Mitigation Assessment of Vegetated Drainage Ditches for Collecting Irrigation Runoff in California

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Widespread contamination of California water bodies by the organophosphate insecticides diazinon and chlorpyrifos is well documented. While their usage has decreased over the last few years, a concomitant increase in pyrethroid usage (e.g., permethrin) (replacement insecticides) has occurred. Vegetated agricultural drainage ditches (VADD) have been proposed as a potential economical and environmentally efficient management practice to mitigate the effects of pesticides in irrigation and storm runoff. Three ditches were constructed in Yolo County, California for a field trial. A U-shaped vegetated ditch, a V-shaped vegetated ditch, and a V-shaped unvegetated ditch were each amended for 8 h with a mixture of diazinon, permethrin, and suspended sediment simulating an irrigation runoff event. Water, sediment, and plant samples were collected spatially and temporally and analyzed for diazinon and permethrin concentrations. Pesticide half-lives were similar between ditches and pesticides, ranging from 2.4 to 6.4 h. Differences in half-distances (distance required to reduce initial pesticide concentration by 50%) among pesticides and ditches were present, indicating importance of vegetation in mitigation. *Cis*-permethrin half-distances in V ditches ranged from 22 m (V-vegetated) to 50 m (V-unvegetated). Half-distances for trans-permethrin were similar, ranging from 21 m (V-vegetated) to 55 m (V-unvegetated). Diazinon half-distances demonstrated the greatest differences (55 m for V-vegetated and 158 m for V-unvegetated). Such economical and environmentally successful management practices will offer farmers, ranchers, and landowners a viable alternative to more conventional (and sometimes expensive) practices.

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Diazinon [O,O-diethyl 0-2-isopropyl-6-methyl (pyrimidine-4-yl) phosphorothioate] and chlorpyrifos (O,O-diethyl O-3,5,6-trichloro-2-pyridyl phosphorothioate) are two commonly identified organophosphate (OP) insecticides found in California water samples. From 1990 to 2001, more than 267,000 kg of diazinon (active ingredient) were applied to 37 different California crops (Epstein et al., 2000). According to Epstein et al. (2000), between 1992 and 1998 there was decreased OP usage in the Sacramento and San Joaquin Valleys, but a concomitant increase in pyrethroid use. The USDA National Agricultural Statistics Service (USDA NASS, 2003) reported over 52,000 kg of the pyrethroid permethrin [3-phenoxybenzyl (1RS)-*cis*, *trans*-3-(2,2-dichlorovinyl)-2,2-dimethylcyclopropanecarboxylate] were applied to 23 of California's crops from 1990 to 2001. According to de Vlaming et al. (2000), over the last decade, pulses of diazinon toxicity (as measured by *Ceriodaphnia dubia* survival) have occurred in California's Central Valley from dormant orchard drainage. Domagalski (1996) discussed storm concentrations of OPs in the Sacramento River Basin. As the storm hydrograph increased, a concomitant increase in OP concentration occurred.

Lee and Jones-Lee (2002) emphasized the urgent need for quantitative information on best management practice (BMP) efficiency for agricultural runoff and discharge situations, particularly within California's Central Valley. Along these same lines, USEPA (2002) published a "Twenty Needs Report" on how research can enhance the total maximum daily load (TMDL) process. Current research described here addresses two of those needs: "Improve watershed and water quality monitoring" and "Improve information on BMPs, restorations or other management practice effectiveness, and the

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Abbreviations: OP, organophosphate; BMP, best management practice; TMDL, total maximum daily load.

related processes of system recovery." The Central Valley Regional Water Quality Control Board (2002) OP TMDL implementation report suggests that focus be placed on agricultural management practices likely to be effective in reducing offsite movement of pesticides. Currently, there is also a substantially increased emphasis on TMDLs and improvement of water quality, especially for nonpoint-source pollution. Several BMPs currently used and encouraged by the U.S. Department of Agriculture's Natural Resource Conservation Service (USDA NRCS) include, but are not limited to, buffer and filter strips, riparian buffers, reduced and conservation tillage, grassed waterways, and constructed wetlands. Except for tillage, all of these BMPs require farmers to remove acreage from production landscape to meet physical BMP requirements. Unfortunately, some farmers are unable to sacrifice acreage, especially in small production plots. In these situations, it is important to find an economical, yet environmentally sound alternative to traditional BMPs such as vegetated agricultural drainage ditches (Moore et al., 2001). Drainage ditches are forgotten links between agricultural fields and aquatic receiving systems. Drent and Kersting (1992) reported the use of experimental ditches in the Netherlands for a variety of ecotoxicological evaluations. These unique ditch ecosystems provide a myriad of potential services other than water conveyance, including sediment trapping and nutrient and pesticide mitigation.

