Environ. Sci. Technol. **2003**, *37,* 2139−2144

Fate and Effects of Azinphos-Methyl in a Flow-Through Wetland in South Africa

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Our knowledge about the effectiveness of constructed wetlands in retaining agricultural nonpoint-source pesticide pollution is limited. A 0.44-ha vegetated wetland built along a tributary of the Lourens River, Western Cape, South Africa, was studied to ascertain the retention, fate, and effects of spray drift-borne azinphos-methyl (AZP). Composite water samples taken at the inlet and outlet during five spray drift trials in summer 2000 and 2001 revealed an overall reduction of AZP levels by 90 ± 1% and a retention of AZP mass by 61 ± 5%. Samples were collected at the inlet, outlet, and four platforms within the wetland to determine the fate and effect of AZP in the wetland after direct spray drift deposition in the tributary 200 m upstream of the inlet. Peak concentrations of AZP decreased, and the duration of exposure increased from inlet (0.73 μg/L; 9 h) via platforms 1 and 4 to outlet (0.08 μg/L; 16 h). AZP sorbed to plants or plant surfaces, leading to a peak concentration of 6.8 μg/kg dw. The living plant biomass accounted for 10.5% of the AZP mass initially retained in the wetland, indicating processes such as volatilization, photolysis, hydrolysis, or metabolic degradation as being very important. AZP was not detected in sediments. Water samples taken along two 10-m transects situated perpendicular to the shore indicated a homogeneous horizontal distribution of the pesticide: 0.23±0.02 and 0.14±0.04 μg/L (n = 5), respectively. Both Copepoda (p = 0.019) and Cladocera (p = 0.027) decreased significantly 6 h postdeposition and remained at reduced densities for at least 7 d. In parallel, the chlorophyll *a* concentration showed an increase, although not significant, within 6 h of spray deposition. The study highlights the potential of constructed wetlands as a risk-mitigation strategy for spray drift-related pesticide pollution.

Introduction

It has recently been shown that constructed wetlands have the potential to retain runoff-related insecticide pollution, preventing it from entering downstream aquatic habitats (*1*, *2*). The implementation of retention ponds in agricultural watersheds was mentioned by Scott et al. (*3*) as one strategy to reduce the amount and toxicity of runoff-related insecticide pollution discharging into estuaries. The usefulness of aquatic plants for removal of insecticides from water has been shown in an indoor microcosm study (*4*), and the effects of the organophosphate phorate have been assessed using littoral mesocosms in South Dakota wetlands (*5*). However, information about the fate or effects of spray drift-borne insecticide input in constructed wetlands is limited.

Processes important for removal of nonpoint-source pesticide pollution in wetlands may include adsorption, decomposition, hydrolysis, microbial metabolism, photolysis, and volatilization (*6*). The macrophytes present in the wetland may play an important role in providing an increased surface area for sorption as well as for microbial activity (*7*). Furthermore, they may contribute directly to metabolism (*8*).

Spray drift is an important route for nonpoint-source pesticide pollution of aquatic habitats (*9*, *10*). Specifically, orchard applications result in a large amount of drift due to small droplet size and the trajectory of release (*11*). Spray drift has been shown to be a significant route of insecticide entry into tributaries of the Lourens River in South Africa (*12*).

The organophosphate pesticide azinphos-methyl (AZP) [*O*,*O*-dimethyl-*s*-[(4-oxo-1,2,3-benzotriazine-3(4*H*)-yl)methyl)]phosphorodithioate], in comparison to other insecticides, has a relatively low K_{OC} of 1000 L/kg; a high water solubility of 29 mg/L at 25 °C (*13*); and a Henry's law constant of 9.5 × 10^{-11} atm m^3 mol^{-1} (*14*). It has been shown to persist in pond water with a half-life of about 2.4 d (*15*). AZP is frequently applied to apple, pear, and plum orchards in the catchment and has been regularly detected following runoff and spray drift activity in the Lourens River and its tributaries (*12*, *16*). The estimated total application in fruit orchards of the Western Cape is 52 000 kg of active ingredient per year. It is also one of the most heavily applied pesticides in the United States, and in 1997 almost 950 000 kg of active ingredient was applied throughout the entire country (*17*).

