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Diazinon transport through inter-row vegetative filter strips: micro-ecosystem modeling

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Abstract

The efficacy of inter-row vegetative filter strips (VFS) for controlling runoff of the commonly used organo-phosphate insecticide (diazinon) from dormant-sprayed orchards was investigated through development of physical (micro-ecosystem) models. The micro-ecosystem consisted of a pesticide sprayer, rainfall simulator and orchard floor model with and without VFS. Diazinon was sprayed at a rate of 2.8 kg/ha, 24 h prior to rainfall simulation. Rainfall, at an intensity of 50 mm/h and 44% of the natural rainfall energy, was simulated for 60 min. Experiments were conducted for 0, 50 and 100% VFS soil coverage. Diazinon concentrations in runoff, interflow and baseflow, and also in soil and vegetative samples were measured in order to quantify transport/adsorption processes.

Total diazinon losses as a fraction of applied pesticide mass from the orchard floor following rainfall-runoff simulation were 8.6, 5.8 and 2.3%, respectively, for the 0, 50 and 100% VFS cover treatments. Diazinon runoff concentrations decreased with time during the rainfall simulation, but at a slower rate in the VFS treatments as compared to the bare soil treatment apparently due to washoff from the sod leaves. The principle mechanism of diazinon runoff control in VFS was diversion of runoff, the primary pesticide carrier, into interflow through the rootzone and mainly vertical infiltration (baseflow) such that the diazinon was trapped on the VFS surface and in its rootzone. In fact, 37 and 88% of the applied diazinon remained as residue in the VFS vegetative matter and rootzone for the 50 and 100% VFS treatments, respectively, following rainfall simulation. Results from the micro-ecosystem suggest that inter-row VFS should be effective in reducing diazinon runoff from dormant-sprayed orchards. These results are used to calibrate a field-applicable numerical model for development of pesticide runoff control strategies, or best management practices (BMP's).

Keywords: Vegetative filter strips; Pesticides; Runoff; Orchards; Water quality

1. Introduction

Pesticide transport via field surface and subsurface runoff is of growing concern in both agricultural and urbanized watersheds across the country. In California, thousands of hectares of deciduous fruit and nut orchards (primarily almond and stone fruit) in the Central Valley are subject to dormant-season application of organo-phosphate insecticides (e.g. diazinon) each year. Total diazinon use for agriculture alone was over 6,12,000 kg in 1992 (Department of Pesticide Regulation, 1992). Of this total, the largest fraction (23.9%) was applied as dormant spray to almonds, of which approximately 46% was applied in the counties along the San Joaquin River. However,

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the Mediterranean climate in California results in most precipitation occurring in December through March. Consequently, pesticides were often found at toxic levels in aquatic environments of the Central Valley during the winter rainy season (Foe, 1990; Foe and Connor, 1991; Crepeau et al., 1991; Foe and Sheipline, 1993; Kratzer, 1999).

Damagalski et al. (1997) measured pesticides used in dormant-sprayed orchard in the San Joaquin River during 1993 winter. Diazinon concentrations in the creeks and canals varied from 0.12 to 7.0 μg/l and often exceeded LC_{50}, the toxic concentration level of diazinon for *Ceriodaphnia dubia* of 0.35 μg/l (Amato et al., 1992) during winter storm. Panshin et al. (1998), of the US Geological Survey, noted that the maximum winter concentrations of chlorpyrifos, simazine and diazinon in streamflow are associated with large discharges generated by precipitation following the application to dormant orchards.

Diazinon has a limited water solubility of 68.8 mg/l at 20° (Howard, 1989) while experimental K_{oc} values vary from 180 to 430 ml/g (Montgomery, 1993; Sharom et al., 1980). These values imply that diazinon is not expected to be strongly bound to the soil and will have moderate to high mobility in soils (Howard, 1989). However, diazinon exhibits significant adsorption to organic matter. About 95% of the diazinon in the sand (0.7% OM) was leached, however only 50% of the diazinon in organic soil (75% OM) was leached after ten 200 ml rinses with distilled water (Sharom et al., 1980). For adsorption kinetics of diazinon, the maximum adsorption occurred within 10 h of initial exposure and was 80, 77 and 85%, respectively, for the Ella, Keaunee, and Poygan soils in Wisconsin (Konrad et al., 1967).

Diazinon degradation in the soil occurs under both biotic and abiotic conditions. Reported half-lives for diazinon in sterile and non-sterile soils were 12.5 weeks and less than 1 week, respectively, in sandy loam, and 6.5 and 2 weeks, respectively, in organic soil and hydrolysis is the primary mechanism of abiotic degradation in soil water environments (Howard, 1989). For three Wisconsin soils, Ella, Keaunee, and Poygan soils, the microbial degradation was not a factor contributing to breakdown of diazinon (Konrad et al., 1967). Volatilization losses under field conditions could be significant (up to a 51% volatilization loss of applied diazinon from a dormant-sprayed peach orchard within one day) because of the method of spray application, wind effects and high surface temperatures (Glotfelty et al., 1990).

Vegetative filter strips (VFS) are known to have potential for reducing pesticide runoff and leaching. VFS promote interflow within the VFS root zone where significant amounts of organic matter exist such that pesticide adsorption and degradation is enhanced. In orchards, VFS established on inter-row contours are currently used to reduce soil erosion and production costs associated with herbicide application (Ross, 1993), but their efficacy for diazinon removal from rainfall-runoff of dormant-sprayed orchards is unknown. VFS filter suspended soil particles by reducing runoff flow velocity and increasing infiltration rates, and can be effective in controling some non-point source pollutants such as sediment and sediment-bound agricultural chemicals and pesticides (Dillaha et al., 1989). However, little study has been done to identify the filtering mechanisms of the VFS and its root zone.

VFS functionally consist of three distinct layers (surface vegetation, root zone, and subsoil horizon) such that water flow and pesticide transport through the VFS is a complex process. Once flow enters the VFS, infiltration occurs. When the subsurface is saturated, or the inflow rate exceeds the infiltration capacity, overland flow begins. In the root zone, some water infiltrates deeper into the soil matrix while the remainder becomes lateral subsurface flow or interflow (Fig. 1). Barfield et al. (1992) considered the major pollutant trapping mechanism of VFS to be infiltration through the rootzone, followed by storage in the surface layer. The effectiveness of VFS in removing pollutants depends on soil type, rainfall intensity, field slope and micro-topography, infiltration capacity of the vegetated area, and the VFS width in the direction of flow.

