

Behavior Assessment Model for Trace Organics in Soil: IV. Review of Experimental Evidence¹

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ABSTRACT

Experimental evidence supporting the environmental screening model of Jury et al. (1983) is reviewed and discussed. For several laboratory studies of volatilization, initial and boundary conditions matched those used in our model and simulations were run. In all cases, good agreement was found between predicted and measured volatilization losses, with and without accompanying water evaporation. When five chemicals of widely differing volatility were exposed to identical experimental conditions, the model correctly predicted the relative loss behavior observed.

The convective mobility predictions of the model were shown to be consistent with several laboratory studies of compound leaching, as long as the water flow rate was slow enough (< 0.01 cm/s) to ensure equilibrium between the solution and adsorbed phases.

The Millington and Quirk tortuosity model used in our representation of the soil diffusion coefficient was found to give a good prediction of the water content dependence of the effective diffusion coefficient observed in several studies. The thickness of the stagnant boundary layer predicted from our similarity assumption was shown to be consistent with the apparent thickness inferred from several laboratory and field measurements of volatilization.

Additional Index Words: chemical movement, diffusion, volatilization, leaching.

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In a previous series of papers (Jury et al., 1983, 1984a,b) we introduced a model for screening large numbers of chemicals for their relative volatility, mobility, and persistence in the soil environment. The model is intended to be used with standard conditions in an ideal soil environment in order to assess the relative behavior of chemicals exposed to those identical soil and environmental conditions, rather than to be used for simulation of a given transport process. The model is constructed so as to require only knowledge of the Henry's constant, K_H , organic C partition coefficient, K_{OC} , and degradation half-life, $T_{1/2}$, for a given chemical, which would enable an assessment of potential environmental risk to be made on large numbers of new chemicals at the time of their development—provided that these benchmark properties could be measured or estimated.

The philosophy of the model, as explained in earlier papers in this series, is to group chemicals together into similar mobility, persistence, or volatility categories, enabling chemicals for which substantial in situ experimental information is available to serve as a representative for a large number of chemicals that have been classified as similar by the screening model.

The purpose of this paper is to provide experimental verification of the model by comparing its output with data from the literature. The experimental evidence reviewed in this paper to verify the model and its assumptions is of three types. First, there are a limited number of laboratory studies conducted under conditions that closely match the ideal scenario represented in the model. For these cases, exact simulation is possible. Second, a limited number of experiments have been conducted in which groups of chemicals are simultaneously studied under identical conditions. For these cases, the screening model will produce a rank ordering of the chemicals that can be compared with the experimental results. Third, there are a number of assumptions in our screening model that have been tested experimentally and will also be discussed in this paper.

VOLATILIZATION STUDIES³

In 1980, Jury et al. published the results of a laboratory chamber experiment measuring volatilization of triallate from two soils, a San Joaquin sandy loam (Abruptic Durixeralfs) (1.2% organic matter) and a Flanagan silt loam (Aquic Argiudolls) (5% organic matter), together with a successful model simulation of volatilization with and without water evaporation. The triallate model, which assumed zero concentration at the soil surface and infinite depth of incorporation of chemical, makes surface volatilization predictions that are virtually identical to those of our screening model for a compound like triallate with a large Henry's constant, K_H , and high adsorption because volatilization of such compounds is not restricted by the stagnant boundary layer (Jury et al., 1984a).

In a similar chamber experiment, Spencer and Cliath (1973) studied volatilization of dieldrin and lindane from Gila silt loam (Typic Torrifluvents) (0.6% organic matter). Figure 1 shows their experimental results together with the simulations of the screening model. The soil and chemical parameters used in the model calculation, given in Table 1, were taken from the article by Spencer and Cliath (1973). In these simulations, the boundary layer thickness, d , was estimated from the water evaporation rate, as discussed in Jury et al. (1983) (see their Eq. [28]), and the effective soil diffusion coefficient, D_E , was calculated using the Millington and Quirk model formulation, which is part of our screening model (Jury et al., 1983) (see their Eq. [18]). No calibrations were made using the data.