The current study involved a comprehensive research effort to determine efficiency of vegetated drainage ditches for mitigation of pesticide-associated runoff from tomato fields. Results from earlier simulated runoff studies (1998 to 2000) indicated substantial sorption of atrazine (2-chloro-4-ethylamine-6-isoprypylamino-S-triazine), lambda-cyhalothrin [(RS)-alpha-cyano-3-phenoxybenzyl 3-(2-chloro-3,3,3-trifluoropropenyl)-2,2-dimethylcyclopropane-carboxylate], bifenthrin [(2-methyl-1,1-biphenyl-3-y1)-methyl-3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethylcyclopropane-carboxylate)], and esfenvalerate [(S)-alpha-cyano-3-phenoxybenzyl (S)-2-(-4-chlorophenyl)-3-methylbutyrate] by ditch vegetation from agricultural fields in Mississippi. Three hours following initiation of simulated storm events, 38% of atrazine and 97% of lambda-cyhalothrin were associated with plant material. Of the measured bifenthrin and esfenvalerate, 52 and 66%, respectively, were associated with vegetation (Moore et al., 2001; Cooper et al., 2002a; Cooper et al., 2002b; Bennett et al., 2005).

Three main objectives involved in the current study were to (i) evaluate mitigation efficiency of two types of ditch design—U (typical in Mississippi Delta) versus V (typical in California)—with the organophosphate diazinon and the pyrethroid permethrin; (ii) evaluate benefit of vegetation in typical California V ditch by comparing its pesticide mitigation efficiency to an unvegetated V-ditch as a control; and (iii) determine pesticide mass distribution within the water, sediment, and (if applicable) plants located in the U-vegetated, V-vegetated, and V-unvegetated ditches to estimate pesticide half-lives, half-distances, and providing data for future modeling efforts.

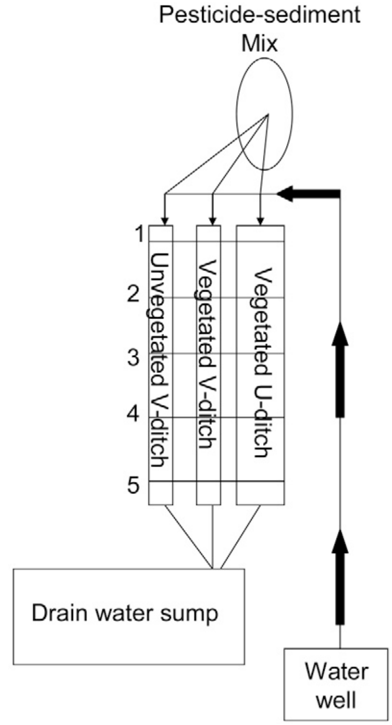


Fig. 1. Schematic of irrigation distribution system and sampling sites.

Materials and Methods

Ditch Design

Three ditches, each 116 m in length, were constructed on a farm in Yolo County, California (Fig. 1). Farm soils were a mix between Yolo silt loam and Reiff very fine sandy loam. Ditch soil analysis indicated a pH of 7.6; CEC = 9.95 cmolc/kg, and 0.025 g/kg organic matter. The field had a 0.2% north to south slope in addition to a 0.15% slope from west to east.

Two different ditch designs ("U" and "V") were employed in this research. Although the "V" ditch design is most common throughout the county, researchers also wanted to compare the broader "U" shape design for pesticide mitigation efficiency. All ditches were wetted less than 24 h before experiment initiation; however, no standing water was present when pesticide amendment began. One U-shaped ditch was constructed with a 3 m top width, 3 m width at maximum water holding capacity, and a water depth of 0.37 m. Two V-shaped ditches were identically constructed with top widths of 1.8 m, 0.6 m water height at maximum holding capacity, and water depths of 0.24 m. While one V-ditch remained unvegetated as a control ditch, the other V-ditch and single U-ditch were planted with *Hordeum vulgare* (barley) and *Lolium multiflorum* (annual ryegrass). Lamb's quarter (*Chenopodium album*) was an invasive prevalent weed within the vegetated ditches. Identical sampling sites were established within all three ditches at the simulated runoff inlet (0 m) (site 1), 42 m (site 2), 51 m (site 3), 88 m (site 4), and 108 m (site 5). One day before the simulated irrigation runoff event, ditch vegetative cover and dominant plant species were determined by sampling multiple 0.23 m^2 quadrats at each sampling site (Table 1). Any runoff leaving the ditches was routed into a

Table 1. Plant density survey for experimental ditches, Yolo County, California.

Distance	V ditch	U ditch
0–42 m	80% coverage	65% coverage
	457 *Hordeum*/m^2	261 *Hordeum*/m^2
43–51 m	100% coverage	100% coverage
	22 *Chenopodium*/m^2	22 *Chenopodium*/m^2
52–89 m	100% coverage	100% coverage
	22 *Chenopodium*/m^2	22 *Chenopodium*/m^2
	717 *Lolium*/m^2	717 *Lolium*/m^2
90–110 m	90% coverage	45% coverage
	22 *Lolium*/m^2	22 *Lolium*/m^2

vegetated sump pond to prevent direct release into the aquatic receiving system. The small volume of water entering the sump filtered through the soil column within 16 h of entry.

