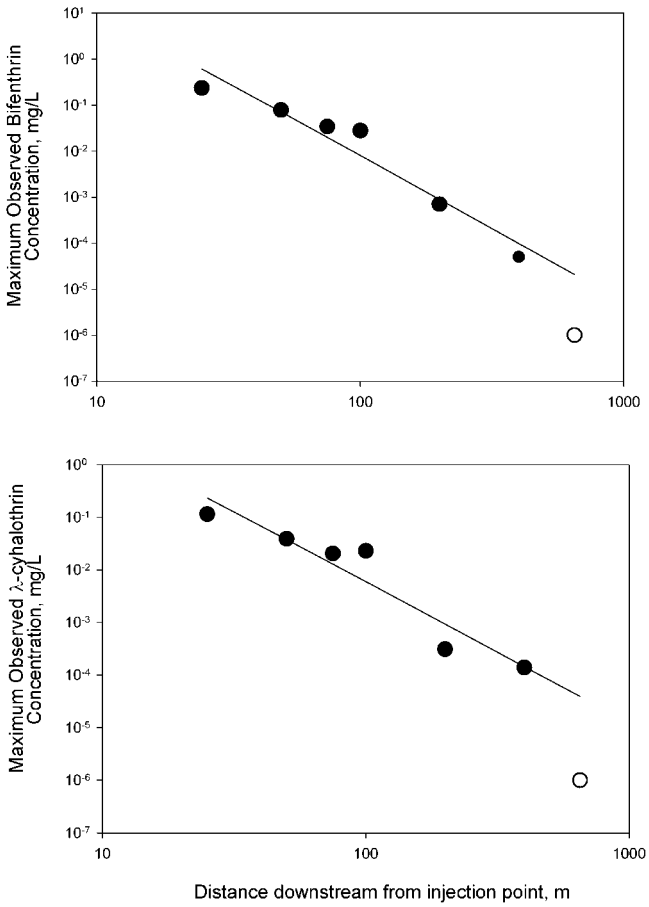


Fig. 3. Least-squares regression relationships fit to log-transformed maximum observed pesticide concentration in water versus distance downstream from injection point. Concentrations at $x = 650$ m (open circles) are detection limits.

Table 4. Estimated mass (g) of bifenthrin and lambda-cyhalothrin in the water, sediment, and plant compartments relative to each sampling time (*total active ingredient amended to ditch at time zero). E-02, E-03, and E-04 represent 10^{-2}, 10^{-3}, and 10^{-4}, respectively, for significant digit purposes

Bifenthrin (g)					Lambda-cyhalothrin (g)				
Time	Water	Plants	Sediment	Total	Time	Water	Plants	Sediment	Total
0 h	—	—	—	*11.4	0 h	—	—	—	*5.70
3 h	5.78	6.29	3.85E-02	12.1	3 h	3.10	6.13	6.24E-02	9.29
12 h	0.718	7.22	3.30E-02	7.97	12 h	0.353	3.76	1.12E-02	4.13
24 h	0.191	4.03	1.13E-02	4.24	24 h	0.106	1.59	3.68E-03	1.70
7 d	0.134	1.93	6.31E-02	2.13	7 d	4.05E-02	6.74E-02	1.79E-02	0.126
14 d	4.48E-02	3.00	5.25E-02	3.10	14 d	6.90E-03	2.09E-01	129E-02	0.229
30 d	4.37E-03	0.199	1.93E-03	0.206	30 d	1.05E-03	3.41E-02	4.24E-02	7.75E-02
44 d	8.82E-04	4.89E-02	8.31E-04	5.07E-02	44 d	6.22E-03	9.10E-02	5.04E-02	0.148

et al. [11] showed that 1.8 to 6.5% of the original dose of lambda-cyhalothrin applied into vegetated ditch enclosures remained after 3 d. These results indicate that a rapid reduction of both pesticides occurred within the first day of application. This rapid decrease in water concentrations is of great importance since pyrethroids have been shown to elicit toxic effects at extremely low concentrations in nontarget aquatic organisms [17,21,22]. To reduce loadings and toxic effects in receiving water bodies after a runoff event, an effective vegetated drainage ditch length is required. Ditch lengths of 120 and 280 m were estimated from this study to reduce bifenthrin and lambda-cyhalothrin to 1.00 and 0.100% of initial values, respectively. These values are based on a worst-case scenario, indicating that shorter ditch lengths may be effective, but depending on space limitations, more conservative distances would likely be more effective.

