Research Article

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Diazinon reduction and partitioning between water, sediment and vegetation in stormwater runoff mitigation through rice fields

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Abstract

BACKGROUND: Contamination of surface waters by pesticides is a concern in the United States and around the world. Innovative mitigation strategies are needed to remediate this potential environmental contaminant. One potential solution is to divert pesticide-laden drainage or surface water through agricultural rice fields. With a hydroperiod, hydrosoil and hydrophyte (rice), these systems serve essentially as a type of constructed wetland. In both summer and fall experiments, diazinon-amended water was diverted through two rice ponds at the University of Mississippi Field Station. Likewise, a non-vegetated control pond was amended with diazinon-laden water. Water, sediment and plant samples were taken spatially and temporally to determine the distribution of diazinon within systems.

RESULTS: Outflow diazinon concentrations decreased significantly ($P < 0.05$) from inflow in both vegetated ponds for both preharvest and post-harvest experiments. Although sorption to rice plants was minimal in the overall mass distribution of diazinon (1–3%), temporal data indicated that diazinon concentrations reached the outflow sediment of the non-vegetated control twice as fast as in either vegetated (rice) system. In both vegetated systems, sediment diazinon concentrations decreased (77 and 100%) from inflow to outflow, while a decrease of <2% was noted in the non-vegetated control.

CONCLUSIONS: Diversion of pesticide-contaminated water through rice fields demonstrated potential as a low-cost, environmentally efficient mitigation practice. Studies on these systems are continuing to evaluate the optimal chemical retention time for rice field mitigation, as well as diazinon transfer to rice grain seeds that may be used as a food source.
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Keywords: pesticide; BMP; wetland; vegetation

1 INTRODUCTION

In California, diazinon, a dormant-spray organophosphate insecticide, has been frequently used in stone fruit and almond orchards to control certain pest insect populations. Winter rainfall and dormant-season pesticide applications resulted in high diazinon toxicity to *Ceriodaphnia dubia* Richard (water flea) in the San Joaquin[1] and Sacramento Rivers,[2,3] thus requiring innovative solutions for *in situ* remediation. A potential solution to contaminated drainage waters is to utilize adjacent agricultural fields in pesticide remediation. Flooding rice fields (pre- and/or post-harvest) with contaminated drainage could allow ecological processes such as biodegradation and plant sorption to reduce aquatic diazinon concentrations and provide efficient tailwater recovery. This system could be especially beneficial to tailwater recovery and water reuse in California, an agricultural area already severely limited by water restrictions.

Rice production in the United States is limited primarily to three areas: Grand Prairie and Mississippi River Delta of Arkansas, Mississippi, Louisiana and Missouri; Florida–Louisiana–Texas Gulf Coast; and California's Sacramento Valley. Between 1995 and 2005, over 1 806 000 t of rice was produced annually in California.[4] With the proximity of California's rice production to the diazinon surface water contamination problem in the Sacramento Valley, this potential management practice would offer an economical solution to an environmental problem. There are several studies documenting California's concerns with diazinon contamination and impairment of streams and drainage ways.[2,3,5,6] Studies reported that the quantity of diazinon applied within a watershed was highly correlated with the extent of *C. dubia* toxicity.[6] An additional study examined both urban and rural aquatic environments for diazinon.[7] Data suggested that diazinon was consistently (93%) detected in urban streams, and the greatest concentrations of diazinon in stormwater runoff occurred with fields and orchards that received an organophosphate pesticide application.

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Certain best management practices are being utilized to reduce diazinon contamination of surface waters. Surface runoff diazinon losses were reduced after heavy rainfall by the use of a granular diazinon formulation rather than liquid spray.[8] Pesticide retention of over 60% was reported within a vegetative filter strip.[9] A 2001 study reported that, the greater the vegetative cover, the smaller is the loss of diazinon, 37–88% of the applied diazinon remaining as residue in vegetative matter and the root zone.[10] However, few, if any, studies have examined utilizing adjacent rice fields specifically for diazinon load reduction, potential tailwater recovery and water reuse.