To minimize the input of sediment into the Lourens River, a 0.44-ha vegetated siltation pond was constructed in 1991 along one of the tributaries. Its effectiveness in retaining runoff-related insecticide input was the subject of an earlier study based on inlet and outlet measurements (*1*). The present study describes the retention, the fate, and the biological effects of spray drift-borne aqueous-phase AZP in this flow-through wetland.

Materials and Methods

Study Region and Study Period. The study area is located in the catchment of the Lourens River, which originates at an altitude of 1080 m in fynbos, a naturally sclerophyllous vegetation, and flows in a southwesterly direction for approximately 20 km before discharging into False Bay at the Strand (34°06′ S, 18°48′ E). The Lourens River has a total catchment area of 92 km^2 and receives an annual mean rainfall of 915 mm. Approximately 87% of its 35 × 10^6 m^3 mean annual discharge occurs during the winter months between April and October (*18*), as is characteristic of the region's Mediterranean climate.

The 400-ha orchard area in the middle reaches of the Lourens River is mainly covered by pears, plums, and apples. The pesticide application period in the study area's orchards begins in early August and continues until the end of March. Organophosphorus (OP) insecticides, such as AZP and

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10.1021/es026029f CCC: $25.00
Published on Web 04/12/2003

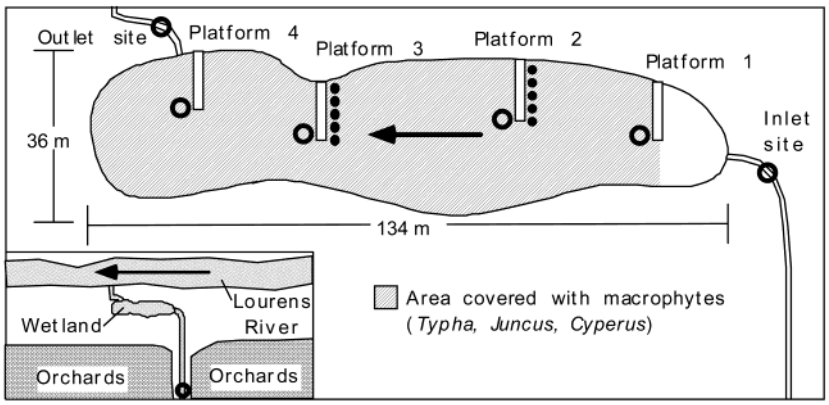

FIGURE 1. Schematic view of the wetland showing size, vegetation coverage, and six main sampling sites (large circles) as well as sites for transect sampling at platforms 2 and 3 (small circles). The inset (not to scale) shows the orientation of the wetland to the Lourens River and to the orchard areas as well as the additional site (spray deposition site, circle) about 200 m upstream of the wetland. The arrow indicates the flow direction.

chlorpyrifos, are applied quite frequently to pears and plums between October and February. Endosulfan is applied mainly in apple orchards (*19*).

The constructed wetland studied in the present investigation is located along one of the tributaries 15 m before its entry into the Lourens River (*1*). This tributary has an average width and depth of 0.89 m × 0.30 m, has a current velocity of approximately 0.1 m/s in summer, and approximately 50% of its surface is covered with emergent vegetation. It has a total length of approximately 1.8 km and flows from a dam through a forest area for 800 m and then into pasture land for 400 m before flowing through the orchard area for a further 600 m. Average discharge in the tributary is 0.03 m^3/s in January and 0.32 m^3/s in July. The discharge during the study period was 0.043 ± 0.002 m^3/s; all studies were done during the dry summer season.

The retention of spray drift-borne AZP in the wetland was derived from measurements undertaken in January and February 2000 and 2001 during the dry summer periods with average monthly rainfall <25 mm. A detailed study of the fate and transport of AZP was done on February 5, 2001, during the last pesticide application of the spraying season 2000/2001, and the resulting biological effects were monitored until May 17, 2001. All studies were done while AZP was being applied in the conventional manner to fruit orchards bordering the tributary, which flows through the wetland. Such application resulted in spray deposition into the tributary about 200 m upstream of the wetland, as described in an earlier study (*12*).