In a soil column volatilization experiment, Yang (1978) studied volatilization and degradation of parathion in two soils— a Panoche clay loam (Typic Torriorthents) (0.9% organic matter) and a Hanford sandy loam (Typic Xerorthents) (2.5% organic matter)—over a 6-d period. This compound, which degrades quite rapidly in soil, also has a relatively low Henry's constant and thus has a smaller volatilization flux than dieldrin or lindane under equivalent conditions. As discussed in

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³ See the Appendix for a listing of scientific names of chemicals used in this article.

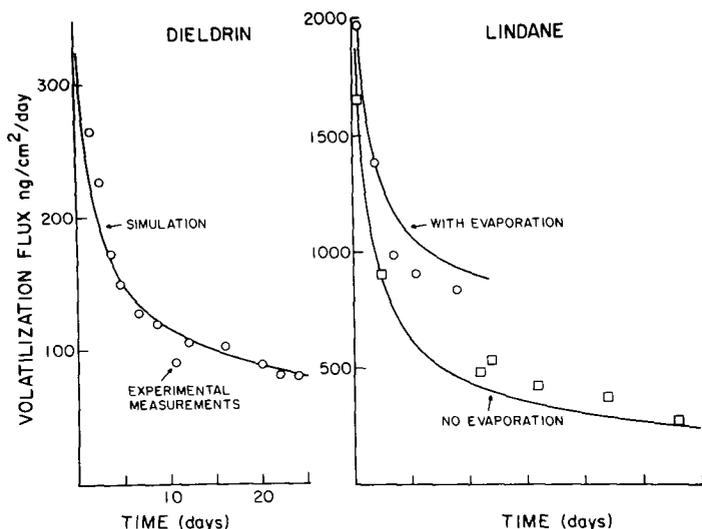


Fig. 1—Measured volatilization fluxes (data points) taken from Spencer and Cliath (1973) compared with predicted fluxes (solid lines) obtained using the screening model.

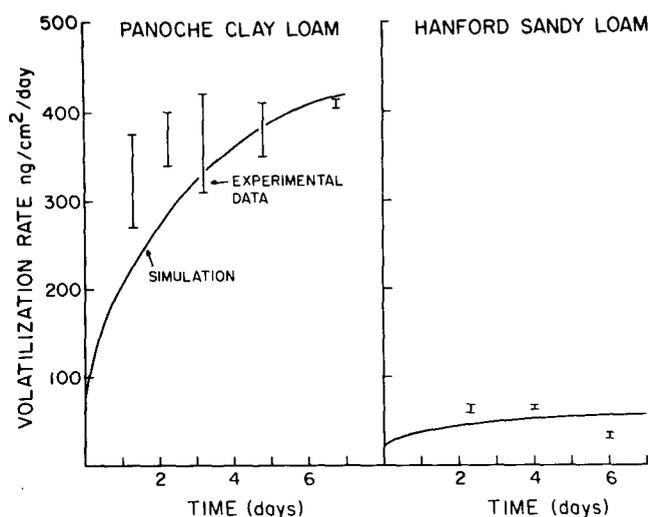


Fig. 2—Measured volatilization fluxes (vertical lines) taken from Yang (1978) compared with predicted fluxes (solid lines) obtained using the screening model. The vertical lines represent the spread between six replicates.

Jury et al. (1984a), a compound such as parathion should have a noticeably enhanced volatilization flux in the presence of water evaporation. Figure 2 shows the experimental results of Yang (1978) together with the simulation of the screening model for conditions given in Table 1.