This study aimed to examine the potential of using preharvest and senesced (post-harvest) rice fields to help reduce diazinon concentrations and loads prior to reintroduction into surface drainage systems. Rice fields were compared against a fallow, non-vegetated control treatment to determine the influence of vegetation on contaminant reduction potential.

2 EXPERIMENTAL METHODS

Diazinon mitigation studies were conducted at the University of Mississippi Field Station (UMFS), Abbeville, MS, in 2006 and 2007. The first study took place on post-harvest rice ponds in November 2006 (fall), while the second study took place on green, emergent preharvest rice ponds prior to the crop heading out in August 2007 (summer). Ponds with no vegetation were used as controls in February 2007 and October 2007, respectively, for the two studies. Each study had identical experimental designs and agricultural practices for field preparation. The mean temperatures for the summer and fall experiments were 23.8 °C (75 °F) and 6.6 °C (44 °F) respectively. There were no significant temperature differences between controls and treated ponds.

2.1 Experimental design

Two ponds were planted with equal densities of common rice (*Oryza sativa* L.) and amended with a diazinon exposure simulating a stormwater runoff event. These ponds were compared against a non-vegetated (NV) control pond. The NV control pond had no crop, but had the same pond dimensions (60 × 45 m) as one of the cultivated ponds (Fig. 1). Diazinon 475 g L^{-1} EC (Diazinon 4E™) was applied over a 3 h drainage water flow period through a PVC diffuser at the inflow of each pond. Carrier water was spring fed from the UMFS and transported to experimental ponds through a series of ditches. Diazinon was discharged to the inflow diffuser through vinyl 1.27 cm ID tubing from a prepared mixing chamber. The mixing chamber had a maximum concentration of 168 mg diazinon L^{-1}, with a mean target discharge concentration of 0.68 mg L^{-1} throughout the duration of the 3 h exposure. The targeted concentration was determined as 0.05% of the recommended field application rates for a 32.3 ha field (approximately 4.8 L ha^{-1}). The mean inflow rate was 0.003 m^3 s^{-1}, with a total discharge effluent of 32 400 L. Outflow discharge rates peaked at 3–3.5 h post-exposure, with outflow volumes of 12 445 and 11 139 L for vegetated ponds 1 (W1) and 2 (W2) respectively. Inflow and outflow rates for the NV control were similar to those of the vegetated ponds, with identical diffuser and delivery set-ups.

Sampling was stratified longitudinally within each pond. Sampling strategies were matched between vegetated ponds and NV control. Baseline samples were taken pre-amendment to determine background levels of diazinon in pond water, carrier water, pond sediment and rice plant samples. Five sampling transects were established within each pond (Fig. 1): inflow (0 m), 12 (8) m, 24 (16) m, 36 (24) m and outflow (50/28 m). Water, sediment and plant samples taken at each transect were bulk samples, constituting a grab sample approximately every 6 m across the entire width of the pond. Inflow and outflow water samples constituted a sample from the respective inflow and outflow pipes. Inflow and outflow plant and sediment samples were again bulk samples collected at either pond edge. Water samples were taken every 30 min for the first 6 h, and thereafter at 8, 12, 24, 48 and 72 h post-amendment if water was still present. Plant and sediment samples were taken every hour for the first 6 h and at 8, 24, 48 and 72 h post-amendment. Water samples were taken in acetone/hexane-washed 1 L amber bottles, while whole-plant and sediment samples were wrapped in heavy-duty aluminum foil and placed in labeled Ziploc™ freezer bags.

