Model Ecosystem Studies of the Environmental Fate of Six Organochlorine Pesticides

by Robert L. Metcalf,* Inder P. Kapoor,* Po-Yung Lu,* Carter K. Schuth,* and Patricia Sherman*

Despite more than 20 years of intensive use there are major uncertainties about the environmental distribution and degradative fate of the various organochlorine pesticides in food web organisms. The problem is both controversial and momentous, as 88,641,000 pounds of cyclodienes and toxaphene and 59,316,000 pounds of DDT were produced in the United States in 1970 *(1)*. The laboratory model ecosystem devised in this laboratory *(2)* has been used to estimate the comparative environmental properties of DDT, methoxychlor, and other DDT analogs *(3-5)*. These investigations have demonstrated environmental degradative pathways and have provided quantitative data on ecological magnification and biodegradability index *(6)*. This paper presents data on six additional organochlorine pesticides which should be of value in judging their overall effects on environmental quality. Furthermore, the data provide a realistic background of the environmental toxicology of standard substances against which screening data for new candidate pesticides can be weighed.

*Department of Entomology, School of Life Sciences, University of Illinois, Urbana-Champaign, Illinois 61801.

Experimental Procedures

The basic model ecosystem methodology utilizes radiotracer techniques. The following ^{14}C-labeled pesticides were obtained: aldrin, ring-labeled, 68.0 mCi/mmole (Amersham Searle), dieldrin, ring-labeled, 1.03 mCi/mmole (Shell Chemical); endrin, ring-labeled, 2.3 mCi/mmole (mallinkrodt); mirex, ring-labeled, 1.74 mCi/mmole (Mallinkrodt); lindane, ring-labeled, 54 mCi/mmole (Amersham Searle); hexachlorobenzene, ring-labeled, 4.61 mCi/mmole (National Institute of Environmental Health Sciences); DDT, ring-labeled, 5.48 mCi/mmole (Amersham Searle). DDT was converted to DDE (2,2-bis-p-chlorophenyl-1,1-dichloroethylene), 5.48 mCi/mmole, by treatment with ethanolic potassium hydroxide and purification by column chromatography. Several of the radiolabeled pesticides contained impurities visible after TLC and autoradiography, and these products were repurified by column chromatography through silicic acid with petroleum ether (60–68° C) as the eluent until they were at least 99+% radiochemically pure.

The model ecosystem evaluation was carried out in a small glass aquarium with a sloping terrestrial-aquatic interface of pure white sand exactly as described previously *(2)*. A 5-mg portion of radiolabeled pesticide

was applied to sorghum seedlings grown on the terrestrial portion from an acetone solution by using a micropipet. The dosage is equivalent to 1.0 lb/A (1.1 kg/hc). Fourth instar salt marsh caterpillars *(Estigmene acrea)* were fed on the leaves until these were consumed, and their fecal products and the larvae themselves contaminated the aquatic portion of the system. The radiolabeled products were transferred through several food chains, e.g., alga *Oedogonium cardiacum* ⟶ snail *Physa;* plankton ⟶ water flea *Daphnia magna* ⟶ mosquito *Culex pipiens quinquefasciatus* ⟶ fish *Gambusia affinis.* After 33 days in an environmental plant growth chamber at 80 ± 1°F and a 12-hr photoperiod, the experiment was terminated, and the amount and nature of the ^{14}C determined by homogenization of the organisms, extraction with acetonitrile, TLC autoradiography, and liquid scintillation counting *(2-4).* TLC was carried out on silica gel containing fluorescent marker (E. Merck GF-254) with the developing mixtures shown in Tables 1-6, and autoradiography was performed on no-screen x-ray film (Eastman Kodak). Wherever possible, identity of degradation products was determined by cochromatography with known standards. Liquid scintillation counting was done in Cocktail D (7 g PPO, 100 g naphthalene in dioxane to make one liter), and counts were corrected to dpm by using channel ratio-quenching corrections.