Description of the Wetland. The wetland was built in 1991 to prevent nonpoint-source input of suspended sediment into the Lourens River (*1*). The catchment area of the wetland comprises 15 ha of pear and plum orchards, 10 ha of pasture land, and 18 ha of forest. The wetland has a length of 134 m and a width of 36 m, giving a total area of 0.44 ha (Figure 1). The water depth varied between 0.3 and 1 m in different parts of the wetland during the study period; this remarkable shallowness, in contrast to the initial depth of up to 1.5 m at time of construction, indicates the extent of particle sedimentation within the wetland. Using the summer flow rates of the tributary, the theoretical water-renewal time of the wetland is 27 h. The first 15 m of the wetland are free of vegetation, and the remaining area is covered mainly with *Typha capensis* Rohrb. (80% coverage), *Juncus kraussii* Hochst (15% coverage), and *Cyperus dives* Delile (5% coverage).

Sampling Procedure and Analysis. Six main sampling stations were used. The inlet and outlet sampling was carried out in the tributary above and below the wetland. Four platforms raised above the water surface at distances of 17, 45, 85, and 110 m from the inlet were used as sampling stations in the wetland to ensure that sampling would not cause unnecessary damage to the macrophytes and/or sediments (Figure 1). An additional seventh site, representing the spray drift deposition site, was located in the tributary about 200 m upstream of the wetland inlet.

TABLE 1. Mean (± SE; $n = 14$) of Various Parameters Measured at the Wetland Inlet between January and May

parameter	mean (± SE)
discharge (m^3/s)	0.041 ± 0.004
total suspended solids (mg/L)	43.7 ± 5.2
temperature (°C)	19.3 ± 0.8
dissolved oxygen (mg/L)	8.6 ± 0.4
pH	7.1 ± 0.1
orthophosphate (mg/L)	0.16 ± 0.02
nitrate (mg/L)	3.6 ± 0.4

Discharge of the tributary was calculated, on the basis of standard formulas (*20*), from velocity measurements along cross-sectional profiles. Total suspended solids (TSS; ±0.1 mg/L) were measured using a turbidity meter (Dr. Lange, Duesseldorf, Germany). Turbidity readings were calibrated as described by Gippel (*22*). Temperature (±0.1 °C), dissolved oxygen (±0.1 mg/L), and pH (±0.1) were measured with portable electronic meters (WTW, Weilheim, Germany). Orthophosphate (molybdenum blue; ±0.01 mg/L) and nitrate (dimethylphenol; ±0.1 mg/L) levels were measured with photometric test kits from Merck (Ingelheim, Germany). The mean values ($n = 14$ between January and May) of various parameters are summarized in Table 1. Details of the retention of nutrients and suspended particles under different hydrological conditions were described in an earlier study (*1*).

To describe the overall AZP retention in the wetland, composite samples were taken in 3-L glass jars at the inlet and the outlet station during five spray drift trials on the following dates: January 28, February 2, and 14, 2000, as well as on January 12 and February 5, 2001. Average discharge through the wetland was 0.043 ± 0.002 m^3/s during these five trials. A previous experiment with the introduction of a tracer dye was undertaken during similar hydrological conditions (discharge: 0.042 m^3/s) to determine the starting point and duration of sampling intervals at the inlet and outlet station of the wetland used for the pesticide sampling. The tracer Rhodamin B was introduced, at a rate of 15 g dissolved in 200 mL of water, into the tributary at a site 200 m above the wetland where spray drift deposition occurs during application to the adjacent pears. Water samples were