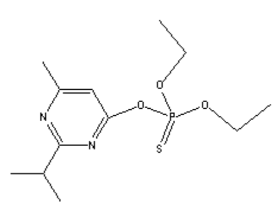
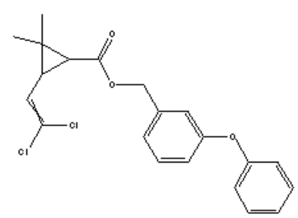
Simulated Irrigation Runoff Event

An irrigation runoff event was simulated in each of the three drainage ditches in July 2005. A mixture of 112 g of diazinon (Diazinon AG500), 12.2 g of permethrin (Pounce 3.2 EC), and 45 kg of dry soil (previously collected from ditch bottoms) was added to a 3800 L steel water tank filled with ground water and kept in suspension using a small submersible pump. Physical and chemical properties of these two pesticides are presented in Table 2. Pesticide concentrations in simulated runoff (0.19 mg/L diazinon and 0.02 mg/L permethrin) were based on recommended application rates for a 32 ha contributing area of tomatoes (4.9 L/ha for diazinon and 0.37 L/ha for permethrin) and an assumed 0.05% diazinon and 0.09% permethrin runoff with a targeted discharge of 7 L/s into experimental ditches (Spencer and Cliath, 1991). Simulated runoff was pumped from the tank into ditches via 1.9 cm tubing using an Atwood V450 submersible pump (maximum flow rate of 1703 L/h) for 8 h. Irrigation pipe (30 cm diameter) carried dilution water from a nearby pump to constructed ditches. The irrigation pipe was reduced to 5 cm and split three ways (one entry for each ditch). A calibrated pesticide delivery system was placed in the water split to incorporate the pesticide-sediment mixture. The theoretical total flow (including dilution water) from 0–8 h was 201,600 L for each ditch; however, the water was "ramped up" for the first 15 min to reach the desired outflow, and it was "ramped down" the last 15 min before the dose ended. Therefore, only about 198,000 L of water actually were added to the system. At 8 h after dose, the water was turned off and there was no further flow addition to the ditches.

Collection of Water, Sediment, and Plant Samples

Water quality data was collected at inflow (site 1) and near the outflow (site 5) of each of the three constructed ditches at times 0, 0.5, 1, 4, 8, and 16 h. Velocity (m/s), temperature (°C), pH, dissolved oxygen (mg/L), and electrical conductivity (dS/m) were measured using calibrated hand-held field meters (Table 3). Grab samples of water were collected in pre-cleaned, certified 1 L amber Boston round, narrow mouth glass bottles with Teflon-lined closures at 0, 0.5, 1, 4, 8, 16, 24, and 48 h, and 5 d post-application from each site. Background sediment samples were collected from each of the three ditches and analyzed for a suite of 17 pesticides including atrazine, methyl parathion (O,O-dimethyl O-4-nitrophenyl phosphorothioate), chlorpyrifos, pp'-DDE (pp'-dichlorodiphenyldichloroethylene), pp'-DDD (pp'-dichlorodiphenyldichloroethane), pp'-DDT (pp'-dichlorodiphenyltrichloroethane), bifenthrin, and lambda-cyhalothrin. Sediment samples were collected in 120 mL wide mouth glass bottles with Teflon-lined closures at times identical to water collection (including 24, 48, and 120 h samples). Plant samples were also collected along the same time schedule as sediments. 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 plant material exposed in the water column (between sediment-water surface) was collected for analysis. Plant samples were wrapped in aluminum foil and placed in pre-labeled 3 L freezer bags. All samples were preserved on wet ice from collection through transport to the Aquatic Toxicol-

Table 2. Physical and chemical properties of diazinon and permethrin.

	Diazinon	Permethrin
Structure		
Molecular weight (g/mol)†	304.35	391.30
Vapor pressure (mm Hg)†	7.28×10^{-7}	3.38×10^{-7}
Water solubility (mg/L)†	40	0.2
Henry's Law Constant (P m^3/mol)‡	0.072	0.189
log K_{ow}‡	3.30	6.1
log K_{oc}‡	1007–1842	10,471–86,000
Hydrolysis half-life (days)§	138	37.7
Photolysis half-life (days)¶#	88	110
Soil half-life (days)‡		
Sandy loam aerobic	39	30
Sandy loam anaerobic	17	108

† EXTOXNET, 1996.

‡ USDA ARS, 1995.

§ Kegley et al., 2007.

¶ Frank et al., 1991.

Laskowski, 2002.

ogy Laboratory (ATL) at the University of California, Davis. Before transport of water samples to the California Department of Fish and Game Water Pollution Control Laboratory (DFG-WPCL), samples were kept in the dark at 4°C. Sediment samples (also transported to DFG-WPCL) were frozen and kept in the dark until transport. Plant material was frozen immediately on receipt at the ATL and shipped overnight to the USDA Agricultural Research Service National Sedimentation Laboratory (NSL) for sample preparation. Upon arrival at the NSL, plant samples were dried and ground using a Thomas-Wiley Model 4 laboratory mill. After preparation, samples were placed in glass vials and shipped to DFG for pesticide analyses.