The results are expressed as equivalent ppm values in Tables 1-6 for the pesticides and their degradation products. The ecological magnification (EM), defined as ppm parent compound in organism/parent compound in water, and biodegradability index (BI), defined as ppm polar products in organism/nonpolar products *(6)*, are used to provide quantitative characterization of the environmental behavior of these organochlorine pesticides.

Results and Discussion

The overall quantitative degradation and accumulation of the six pesticides investigated is shown in Tables 1-6.

Aldrin (1,2,3,4,10,10-hexachloro-1,4,4a,5,8,8a-hexahydro-1,4-*endo*,*exo*-5,8-dimethanonaphthalene) is well known to be converted environmentally to the 6,7-epoxide dieldrin. The data of Table 1 indicate that this conversion occurred rapidly and nearly completely in the model ecosystem and that dieldrin was stored in the various organisms as 95.9% of the total ^{14}C in the fish *Gambusia*, 91.6% in the snail *Physa*, and 85.7% in algae *Oedogonium* (Fig. 1). In *Gambusia*, at the top of the food chain, only 0.5% of the ^{14}C was stored as aldrin. Three minor degradation products were clearly visible in the autoradiographs of TLC plates developed in ether–hexane 1:1, R_f 0.63, 0.45, and 0.34 (Fig. 1). These three degradation products

Table 1. Distribution of ^{14}C aldrin and degradation products.

	Aldrin equivalents, ppm				
	H_2O	*Oedogonium* (alga)	*Physa* (snail)	*Culex* (mosquito)	*Gambusia* (fish)
Total ^{14}C	0.0117	19.70	57.20	1.13	29.21
Aldrin (R_f = 0.81)[a]	0.00005	1.95	2.23	—	0.157
Dieldrin (R_f = 0.71)	0.0047	16.88	52.40	1.1	28.00
Unknown I (R_f = 0.63)	—	0.57	2.05	—	0.612
9-OH Dieldrin (R_f = 0.45)	0.00052	0.12	0.17	—	0.322
9-C=O Dieldrin (R_f = 0.34)	0.0004	0.079	0.217	—	0.088
Unknown II (R_f = 0.08)	0.00039	0.015	—	—	—
Polar (R_f = 0.0)	0.0040	0.015	0.097	—	0.004
Unextractable	0.00155	—	—	—	—

[a]TLC with hexane-diethyl ether, 1:1.

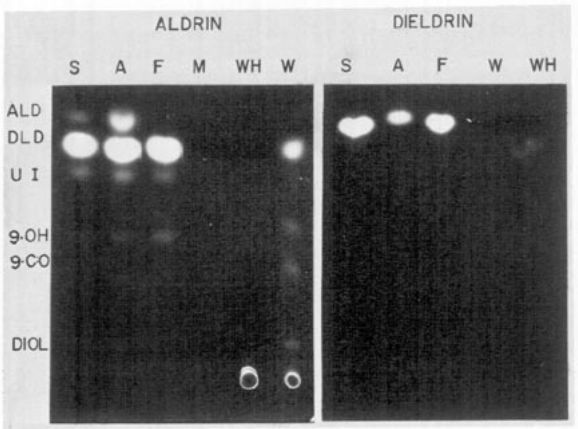

FIGURE 1. Autoradiograph of extracts from model ecosystems treated with ^{14}C aldrin and dieldrin. Abbreviations are S (snail), A (alga), F (fish), M (mosquito), W (water), WH (water hydrolyzed with HCl).

were also found in the model ecosystem run with ^{14}C dieldrin. The compound having R_f = 0.45 cochromatographed with an authentic sample of 9-hydroxydieldrin and that at R_f = 0.34 with an authentic sample of 9-ketodieldrin (Shell Chemical Company). The unknown II, R_f = 0.08 is probably *trans*-dihydroxydihydroaldrin formed by hydration of the 6,7-epoxide of dieldrin.