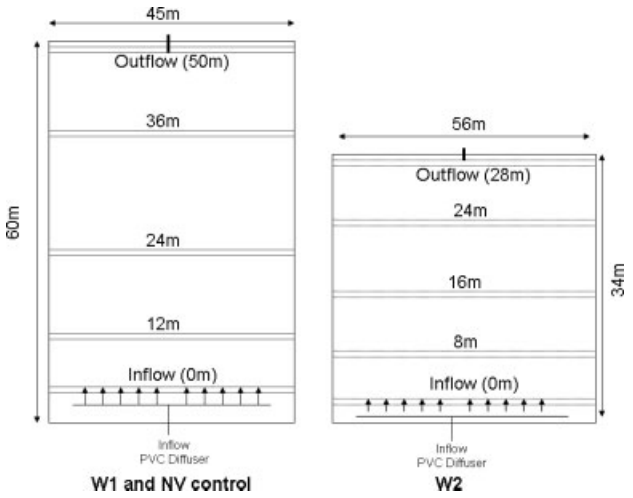

Figure 1. Schematic of transect layout and pond dimensions for vegetated (W1 and W2) and non-vegetated control (NV) ponds. Five transects were sampled on a temporal basis. The PVC inflow diffuser delivered a constant diazinon runoff amendment for 3 h.

2.2 Sample preparation and analysis

Water, plant and sediment samples were transported in ice chests to the USDA Agricultural Research Service National Sedimentation Laboratory in Oxford, MS, for analysis within 2 h of collection. Water samples were fixed with 500 mg of potassium chloride and 25 mL of distilled ethyl acetate, and analytes were prepared within 48 h for analysis.[11–14] Plant tissue and sediment samples were frozen, air dried in a greenhouse to constant weight and ground with a Wiley mill prior to pesticide analysis.[11–13] Quantities of 2 g of plant material and 5 g of sediment were used in pesticide extraction analysis. Further details on sample preparation and extraction techniques can be found in work by Bennett *et al.*,[11] Smith and Cooper[12] and Smith *et al.*[13] All extracts (water, plant and sediment) were analyzed by gas chromatography–electron capture detection using an HP model 6890 gas chromatograph equipped with dual HP 7683 ALS autoinjectors, dual split–splitless inlets, dual capillary columns, a HP Kayak XA chemstation[12] and a main 30 m × 0.25 mm ID (0.25 µm film thickness) HP 5 MS capillary column. Two Agilent electron capture detectors analyzed analytes at 325 °C with UHP nitrogen make-up gas. The retention time for diazinon under these conditions was 11.19 min over a single run time of 20 min. Diazinon runs were calibrated using a multilevel

calibration of three standards, with 100% recovery updated every tenth sample. Diazinon extraction efficiency was greater than 95%, based on fortified samples.

All data were log transformed to meet assumptions of normality and unequal variances set forth by analysis of variance (ANOVA). One-way ANOVAs compared vegetated and non-vegetated treatments. Two-tail Student *t*-tests compared preharvest summer to fall experiments. Pearson's linear correlations and polynomial regressions determined concentration and mass relationships with distance within each pond. Diazinon concentration was calculated as mass by the multiplication of the concentration by the known water volume entering each pond. Diazinon mass and concentrations in sediments and plants comprised the total sorption value. Often, the diazinon concentration and load of a respective treatment transect through time were summed to provide an overall concentration or load for that transect. This overall diazinon result allowed a direct comparison with overall inflow concentrations or loads for each respective treatment. Alpha was set at 0.05 for all analyses except where stated otherwise.

3 RESULTS

3.1 Preharvest experiment (summer)

An ANOVA comparison of inflow water concentrations between treatments showed no significant difference ($F = 0.8580$; $P = 0.4131$) between vegetated and NV control amendments. Vegetated replicates, W1 and W2, reduced overall outflow diazinon water concentrations by 67.5 and 93.9% respectively, while the NV control treatment reduced outflow concentrations by 47.9% (Fig. 2). Outflow concentrations varied through time for all treatments (Fig. 2), with all outflow concentrations for all treatments significantly ($P \leq 0.1$) lower than inflow concentrations (1 h – outflow).