Pesticide Extraction– Water

Unfiltered water samples were extracted according to USEPA Method 3510C– Separatory Funnel Liquid-Liquid Extraction. One-liter water samples were fortified with triphenyl phosphate and dibromooctafluorobiphenyl to monitor extraction proficiency and extracted twice with dichloromethane (DCM) using a mechanical rotating extractor. Extracts were dried using sodium sulfate, concentrated, and solvent exchanged with petroleum ether (PE) using Kuderna-Danish (K-D) evaporative glassware equipped with a 3-ball Snyder column followed with a micro-Snyder apparatus and adjusted to a final volume of 2 mL in iso-octane. Concentrations of *cis* and *trans* isomers of permethrin were reported separately, since *cis*-permethrin is generally considered more toxic than *trans*-permethrin.

Pesticide Extraction and Cleanup– Sediment and Vegetation

Sediment and vegetation sample extraction followed USEPA Method 3545A– Pressurized Fluid Extraction. Homogenized sediment (10 g) and dried vegetation (2.5 g) samples were mixed with pre-extracted Hydromatrix (7 g, Varian Corp.) and fortified with triphenyl phosphate, dibromooctafluorobiphenyl, and dibutylchlorendate. Samples were extracted twice with acetone/DCM (50/50, v/v) using a Dionex Accelerated Solvent Extractor (ASA 200, 100°C, 1500 psi). Extracts were dried using sodium sulfate, concentrated, and solvent exchanged with PE using K-D evaporative glassware equipped with a 3-ball Snyder column followed with a micro-Snyder apparatus and adjusted to final volume of 2 mL in *iso*-octane. Clean up of sulfur, chlorophyll, and other matrix interferences followed USEPA Method 3600C– Cleanup guidelines, as needed.

Instrument Analysis

Water, sediment, and vegetation sample final extracts were analyzed for diazinon using USEPA 8141B (Organophosphate Compounds by Gas Chromatography) guidelines, while USEPA 8081B (Organochlorine Pesticides by Gas Chromatography) guidelines were used for permethrin analysis. Diazinon was analyzed by dual column high resolution gas chromatography with flame photometric detectors in phosphorus mode. Permethrin was analyzed using dual column high resolution gas chromatography equipped with electron capture detectors. Aqueous reporting limits for *cis*-permethrin, *trans*-permethrin, and diazinon were 0.005, 0.005, and 0.020 μg/L, respectively. Sediment reporting limits (dry weight) for *cis*-permethrin, *trans*-permethrin, and diazinon were 4.00, 4.00, and 10.0 μg/kg, respectively. Vegetation reporting limits (fresh weight) were 5.00, 5.00, and 10.0 μg/kg for *cis*-permethrin, *trans*-permethrin, and diazinon, respectively.

Table 3. Inflow and outflow water quality of three constructed drainage ditches, 2005. A dash (–) indicates no water available for sampling.

Site	Time	Velocity	Temperature	pH	Dissolved O_2
		m/s	°C		mg/L
U-1†	0.0 h	5.19	18.8	7.2	9.2
	0.5 h	2.59	19.3	7.2	8.1
	1.0 h	3.17	21.3	7.2	6.0
	4.0 h	3.40	23.4	7.1	7.1
	8.0 h	0.00	25.0	7.1	6.8
	16 h	–	–	–	–
U-5	0.0 h	–	–	–	–
	0.5 h	1.07	19.3	7.5	4.7
	1.0 h	0.00	20.0	7.7	6.7
	4.0 h	1.16	26.2	7.8	6.6
	8.0 h	0.00	25.4	7.6	5.4
	16 h	0.00	21.3	7.4	4.7
V_1–1	0.0 h	3.31	19.0	7.3	8.7
	0.5 h	3.00	18.4	7.1	7.5
	1.0 h	5.36	20.1	7.3	7.1
	4.0 h	4.11	23.2	7.1	6.2
	8.0 h	0.00	22.6	7.3	9.2
	16 h	–	–	–	–
V_1–5	0.0 h	–	–	–	–
	0.5 h	–	–	–	–
	1.0 h	1.61	20.1	7.6	6.4
	4.0 h	0.63	21.9	7.4	6.4
	8.0 h	0.00	21.3	7.5	6.4
	16 h	0.00	19.5	7.2	4.9
V_2–1	0.0 h	2.59	19.1	7.2	8.5
	0.5 h	2.68	18.4	7.2	7.2
	1.0 h	3.58	19.3	7.1	8.4
	4.0 h	3.58	25.5	7.1	8.0
	8.0 h	0.00	22.8	7.4	9.5
	16 h	–	–	–	*
V_2–5	0.0 h	–	–	–	–
	0.5 h	1.56	19.3	7.6	7.0
	1.0 h	0.00	20.9	7.6	6.5
	4.0 h	0.40	22.6	7.4	6.5
	8.0 h	0.00	22.2	7.2	6.9
	16 h	–	–	–	–

† U-1 (U-ditch inflow), U-5 (U-ditch outflow), V_1–1 (Vegetated V-ditch inflow), V_1–5 (Vegetated V-ditch outflow), V_2–1 (Unvegetated V-ditch inflow), V_2–5 (Unvegetated V-ditch outflow).