The biodegradability index (BI) of aldrin for fish is 0.00014 and for snail is 0.0017 and the ecological magnification (EM) in fish is 3140 and in snail is 44,600. Aldrin is clearly an undesirable micropollutant in its own right, apart from its easy conversion to dieldrin. The EM values for the other degradation products in Table 1 provide useful information about their relative environmental hazards. For dieldrin, the EM in fish is 5957 and in snail is 11,149. The 3-OH dieldrin has EM values of 617 in fish and 327 in snail and 3-C=O dieldrin of 220 in fish and 542 in snail. Thus aldrin and dieldrin provide the major environmental hazards from bioconcentration and the other compounds including aldrin *trans* diol (unknown II) are on the degradation pathways and are slowly being eliminated.

When the model ecosystem run was begun with dieldrin (1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4a,5,6,7,8,8a-octahydro-1,4-*endo*-*exo*-dimethanonaphthalene), the results were very similar to those found with aldrin (Table 2). The extraordinary environmental stability of dieldrin is shown by its storage as 95.5% of the ^{14}C in *Gambusia*, 95.8% in *Physa*, and 87.1% in *Oedogonium*. Three minor degradation products scarcely visible on the radioautographs (Fig. 1) were unknown II having R_f = 0.59, R_f = 0.45 (probably 9-hydroxydieldrin), and R_f = 0.34 (probably 9-ketodieldrin). These are apparently the same degradation products found in the aldrin model ecosystem experiments. The unknown with R_f = 0.09 is probably *trans*-

Table 2. Distribution of dieldrin and degradation products.

	Dieldrin equivalents, ppm				
	H_2O	*Oedogonium* (alga)	*Physa* (snail)	*Culex* (mosquito)[a]	*Gambusia* (fish)
Total ^{14}C	0.0039	0.73	90.09	—	3.96
Unknown I (R_f = 0.72)[b]	0.00009	0.033	0.83	—	0.01
Dieldrin (R_f = 0.65)	0.0014	0.64	86.32	—	3.78
Unknown II (R_f = 0.59)	0.0009	0.012	0.98	—	0.041
9-OH Dieldrin (R_f = 0.45)	0.00009	0.030	0.51	—	0.070
9-C=O Dieldrin (R_f = 0.34)	—	—	0.42	—	0.023
Unknown III (R_f = 0.26)	—	—	0.32	—	—
Unknown IV (R_f = 0.20)	0.00012	—	—	—	—
Unknown V (R_f = 0.09)	0.00038	0.0031	0.090	—	0.0008
Polar (R_f = 0.0)	0.00020	0.0010	0.081	—	0.007
Unextractable	0.0013	—	—	—	—

[a]All were killed throughout the experiment.
[b]TLC with hexane-diethyl ether, 1:1.

dihydroxydihydroaldrin. Thus the degradation pathways of dieldrin in the model ecosystem are essentially those found in the rat *(7)* and in the sheep *(8)* where these compounds were identified as degradation products.

The exceptional environmental stability of dieldrin is indicated by the BI values of 0.0018 in fish and 0.0009 in snail, and the EM values of 2700 in fish and 61,657 in snail. As with the aldrin experiment (Table 1), the degradation products had lower EM values, 777 in fish and 5666 in snail for 3-OH dieldrin.

Endrin or 1,2,3,4,10,10-hexachloro-6,7-epoxy-1,4,4a,5,6,7,8,8a-octahydro-1,4-*endo*, *endo*-5, 8-dimethanonaphthalene is the *endo,endo* isomer of dieldrin. As shown in Table 3, it is somewhat less stable in the model ecosystem than dieldrin and is stored as 75.8% of the total ^{14}C in *Gambusia*, 82.8% in *Physa*, and 84.9% in *Oedogonium*. The major degradation products were unknowns I (R_f = 0.83) and II (R_f = 0.52) (Fig. 2). The latter compound did not cochromatograph with an authentic sample of Δ-ketoendrin (Shell Research). By analogy with dieldrin, unknown II is probably 9-hydroxyendrin, one of the chief degradation products found in the rat together with 9-ketoendrin *(9)* and 5-hydroxyendrin (M. K. Baldwin, Shell Research, Tunstall Laboratory, Sittingbourne, Kent, England; unpublished data). Reference samples of these compounds were unfortunately not available for comparison. The BI for endrin was 0.009 in fish and 0.0124 in snail, and the EM values were 1335 in fish and 49,218 in snail. The EM value for unknown II was 270 in fish and 1701 in snail, supporting the conclusion that it is a hydroxylated degradation product with a lower lipid/H_2O partition coefficient.