Mean overall sediment diazinon concentrations decreased by 77.2 and 100% in both vegetated treatments, while only decreasing 1.5% in the NV control treatment. Furthermore, diazinon concentrations in the vegetated replicates were not detected in outflow sediments until 8 h post-storm amendment, while diazinon was detected in NV control outflow sediments 4 h post-amendment. The NV control treatment had significantly higher overall mean mass ($32.5 \pm 6.9\ \mu g\ kg^{-1}$) of sorbed diazinon in sediments than both W1 and W2 ($8.5 \pm 3.1\ \mu g\ kg^{-1}$ and $3.1 \pm 5.8\ \mu g\ kg^{-1}$).

Although total diazinon mass accumulated by plant tissue over time did not differ between vegetated treatments ($t = 0.210$; $P = 0.8357$), the spatial patterns of diazinon retention by plant tissue varied between the two replicates.

Table 1 presents the relative percentage mass contributions that the water column, sediment and plant compartments made to diazinon mitigation in the preharvest summer experiment. Throughout the experiment's duration, the water column had >90% of the diazinon mass for the vegetated replicates. This mass percentage in the water column decreased rapidly in the NV control treatment. The rapid decrease in water diazinon mass was concomitant with an increased percentage of sediment diazinon mass. The percentage of diazinon in sediment remained low during the amendment (0–4 h, <7%) for all treatments. Overall, mean sediment diazinon mass percentages were significantly ($F = 5.786$; $P = 0.003$) higher in the NV control treatment ($62\% \pm 11.1$) than in both W1 ($8.5\% \pm 4.5$) and W2 ($3.1\% \pm 1.5$). In terms of relative plant mass contribution, <1% of relative diazinon mass had accumulated in plant tissue for each vegetated replicate (Table 1).

3.2 Post-harvest experiment (fall)

ANOVA comparisons of inflow concentrations for the fall experiment showed no overall significant differences between treatments ($F = 0.172$; $P = 0.843$). Overall, there were no significant differences in aqueous outflow concentrations between the vegetated and NV control.

Sediment diazinon concentrations at the inflow were similar among treatments but varied considerably with distance from

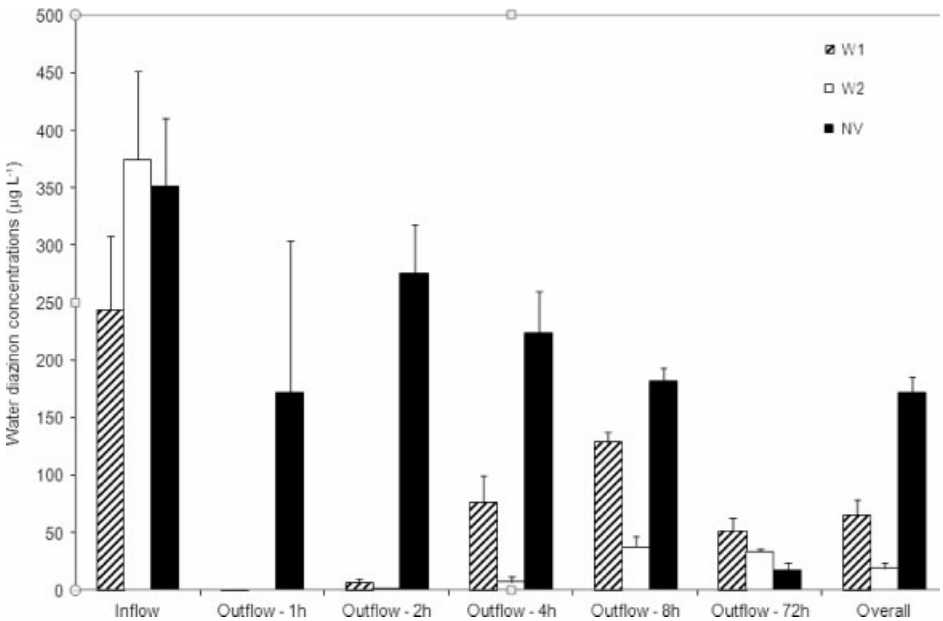

Figure 2. Average aqueous inflow, temporal outflow and overall diazinon concentrations ($\mu g\ L^{-1}$) post-amendment for the duration of the summer experiment. W1, W2 and NV are the two vegetated treatments and the non-vegetated treatment respectively. The overall concentration is the mean concentration of all spatial and temporal samples for the outflow of each respective treatment.