The use of VFS has been studied for controling pollutant runoff from cropland and feedlots (Dillaha et al., 1987; Dillaha et al., 1989; Bingham et al., 1980; Young et al., 1980), as well as, for removing sediment from surface mining and urban drainage (Albercht and Barfield, 1981, Glick et al., 1992). Results from these studies are summarized in Table 1. The sediment trapping efficiency was frequently greater than 90% and the nutrient trapping efficiency varied from about 50–80%. The trapping efficiency for the several

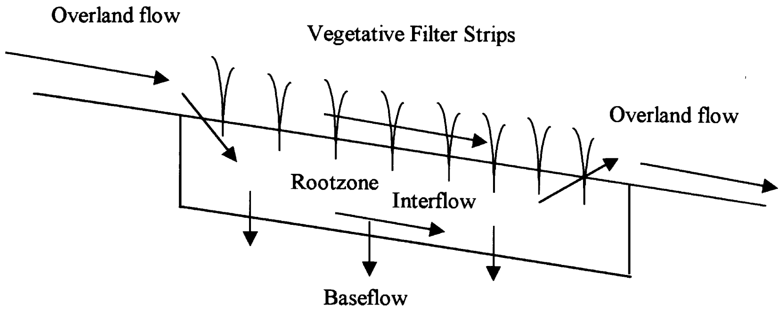

Fig. 1. Conceptualized flowpaths in Vegetative Filter Strips.

herbicides ranges about 50–100% depending on the experimental conditions (Barfield et al., 1992; Webster and Shaw, 1996; Patty et al., 1997; Murphy and Shaw, 1997; Tingle et al., 1998). The key VFS design elements are also discussed at the U.S EPA's WWW site in 'Management Measure for Vegetated Treatment Systems' (USEPA, 2000).

While several studies have considered measurement of specific parameters affecting diazinon transport/adsorption, and general field investigations of diazinon loss rates or pesticides losses through VFS, there is little integrated information available about the various pesticide transport mechanisms in VFS. The purpose of this research was to develop a micro-ecosystem physical model and subsequent experiments to provide insight into pesticide transport mechanisms such that adequate field-applicable numerical models to assist in design of VFS for control of organo-phosphate pesticide runoff from dormant-sprayed orchards can be developed.

2. Experimental methods and procedures

2.1. Micro-ecosystem model development

The micro-ecosystem model (with and without VFS) was developed to simulate the soil environment of a dormant-spray orchard (Figs. 2 and 3). The system (1.0 m × 2.0 m × 2.0 m) consisted of a rainfall simulator with a water tank and a pump, a pesticide sprayer, and three rectangular orchard floor models (for triplicate replication). The orchard floor models included individual collection devices for overland flow, interflow, and baseflow.

The rainfall simulator (1.0 m × 2.0 m) was located 1.0 m above the soil surface. The simulator consisted of two adjacent 1 m^2 by 7.0 cm tall lucite reservoirs with a capacity of 45.9 l of water. At the base of each rainfall reservoir, 841 (29 × 29 grid) raindrop emitters were installed on a 3.4 cm square spacing. Each emitter was constructed from the lower 3.0 cm of a 3 cc plastic syringe equipped with a 23 gage, 2.5 cm long needles (Becton and Dicson & Co.). The pump-valve assembly allowed the reservoirs to be filled quickly and the rainfall intensity adjusted to the desired level. A large floor fan was used to randomize the raindrop pattern.

We calibrated the rainfall simulator for a 50 mm/h rainfall intensity, or a flow rate of 73.2 l/h following the procedures described by Battany and Grismer (1999). This intensity was used to develop sufficient runoff, interflow and baseflow for the evaluating filtering mechanisms of the VFS systems. The average emitter raindrop diameter was 2.6 mm resulting in a drop velocity of approximately 4.3 m/s and an estimated drop kinetic energy of 0.128 J/s/m^2 (Adams et al., 1957). This energy is equivalent to 44% that for natural rainfall with 50 mm/h intensity (Smith, 1992). Designed rainfall intensity of 50 mm/h is considered to be an extreme condition since the 6 h-rainfall for 10 year return period in the Central Valley ranged from about 30 to 55 mm (NOAA, 1973). The variability in rainfall intensity between submodels was sufficiently small such that each submodel could be considered a replicate experiment with respect to rainfall (Watanabe, 1999).

A portable pesticide sprayer was located between the orchard floor and rainfall simulator. Three Teejet even flat spray tips (8001EVS, Spraying Systems Co.)

Table 1
Pollutant trapping efficiency of VFS from various studies

Reference	VFS type	Nutrient source	Plot length (m)	Pollutant	Trapping efficiency (%)
Young et al., 1980	Grass mixture (4% slope)	Feedlot waste	13.7	P-total	88
				N-total	87
	Grass mixture (4% slope)		13.7	P-total	81
				N-total	84
Dillaha et al., 1988	Orchard grass (5–16% slope)	Simulated feedlot waste	4.6	P-total	39
				N-total	43
			9.1	P-total	52
				N-total	52
Dillaha et al., 1989	Orchard grass (5–16% slope)	Cropland runoff	4.6	P-total	75
				N-total	61
			9.1	P-total	87
				N-total	61
Parsons et al., 1991	Grass mixture (slope 2.5–4%)	Cropland runoff	4.3–8.5	P-total	26
				N-total	50
Barfield et al., 1992	Grass and fescue (9% slope)	Cropland runoff	4.6	NH_4–N	92
				Atrazine	93
			9.1	NH_4–N	100
				Atrazine	100
			13.7	NH_4–N	97
				Atrazine	98
Webster and Shaw, 1996	Tall fescue	Cropland runoff	22.0	Metolachlor	55–90
				Metribuzin	50–90
Patty et al., 1997	Rye-grass	Cropland runoff	6.0–18.0	Lidane	72–100
				Atrazine	44–100
				Isoproturon	99
				Diflufenican	97
Murphy and Shaw, 1997	Tall fescue	Cropland runoff	0.5–1.0	Fluometuron	48
				Norflurazon	50