Burkhard and Guth (1981) reported results of a 24-h volatilization experiment for five different compounds of low Henry's constant on two different soils, a Collombey sand (2.2% organic matter) and a Les Evouettes silt loam (3.6% organic matter). Measured volatilization rates for these five compounds, subjected to identical conditions, differed by orders of magnitude. In the experiment, they maintained a saturated air atmosphere above the volatilization chamber so that no water evaporation occurred, and they exchanged the soil air above the soil surface only once every 36 s, so that stagnant conditions prevailed. Our model represents stagnancy with a boundary layer above the soil surface, which in the absence of water evaporation must be estimated by calibration. Using the conditions specified in the experiment by Burkhard and Guth (given for diazinon in Table 1), we varied the thickness, d , for our boundary layer for diazinon in the experiment on

Collombey sand until a reasonable agreement was obtained between measured and calculated volatilization over the 24-h period. This occurred with $d = 0.75$ cm. This same boundary layer thickness was subsequently used for all other chemicals and in both soils. Results of the simulation and experimental measurements are given in Table 2. It should be noted that the agreement between their measurements and our calculations is considerably better than the agreement they achieved using a simple partitioning model.

Farmer et al. (1972) measured volatilization of dieldrin, lindane, and DDT over 7 d in shallow, 0.5-cm trays. The volatilization fluxes were high because of shallow incorporation, but the surface was exposed to a very slow air flow rate, which exchanged the chamber air every 30 s. The simulation was run for the conditions given in Table 1 using a boundary layer thickness of 1.5 cm, which was obtained by calibration with the lindane flux, to represent the stagnant surface. The adsorbed-liquid distribution coefficient, K_D , of DDT was calculated from the measured K_D of dieldrin and their K_{oc} values given in Jury et al. (1984b). Results of the simulation, expressed as a percent of applied chemical, are shown in Table 3.

Table 1—Experimental conditions used in simulations.

Chemical	Soil	Type	K_H	K_D	$T_{1/2}$	C_{T_0} †	L	θ	T	E	d	Ref. no.
				$m^3/kg \times 10^3$	days	g/m^3	cm		°C	cm/day	cm	
Dieldrin	Gila	Silt loam	1.3×10^{-3}	125	868	14	10	0.27	30	0.26	1.2	30
Lindane	Gila	Silt loam	2.6×10^{-4}	2.2	266	14	10	0.23	30	0.13	1.2	30
Parathion	Panoche	Clay loam	3.0×10^{-6}	4.0	10	36	13	0.15	20	0.3	1.0	36
Parathion	Hanford	Sandy loam	3.0×10^{-6}	13.5	10	34	13	0.15	20	0.3	1.0	36
Diazinon	Collombey	Sand	3.0×10^{-5}	5.6	50	60	0.9	0.12	20	0	0.75	4
Dieldrin	Commerce	Silty clay	3.3×10^{-4}	250	868	14	1	0.3	20	0	0.5	35
Dieldrin	Commerce	Silty clay	3.3×10^{-4}	250	868	14	1	0.5	20	0	33‡	35
Dieldrin	Gila	Silt loam	1.3×10^{-3}	125	868	7.5	0.5	0.1	30	0	1.5	7
Lindane	Gila	Silt loam	2.7×10^{-4}	2.2	266	7.5	0.5	0.1	30	0	1.5	7
DDT	Gila	Silt loam	4.2×10^{-3}	2500	3837	7.5	0.5	0.1	30	0	1.5	7
Dimethoate			~0	0.29	--	--	--	--	25	--	--	10
Triallate	San Joaquin	Sandy loam	7.9×10^{-4}	32	∞	12.1	10	0.28	25	0.60	0.4	15

† C_{T_0} = initial total concentration.

‡ Equivalent vapor thickness of 10 cm water barrier (see Eq. [1]).

Table 2—Twenty-four-hour average volatilization fluxes measured in experiment in Burkhard and Guth (1981).