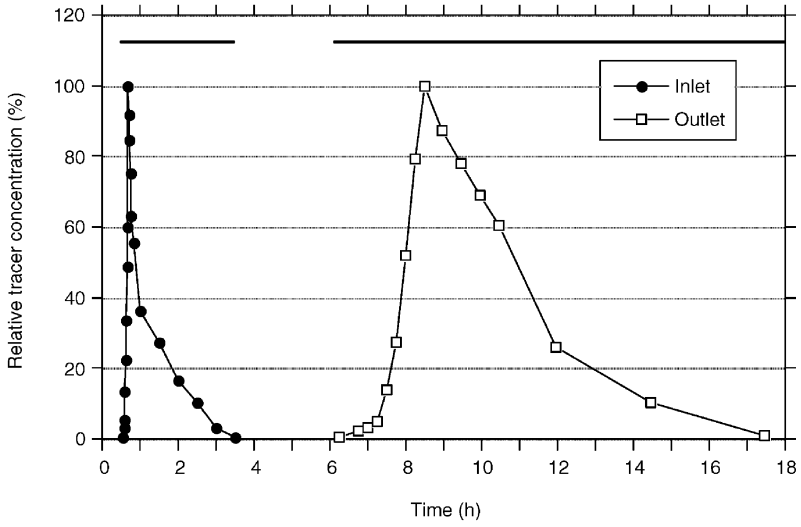

FIGURE 2. Time course of relative concentration of a tracer dye (Rhodamin B) at the two sampling stations at the inlet and outlet of the constructed wetland. The maximum concentration at both stations was set at 100%. The bold horizontal lines indicate the sampling intervals used.

taken at various time intervals at the inlet and outlet stations. Samples were stored at 4 °C in the darkness. They were analyzed within 24 h using a fluorescence photometer (TD700; GAT Bremerhaven, Germany) with a detection limit of 2.5 ng/L. The concentrations of the tracer are expressed in relative terms because degradation in the presence of sunlight decreases the absolute concentrations of Rhodamin B with a field half-life of approximately 3 h (*21*). In accordance with the results of this tracer experiment (Figure 2), 3-h composite sampling commenced at the inlet station 30 min after introduction of spray drift 200 m above the wetland, while 12-h composite sampling commenced at the outlet station 6 h after introduction of spray drift. This sampling design potentially enabled collection of the same parcel of water with >95% of the pesticide mass at both stations via composite samples. Each composite sample (1 L) was combined from 10 subsamples (each 100 mL) taken every 20 min at the wetland inlet and every 80 min at the wetland outlet. As the discharge did not vary over time and showed only very little variation between the five spray drift trials, the concentrations obtained from the composite samples were used for comparisons between inlet and outlet and (along with the discharge data) for calculating AZP mass. However, it should be noted that these concentrations do not represent peak levels but represent average levels based on the 10 subsamples from the composite sampling. As no time-dependent hydraulic residence time distributions were defined, the direct comparison of concentrations between inlet and outled should be interpreted with care.

A detailed description of AZP distribution and fate over time was enabled during the spray drift trial on February 5, 2001, by taking discrete (separate) water samples at the inlet and outlet stations and water, plant (rinsed subsurface sections of living *T. capensis* shoots without roots including the attached biofilm), and sediment (upper 2 cm) samples at platforms 1 and 4 between time zero at the commencement of spray deposition and day 7 following spray drift. Timing of sampling depended on the expected duration of exposure (i.e., shorter time intervals were used at the inlet station in comparison to the outlet station). In addition, four replicate water samples were taken at the spray deposition site about 200 m upstream of the wetland inlet, according to methods described in Schulz et al. (*12*). For a description of the horizontal distribution of the pesticide in the vegetated wetland area, five water samples were taken at each of the platforms 2 and 3, at 2-m intervals along 10-m transects perpendicular to the wetland shore (Figure 1). The transect used at platform 3 covered more than half of the width of the wetland at this site.

A mass balance was established to estimate the total amount of AZP sorbed to plants or plant surfaces on the basis of analytical data obtained from the plant samples related to the estimated average inlet mass. The mean (±SE) dry biomass of subsurface shoots sections of living *T. capensis* excluding root material was 1086 ± 57 g/m^2, and the moisture content was $92 \pm 0.5\%$ ($n = 6$). Only 50% of the total area covered by plants was considered in the mass balance as AZP was detected in plants taken at platform 1 but not in those from platform 4. The resulting total biomass was 2280 kg dry weight (dw).