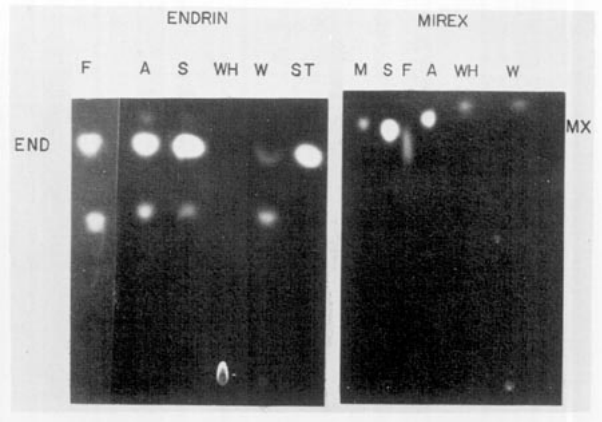

FIGURE 2. Autoradiograph of extracts from model ecosystems treated with ^{14}C endrin and mirex. Abbreviations are F (fish), A (alga), S (snail), M (mosquito), W (water), WH (water hydrolyzed with HC1).

Biological observations on the model system experiment with endrin were particularly informative. The compound was highly toxic to the salt marsh caterpillar, which had difficulty consuming it. As the level in the water rose to 0.06 ppm the *Daphnia* and

Table 3. Distribution of endrin and degradation products.[a]

	Endrin equivalents, ppm				
	H_2O	*Oedogonium* (alga)	*Physa* (snail)	*Culex* (mosquito)[b]	*Gambusia* (fish)
Total ^{14}C	0.0134	13.62	150.58	—	4.48
Unknown I (R_f = 0.81)[c]	—	0.48	5.07	—	—
Endrin (R_f = 0.73)	0.00254	11.56	125.00	—	3.40
Unknown II (R_f = 0.53)	0.00385	1.58	6.55	—	1.04
Unknown III (R_f = 0.42)	+[d]	+	5.87	—	—
Unknown IV (R_f = 0.31)	+	+	2.69	—	—
Polar (R_f = 0.0)	0.00436	—	1.85	—	0.04
Unextractable	0.0027	—	—	—	—

[a]Experiment terminated after 63 days.
[b]All were killed throughout experiment.
[c]TLC with ether-hexane, 1:1.
[d]Trace quantity

mosquito larvae in the aquarium were repeatedly killed and had to be reinfested. Most importantly, the water phase was incredibly toxic to *Gambusia*, which began violent convulsions within a few minutes of being added to the aquarium and died within a few hours. The high toxicity of the water persisted for more than 60 days from the beginning of the experiment and was associated with water concentrations of endrin ranging from 1 to 2 ppb. This toxicity delayed the termination of the experiment for twice the usual 33 days and provided a striking parallel to the Mississippi River fish kills associated with the leaching of endrin wastes *(10)*. This experiment demonstrated the substantial predictive value of biological observations during the model ecosystem evaluation.