Chemical	K_H	Volatilization (g/m ²) × 10 ⁴					
		Collombey sand Organic matter = 2.2%			Les Evouettes silt loam Organic matter = 3.6%		
		K_D	Mea- sured	Calcu- lated	K_D	Mea- sured	Calcu- lated
		(m ² /kg) × 10 ³			(m ² /kg) × 10 ³		
Diazinon	3.0×10^{-5}	5.6	1224	1329	11.7	348	703
Isazophos	3.6×10^{-6}	1.25	816	754	2.25	194	456
Metolachlor	3.7×10^{-7}	1.54	108	76	3.18	36	38
Methidathion	6.8×10^{-8}	2.35	20	10	3.89	6.7	5.9
Metalaxyl	4.4×10^{-9}	0.43	8.4	1.5	0.87	2.9	1.5

Indirect Evidence

Many studies reported in the literature either did not provide enough information for direct simulation or had conditions that differed from those of our model. In these cases, only qualitative comparisons were made. In a comparative volatilization study, Kearney et al. (1964) examined the relative volatilization of different striazine compounds, and found that the volatilization of atrazine > prometryne > simazine. This corresponds to the relative volatilization rank ordering of these three compounds based on our screening model (Jury et al., 1984b).

Willis et al. (1972) conducted field experiments on dieldrin in saturated and moist soil over a 150-d period on a Commerce silty clay loam soil (Aeric Fluvaquents). The moist plot was sprinkler-irrigated to maintain soil-water potential between 0.33 and 1 bar, which we approximated as 60% of saturated water content. The model predicted that 21.5 and 3.0% of the dieldrin should have volatilized during the 150-d period, compared with 18 and 2% measured losses for the moist and flooded conditions, respectively. The K_D of dieldrin was calculated from K_{oc} and the equivalent vapor thickness, d_v , of the water barrier of thickness d_L was calculated from Eq. [1].

$$D_L^{\text{water}}/d_L = K_H D_G^{\text{air}}/d_v \text{ or } d_v = K_H D_G^{\text{air}} d_L/D_L^{\text{water}}, \quad [1]$$

where D_L^{water} is the liquid diffusion coefficient in water and D_G^{air} is the vapor diffusion coefficient in air.

In a comparative study, Caro et al. (1976) examined volatilization of dieldrin and carbofuran under field and laboratory conditions. The dieldrin volatilization flux was found to continually decrease with time, whereas the carbofuran volatilization flux remained relatively constant. In one experiment, when no water was evaporating, the volatilization of dieldrin greatly exceeded that of carbofuran, whereas in the other experiment where the soil surface dried and water was evaporating, they had comparable volatilization rates. These

Table 3—Cumulative volatilization after 7 d expressed as a percent of initially incorporated chemical, from the experiment of Farmer et al. (1972).

Chemical	Measured	Simulated
Lindane	63	66
Dieldrin	40	27
DDT	9	7

observations are consistent with the predicted behavior of a category I compound (dieldrin), and a category III compound (carbofuran) under identical situations as discussed in Jury et al. (1984b).

Cliath and Spencer (1971) studied persistence of field-applied dieldrin and lindane over a 2-yr period. They observed effective half-lives, including volatilization, of between 3 and 5 yr for dieldrin and between 240 and 300 d for lindane. These figures are consistent with the persistence categories we defined for these two compounds in Jury et al. (1984b).

Boundary Layer Model

An important feature of our screening model is the assumption that a stagnant air boundary layer exists above the soil surface through which organic chemical vapor and water vapor must move by molecular diffusion. The thickness of this boundary layer is a complicated function of wind speed, fetch, and surface roughness, which we do not attempt to model. Rather, we assume a constant thickness of the boundary layer, which is calculated from measured water evaporation rates and assuming an analogy between water vapor movement and pesticide vapor movement. Thus, as shown in Jury et al. (1983), an evaporation rate of 2.5 mm/d into an atmosphere of 50% relative humidity and a temperature of 25°C implies a stagnant boundary layer thickness of approximately 5 mm. Since no direct measurements can be made of this boundary layer thickness, experimental tests of the model must be made in an indirect manner.