Water samples (500−900 mL) were solid-phase extracted (SPE) within 10 h after sampling using C18 columns (Chromabond). The columns were air-dried for 30 min and kept at −18 °C until analysis. Analyses were performed at the Forensic Chemistry Laboratory of the Department of National Health, Cape Town. Sediment, plant, and water samples were analyzed as documented in Bennett et al. (*23*) and Schulz et al. (*19*). Measurements were done using gas chromatography (HP 5890's) fitted with standard HP electron-capture, nitrogen−phosphorus, and flame-photometric detectors. Concentrations for sediments and plants were expressed as micrograms per kilograms dry weight (μg/kg dw). Identity of AZP was confirmed by matching retention times on three different stationary phases. Method validation employed water matrixes that were found to have no detectable levels of the investigated pesticides and consisted of spiking water at eight spiking levels over the range of concentrations found in the actual samples. Overall mean recoveries were between 79 and 106%. For quality control, a matrix blank was analyzed with each extraction set. The investigated pesticides were never detected in matrix blanks. The detection limits were 0.02 μg/L for water, 0.2 μg/kg for sediments, and 0.2 μg/kg for plants.

Pesticide Effects. Four replicate zooplankton samples were taken at platform 3 at various times between day −7 and day 7 of the spray drift trial on February 5, 2001. Each water sample (9 L) was filtered through an 80-μm mesh, and the zooplankton was preserved in 70% EtOH. Numbers of copepods and cladocerans were counted. Chlorophyll *a* was measured in four replicate samples taken at platform 3 at various times in conjunction with the zooplankton sampling. Volumes of 750 mL each were filtered in the darkness through

TABLE 2. Concentration and Mass of AZP during Spray Drift at the Inlet and Outlet of the Constructed Wetland[a]

	AZP concentration			AZP mass		
trial	inlet (μg/L)	outlet (μg/L)	reduction (%)	inlet (mg)	outlet (mg)	retention (%)
1	0.41	0.04	90	159.4	62.2	61
2	0.36	0.04	89	186.6	82.9	56
3	0.34	0.02	94	143.2	33.7	77
4	0.51	0.04	92	225.8	70.8	69
5	0.27	0.04	85	145.8	86.4	41
avg ($\pm$ SE; $n = 5$)	0.4 $\pm$ 0.03	0.04 $\pm$ 0.003	90 $\pm$ 1	172.2 $\pm$ 12.6	67.2 $\pm$ 7.7	61 $\pm$ 5

[a] Inlet values are from 3-h composite samples ($n = 1$), whereas those for the outlet are from 12-h composite samples ($n = 1$) during spray drift trials performed in January and February 2000 and 2001.

glass fiber filters immediately following sampling using a vacuum pump. Filters were stored on ice until acetone extraction and analyzed according to methods described in Wetzel and Likens (*24*).

Results and Discussion

Overall Pesticide Retention. Concentrations of AZP derived from composite samples taken during five independent spray deposition trials averaged 0.40 $\pm$ 0.03 μg/L at the inlet and 0.04 $\pm$ 0.003 μg/L at the outlet station (Table 2). Average reduction was 90 $\pm$ 1%. The same calculations were undertaken with the loading rates, giving a retention of AZP mass of 61 $\pm$ 5% from 172.2 $\pm$ 12.6 mg at the inlet to 67.2 $\pm$ 7.7 mg at the outlet (Table 2).

The retention of insecticides in constructed wetlands has so far been described in a few studies dealing with runoff scenarios. According to previous results obtained during a runoff event in the same wetland, AZP, chlorpyrifos, and endosulfan at inlet peak levels of 0.85, 0.02, and 0.2 μg/L, repectively, were retained at rates >77% (*1*). Approximately 47−65% by mass of chlorpyrifos was retained in plants and sediments within the first 30−36 m of wetland mesocosms in Mississippi (*2*). A decrease of λ-cyhalothrin from 460 to <0.02 μg/L within a 50-m stretch was predicted from studies undertaken in a slow-flowing vegetated agricultural drainage ditch (*25*). For the herbicide atrazine, a removal rate between 25 and 95% was demonstrated in the Des Plaines Wetland cells (*26*), while reduction of atrazine concentration in water was 11−14% in 230-m flow-through wetland mesocosms in Minnesota (*27*).