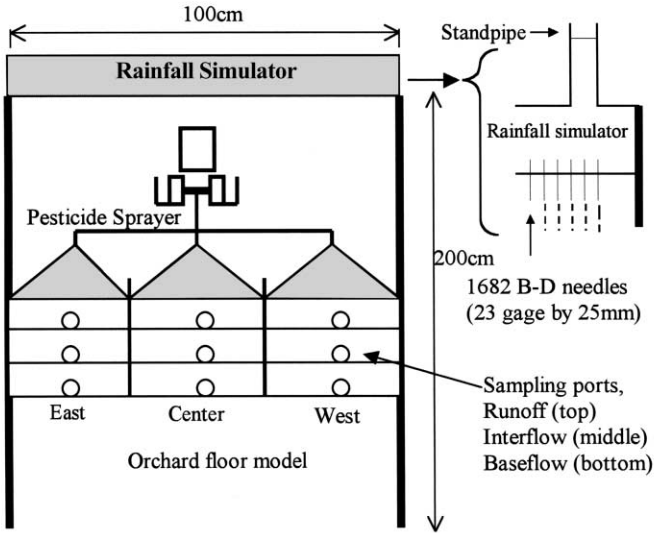

Fig. 2. Cross section view of micro-ecosystem model and pressure head standpipe.

were mounted on a four-wheel cart (25 cm × 35 cm), and connected via tygon tubing through the diazinon tank on the cart to the pressurized CO_2 tank placed at the side of the system. The sprayer cart placed on two rails mounted on the soil box frame moves at a constant speed of 22.0 cm/s while spraying diazinon over the soil surface. The pesticide was sprayed at a rate of 935 l/ha of pesticide solution (100 gal/acre), or 2.80 kg active ingredient/ha (2.5 lb/acre); the loading rate commonly recommended for dormant spray of diazinon in the Central Valley. Prior to the experiment, the pesticide sprayer was calibrated for the designed application rate, and tested for the actual application rate and the spray distribution using manganese sulfate ($MnSO_4H_2O$) as a tracer in the spray solution following the procedures developed by Giles, (1995). The overall variability (*CV*'s) in the application rates were 2.1, 5.4 and 5.6%,

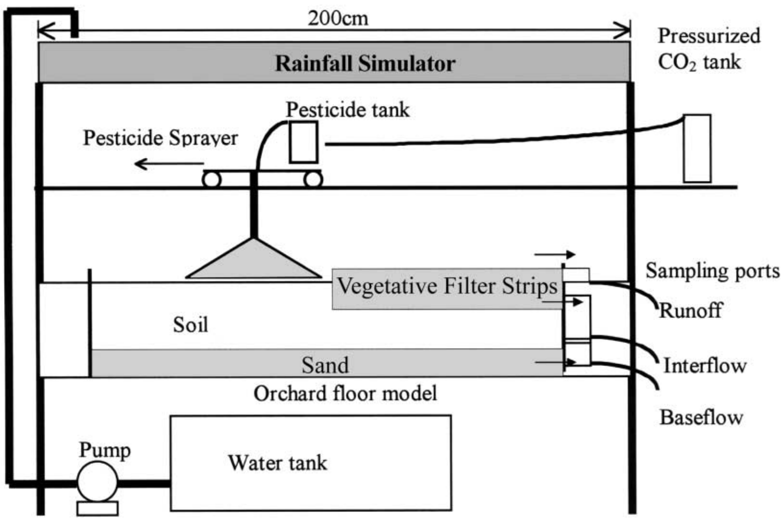

Fig. 3. Side view of micro-ecosystem.

respectively, for the east, center and west submodels. The application rates were easily replicated, thereby enabling the use of submodels as replications for each experiment (Watanabe, 1999).

The three orchard floor submodels (30.5 cm wide × 160 cm long × 25 cm deep compartments) used as replicates were packed in the soil box set on a 3% slope. Water sample collection devices for overland flow, interflow within the root zone and vertical infiltration or baseflow below the soil were installed at the downslope end of each compartment. The orchard floor submodels were constructed in the soil box by first packing a 3.0 cm thick sand layer on the bottom, followed by a 12 cm thick Yolo silt loam layer packed in four 3.0 cm thick lifts. The silt loam soil was packed homogeneously on the sand floor maintaining a bulk density of 1.40 g/cm^3. To simulate the orchard floor with VFS, established fescue sod (2–3 months old) with a 3.0 cm thick root zone was used. Sod was cut to fit as needed and placed on a 9.0 cm thick layer of silt loam packed on the 3.0 cm thick sand layer. A new orchard model was packed for each experiment.

Prior to the rainfall-runoff experiments, the sand, silt, and clay fractions, organic matter contents, and pH of soil solutions for both the silt loam and the root zone soils were measured. These physical characteristics of the soils are listed in Table 2. We also measured the soil hydraulic and water retention properties using a constant-head permeameter for saturated hydraulic conductivity (K_s) and the multi-step outflow method of Eching and Hopmans (1993) for the unsaturated hydraulic properties. The saturated hydraulic conductivity for the sod rootzone ranged from 15 to 333 mm/h however, the variability of the bulk density was relatively small with standard deviations of 0.02 and 0.04, respectively, for 50 and 100% VFS cover treatments. The large variability in K_s for the sod samples was due to an apparent macro-pore structure. Watanabe (1999) describes the unsaturated hydraulic parameters for these soils.

2.2. VFS treatment simulations

Three rainfall-runoff simulations (i.e. experiments) were performed. Simulation #1 was set up as a base condition and conducted using bare soil with no VFS. Simulation #2 was conducted using 50% VFS coverage of the lower half of the submodel and simulation #3 was conducted using 100% VFS coverage of the soil. The 50 mm/h, 1 h storm was applied for the three simulations. Each experiment was conducted in triplicate using the three submodels as described above. Prior to each experiment, the orchard floor submodels were initially saturated and then allowed to dry for about two weeks to obtain structural stability in the soil and at the soil-root interface (Watanabe, 1999). The soil surface was covered by plastic until measured soil matric potential became constant (about 2days). At this point, it was assumed that an equilibrium water content distribution was achieved across the soil profile to set the initial condition of each experiment.

Diazinon was applied using the pesticide sprayer described above at the rate of 2.80 kg/ha (active ingredient) 24 h prior to rainfall initiation. An aluminum sheet was used to cover the orchard floor model box immediately after the diazinon application in order to minimize volatilization loss of the pesticide. After setting the rainfall simulator for a 50 mm/h intensity, the aluminum sheet was removed and the rainfall-runoff experiment initiated. Samples of overland flow, interflow, and vertical infiltration as a baseflow were collected in individual bottles (3.8 l for overland flow, and 1.9 l for both interflow and baseflow) at 10 min intervals following onset of rainfall. The times when overland flow, interflow and baseflow began were also recorded.