Mirex (1,2,3,4,5,5,6,7,8,9,10,10-dodecachloro-octahydro-1,3,4-metheno-2H-cyclobuta-[c,d]-pentalene) is one of the most stable compounds yet evaluated in the model system. Autoradiographs of the TLC plates containing extracts of organisms showed only unchanged mirex and traces of polar material at the origin of the chromatograms (Fig. 2). Table 4 indicates that mirex was stored as 98.6% of the total ^{14}C in *Gambusia*, 99.6% in *Culex*, 99.4% in *Physa*, and 97.9% in *Oedogonium*. The BI values were 0.0145 in fish and 0.006 in snail, and the EM values were 219 for fish and 1165 for snail. A similar degree of stability was found when mirex was fed to the rat *(11)*, where it was stored and slowly excreted virtually unchanged. These data suggest that mirex is a highly persistent micropollutant.

The data of Table 5 indicate the model ecosystem fate of γ-1, 2, 3, 4, 5, 6-hexachlorocyclohexane or lindane. Lindane was stored as 91.5% of the ^{14}C in *Gambusia* and 20.6% in *Physa*. The compound with R_f=0.55 found as a major degradation product in *Physa* had the same R_f as the major impurity in the ^{14}C preparation of lindane before it was repurified. This degradation product cochromatographed with γ-pentachlorocyclohexene prepared from lindane by reaction with sodium

Table 4. *Distribution of mirex and degradation products.*

	Mirex equivalents, ppm				
	H_2O	*Oedogonium* (alga)	*Physa* (snail)	*Culex* (mosquito)	*Gambusia* (fish)
Total ^{14}C	0.018	9.70	18.40	13.60	3.50
Mirex (R_f = 0.95)[a]	0.0157	9.49	18.29	13.54	3.45
Polar (R_f = 0.0)	0.0023	0.21	0.11	0.06	0.05

[a]TLC with chloroform.

Table 5. Distribution of lindane and degradation products.

	Lindane equivalents, ppm				
	H_2O	*Oedogonium* (alga)	*Physa* (snail)	*Culex* (mosquito)	*Gambusia* (fish)
Total ^{14}C	0.0232	0.375	3.70	0.75	1.02
Pentachloro-cyclohexene (R_f=0.55)[a]	—	—	2.50	—	—
Lindane (R_f=0.47)	0.00167	—	0.762	—	0.935
Unknown II (R_f=0.27)	0.000084	—	—	—	—
Unknown III (R_f=0.19)	0.00304	—	—	—	—
Unknown IV (R_f=0.14)	0.00276	—	—	—	—
Unknown V (R_f=0.09)	0.00636	—	0.248	—	—
Polar (R_f=0.0)	0.00877	0.375	0.185	—	0.085
Unextractable	0.0006	—	—	—	—

[a]TLC with hexane-acetone, 9:1.

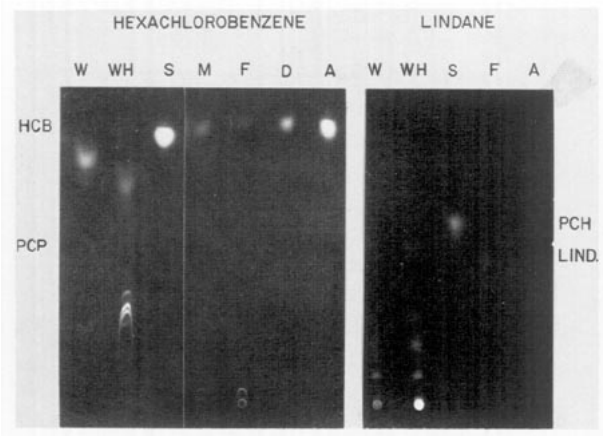

FIGURE 3. Autoradiographs of extracts from model ecosystems treated with ^{14}C hexachlorobenzene and lindane. Abbreviations are A (alga), F (fish), S (snail), M (mosquito), D (daphnia), W (water), and WH (water hydrolyzed with HCl). PCP is pentachlorophenol and PCH is pentachlorocyclohexene.

hydroxide *(11)*, which is the key degradation product of lindane in the rat *(12)*. *Gambusia* contained only lindane and a substantial amount of polar radioactivity. The four unknowns of decreasing R_f found in the water phase (Fig. 3) may be trichlorothiophenols as found by Grover and Sims *(13)* to be further degradation products of lindane in the rat.