In summary, the wetland studied in the present investigation has very positive effects in reducing the concentration and mass of aqueous-phase AZP entering the surface water via spray drift. Spray deposition resulting in short-term peak AZP levels of 2.5 $\pm$ 2.6 μg/L at the deposition site 200 m upstream of the wetland inlet was comparable with results (1.7 $\pm$ 0.2 μg/L) obtained from previous spray drift studies undertaken in the same catchment in 1999 (*12*).

On the other hand, the removal of water-borne toxicants by wetlands could potentially lead to unwanted long-term accumulation of chemicals, as documented for natural wetland areas (*28*). The effectiveness of pesticide retention in wetlands may differ with season due to fluctuations in water temperature and flow as well as wetland abiotic and biotic conditions (*29*). There is still a need for further studies to demonstrate the long-term fate of insecticides in constructed wetlands.

Pesticide Fate and Distribution. During the spray drift trial on February 5, 2001, AZP was present in the water at the inlet between 40 min and 9 h and at the outlet between 8 h and 1 d postdeposition, and similar durations of detection applied to platforms 1 and 4 (Figure 3). Measured peak concentrations decreased from the inlet (0.73 μg/L) via platforms 1 (0.65 μg/L) and 4 (0.12 μg/L) to the outlet (0.08 μg/L). Plants taken at platform 1 contained about 2 μg/kg AZP at 0 h, 3 h, 12 h, and 7 d and peaked at 6.8 μg/kg at 6 h after commencement of spray deposition (Table 3). AZP was not detectable in plants at platform 4. None of the sediment samples taken at platforms 1 and 4 between 0 h

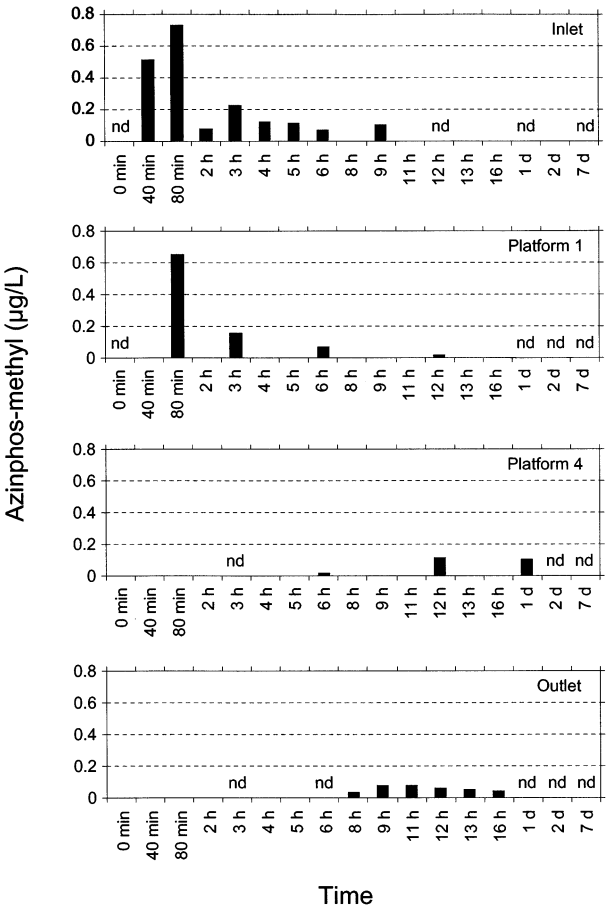

FIGURE 3. Concentrations of azinphos-methyl (nd, not detectable) at various sites during a spray drift event on February 5, 2001. Time zero refers to the commencement of spray deposition about 200 m upstream of the wetland inlet where peak azinphos-methyl levels were 2.5 $\pm$ 2.6 μg/L.