Following each experiment, collected water

Table 2
Summary of soil physical properties used in the micro-ecosystem experiments

Soil	OM (%)	pH	Sand (%)	Silt (%)	Clay (%)	ρ_b[a] (g/cm^3)	K_S[a] (mm/h)
Silt loam	1.61	6.9	24	58	18	1.37	9.0
Rootzone	1.62	6.9	41	45	14	1.09/1.27[b]	125

[a] ρ_b: Soil bulk density, K_S: Saturated Hydraulic conductivity.
[b] 1.09 for 50%VFS cover treatment and 1.27 for 100% VFS cover treatment.

samples were refrigerated at less than 4° and divided into sub-samples within 48 h for later analyses. Prior to sample division, the weight of each sample was recorded. For runoff water samples, only the treatment without VFS had sufficient sediment for later analyses. Each runoff sample from that treatment without VFS was subdivided following the 3/10th and 7/10th method using a US Geological Survey sample splitter. The 3/10th portion of the samples was used to measure sediment concentration of the runoff water after filtration and oven drying. Diazinon concentrations in sediment were not measured since there were not enough sample volumes for the chemical analysis. The remaining 7/10th of the samples were put into 1.9 l jars and left for several hours until sediment settled. About 100 ml samples were drawn into 200 ml amber bottles for analysis of diazinon concentrations in water.

Soil cores were also collected for diazinon analysis following completion of each experiment to determine the diazinon residue in the soil, grass litter and grass leaves. Soil samples from the bare soil area were collected using a 48 mm diameter aluminum sampling rod driven 30 mm into the soil and 'subsequent extraction' of the soil core into a wide-mouthed jar. After washing the sampling rod with distilled water, the next 90 mm section was taken in the same manner and placed into another different jar. For the VFS area, the first 30 mm rootzone core was taken from the sod surface. The core was then divided into leaf and surface plant residues leaving 10 mm of plant stems. Each root zone sub-sample was placed into separate jars. After washing the sampling rod, next 90 mm section of soil sample was taken. Three sampling locations were chosen randomly from each submodel. All samples were kept frozen for later pesticide analysis. Diazinon concentrations of the samples were measured using gas chromatography following extraction by methods appropriate for each sample type (e.g. water, soil, plant). The detection limit of the diazinon analyses for water was 0.05 mg/l. Watanabe (1999) describes the diazinon extraction and analytical methods in detail.

3. Results and discussion

3.1. Hydrological response of micro-ecosystem

Before considering the experimental results, the hydrologic conditions associated with each experiment were discussed so that their results can be appropriately compared. Average rainfall intensities for the 0, 50 and 100% VFS cover treatment experiments were 49.9, 50.7 and 48.8 mm/h, respectively, with the corresponding *CV*'s between submodels of 1.8, 1.8 and 3.1%. Table 3 shows initial volumetric water contents of the orchard floor for 0, 50 and 100% VFS cover treatments. Soil moisture variability between the submodels for the VFS area for 50% VFS cover treatment appeared to result from local differences in the VFS sod vigor and affected the rates of runoff and infiltration during the experiment. Overall, the initial and rainfall conditions associated with each experiment were nearly the same with the exception of some variability in the initial soil-water contents of the submodels in the 50% VFS coverage experiment.

We first consider the effects of the VFS treatments on the hydrologic response of the micro-ecosystem summarized in Table 4. Increased VFS coverage of

Table 3
Initial volumetric water content of orchard floor model

VFS cover	Area	Depth (cm)	Volumetric water content		
			East	Center	West
0%	Bare soil	0–12	0.26	0.26	0.27
50%	Bare soil	0–3	0.23	0.25	0.25
50%	Bare bare soil	3–12	0.29	0.30	0.33
50%	VFS (rootzone)	0–3	0.27	0.35	0.42
50%	Soil below VFS	3–12	0.33	0.36	0.38
100%	VFS (rootzone)	0–3	0.36	N.A.[a](0.35)	0.34
100%	Soil below VFS	3–12	0.30	0.30	0.31

[a] N.A.: Data not available and (0.35) is an average value of the East and West sub-basins.

Table 4
Summary of hydrologic response variables from micro-ecosystem experiments

Experiment	Runoff					Interflow				Baseflow				Net infiltration[a]		
	T_i[b] (min)	T_p (min)	Q_p (mm/h)	R_d (mm)	C_{sed} (g/l)	T_i (min)	T_p (min)	Q_p (mm/h)	IF_d (mm)	T_i (min)	T_p (min)	Q_p (mm/h)	B_d (mm)	T_{ss} (min)	Q_{ss} (mm/h)	I_d (mm)
Bare Soil																
East basin	2.6	20	4.50	42.1	2.33	ND[c]	ND	0	0	ND	ND	0	0	15	0.48	7.44
Center basin	2.9	20	4.38	40.1	2.87	ND	ND	0	0	ND	ND	0	0	20	0.54	9.14
West basin	2.5	20	4.56	42.2	3.00	ND	ND	0	0	ND	ND	0	0	20	0.57	8.74
Average	*2.7*	*20*	*4.44*	*41.5*	*2.77*	*ND*	*ND*	*0*	*0*	*ND*	*ND*	*0*	*0*	*20*	*0.53*	*8.44*
50% VFS																
East basin	25.2	60	2.36	9.8	T[c]	17.3	25	1.00	5.61	45	45	1.20	7.42	45	2.16	35.6
Center basin	11.2	60	2.91	19.6	T	13.0	25	0.28	17.7	38.1	45	1.00	6.47	45	1.86	28.3
West basin	7.5	60	2.92	20.3	T	9.1	30	0.40	35.2	42.0	45	0.97	5.55	45	1.80	27.6
Average	*14.6*	*60*	*2.73*	*16.5*	*T*	*13.1*	*25*	*0.56*	*36.3*	*41.7*	*45*	*1.07*	*6.48*	*45*	*1.98*	*30.5*
100% VFS																
East basin	31.9	60+	3.06	11.3	ND	21.7	60+	0.66	0.16	34.5	60	1.60	6.53	60+	1.92	39.0
Center basin	42.0	60+	1.50	4.0	ND	18.0	60+	1.25	4.36	35.3	60	2.28	8.69	60+	2.04	39.7
West basin	30.1	60+	2.10	5.6	ND	23.0	60+	0.43	1.29	36.3	50	1.63	6.08	60+	2.28	40.8
Average	*34.7*	*60+*	*2.22*	*7.0*	*ND*	*20.9*	*60+*	*0.58*	*1.94*	*35.3*	*60*	*1.84*	*7.10*	*60+*	*2.10*	*39.9*

[a] Net infiltration = Rainfall − (Runoff+Interflow).
[b] All times are relative to initiation of rainfall at time zero. T_i = time initiated, T_p = time to peak, Q_p = peak rate, R_d, IF_d, B_d, and I_d = equivalent total depth, T_{ss} and Q_{SS} = time and flow to steady-state rate, C_{sed} = sed. conc.
[c] ND = No detection and T = Trace.