Table 1. Relative percentage mass contributions of water, sediment and plant compartments in summer to diazinon mitigation. Percentages are based on the average contribution of mass throughout the entire wetland cell for the respective time period. W1, W2 and NV are the two vegetated treatments and the non-vegetated treatment respectively

	Relative % water			Relative % sediment			Relative % plant		
Time (h)	W1	W2	NV	W1	W2	NV	W1	W2	NV
1	99.3	99.1	98.	0.4	0.9	1.2	0.2	0.04	–
2	99.3	99.1	97.6	0.5	0.6	2.3	0.1	0.2	–
3	98.1	99.5	95.7	1.4	0.03	4.2	0.5	0.4	–
4	99.4	99	93.9	0.3	0.5	6.1	0.3	0.4	–
5	97	97.2	90.6	2.5	1.7	9.4	0.5	1.0	–
6	98.9	99.5	66.5	0.7	0.3	33.5	0.4	0.2	–
8	94.3	97.1	71.1	5.1	2.3	28.8	0.5	0.6	–
24	93.8	90.1	46.4	5.4	8.4	53.6	0.8	1.4	–
48	55.8	98.8	9.5	41.9	0.8	90.5	2.3	0.3	–
72	70.9	82.2	9.1	27.1	15.5	90.9	1.8	2.2	–
Average	90.7	96.2	67.9	8.5	3.1	62.1	0.7	0.7	–
SE	4.7	1.78	11.1	4.5	1.58	11.1	0.23	0.21	–

Table 2. Relative percentage mass contributions of water, sediment and plant compartments in the fall to diazinon mitigation. Percentages are based on the average contribution of mass throughout the entire wetland cell for the respective time period. W1, W2 and NV are the two vegetated treatments and the non-vegetated treatment respectively

	Relative % water			Relative % sediment			Relative % plant		
Time (h)	W1	W2	NV	W1	W2	NV	W1	W2	NV
1	49.9	86.5	58.1	47.8	13.4	41.8	2.2	0	–
2	54.7	70.4	46.2	44	26.6	53.7	1.0	3.0	–
3	55.1	68.1	63.9	42.9	29	36.1	1.8	2.8	–
4	69.2	57.3	26.6	28.5	39.7	73.3	2.3	2.9	–
5	37.4	57.1	50.7	60.7	37.7	49.2	1.8	5.0	–
6	59.2	48.6	19.7	36.8	46.9	80.3	3.8	4.5	–
8	51.1	61.1	44.3	44.6	32.9	55.6	4.3	5.9	–
48	47.5	55.6	2.9	46.6	40	97	5.8	4.3	–
72	46.8	37.8	4.7	51.2	58.5	95.3	1.9	3.7	–
Average	52.4	60.3	32.7	44.8	36.1	67.7	2.8	3.6	–
SE	2.8	4.4	7.4	2.8	4.1	7.4	0.5	0.5	–

the diffuser. Both vegetated replicate sediment concentrations decreased with distance from the diffuser (W1 = 54%; W2 = 71%), while the NV control treatment had a 29.7% increase in sediment diazinon concentration. Mean sediment diazinon concentrations at the outflow (Fig. 3) show detection of diazinon in the sediment 3–4 h earlier in the NV control treatment than in vegetated treatments. A mass comparison of sediment diazinon showed that the NV control treatment had a significantly higher diazinon accumulation in sediments ($F = 7.69$; $P = 2.6 \times 10^{-3}$) than vegetated replicates (Table 2). Similar to the summer experiment, plant tissue diazinon concentrations in the fall were variable between vegetated replicates. There were no significant differences in the plant tissue diazinon mass accumulated between vegetated replicates ($t = -0.617$; $P = 0.495$).