TABLE 3. AZP Concentrations in Plant Samples (μg/kg dw; $n = 1$) Taken at Platforms 1 and 4 of the Wetland[a]

time	platform 1	platform 4
0 h	2.0	nd
3 h	1.6	nd
6 h	6.8	nd
12 h	2.2	nd
7 d	2.2	nd

[a] nd, not detectable. Time zero refers to the commencement of spray deposition about 200 m upstream of the wetland inlet on February 5, 2001.

TABLE 4. AZP Concentrations in Water (μg/L; $n = 1$) along Transects across Platform 2 at 3.5 h and Platform 3 at 6 h following Commencement of Spray Deposition about 200 m Upstream of the Wetland Inlet on February 5, 2001[a]

sample	platform 2	platform 3
a	0.17	0.15
b	0.22	0.21
c	0.20	0.16
d	0.29	0.19
e	0.27	0.01

[a] Five samples (a–e) were taken at each platform at 2-m intervals along 10-m transects perpendicular to the wetland shore.

and 7 d contained any detectable AZP. At both platforms 2 and 3, AZP was distributed virtually uniformly along the transects situated perpendicular to the wetland shore. Average (±SE; $n = 5$) levels were 0.23 ± 0.02 and 0.14 ± 0.04 μg/L, with coefficients of variance of 22% and 55% (Table 4).

Detailed measurements of AZP transport through the wetland during the spray drift trial on February 5, 2001, confirmed the positive effect of the wetland in reducing aqueous-phase concentrations. Macrophyte samples contained up to 6.8 μg/kg AZP in comparison to pre-event levels ≤ 2 μg/kg, which are presumably due to AZP applications earlier in the season. AZP concentrations in plants decreased rapidly and were in the range of 2.2 μg/kg 12 h and 7 d post-spray deposition, while the pesticide was not detected in any sediment samples. As studies with chlorpyrifos, which has a much higher K_{OC} value, showed much longer residence times in plants and sediments (*2*), it is assumed that the relatively high water solubility of AZP along with the fact that a flow-through wetland with a theoretical water renewal time of 27 h was used in this study are of importance. On the basis of the present data, the water–plant (surface) distribution coefficient would be 17 for AZP. Plant samples taken at platform 4 next to the outlet contained no detactable AZP levels, indicating that aqueous-phase AZP concentrations are generally much lower at this site. A mass balance revealed that in total about 11 mg of AZP was sorbed to the living water-exposed parts of *T. capensis* in the wetland. This equals 10.5% of the total mass of AZP initially retained in the wetland. It should be noted that senesced plant matter and other plant species are not included in this mass balance, which might increase the relative contribution of the aquatic vegetation. However, the majority of the initial loss of AZP in the wetland may be due to processes such as volatilization, photolysis, hydrolysis, or metabolic degradation (*6*, *7*). On the other hand, the results highlight the importance of aquatic plants for the sorption of insecticides, even if they have relatively low K_{OC} values and high water solubilities. On the basis of the present data, it is not possible to decide whether the vegetation itself or microbial communities attached to the plant surface are relevant for the AZP sorption. The latter process seems more likely, particularly with respect to the short time periods involved. However, the role of plants including attached algal and microbial communities for insecticide sorption has already been demonstrated by various other workers (*2*, *4*, *25*, *30*, *31*).

Another important effect of the plant coverage is related to the flow conditions in the wetland. As was shown by the results from transects along platforms 2 and 3, measured AZP was relatively evenly distributed between sites, indicating that it did not flow in a single main channel through the wetland. It is likely that the dense vegetation coverage in the wetland influences the hydraulic conductivity (*32*) and thus homogenizes the horizontal pesticide distribution along trans-sectional profiles. Since both transects covered only up to half of the wetland width, however, it is possible that concentrations differed in the other half.