The BI values for lindane were 0.091 in fish and 0.052 in snail, and the EM values were 560 in fish and 456 in snail. Thus lindane is considerably more biodegradable than the cyclodiene compounds.

Hexachlorobenzene has been used as a substitute fungicide and seed dressing to replace the alkylmercury compounds. Abbott et al. *(14)* have found residues in human adipose tissues in Great Britain ranging up to 0.29 ppm. It is evidently a persistant environmental micropollutant. In the model system it was found in substantial quantities in the various organisms with little evidence of degradation products except highly polar materials and conjugates (Table 6). Hexachlorobenzene comprised 85.1% of the total radioactivity in *Oedogonium*, 90.8% in the snail, 87.2% in *Daphnia*, 58.3% in *Culex*, and 27.2% in *Gambusia*. The water phase, however, contained an appreciable quantity of pentachlorophenol (R_f = 0.5), identified by cochromatography. This compound was not found in free form in any of the organisms of the system. Hydrolysis of the polar products in the water, with HCl, showed a family of related compounds with R_f values of 0.27, 0.22, 0.17, and 0.10 (Fig. 3) which are other chlorinated phenols. Mehendale *(15)* has tentatively identified, 2,4,5-trichlorophenol along with pentachlorophenol as urinary degradation products of hexachlorobenzene in the rat. The BI values for hexachlorobenzene were 0.46 in fish and 0.10 in snail, and the EM values were 287 in fish and 1247 in snail.

Ecological magnification in organisms of the food chain appears to be the most pernicious environmental effect resulting from the general usage of the organochlorine pesticides. This phenomenon, which is too well-known to need documentation here, is clearly revealed in the values of Table 7, showing the bioconcentration of the organochlorine pesticides from water to *Physa* snail and *Gambusia* fish. Bioconcentration clearly results from two important properties of

Table 6. Distribution of hexachlorobenzene and degradation products.

	Hexachlorobenzene equivalents, ppm					
	H_2O	*Oedogonium* (alga)	*Physa* (snail)	*Culex* (mosquito)	*Daphnia* (water flea)	*Gambusia* (fish)
Total ^{14}C	0.00644	1.827	4.099	0.737	0.696	3.154
Hexachloro-benzene (R_f = 0.80)[a]	0.00298	1.556	3.72	0.429	0.598	0.857
Pentachloro-phenol (R_f = 0.50)	0.00034	—	—	—	—	—
Unknown I (R_f = 0.10)	0.00023	—	—	—	—	0.446
Unknown II (R_f = 0.05)	—	—	—	0.0385	—	0.857
Polar (R_f = 0.0)	0.00143	0.271	0.378	0.269	0.098	0.995
Unextractable	0.00197	—	—	—	—	—

[a]TLC with benzene:acetone, 1:1.

these organochlorine micropollutants, their high lipid solubility and water insolubility, i.e., a large lipid/water partition coefficient, and their resistance to degradation by the multifunction oxidase enzymes through which organisms protect themselves against xenobiotic pollutants. All of the organochlorine pesticides studied here have a high lipid/water partition coefficient and this factor does not relate convincingly to values for ecological magnification which cover three orders of magnitude (Table 7). As suggested by Hamelink et al. *(16)*, the water insolubility of these highly lipid soluble compounds is likely to be the key driving force in producing lipid storage, e.g., in fish from successive partitionings from water to blood and blood to tissue lipids.

This relationship holds surprisingly well for the organochlorine pesticides studied here, together with DDT, DDE, DDD, methoxychlor, ethoxychlor, and methylchlor previously investigated *(2-4)*. As shown in Figure 4, when the logarithm of ecological magnification in *Gambusia* from the model ecosystem was plotted against the logarithm of water solubility, a clearly discernable linear relationship was found. Water solubility values were taken from Gunther et al. *(17)* and those for which no literature values could be found (Table 7) were measured at 25°C by using ^{14}C-labeled compounds shaken in distilled water for 24 hr, centrifuged, and filtered to remove particulate material before radioassay. On disregarding hexachlorobenzene, which is obviously aberrant, a line fitted to the remaining 11 points by least-squares analyses had the equation:

$$y = 5.99 - 1.176\,x$$

The correlation coefficient $r = 0.87$, and an F-test of significance gave a p value of <0.0005.