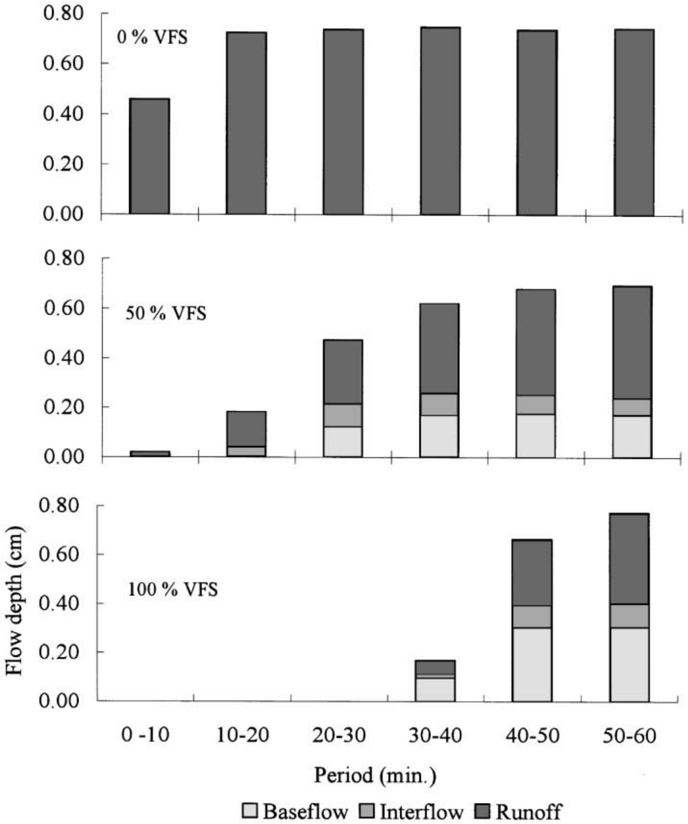

Fig. 4. Runoff, interflow and baseflow depth for 0, 50 and 100% VFS cover treatment as average of three submodels.

the orchard floor increased baseflow and interflow while decreasing surface runoff by delaying when it began and reducing the peak discharge (Fig. 4). Expressed as fractions of the simulated rainfall depth, runoff accounted for 83, 33 and 14%, respectively, for the 0, 50 and 100% VFS cover treatments. Interflow accounted for 7 and 4%, and baseflow for 13 and 15% of the applied rainfall, respectively, for the 50 and 100% VFS cover treatments.

For 0% VFS cover treatment, runoff variability between sub-models was very small since the variability in initial moisture content was quite small. There was no interflow and baseflow occurrences within 60 min of simulation from the bare soil (0% VFS coverage) treatment. For the 50% VFS cover treatment, large variability in time required for the runoff initiation between submodels corresponded to the initial water content of each sub-basin. Similarly, the time required for initiation of interflow from sub-models decreased with increasing initial water content. Note that interflow from the east submodel preceded its runoff by 7.9 min. This early initiation of interflow as compared to runoff appeared to be due to edge flow between the VFS rootzone and the submodel walls. For 100% VFS cover treatment, the time required for runoff initiation was consistent between the east and west submodels, whereas an additional 10 min was required before runoff occurred from the center submodel. However, interflow from the center submodel began nearly 4 min prior to its runoff probably because of edge flow. Nonetheless, the time required for initiation of interflow and baseflow generally increased with increasing VFS coverage. Earlier initiation of interflow and baseflow in the 50% relative to the 100% VFS cover treatment was due to the migration of runoff water from the upstream bare soil area into the VFS rootzone.

Using experimental plots on a 5% slope, Dillaha et al. (1989) found that the fractions of simulated rainfall (200 mm) appearing as runoff during a 7-day period were 30, 8 and 8%, respectively, for 0, 20 and 33% VFS cover treatments. The effect of the VFS coverage on runoff was obvious, however, the effect of the length of VFS was not observed. In the micro-ecosystem simulations, infiltration from the bare soil area was very small as compared to that observed by Dillaha et al. (1989) and the difference in net infiltration between bare soil and VFS areas was greater. As such, results from the micro-ecosystem experiments better indicate the significant effects of both the presence and length of VFS on runoff and infiltration. The peak baseflow from the 100% VFS cover treatment was about twice that from the treatment with 50% VFS cover, which also indicates the distinct difference in infiltration rate between the bare soil and VFS areas.

The net infiltration depths estimated by subtracting the depth of runoff and interflow from the rainfall depth were 8.4, 30.5 and 39.9 mm, or 17, 60 and 82% of the total rainfall, respectively, for the 0, 50 and 100% VFS cover treatments. For bare soil conditions, the estimated infiltration rates decreased and approached a constant value within 10 min after initiation of simulated storm, whereas over 40 min were required for infiltration rates to decrease and level off in the 50 and 100% VFS coverage experiments. In comparison, Dillaha et al. (1989) found that infiltration accounted for 70, 92 and 92 of the applied rainfall, respectively, for 0, 20 and 33 VFS cover treatments on 5% slope plots. Barfield et al. (1992) reported that more than 90% of incoming runoff infiltrated into the VFS area on a 9% slope. Again, the influence of the VFS on infiltration is obvious and the VFS diverts significant amount of water from surface to the subsurface.

Average sediment concentrations varied from 2.5 g/l to 3.1 g/l (or an average total sediment loss of 1.13 ton/ha) for the 0% VFS coverage experiment and were relatively constant during the simulation. For simulated field runoff, equilibrium sediment concentrations were 76, 42 and 12 g/l, respectively, for plow, disk and no till plots (Watermeier et al., 1992). Sediment concentrations from the orchard floor model were very small probably because the model simulated the conventional orchard floor with a relatively compacted soil surface. Not surprisingly, only trace sediment concentrations were resulted in the 50% VFS coverage treatment and were too small to be measured in the 100% VFS cover treatment. As noted above, many studies have reported VFS sediment trapping efficiencies in excess of 90% (Neibling and Alberts, 1979; Young et al., 1980; Hayes and Hairton, 1983; Hayes et al., 1984; Dillaha et al., 1989; and Barfield et al., 1992).