Data Analysis

Ordinary least-squares linear regression analyses (Sokal and Rohlf, 1981) were used to fit curves to log-transformed diazinon and permethrin water concentrations (*y*) versus the log of the distance down ditch from the inlet (*x*). Mass balances were performed using data on water, plant, and sediments collected along transects of the ditch length for each sample time point (0.5, 1, 4, 8, 16, 24, 48, and 120 h). This enabled quantitative evaluation of chemical partitioning that occurred over the study duration.

Table 4. Selected aqueous pesticide concentrations (μg/L) in inflow and outflow (site 5) of three experimental drainages ditches following a simulated irrigation event in Yolo County, CA. A dash (–) indicates no water available for sampling.

	U-vegetated			V-vegetated			V-unvegetated		
	CP†	TP	D	CP	TP	D	CP	TP	D
Inflow (0 h)	27.0	31.6	580	225	275	1.32E03	117	110	1.10E03
Inflow (1 h)	2.31	2.20	78.0	2.12	2.20	98.0	1.98	2.20	84.8
Site 5 (1 h)	18.2	17.9	520	9.63	10.1	384	13.5	14.3	500
Inflow (4 h)	1.22	1.21	37.8	1.35	1.45	48.2	1.80	1.87	63.0
Site 5 (4 h)	6.80	7.48	453	1.37	1.43	50.4	1.35	2.75	60.0
Inflow (8 h)	16.7	19.0	300	5.27	6.20	220	5.24	5.65	297
Site 5 (8 h)	1.08	1.10	87.0	1.89	1.86	70.0	1.17	1.21	42.0
Inflow (16 h)	–	–	–	–	–	–	–	–	–
Site 5 (16 h)	0.981	1.05	112	–	–	–	–	–	–

† CP = cis-permethrin, TP = trans-permethrin, D = diazinon.

The mass balance at a given time point was determined as:

$$M_{(\text{total})} = M_{\text{w (0–108m)}} + M_{\text{p (0–108m)}} + M_{\text{s (0–108m)}} \quad [1]$$

where $M_{w(0–108)}$, $M_{p(0–108)}$, and $M_{s(0–108)}$ reflect the total chemical mass (g) in water, plants and sediments over the 108 m ditch length. Other abiotic and biotic factors such as leaching, volatilization, and microbial activity were not measured; therefore, they are not represented in the mass balance equation.

Ditch chemical depuration rate constants (k_2) were determined for water in each of the three ditches. This was accomplished by plotting the ln (maximum observed concentration; microgram per liter) as a function of time and, through linear regression analysis, determining the slope. Pesticide half-lives ($t_{1/2}$) in water were estimated using the equation ln (2)/k_2. Using the same premise, ditch half-distances were determined by plotting the ln (total concentration) as a function of ditch sample distance, determining the slope, and using the ln(2)/k_2 equation.

Results

Background Sediment Samples

In the V-vegetated ditch sediment, only DDT and its metabolites were detected among the 17 pesticides analyzed. DDT was measured at 7.12 μg/kg, while metabolites DDD and DDE were each below 1 μg/kg. Sand, silt, and clay fractions for this ditch sediment were 69, 30, and 1%, respectively. Sediment in the V-unvegetated ditch had concentrations of 26.3, 0.42, and 5.04 μg/kg for atrazine, DDE, and DDT, respectively. It possessed 54% sand, 44% silt, and 2% clay. U-vegetated sediment samples had only DDE (0.19 μg/kg) and DDT (5.04 μg/kg) present out of the 17 analyzed pesticides. This sediment was 78% sand, 21% silt, and 1% clay. Organic matter content in all three ditches was ≤0.025 g/kg.

Table 5. Distribution of mean measured mass (% ± SE) in water, sediment, and plant for *cis*-permethrin, *trans*-permethrin, and diazinon from experimental ditches in Yolo County, California over the 8 h experimental dose.

	U-vegetated	V-vegetated	V-unvegetated
Cis-permethrin			
Water	14 ± 6	16 ± 8	20 ± 6
Sediment	64 ± 5	52 ± 2	80 ± 6
Plant	23 ± 7	33 ± 5	–
Trans-permethrin			
Water	16 ± 7	18 ± 10	23 ± 6
Sediment	64 ± 6	49 ± 3	77 ± 6
Plant	20 ± 6	33 ± 7	–
Diazinon			
Water	38 ± 9	38 ± 9	50 ± 5
Sediment	54 ± 8	57 ± 9	50 ± 5
Plant	8 ± 0.3	5 ± 0.3	–

Cis-permethrin

Initial measured aqueous concentrations of *cis*-permethrin in each of the three ditches differed substantially. Although all ditch delivery systems were calibrated and re-checked before the simulated irrigation event, variability of inflow concentrations of all pesticides (*cis*-permethrin, *trans*-permethrin, and diazinon) still occurred. Samples collected from the inflow pipe at time 0 (test initiation), indicated *cis*-permethrin concentrations of 27.0, 225, and 117 μg/L for the U-ditch, V-vegetated, and V-unvegetated ditches, respectively. Inflow samples collected from 0 to 8 h had mean (± SE) *cis*-permethrin concentrations of 11.2 ± 4.82, 49.6 ± 43.9, and 28.7 ± 22.3 μg/L for the U-ditch, V-vegetated, and V-unvegetated ditches, respectively. At V-vegetated ditch site 5 (108 m down-ditch), final *cis*-permethrin concentration decreased 80% from the 1 h sampling to the 8 h sampling (Table 4).