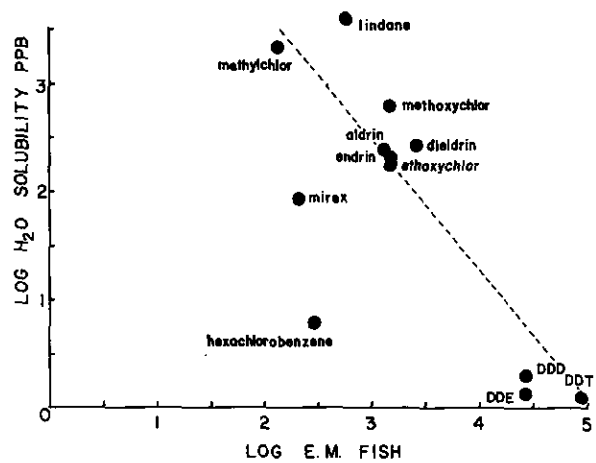

FIGURE 4. Plot of ecological magnification (EM) in fish from model ecosystem vs. water solubility in ppb. The dotted line was computed by method of least squares.

Table 7. Quantitative values for ecological magnification (EM) and biodegradability index (BI) for eight organochlorine pesticides in fish and snail.

	H_2O Solubility, ppm[a]	*Gambusia*		*Physa*	
		EM	BI	EM	BI
Aldrin					
As aldrin	0.20	3140	0.00014	44,600	0.0017
As dieldrin		5957	0.00013	11,149	0.00016
Dieldrin	0.25	2700	0.0018	61,657	0.0009
Endrin	0.23	1335	0.009	49,218	0.0124
Mirex	0.085	219	0.0145	1,165	0.006
Lindane	7.3	560	0.091	456	0.052
Hexachlorobenzene	0.006	287	0.46	1,247	0.10
DDT	0.0012	84,545	0.015	34,545	0.044
DDE	0.0013	27,358	0.032	19,529	0.017
DDD or TDE	ca. 0.002	83,500	0.054	8,250	0.024

[a]Values selected from Gunther et al. *(17)* or determined at 25° C. by radioassay for mirex, hexachlorobenzene, DDE.

Biomagnification of Organochlorine Pesticides by Aquatic Organisms of Model System

The question arises as to the extent to which the ecological magnification found in the organisms of the ecosystem (Table 7) is the result of successive concentration through food chains or of direct absorption through cuticle and gills. Some indications may be gained from experiments to determine the relative rates of concentration occurring in *Daphnia*, *Culex* and *Gambusia* exposed for 24-72 hr in one liter of standard reference water *(18)* containing approximately 1-3 ppb of the radiolabeled pesticide. The results in Table 8 show the extent of biomagnification of the pesticide in the three organisms. The various compounds are clearly concentrated manyfold directly from water into the organism and the degree of concentration varies with the individual pesticide. Bioconcentration is greatest with *Daphnia*, which has the largest ratio of surface to volume. There is a substantial trend toward increased accumulation with time, but some of the results show an apparent decrease with time suggesting degradation and excretion. The data with *Gambusia* suggest that a 3-day exposure to levels of pesticide contamination of the same general order of magnitude as found in the model ecosystem experiments does not produce levels of pesticide storage as high as those from ingestion of food chain organisms containing the pesticide. For the fish *Gambusia*, comparison of the values in Tables 7 and 8 suggests the following ratios for ecological magnification through food to biomagnification by direct uptake from water: aldrin 6.7X, endrin 2X, mirex 0.41X, lindane 3.3X, hexachlorobenzene 3.1X, DDT 250X, and DDE 125X. Biodegradation is, of course, materially enhanced by successive passage through food chain organisms.