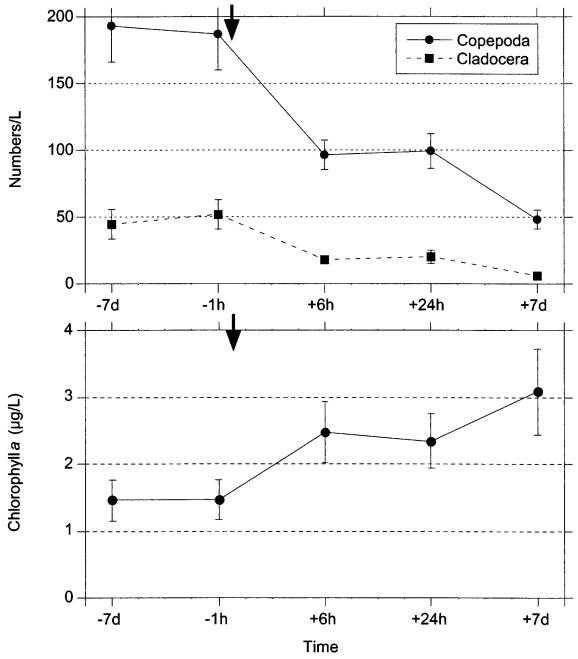

FIGURE 4. Mean (± SE; $n = 4$) densities of zooplankton and chlorophyll *a* concentrations in the wetland at platform 3. The arrows indicate the spray drift related input of azinphos-methyl on February 5, 2001, at time zero causing a significant decrease in density of copepods and cladocerans.

Pesticide Effects. Both Copepoda and Cladocera decreased significantly (*t*-test) 6 h following the spray deposition, from approximately 180 to 100 animals/L ($p = 0.019$; $n = 4$) and from approximately 50 to 20 animals/L ($p = 0.027$; $n = 4$), respectively. They remained at these levels until the end of the sampling period on day 7 (Figure 4). Chlorophyll *a* concentration showed an increase from 1.5 μg/L prior to AZP contamination to >2.5 μg/L between 6 h and day 7, which was not significant. These generally low concentrations indicate a low phytoplankton density, presumably due to the shading effect and nutrient consumption of the emergent macrophyte vegetation.

Cladocerans and copepods were significantly affected at platform 3 where average AZP concentrations were 0.14 ± 0.04 μg/L at 6 h following spray deposition. It has been reported from littoral enclosures that AZP levels as low as 1 μg/L exhibit short-term effects on zooplankton communities (*15*, *33*), with cladocerans being the most sensitive group (*33*). The increased chlorophyll *a* concentrations, although not significant, may indicate an indirect positive effect of the insecticide on the phytoplankton communities. As the grazing pressure may decrease with reduced zooplankton densities, the algae density (expressed as chlorophyll *a*) increases. A similar effect was assumed to be responsible for an increase of chlorophyll *a* concentration in ponds exposed to the organophosphate insecticides temephos and chlorpyrifos (*34*).

AZP concentrations of 0.73 μg/L detected in the inlet water were higher than acute toxic concentrations for various species of crustaceans such as *Gammarus fasciatus* Say (*35*) and *Hyalella azteca* Saussure (*36*). They also exceeded the 96-h LC_{50} of *Chironomus tentans* Fabricius, which is 0.37 μg/L (*36*). In addition, in situ exposure of chironomids during runoff events with 0.85 and 0.06 μg/L AZP at the inlet and outlet stations, respectively, revealed an 89% reduction in toxicity below the same wetland in an earlier study (*1*).

In summary, this study suggests that macrophyte-vegetated wetlands have the potential to contribute to aqueous-phase pesticide risk-mitigation. However, further

studies are needed focusing on the long-term processes in constructed wetlands. These results confirm the importance of vegetated buffer zones, in the form of either wetland areas supplied by streams or ditches, or vegetation coverage in the streams or ditches. It can be concluded that the conservation and management of vegetation in small drainage channels may be an effective tool to avoid agricultural pesticide contamination of larger receiving water bodies.

Acknowledgments

This study is part of the ENVIROMAP project (http://www.tu-bs.de/enviromap) and is funded by the Volkswagen Stiftung, Hannover (I/76 177) and the National Research Foundation, Pretoria. The authors thank A. J. Reinecke, J. A. Day, P. Benecke, C. Coulsen, H. Malan, and M. Sauerman. The manuscript benefited from suggestions made by A. Thorson, J. Wogram, and three anonymous reviewers.

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Received for review August 6, 2002. Revised manuscript received January 6, 2003. Accepted March 10, 2003.

ES026029F