Even though initial inflow water concentrations of pesticides differed between ditches, mass balance calculations allowed for a correction factor, making ditches comparable to one another. Based on both the theoretical application and recovered mass from sampling, 65, 56, and 47% of overall permethrin mass was accounted for in the U-ditch, V-vegetated, and V-unvegetated ditches:. Mass balances of *cis*-permethrin in water shifted from the 1 h sample as compared to the 8 h sample. In the U-ditch, 26 ± 11% of measured *cis*-permethrin mass was located in the water at 1 h; however, only 4 ± 1% of the mass was in the water column at the 8 h sample. Similar trends were evident for the same time periods in the V-vegetated (31 ± 8% and 9 ± 3%) and V-unvegetated (32 ± 7% and 17 ± 3%) ditches. Examination of each ditch indicated 14 ± 6, 16 ± 8, and 20 ± 6% of measured *cis*-permethrin mass during the actual exposure period (8 h) was located in water of the U-ditch, V-vegetated, and V-unvegetated ditches, respectively (Table 5). When assessing all water samples collected over the experiment duration, 19 ± 10, 29 ± 14, and 27 ± 8% of measured *cis*-permethrin mass

was located in the water column of the U-ditch, V-vegetated ditch, and the V-unvegetated ditch, respectively. Summing water, sediment, and plant *cis*-permethrin masses at each individual sampling time during the entire experiment yielded mass ranges from 225 to 1901 mg for the U ditch, 192 to 843 mg for the V-vegetated ditch, and 206 to 2149 mg for the V-unvegetated ditch.

Several sites in the drainage ditches were dry before the 16 h sampling due to soil conditions. As a result, sediment-pesticide masses shifted. When examining the overall experiment, *cis*-permethrin mass percentage in sediment for the U-ditch, V-vegetated ditch, and V-unvegetated ditch was 70 ± 3, 58 ± 6, and 86 ± 6%, respectively. By analyzing data where no water was present, the *cis*-permethrin sediment mass percentages changed to 75 ± 3, 72 ± 3, and 100 ± 0% respectively, for the U-ditch, V-vegetated, and V-unvegetated ditches. *Cis*-permethrin half-lives in ditch water ranged from 2.4 h (V-vegetated) to 4.1 h (U-ditch), while pesticide half-distances ranged from 22 m (V-vegetated) to 347 m (V-unvegetated) (Table 6).

Trans-permethrin

Inflow concentrations of trans-permethrin at time 0 were 31.6, 275, and 110 µg/L for the U, V-vegetated, and V-unvegetated ditches, respectively. Inflow samples collected from 0 to 8 h had mean (± SE) *trans*-permethrin concentrations of 12.7 ± 5.70, 60.0 ± 53.8, and 27.6 ± 20.8 µg/L for the U-ditch, V-vegetated, and V-unvegetated ditches, respectively. Between 1 and 8 h samples, aqueous trans-permethrin concentrations decreased by 94, 82, and 92% at site 5 for the U, V-vegetated, and V-unvegetated ditches, respectively (Table 4). Water was not present after 8 h at site 5 in either the V-vegetated or V-unvegetated ditches.

Trans-permethrin mass in ditch water showed similar shifts to that of *cis*-permethrin. In the U-ditch, 30 ± 11% of trans-permethrin mass was in the water column at 1 h, but by 8 h, only 5 ± 1% of the mass was present in water. In the V-unvegetated ditch, 34 ± 7% of the mass was present at 1 h, but had been reduced to 19 ± 4% by 8 h. The largest shift occurred in the V-vegetated ditch, with 37 ± 8% mass in water at 1 h, but only 9 ± 3% mass in water at 8 h. Examination of each individual ditch indicated that 33, 44, and 39% of the measured *trans*-permethrin mass during the actual exposure period (8 h) was located at site 4 (88 m) in the U-ditch, V-vegetated, and V-unvegetated ditches, respectively. During the 8 h exposure, 16 ± 7%, 18 ± 10%, and 23 ± 6% of *trans*-permethrin mass was measured in water from the U-ditch, V-vegetated ditch, and V-unvegetated ditch, respectively (Table 5). Summing water, sediment, and plant *trans*-permethrin masses measured at each individual sampling time during the entire experiment yielded *trans*-permethrin ranges from 394 to 1613 mg for the U ditch, 132 to 738 mg for the V-vegetated ditch, and 209 to 2226 mg for the V-unvegetated ditch.