Water, plant, and sediment samples were collected throughout the study to determine the relative importance of each compartment. Since it is difficult to directly compare concentrations between compartments, mass balance calculations were performed to better understand the distribution and fate of bifenthrin and lambda-cyhalothrin in this system. After 12 h, the majority of pesticide mass was found in the plant compartment, giving evidence that this compartment was the most important and effective compartment in the mitigation of these insecticides. Recent reports [3,10–12] have also shown the importance of aquatic vegetation in pesticide mitigation. It would be expected that sediments would also play an important role in this mitigation process since pyrethroids have a relatively high K_{oc} (Table 1). In this study, sediments were a minor sink because of the dense plant community that limited the movement and/or partitioning of bifenthrin and lambda-cyhalothrin to the sediment compartment. Similar results were found in a microcosm study by Hand et al. [10], where aquatic plants significantly reduced the amount of lambda-cyhalothrin reaching the sediments. Moreover, in the same study, it was shown that lambda-cyhalothrin plant adsorption was virtually irreversible, in turn reducing sediment partitioning. The results of the present study suggest that a small amount of bifenthrin and lambda-cyhalothrin was mobilized from the more concentrated upstream portion of the ditch, as evidenced by an increase in dispersion distance of relative water and sediment contamination along ditch length at the 30- (Fig. 2) and 44-d sampling intervals. An increase in pesticide dispersion distance was not evident for plants at later time points.

Differences in stability between bifenthrin and lambda-cyhalothrin in this test system were evident from the half-lives calculated for each pesticide. Bifenthrin exhibited a half-life of 6.12 d, while the half-life for lambda-cyhalothrin was only 1.35 d. Hand et al. [10] reported similar results in a study investigating the route of metabolism of [^{14}C]lambda-cyhalothrin following adsorption to aquatic plants where lambda-cyhalothrin quickly bound to the plant surface and was readily degraded by ester cleavage. This was evident because of the rapid increase in the cyclopropane acid metabolite and lack of parent compound present in the water of their test system. Alternatively, this shorter half-life may have been attributed to alkaline hydrolysis. Studies have shown that the pH in surface waters can exceed 9 because of the photosynthesis by plants and algae [23,24]. Lambda-cyhalothrin has been shown to be unstable under these basic conditions, while bifenthrin has been shown to be stable [25].

The fate of bifenthrin and lambda-cyhalothrin reflects a complex combination of processes, and other important factors also affect their fate in drainage ditches, especially in the Mississippi Delta region. In this region, water temperatures can be greater that than 30°C, while air temperatures near the water surface can reach 40°C. Even although pyrethroids have low volatility (Table 1), these higher temperatures may play a role in their degradation and volatilization from the ditch system. Other factors that are related to this process include plant growth and water level fluctuation, which expose plant material to ambient air and direct sunlight, leading to possible volatilization and photolysis. Rudel [26] reported that pesticide volatilization from plant surfaces was higher as compared to soil surfaces for a range of pesticides, including lambda-cyhalothrin, because of increased airflow and exposure to these surfaces. In relation to water and submerged plant surfaces, these air-exposed plant surfaces previously submerged may be an important vector for pesticide loss from the ditch system. Furthermore, many of the drainage ditches in the Mississippi Delta are ephemeral and by late summer/early fall are often void of water. This may play an important role in the cycling or degradation of compounds that remain sorbed to remaining plant material and sediments. With the loss of water, light and air have direct contact with these surfaces, possibly increasing the photo- and aerobic degradation of agrichemical compounds. This annual cycling may act as a self-cleaning mechanism for these ditch systems and reduce pesticide loadings into receiving water bodies during the winter and spring, when water returns to these systems.

CONCLUSION

Ditch lengths of less than 300 m are required to reduce loadings into receiving water bodies during worst-case-scenario runoff events. This demonstrates the importance and

effectiveness of vegetated drainage ditches as a best management practice for the mitigation of pyrethroid runoff. For this tool to be optimally effective, it would need to work in parallel with other existing best management practices and farming practices to facilitate single entry points into surrounding ditch systems. Since most fields are surrounded by these ditch systems in the Mississippi Delta, multiple entry points may be required, especially in larger-acreage fields. Buffer strips on the fringes of agricultural fields would also help in the funneling of runoff into these ditch entry points. In many cases where other best management practices are not available, simple vegetated drainage ditches would still be an effective tool.

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