Degradation in Salt Marsh Caterpillar

The contribution of the second element in the food chain, *Estigmene acrea*, to the degradation of each pesticide was measured by feeding individual larvae approximately 100 μg of radiolabeled pesticide and analyzing excreta and body homogenate by TLC *(3)*. The organochlorine pesticides are very resistant to metabolism in the insect body as shown by the radioautographs of excreta and body homogenate shown in Figure 5. Diel-

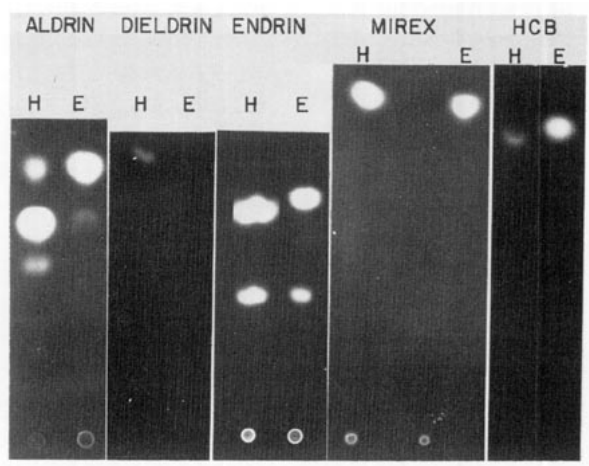

FIGURE 5. Autoradiographs of extracts from excreta (E) and body homogenates (H) of salt marsh caterpillars fed radiolabeled organochlorine pesticides.

Table 8. Biomagnification of pesticides from water into water flea, mosquito, and fish.

Pesticide	Pesticide concentration in H_2O ppb	Biomagnification *Daphnia* 1 day	*Daphnia* 2 days	*Daphnia* 3 days	*Culex* 1 day	*Culex* 2 days	*Culex* 3 days	*Gambusia* 1 day	*Gambusia* 2 days	*Gambusia* 3 days
Aldrin	1.7	1829	9118	—	970	1062	853	265	459	461
Endrin	1.04	2600	1200	330	2100	500	310	800	1000	680
Mirex	0.9	4900	—	2200	205	345	210	130	290	530
Lindane	1.4	430	800	810	125	120	95	233	177	166
Hexachlorobenzene	0.85	151	322	236	20	11	16	172	66	93
DDT	2.0	1170	1570	1330	515	165	110	300	300	344
DDE	0.90	282	984	—	187	132	74	111	108	217

drin, mirex, and hexachlorobenzene passed through the caterpillar essentially without change. Aldrin was converted to dieldrin, 4.6% in the excreta and 85% in the body homogenate. Endrin was substantially less stable than dieldrin and only 77% survived passage through the gut and 69% remained intact in the body homogenate. DDT in *Estigmene acrea* comprised >99% of the excreta but was converted 44.9% to DDE in the body homogenate *(3)*. DDE itself remained unchanged in both excreta and homogenate.

Conclusions

The results of laboratory model ecosystem investigations with radiolabeled organochlorine pesticides support the value of this technique for study and measurement of the environmental toxicology of micropollutants. The organochlorine pesticides were shown to accumulate in the tissues of fish and snail to levels from 200 to 84,000 times greater than those in the water of the model system. There was a satisfactory inverse correlation between levels of ecological magnification and the water solubility of the compounds investigated. The model ecosystem studies demonstrated the chemical pathways for the environmental degradation of the pesticides and provided estimates of the acute toxicological hazards of the pesticides and their transformation products to the organisms of the ecosystem. The high values for ecological magnification and the low biodegradability indices for the organochlorine pesticides are in marked contrast to the low ecological magnification values and high biodegradability indices found with common organophosphorus and carbamate insecticides. Therefore, the laboratory model ecosystem provides a useful screening device for evaluating the environmental toxicology of new candidate pesticides.

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