In several laboratory and field studies, the volatilization flux, J_v , and the chemical vapor concentration C_G^{air} at a height near the soil surface were measured for experiments where pesticide had been applied at concentrations sufficient to saturate the soil vapor density or where the initial vapor concentration was known (Spencer & Cliath, 1973; Jury et al., 1980; Glotfelty, 1981). For these experiments, the boundary layer thickness, d , may be estimated by Eq. [2]

$$d = D_G^{\text{air}} (C_G^* - C_G^{\text{air}})/J_v \quad [2]$$

where * denotes saturation, provided that C_G^{air} is measured above the boundary layer.

Table 4 presents a summary of a number of different experiments conducted in the field and the laboratory in which the boundary layer thickness, using Eq. [2], was calculated from fluxes measured immediately after application. For several cases in the laboratory, a water evaporation rate was also measured, which allowed us to calculate the boundary layer thickness directly using our model (see Eq. [28] of Jury et al., 1983). As shown in Table 4, all of the boundary layer thicknesses calculated from the volatilization data are consistent with the thicknesses that we use in our model. Furthermore, a correlation was obtained between predicted boundary layer thickness (Eq. [2]) and the boundary layer thickness calculated from measured water evaporation and water vapor density difference. Since the boundary layer model is an idealization, no better agreement than this could be expected.

Table 4—Boundary layer thickness calculated from various laboratory and field experiments.

Compound	Experiment	<i>d</i> Measured	Evaporation	<i>d</i> (E)†	Ref. no.	Comments
		cm	cm/day	cm		
Trifluralin	Field	0.84	--	--	9	Measurements taken at 0900 h. Wind speed 2.0–2.7 m/s (0.2–2.0 m)
Heptachlor	Field	1.3	--	--	9	
Dacthal	Field	0.68	--	--	9	
Chlordane	Field	0.88	--	--	9	
	Average	0.93				
Trifluralin	Field	0.16	--	--	9	Measurements taken at 1200 h. Wind speed 3.8–5.8 m/s (0.34–2.5 m)
Lindane	Field	0.13	--	--	9	
	Average	0.15				
Triallate	Lab	0.37	0.63	0.38	15	50% Relative humidity
Triallate	Lab	0.21	0.60	0.26	15	
	Average	0.28				
Trifluralin	Lab	0.39	--	--	31	100% Relative humidity
Lindane	Lab	0.12	0.27	0.55	30	50–100% Cycled relative humidity
Dieldrin	Lab	0.78	0.25	1.2	30	50% Relative humidity

† Calculated from water evaporation rate.

Effective Diffusion Coefficient

Our model assumes equilibrium partitioning between liquid, vapor, and adsorbed phases and assumes that the variation in diffusion coefficient with water content or air content may be described by the model of Millington and Quirk (1961) as shown in Eq. [3].

$$D_E = (D_G^{\text{air}} K_H a^{10/3} + D_L^{\text{water}} \Theta^{10/3}) / \phi^2 R_L, \quad [3]$$

where *a* = volumetric air content; Θ = volumetric water content; ϕ = porosity; R_L = liquid partition coefficient = C_T/C_L , where C_T = total concentration and C_L = solution concentration. The liquid partition coefficient, R_L , is approximately equal to $\rho_b K_D + \Theta$, where ρ_b = soil bulk density (Jury et al., 1983).

There have been few measurements made of effective diffusion coefficients for organic compounds over large ranges of water content. One such study, however, is the experiment reported in Shearer et al. (1973) in which the effective diffusion coefficient of lindane was measured over a range of water contents from near air dry to saturation on Gila silt loam. The measured values of lindane diffusion coefficient, together with the model calculation using Eq. [3], are shown in Fig. 3, using the appropriate parameters for lindane and Gila silt loam taken from their article (see Table 1). Also shown in Fig. 3 are the measured and predicted diffusion coefficients for dimethoate as measured in the experiment of Graham-Bryce (1969) for the parameters given in Table 1. The good agreement found for each of these compounds of widely differing characteristics using the same model supports the use of the Millington and Quirk model. Additional verification for this model was obtained by Farmer et al. (1980), who found a good agreement between measured and calculated diffusion coefficients for hexachlorobenzene (HCB) over a large range of soil air contents.