As with *cis*-permethrin, mass *trans*-permethrin percentages in sediments shifted after 16 h due to an absence of water. Overall percent *trans*-permethrin masses measured in sediments were 66 ± 5, 58 ± 8, and 85 ± 7% for the U-ditch, V-vegetated, and V-unvegetated ditches, respectively. By analyzing only samples where water was absent, sediment mass percentages changed to 73 ± 6, 75 ± 6, and 100 ± 0%, respectively. Half-lives of trans-permethrin in ditch water ranged from 3.4 (V-vegetated) to 4.1 h (U-ditch, while trans-permethrin half-distances ranged from 21 m (V-vegetated) to 239 m (V-unvegetated) (Table 6).

Table 6. Pesticide water half-lives and half-distances in U, V-vegetated, and V-unvegetated experimental ditches in Yolo County, California.

	U-vegetated	V-vegetated	V-unvegetated
Cis-permethrin half-life (h)	4.1	2.4	3.5
Cis-permethrin half-distance (m)	169	22	50
Trans-permethrin half-life (h)	4.1	3.4	3.7
Trans-permethrin half-distance (m)	124	21	55
Diazinon half-life (h)	6.4	4.5	4.5
Diazinon half-distance (m)	1155	56	158

Diazinon

Diazinon inflow concentrations at the U, V-vegetated, and V-unvegetated ditches at time 0 were 580, 1320, and 1100 µg/L, respectively. Inflow samples collected from 0 to 8 h had mean (± SE) diazinon concentrations of 251 ± 96.5, 402 ± 234, and 397 ± 189 µg/L for the U-ditch, V-vegetated, and V-unvegetated ditches, respectively. Between the 1 and 8 h U-ditch site 5 (108 m) samples, diazinon concentrations had decreased by 83% (Table 4). Concentrations in the V-vegetated and V-unvegetated had decreased by 82 and 92%, respectively, in the same time period (Table 4).

As opposed to *cis*- and *trans*-permethrin masses in water, aqueous diazinon masses varied little. In U-ditch 1 h samples, 55 ± 8% of measured diazinon was in water, and by 8 h, 30 ± 6% was present in water. Aqueous mass percentages slightly increased in the V-unvegetated ditch between 1 h and 8 h (50 ± 8 to 58 ± 9%). In the V-vegetated ditch, 49 ± 5% of the diazinon mass was measured in the water column at 1 h, while 44 ± 12% was measured in water at 8 h. Examination of each individual ditch indicated that 24, 39, and 37% of the measured diazinon mass during the actual exposure period (8 h) was located at site 4 (88 m) in the U-ditch, V-vegetated, and V-unvegetated ditches, respectively. For the 8 h exposure, 38 ± 9, 38 ± 9, and 50 ± 5% of the diazinon mass was measured in water of the U-ditch, V-vegetated ditch, and V-unvegetated ditch, respectively (Table 5). Summing water, sediment, and plant diazinon masses measured at each individual sampling time during the entire experiment yielded diazinon masses ranging from 4533 to 15,970 mg for the U ditch, 1868 to 6637 mg for the V-vegetated ditch, and 1230 to 10,919 mg for the V-unvegetated ditch.

Based on both the theoretical application and recovered mass from sampling, 29, 15, and 14% of overall diazinon mass was accounted for in the U-ditch, V-vegetated, and V-unvegetated ditches. As with both *cis*- and *trans*-permethrin, mass diazinon percentages in sediments shifted after 16 h due to an absence of water. Overall percent diazinon masses measured in sediments were 69 ± 8, 74 ± 10, and 73 ± 10% for the U-ditch, V-vegetated, and V-unvegetated ditches,

respectively. When water was not present in ditches during sampling, sediment mass percentages changed to 93 ± 1, 97 ± 1, and 100 ± 0%, respectively. Half-lives of diazinon in ditch water ranged from 4.5 h (both V-unvegetated and V-vegetated) to 6.4 h (U-ditch). Half-distances of diazinon ranged from 55 m (V-vegetated) to 1155 m (U-ditch) (Table 6).

Discussion

Domagalski et al. (1997) reported at least two potential modes of pesticide entry into surface water within the San Joaquin– Tulare Basin (California) study areas—rainfall and irrigation drainage. Pesticide entry into receiving waters following storm or irrigation events depends on several factors, such as pesticide chemistry, rainfall intensity, time of application, and surrounding soil properties. The united goal of farmers, pesticide applicators, conservation associations, and regulatory agencies is to keep pesticides on agricultural production acreage and out of surrounding water bodies. In efforts to reduce the possibility of this occurring, management practices have been suggested to mitigate pesticide runoff.