Table 2 provides the relative mass percentage contribution of water, sediment and plant to diazinon mitigation in the fall experiment. Relative water mass percentages on average were 50%, with significantly ($F = 5.67$; $P = 9.7 \times 10^{-3}$) different percentages occurring between vegetated (W1 52.4% ± 2.8; W2 60.3% ± 4.4) and NV control treatments (32.7% ± 7.4). Sediment played a significant role in mass accumulation for both vegetated and NV control treatments. NV control treatment sediment (67.7% ± 7.4) played a significantly greater ($F = 7.70$; $P = 2.6 \times 10^{-3}$) role in diazinon accumulation than either vegetated replicate (W1 32.7% ± 7.4; W2 44.8% ± 2.8). Plant

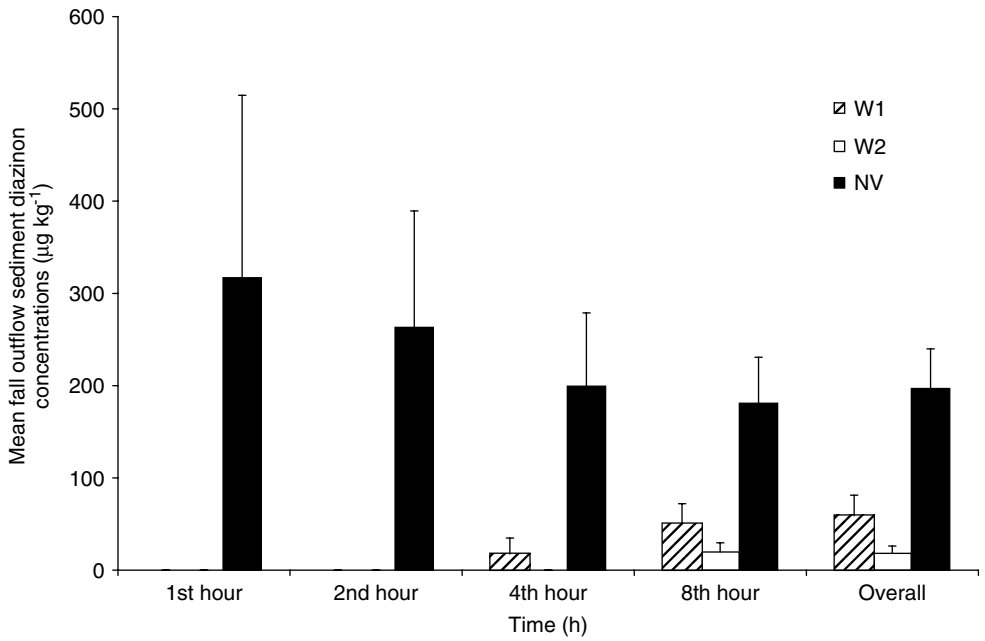

Figure 3. Mean outflow fall sediment diazinon concentrations (µg kg^{-1}) for vegetated (W1 and W2) and non-vegetated treatments. The overall concentration is the mean concentration of all spatial and temporal samples for each respective treatment.

tissue contributed <3% to the overall mass reduction in diazinon. There was also no clear change in water column or sediment mass between the duration of diazinon amendment (0–4 h) and the end of the sampling period (72 h).