In addition to the Millington-Quirk method of representing tortuosity, our model assumes a common value for the air-gas diffusion coefficient, $D_G^{\text{air}} = 4320 \text{ cm}^2/\text{d}$, and liquid-water diffusion coefficient, $D_L^{\text{water}} = 0.432 \text{ cm}^2/\text{d}$, for all organic chemicals in the intermediate molecular weight range. The justification for this assumption was reviewed in Jury et al. (1983) and was also

discussed in Letey and Farmer (1973). In our 1983 article, we alluded to the common values found in the numerous measurements of gas diffusion coefficient reported in the article of Boynton and Brattain (1929) and of liquid diffusion coefficient reported in the article of Bruins (1929). From these two sources, we calculated the average values given above.

It should be noted that the volatilization simulations discussed above all use the Millington and Quirk formulation and the common parameter assumptions. Thus, the good agreement found for the case discussed above is additional verification of the effective diffusion coefficient model.

LEACHING STUDIES

The leaching behavior of compounds in our screening model is described with the same simple distribution coefficient model assumption used over the years by many other authors [see reviews by Bailey & White (1970), Karickhoff et al. (1971), Rao & Davidson (1980), Green et al. (1980)]. For example, we showed that the time re-

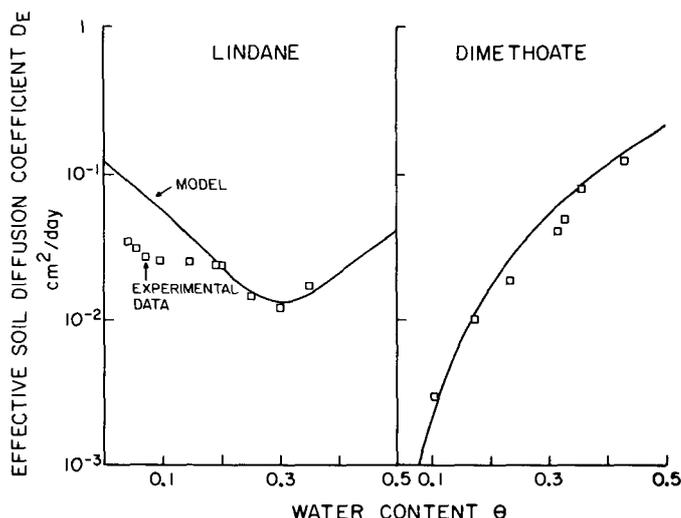


Fig. 3—Measured effective diffusion coefficients for lindane taken from Shearer et al. (1973), and for dimethoate taken from Graham-Bryce (1969), compared with the diffusion model (solid lines) used in our screening model.

quired to leach an adsorbed compound a given distance under continuous leaching was proportional to the distribution coefficient, K_D (Jury et al., 1983). The distribution coefficient model, which assumes linear, equilibrium partitioning between solution and adsorbed chemical phases, has had extensive testing under laboratory conditions with slowly percolating solutions. King and McCarty (1968) obtained distribution coefficients by batch equilibrium and then conducted extensive leaching tests on columns of 4, 15, and 90 cm. Using a total of four soils and six pesticides, they obtained good agreement between predicted and measured effluent concentrations for a chromatography model (formally similar to our own) when degradation was taken into account using a first-order rate constant. Our model would produce equivalent results to theirs when applied on the same data.

Huggenberger et al. (1972, 1973) studied leaching of lindane, diuron, and atrazine in three soils. They measured distribution coefficients in separate experiments and then studied vertical infiltration of pesticide at a constant water rate into dry soil. Although they had difficulty simulating the shape of the breakthrough curves, they achieved a good agreement between observed depth of leaching and predicted depth of leaching using the measured distribution coefficient. Our model would produce equivalent results on the same data.