Vegetation plays a significant role in many of the suggested BMPs. Stiff grass hedges, grassed waterways, and riparian filter strips are just three examples of incorporating vegetation into runoff mitigation strategies. Vegetated drainage ditches are becoming increasingly popular among farmers and landowners with little available production acreage to set aside for potential mitigation purposes. Vegetation has been documented to assist in mitigation of permethrin and diazinon. Filter strips containing trees, shrubs, and grasses at widths of 7.5 and 15 m reduced permethrin-associated contaminants 27 to 83% (Schmitt et al., 1999). In a 50% vegetated filter strip treatment, 37% of applied diazinon was retained in the vegetative matter and root zone. Where vegetated filter strip treatment was 100%, 88% of applied diazinon was found in the vegetative matter and root zone (Watanabe and Grismer, 2001). *Chenopodium album* (an invasive present in the current study) along with three other species retained 81 to 98% of applied diazinon in laboratory studies (Syversen and Haarstad, 2005).

When taking the successful and proven vegetated ditch concept from the Mississippi Delta to Yolo County, California, certain issues needed to be addressed. Foremost was the initial ditch design. Typical drainages in the South and Midwest have a U-shaped design in addition to being permanent systems. In California, on the other hand, field ditches are more temporary and are V-shaped, due to a common implement used by farmers. Due to the temporary nature of California ditches, more variety of soil properties is present. Hardpan clay bottoms are present in many drainage ditches in the Mid-South; however, in the current research, sandy conditions were present in the field. These soil differences might potentially play a role in ditch flow, seepage, and interactions with shallow ground water supplies. Investigations into multiple flow directions were not examined within the current study.

Pesticides entering surface water bodies have the potential to impact aquatic flora and fauna. Because of its relatively high water solubility, the majority of diazinon transported into aquatic systems can be expected to remain in the aqueous phase (Bondarenko and Gan, 2004). Diazinon's aqueous fate is dependent on factors such as pH, temperature, and organic carbon content of water via chemical hydrolysis and microbial degradation (Bondarenko et al., 2004; USEPA, 2005). At pH 7, studies have reported diazinon's stability to be near 6 mo (USEPA, 2005). In the current study, aqueous pH measurements ranged from 7.1 to 7.8 in the three ditches, but diazinon aqueous half-lives were between 4.5 to 6.4 d (Table 6). While no specific microbial studies were conducted as part of this experiment, it is assumed microbial activity played a role in diazinon degradation.

Permethrin's chemical profile, including relatively low water solubility, indicates its preference for binding to sediment or other forms of organic carbon. Using limnocorrals, Solomon et al. (1985) applied permethrin in a small Ontario (Canada) lake to study aqueous dissipation. Permethrin dissipation was rapid and generally 90% was transferred or transformed from the water column within 10 d. Field studies of permethrin runoff from cotton (*Gossypium hirsutum*) indicated that even in abnormally high rainfall years, less than 1% of applied permethrin was detected in runoff samples (Carroll et al., 1981). Rawn et al. (1982) and Solomon et al. (1985) indicated permethrin's rapid sorption to sediment and subsequent slow disappearance. Distributions of mean measured masses in the current study compliment these earlier findings (Table 5).

Although aqueous pesticide concentrations measured in the outflow of each ditch still exceeded established diazinon water quality criteria (0.17 μg/L) (USEPA, 2005), it is not indicative of a lack of ditch efficiency. Most dynamic pesticide reduction systems require combinations of BMPs. Additional measures, such as adding slotted board risers with the ability to retain water, thereby increasing water residence time (and contact time for vegetation) could be easily amended into ditch construction. Also, constructed ditch lengths were limited by available field size of the cooperating landowner; therefore, complete mitigation of pesticides was not a realistic objective. This study was specifically designed to provide preliminary feasibility information for the use of vegetated drainage ditches as a BMP and to assist in determining what ditch length would be necessary to mitigate typical irrigation runoff. Future modeling efforts using data generated from this study will address the necessary ditch length question in greater detail.

Conclusions

The use of vegetative ditches is effective for the mitigation of pesticides, particularly pyrethroids as demonstrated in this project and previous studies (Moore et al., 2001; Cooper et al., 2002a, 2002b; Bennett et al., 2005). Since pyrethroids have shorter environmental half-lives than organochlorines and many of the OP insecticides, there is little concern for pesticide accumulation in ditch water, sediment, and plants. While half-lives were similar among pesticides and different ditches, half-distances among the three ditches indicated the greatest differences. Depending on the pesticide, distances

needed to reduce initial pesticide concentrations by 50% were 2.3 to 2.8 times less in V-vegetated than V-unvegetated. When comparing the V-vegetated to U-vegetated ditches, pesticide half-distances ranged from 6 to 21 times less in V-vegetated than U-vegetated, thus making them most efficient of the three. Although an effective BMP, vegetated ditches should be considered one tool of many available options for mitigation of pesticides, including constructed wetlands, sediment retention ponds, grassed buffers, etc. Site specific needs routinely call for multiple BMPs in sequence to sufficiently address the nonpoint source problem. Additional studies should be conducted utilizing V-vegetated ditch design to further elucidate effectiveness of particular California perennial plant species. In addition, results from this study are being used to develop a model to define optimal ditch length needed for an individual farmer's field based on soil type, pesticide and application rate, cropping pattern, and ditch vegetation.

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