3.3 Preharvest summer versus fall

A comparison of inflow diazinon concentrations between summer ($297 \pm 48\ \mu g\ L^{-1}$) and fall ($189 \pm 18\ \mu g\ L^{-1}$) experiments showed significantly higher concentrations over the summer experiment ($t = 2.54$; $P = 0.015$). This translated into significantly higher mass loading in the summer versus fall experiment ($t = 6.05$; $P < 1 \times 10^{-4}$). Percentage diazinon associated with the water column was distinctly higher when comparing Table 1 and Table 2 for the summer and fall experiments respectively. Interestingly, at the height of the exposure, there were no significant differences in water diazinon concentrations between summer ($202 \pm 28\ \mu g\ L^{-1}$) and fall ($179 \pm 22\ \mu g\ L^{-1}$) experiments ($t = 0.597$; $P = 0.553$). There were also no significant differences in outflow concentrations between summer and fall experiments ($t = -1.51$; $P = 0.135$). Thus, higher summer inflow concentrations were reduced to fall outflow concentrations within the allotted treatment space. In the sediment compartment, diazinon mass percentages were significantly higher in the fall experiment. Sediment diazinon mass was significantly higher in the fall versus summer experiment ($t = -5.69$; $P < 1 \times 10^{-4}$). Sediment concentrations in the fall were an order of magnitude higher (W1 $= 102 \pm 29\ \mu g\ kg^{-1}$; W2 $= 71 \pm 15\ \mu g\ kg^{-1}$; NV $= 200 \pm 30\ \mu g\ kg^{-1}$) than in the summer experiment (W1 $= 7 \pm 1.8\ \mu g\ kg^{-1}$; W2 $= 3 \pm 1\ \mu g\ kg^{-1}$; NV $= 24 \pm 5.2\ \mu g\ kg^{-1}$), even though the summer experiment initially had significantly higher inflow concentrations. Independent soil core samples confirmed this pattern of diazinon accumulation, whereby fall control sediments ($19.6 \pm 2.9\ \mu g\ kg^{-1}$) were significantly higher ($P < 0.05$) than fall vegetated sediments ($5.41 \pm 1.12\ \mu g\ kg^{-1}$), which were greater than summer control ($9.98 \pm 1.17\ \mu g\ kg^{-1}$) and summer vegetated ($8.86 \pm 1.45\ \mu g\ kg^{-1}$) sediments. Accumulated relative plant diazinon mass was also significantly higher in the fall experiment than in the summer experiment ($t = -5.88$; $P = 0.001$). The concentration of diazinon in plant tissue was similarly an order of magnitude higher in the fall ($262 \pm 77\ \mu g\ kg^{-1}$) than in the summer ($40.2 \pm 8\ \mu g\ kg^{-1}$).

4 DISCUSSION

Through drainage water flow, the present study highlighted the use of cultivated rice ponds in both preharvest and post-harvest stages in reducing diazinon concentrations and loads prior to outflows reaching receiving waters. The reduction in water column concentrations for rice over the summer (67–93%) and fall (48–56%) documents the potential for using rice paddies as at least a partial solution in pesticide mitigation in surface waters.

The mere presence of diazinon in aquatic systems is not a compelling argument for remediation. Of greater concern is the level at which diazinon is present in such receiving systems. Many studies have focused on the ecotoxicological impact of aqueous and sediment-bound diazinon relative to fish and benthic macroinvertebrates. A 2003 study conducted on agricultural drainage water from the Salinas River (California) reported acute toxicity due in part to diazinon and other organophosphorus insecticides.[14] Once diazinon is detected in the environment, a variety of factors affect its fate and degradation. With regard to aqueous environments, pH is one of the controlling factors influencing hydrolytic and photolytic processes involved in degradation of diazinon.[15] Diazinon has been reported to degrade more rapidly under acidic conditions than in neutral or alkaline solutions.[16] Acidic conditions (pH 3.5–5.0) were present in both water and sediment of the present study location, as measured by an Oakton pH Tester 2 instrument. It is critically important to realize that both vegetated and NV systems acted similarly as biogeochemical transformers of diazinon. However, the presence of vegetation would provide a greater surface area for sorption (absorption and adsorption) and microbial attachment, and potentially a more diverse microbial community. The fact that a greater amount of diazinon (3%) was associated with senescent vegetation as opposed to green emergent vegetation (<1%) was puzzling. This could be due to the greater surface area of above-ground senescent rice tissue suspended in the water column available for sorption.