Weber and Whitacre (1982) conducted 30 d of leaching on bromacil, buthidazole, atrazine, prometon, and diuron and observed leaching distances that were inversely proportional to the organic C coefficient, K_{oc} . Swartzenbach and Westall (1981) conducted leaching studies on 11 polar organic compounds and found that leaching predicted by the batch equilibrium distribution coefficient agreed with the leaching behavior observed in the columns at low flow rates. They found that non-equilibrium effects began to appear at water velocities of 0.01 cm/s or greater. These rates greatly exceed typical velocities found in the field except under infiltration conditions.

McCall et al. (1980) found an inverse relation between distance leached and organic C partition coefficients for nine different compounds. As we discussed in Jury et al. (1984a), our model predicts an inverse relation between leaching distance and distribution coefficient (or K_{oc}), unless the compound is only slightly adsorbed.

DISCUSSION

Volatilization

The experimental evidence reviewed above offers strong support for the volatilization part of the screening model. In cases where the experiments were conducted under similar conditions to those assumed in the model, a simulation produced good agreement with observation. In cases where a number of compounds of widely differing properties were studied, the model predicted a rank ordering that agreed with the observed order. Significantly, all compounds studied behaved in a manner consistent with the volatilization category predictions made in our earlier paper (Jury et al., 1984a), in which we grouped large numbers of compounds

depending on whether the Henry's constant K_H was significantly greater (category I) or less (category III) than $K_H \cong 10^{-5}$. For example, parathion (category II) in Fig. 2 increases its volatilization rate with time when evaporation is occurring, but dieldrin and lindane (category I) in Fig. 1 decrease with time.

The field studies reviewed above offer indirect support for the volatilization calculations made in our model. In the experiments of Glotfelty (1981), the apparent thickness of the boundary layer inferred from using Eq. [2] with the field measurements was consistent with the thickness calculated from an analogy with water evaporation fluxes to the atmosphere. Furthermore, the boundary layer thickness appeared to decrease as windspeed increased (Table 4). Since—as shown in Mayer et al. (1974)—the thickness assumed for the boundary layer has a critical influence on model calculations, the direct and indirect evidence given in Table 4 to support our method of selecting this thickness is encouraging.

The volatilization flux is strongly influenced by the effective diffusion coefficient, which for the Millington and Quirk (1961) model used in our calculations is a nonlinear function of air or water content. Although this model is empirical, the evidence presented here (Fig. 3 and Farmer et al., 1980) supports its use in homogeneous laboratory soil systems when hydrodynamic dispersion is small. In addition, recent evidence (Sallam et al., 1984) suggests that the Millington and Quirk model may be useful for representing vapor diffusion, even at extremely low air contents.

There are influences on volatilization that are not taken into account in our model. When the soil surface layer dries out sufficiently, adsorption of chemical to the mineral or organic surfaces increases significantly and volatilization rates decrease (Spencer et al., 1969). However, there is experimental evidence, both in the laboratory (Spencer et al., 1969) and in the field (Harper et al., 1976), that this increased adsorption over a wetting and drying cycle is similar to what would have occurred if the soil had not dried.

The experimental studies of Burkhard and Guth (1981) (Table 2) are significant in that they are among the few volatilization experiments conducted on category III (small $K_H \ll 10^{-5}$) compounds, which we predicted to have completely different properties than category I ($K_H \gg 10^{-5}$) compounds (Jury et al., 1984a). There was reasonably good agreement obtained between our model calculations and their measurements, especially considering that the range of volatilization rates was over two orders of magnitude. We would recommend further study of compounds in this category because our model predicts that under certain conditions (high water evaporation, high concentrations) they would volatilize significantly (Jury et al., 1984a).

Leaching

The leaching behavior of adsorbed compounds is reasonably well described by the linear, equilibrium adsorption model used in our calculations for compounds tested in homogeneous laboratory columns at low flow rates. The studies reviewed here all fall into that cate-

gory. Since our model is based on the same assumptions as other earlier work on mobility, the general mobility criteria used by McCall et al. (1980) and others are equivalent to our own. The use of a distribution coefficient does have limitations, however, which are reviewed in Mingelgrin and Gerstl (1983).