3.2. Diazinon losses in runoff, interflow and baseflow

Figs. 5 and 6 illustrate the variation in diazinon concentrations and accumulated mass losses from

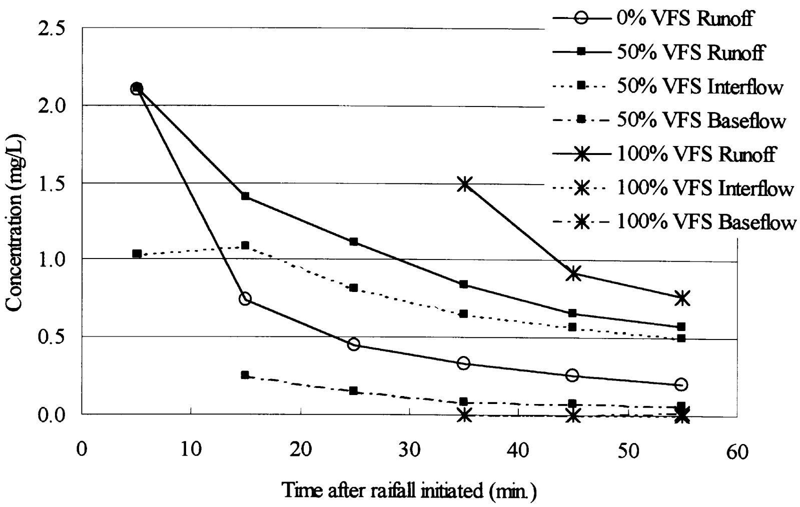

Fig. 5. Diazinon concentrations in micro-ecosystem flows.

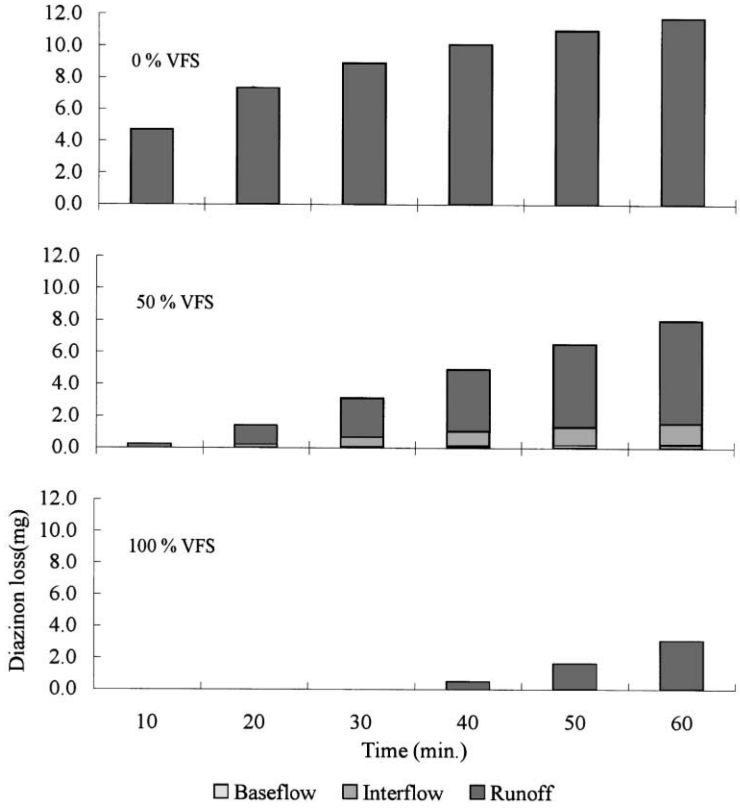

Fig. 6. Cumulative diazinon mass losses from micro-ecosystem flows.

the micro-ecosystem experiments. Each value in Fig. 5 represents the average concentration from volume-averaged flow samples collected during a 10 min period from each of the three sub-models. Note that diazinon concentrations in runoff and interflow for 50% VFS treatment during the first 10 min in Fig. 5 represented data from the west basin. The diazinon losses for the same period on the Fig. 6 was calculated with average flow volume multiplied by the diazinon concentration represented by the west basin. The average diazinon concentrations in runoff for 0% VFS cover treatment ranged from 2.09 to 0.20 mg/l, those for 50% VFS cover treatment ranged from 2.10 to 0.58 mg/l, and those for 100% VFS cover ranged from 1.49 to 0.76 mg/l. The average diazinon concentrations in interflow for the 50% VFS cover treatment ranged from 1.08 to 0.51 mg/l, however diazinon was not detected in interflow for the 100% VFS cover treatment. The average diazinon concentrations in baseflow for the 50% VFS cover treatment ranged from 0.25 to 0.07 mg/l and those of 100% VFS cover treatment were below the detection limit of the analyses (i.e. 0.05 mg/l).

In comparison to other field studies, Ritter et al. (1974) measured a maximum runoff diazinon concentration of 0.082 mg/l from a small agricultural watershed. The relatively low concentration of diazinon in runoff water was due in part to its rapid degradation, incorporation into the soil and lack of significant rainfall events following field application. After applying 4.7 kg/ha diazinon and approximately 96 mm of simulated rainfall to a fescue-covered plot, Evans et al. (1998) measured flow-weighted diazinon runoff concentrations ranging from 0.41 to 0.67 mg/l depending upon the depth of the irrigation applied prior to the rainfall simulation. For the micro-ecosystem experiments, the flow-weighted diazinon runoff concentrations were similar at 0.58, 0.80 and 0.87 mg/l, respectively, for 0, 50 and 100% VFS cover treatments.

In general, diazinon concentration in runoff, interflow and baseflow decreased exponentially with time of simulated rainfall, however, the rate of decrease depended on the type of flow and the length of the VFS cover. Diazinon concentration in runoff for the 0% VFS treatment decreased most rapidly among the treatments. Chemical concentrations in soil pore solution under rainfall have usually been described using exponential curves (Ahuja, 1982; Ahuja et al., 1981; Heathman et al., 1985; Havis et al., 1992). It is supposed that soil-applied diazinon initially infiltrated into the soil matrix, and as surface runoff developed, diazinon dissolved into the saturated surface layer or mixing zone (Williams and Hann, 1978; Haith and Tubbs, 1981), and then was transported by runoff after being transferred from the saturated soil to surface water.