Interestingly, sediment played a significant role in diazinon accumulation in the NV control, a system devoid of vegetation and vegetative organic matter. Soils of both vegetated and NV treatments were high clay (>85%) soils with high organic carbon levels. With K_{oc} values of 1007–1842, diazinon has a low to moderate tendency to remain bound to soil and sediments, which is mainly attributed to organic matter content.[17,18] While diazinon soil half-lives range from 3 to 54 days, a range of 3–13 days is most representative of actual field conditions.[17] Sediment half-life studies conducted in Orange County, California, reported aerobic sediment half-lives ranging from 14.4 to 21.1 days, while anaerobic conditions increased half-lives from 23.7 to 31.7 days.[19] In both aqueous and sediment matrices, microbial activity is reported to be a significant factor in diazinon degradation. Microbial metabolism is also dependent on soil moisture.[20,21] However, no measurements of microbial metabolism were made as part of the present study. Diazinon degradation was a result of microbial metabolism and assimilation, hydrolysis, photolysis and potentially volatilization. High temperatures and low pH over the summer, preharvest experiment could have significantly contributed to the increased reduction in diazinon within the vegetation systems.

While diazinon degradation can take place in the water and soil environment, the objective is to capture the pesticide before it enters aquatic receiving systems. The use of rice paddies as tailwater recovery or pollutant mitigation systems is a viable path to this goal. Regardless of vegetation, diazinon concentrations and loads were significantly decreased prior to outflow effluent reaching receiving waters. The presence of vegetation contributed significantly to decreased diazinon concentrations, although a mass balance shows that the amount of diazinon in plant material was small (1–3%).

Various treatment scenarios have been investigated for the ability to intercept diazinon. Many of these successful treatment practices utilize vegetation in pesticide mitigation. In California, peak pesticide concentration and mass from first-flush runoff was significantly reduced when treatments utilized a vegetative ground cover versus bare soil treatments.[22] Another California study on the persistence of diazinon in water concluded that any management practice aimed at increasing the holding or hydraulic retention time of pesticide-contaminated water (e.g. diversion through wetland systems) may effectively reduce potential damage to aquatic receiving systems.[23] Following this suggestion, a comparative study was conducted on constructed agricultural drainage ditches, with and without vegetation.[24] Study results indicated that the ditch distance required for the

diazinon concentration to be decreased to half its original dose (i.e. half-distance) was roughly 3 times less in a vegetated ditch than in a non-vegetated ditch, demonstrating the benefits of vegetation in pesticide mitigation.[24] Outflow aqueous diazinon concentrations from the present study support the findings of vegetation's mitigation capacity for pesticides associated with runoff, as diazinon was detected in the outflow of the non-vegetated control before it was detected in either vegetated rice pond outflow.

In addition to drainage ditches as mitigation tools, constructed wetlands have also been utilized to remediate diazinon runoff.[25] Diazinon amended runoff was filtered through a 180 m (length) × 30 m (width) series of two constructed wetland cells. Overall results indicated that 43% of the measured diazinon mass was associated with plant material, while 23 and 34% of the measured diazinon mass was associated with the sediment and aqueous phases respectively.[25] Present-study plant sorption data with rice ponds indicated much less plant-associated pesticide sorption. One potential reason for this difference is the increased water depth in the constructed wetland (often >50 cm) as opposed to a minimal (≤10 cm) water depth in rice ponds. Increased water depth resulted in increased detention time, enhancing available contact time for pesticide sorption in constructed wetland scenarios. In the present rice field experiment, only a small fraction of the actual plant was exposed to diazinon-amended water, which may have translated into little plant pesticide sorption. While plant sorption results from the present study may lead some to believe phytoremediation is not productive with diazinon runoff, recent studies indicate otherwise.[9,10]

5 CONCLUSIONS

Diverting pesticide-laden drainage water into rice fields bordering aquatic receiving systems has been proposed as an alternative best management practice. Studies conducted during both summer (early rice-growing stage) and fall (senesced rice stage) indicate the potential for diazinon mitigation when compared with diverting pesticide water through a non-vegetated field only. Further studies are needed to elicit the necessary retention time and water depth for optimal pesticide sorption within the rice field system. Additional studies are also under way to monitor diazinon transfer to rice seeds when exposure occurs during the early growing stage; however, as diazinon is oxidatively metabolized in plants, absorption of small amounts should probably be of minor concern for food safety.

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