Little quantitative information is available about leaching of adsorbed chemicals under field conditions, where soil structure may result in incomplete exposure of adsorbing surfaces to the chemical in solution during transport. It is hoped that the *relative mobility* of different compounds under field conditions will be similar to that predicted from laboratory studies. This would allow field calibrations to be run on a few representative chemicals from each group rather than on all compounds.

SUMMARY AND CONCLUSIONS

The screening model developed in Jury et al. (1983, 1984a,b) has been tested on published experimental data on volatilization, leaching, and persistence where comparisons could be made. The reasonable agreement found in these comparisons offers encouragement for the use of the model to classify and group chemicals into similar loss pathway categories.

We firmly believe that in situ experiments offer the only reliable method for determining the loss pathways of a chemical, particularly under field conditions. However, the expense and time required for such experiments, and the large number of chemicals in need of testing, make it likely that models will play a significant role in such assessments. Our series of papers has developed a number of relationships between the benchmark properties of a chemical and its relative susceptibility to loss, which should be extensively tested in situ prior to use of the model. The evidence offered above in support of the model's predictions is the first step in this testing.

APPENDIX

Scientific names of chemicals used in this article.

Common or trade name	Chemical name
Atrazine	2-chloro-4-(ethylamino)-6-(isopropylamino)-s-triazine
Bromacil	5-bromo-3-sec-butyl-6-methyluracil
Buthidazole	3,5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl-1 hydroxy-1-methyl-1-imidazolidinone
Carbofuran	2,3-dihydro-2,2-dimethyl-7-benzofuranylmethylcarbamate
Chlordane	1,2,4,5,6,7,8,8-Octachloro-2,3,3a,4,7,7a-hexahydro-4,7-methanoindane
Dacthal	Dimethyl tetrachloroterephthalate
DDT	1,1,1-trichloro-2,2-bis(p-chlorophenyl) ethane
Diazinon	O,O-diethyl-O-(2-isopropyl-6-methyl-4-pyrimidinyl) phosphorothioate
Dieldrin	1,2,3,4,10,10-hexachloro-6,7-epoxy,1,4,4a,5,6,7,8,8a,octahydro-1,4-endo-exo-5,8-dimethanonaphthalene

Dimethoate	O,O-Dimethyl S-(N-methylcarbamoylmethyl) phosphorodithioate
Diuron	3-(3,4-dichlorophenyl)-1,1-dimethylurea
Heptachlor	1,4,5,6,7,8,8-Heptachloro-3a,4,6,6a-tetrahydro-4,7-methanoindene
Isazophos	O-(5-chloro-1-{methylethyl}-1H 1,2,4-triazol-3-yl) O,O-diethyl phosphorothioate
Lindane	γ -1,2,3,4,5,6-hexachlorocyclohexane
Metalaxyl	N-(2,6-Dimethylphenyl)-N-(methoxyacetyl)-alanine methyl ester
Methidathion	O,O-dimethyl phosphorodithioate, S-ester with 4-(mercaptomethyl)-2-methoxy Δ^2 -1,3,4-thiadiazolin-5-one
Metolachlor	2-chloro-N-(2-ethyl-6-methylphenyl)-N-(2-methoxy-1-methylethyl) acetamide
Parathion	O,O-diethyl O-(p-nitrophenyl) phosphorodithioate
Prometone	2,4-bis(isopropylamino)-6-methoxy-s-triazine
Prometryne	2,4-bis(isopropylamino)-6-methylthio-s-triazine
Simazine	2-chloro-4,6-bis (ethylamino)-s-triazine
Triallate	S-(2,3,3-trichloroallyl)diisopropylthiocarbamate
Trifluralin	a,a,a-trifluoro-2,6-dinitro-N,N-dipropyl-p-toluidine

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