For 50% VFS cover treatment, diazinon concentrations in runoff and interflow during the early period were probably affected by migration of highly concentrated diazinon runoff from the upstream bare soil area and the diazinon washoff from the sod leaf surface. A slower rate of decrease in diazinon concentrations probably resulted when migrated diazinon from the bare soil area was transported with relatively slower and more advective flow through the thatched layer and rootzone. In the later period, diazinon concentration in both runoff and interflow exhibited prolonged tailing with significant concentrations. The effect from diazinon washoff and desorption of absorbed diazinon probably became more predominant as the diazinon concentration of the migrating runoff became small. As will be discussed later, the diazinon desorption from the VFS rootzone and the thatched layer were apparent by measurements of significantly less diazinon residues in those area of the 50% VFS cover treatment as compared to those of 100% VFS cover treatment. The baseflow diazinon concentrations decreased with time, but were nearly an order of magnitude smaller than those of interflow. This difference in diazinon concentration between interflow and baseflow also indicates the significant adsorption of diazinon in the soil matrix below the rootzone during infiltration.

For 100% VFS cover treatments however, diazinon concentrations in runoff appeared to be largely a result of diazinon washoff from the sod leaf surface. A study by Willis et al. (1982) showed that washoff concentrations of toxaphene insecticide applied on cotton decreased from approximately 0.6 to 0.1 mg/l during 120 min of 13 mm/h simulated rainfall. Though applied at a similar rates in both studies, the lower concentrations of toxaphene measured in the runoff as compared to those of diazinon was due in part to its relatively low solubility in water (~2 mg/l). Diazinon's greater solubility (68.8 mg/l) is expected to result in more pesticide washoff from the sod as compared to that of toxaphene. The different rates of decrease in runoff diazinon concentration from the bare soil orchard floor and those covered with VFS probably reflected the difference in the pesticide detachment processes from the soil surface and that from the sod leaves into runoff. In contrast to runoff, no diazinon was detected in either interflow or baseflow from the 100% VFS cover treatment suggests that its rootzone adsorbed most of the diazinon transported through it. Smith and Bridges (1996) measured comparably small herbicide concentrations in leachate from simulated golf-green lysimeters. Results from both studies imply that pesticide adsorption within and below VFS could have a significant effect on the quality of the infiltrating water.

Total diazinon losses from the micro-ecosystem experiments by combined outflows expressed as a fraction of the applied mass were 8.6, 5.8 and 2.3% for the treatments with 0, 50 and 100% VFS cover, respectively (Fig. 6). Of the total diazinon losses, the surface runoff accounted for 100, 81 and 96% respectively, for the 0, 50 and 100% VFS cover treatments. From Fisher's Least Significant Difference test (Ott, 1988), average total diazinon losses from three VFS cover treatments were statistically different. The presence of the VFS resulted in reductions in total diazinon losses of 33 and 73%, respectively, for the 50 and 100% VFS cover treatments. Significant delay of the initiation of the runoff contributed to the decrease in the total diazinon losses from the system during the simulation.

Smith and Bridges (1996) reported that 9.5, 14 and 13% of applied 2,4-D, dicamba, and mecoprop were lost in runoff from simulated golf course greens. The fescue-covered plot study by Evans et al. (1998) indicated that the runoff loss of applied diazinon (liquid formation) was only 0.13, 0.99 and 0.68%, respectively, for 0, 64 and 127 mm of irrigation applied prior to rainfall simulation. Their diazinon runoff losses were very small compared to the micro-ecosystem experiment because of their small runoff volumes and relatively large volumes of infiltration. In the Evans et al. study, the runoff/rainfall ratios were 1.02, 8.66 and 9.83%, respectively, whereas those ratio for micro-ecosystem simulations were 83, 33 and 14%, respectively, for the 0, 50 and 100% VFS cover treatments. Finally, though some sediment was recovered in the runoff from the bare soil simulation, it was insufficient for analyses of diazinon concentration. Using available partitioning coefficients and measured sediment masses, the diazinon loss by the runoff sediment was estimated and it was only 0.09 mg, or less than 1% of that applied.

3.3. Diazinon fate and distribution in micro-ecosystem

The diazinon residual concentrations remaining in the soil, rootzone, vegetative residue and sod leaves of the micro-ecosystem are summarized in Table 5. No diazinon was detected in pre-simulation sampling of the micro-ecosystem components. The detection limits of the diazinon analyses were 0.86, 3.17, 0.55 and 60.8 mg/kg, respectively for soil, vegetative residue, rootzone soil, and sod leaf samples (expressed as dry mass).

Similar to other studies (e.g. Kuhr and Tashiro, 1978; Horst et al., 1996), the greatest residual diazinon concentrations were found on the vegetative residue and sod leaves. The vertical distribution of diazinon concentrations in orchard floor models of the 50 and 100% VFS cover treatments were quite similar, however diazinon concentrations in vegetative residue and rootzone soil were greater in the 100% VFS cover treatments as compared to the 50% VFS cover treatments. Lower diazinon concentrations in these materials for the 50% VFS cover treatment are probably a result of diazinon desorption by runoff water that migrated from upstream bare soil area into the VFS surface and its rootzone during the later period. As discussed earlier, the diazinon concentration in runoff from the bare soil area decreased quickly. Towards the later period, runoff water with low diazinon concentration probably dissolved and translocated diazinon from vegetative residues and the rootzone.

Table 5
Average residual diazinon concentrations (mg/kg) of the each experiment's orchard floor model components following rainfall simulation

Component	0% VFS	50% VFS	100% VFS
Bare soil 0–30 mm	4.1	5.0	NA
Bare soil 30–120 mm	0	0	NA
Sod leaf	NA	464.4	461.6
Vegetative residue	NA	13.4	20.9
Rootzone soil	NA	0.1	1.1
VFS Soil 30–120 mm	NA	0.2	0.2

A summary of the results of micro-ecosystem simulations in terms of diazinon fate distribution expressed as percentages of applied diazinon mass following rainfall-runoff simulation are shown in Fig. 7. In general, most of the applied diazinon was found on the orchard floor for all treatments since little diazinon was actually transported with flows out of the system. It is expected that diazinon on the orchard floor would be subject to degradation processes. Degraded diazinon in thatched turf after 3 weeks was 90.3, 64.5, 59.2 and 49.6%, respectively, for turf with a thatch layer irrigated daily, turf with thatch irrigated every 4 days, turf without thatch irrigated daily, and turf without thatch irrigated every 4 days (Branham and Wehner, 1985). The reactions of diazinon in soil appeared to be a rapid adsorption followed by degradation at the adsorption sites and the release of degradation products into solution (Konrad et al., 1967). Enhanced infiltration in the VFS relative to bare soil is then an important function of the VFS towards reducing pesticide discharge from the field. Barfield et al. (1992) reported that during the lag time between the initiation of rain and the start of runoff from the VFS, essentially all of the applied water and accompanying contaminant load from upstream areas had already infiltrated. One would anticipate that much of the chemical trapping was a result of infiltration (Barfield et al., 1992).

As VFS cover increased, the diazinon discharge decreased and shifted towards later periods. However, if the rainfall simulation continued, diazinon losses from 50% VFS cover treatment would eventually exceed those from 0% VFS treatment. For 50% VFS cover treatment, diazinon concentrations near the end of the period maintained a significant level and more than 30% of applied diazinon remained on the sod leaf, that is still available for the washoff. The same

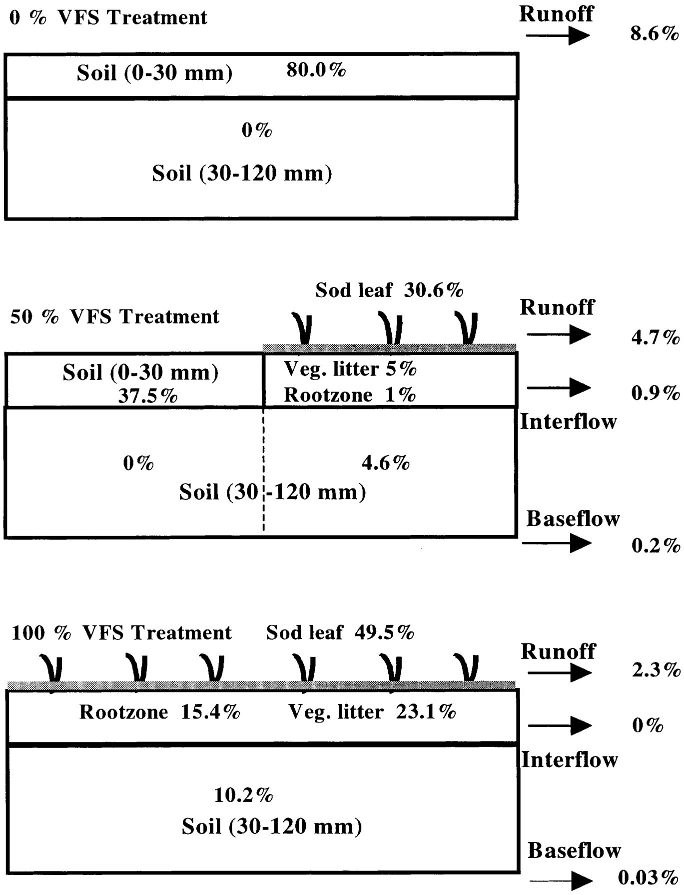

Fig. 7. Diazinon fate distribution expressed as percentages of applied diazinon mass following rainfall-runoff simulations.

occurred for the 100% VFS treatment at longer rainfall simulation. On the other hand, if the rainfall duration is short as expected for natural weather conditions, total diazinon losses with outflows will be significantly reduced by increased VFS coverage. For example, during the first 30 min of simulation, diazinon losses were 6.5, 2.3 and 0% of applied mass, respectively, for the 0, 50 and 100% VFS treatments.

Most diazinon residues were found in the 0–30 mm surface compartment of the orchard floor models regardless of the VFS coverage; accounting for 80, 74 and 88% of the applied diazinon, respectively, for the 0, 50 and 100% VFS cover treatments. Branham and Wehner (1985) also reported that only 0.6–3.4% of applied diazinon was lost by leaching from turfgrass and the majority (96%) of diazinon residue in the 5 cm-turf profile was found in the top 10 mm sampling depth. Sears and Chapman (1979) also found that 98, 2 and <1% of the applied diazinon was in the grass-thatch layer, rootzone and underlying soil, respectively, after application of 27.7 mm of water. Considerable organic matter, in the form of grass and thatch, is present in the surface layer. The thatch, in particular, presents a barrier to movement of insecticides into the underlying soil (Niemczyk et al., 1977).

For the 30–120 mm soil layer, no diazinon was found from the bare soil condition, however, 4.6 and 10.2% of the applied diazinon was found in soil below the rootzone after the simulations, respectively, for the 50 and 100% VFS cover treatments. This difference in location of diazinon residues between the bare soil and VFS areas is probably due to greater infiltration below the sod rootzone as compared to the bare soil area of which infiltration is limited by crusting and sealing during rainfall-runoff simulation (Le Bissonnais and Singer, 1992). In addition, many earthworms were observed in VFS and the soil below VFS during their removal after the simulations. Starrett et al. (1996) reported that only 6.3% of the applied isazofos (having similar solubility and K_{OC} values as diazinon) leached though 0.50 m undisturbed soil columns covered with turfgrass subject to heavy irrigation. They also concluded that this leaching was likely due to the macro-pore system of the undisturbed soil columns. On the other hand, caution should be taken for the application to hydrophilic compounds having longer persistency that greater infiltration for the VFS environment may promote groundwater contamination.

4. Conclusions

In conclusion, the micro-ecosystem study indicated that inter-row VFS have excellent potential for reducing diazinon runoff from dormant-sprayed orchards. The principle mechanism of diazinon runoff control in VFS was diversion of runoff, the primary pesticide carrier, into interflow through the rootzone and vertical infiltration (baseflow) such that the diazinon was trapped on the VFS surface and in its rootzone where further diazinon adsorption, attenuation, and presumably, degradation can occur. Increasing VFS coverage on the orchard floor should reduce runoff and diazinon losses.

The major pathway for diazinon transport into adjacent surface waters is via runoff. While the diazinon concentrations of interflow were comparable to that of runoff water, the interflow volume is much smaller than that of surface runoff. Therefore, interflow is expected to be a minor contributor to stream contamination by dormant-sprayed diazinon. The baseflow from the micro-ecosystem can be treated as groundwater recharge and that may reappear as streamflow. The diazinon concentrations of baseflow were very low and its potential contamination of the stream is also expected to be nearly insignificant as compared to that from surface runoff.

For the field application of these results, further evaluation of key processes is needed. The effects of diazinon washoff kinetics and diazinon adsorption–desorption mechanisms during rainfall-runoff need to be investigated in order to elucidate mechanisms of diazinon transport in VFS. The effects of rainfall intensity and duration on the performance of inter-row VFS also need evaluation.

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