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IMPACT AND AMELIORATION OF SEDIMENT AND AGRO-CHEMICAL POLLUTION IN CARIBBEAN COASTAL WATERS

TOXICITY REVIEW FOR AGRO-CHEMICALS IN ST LUCIA AND JAMAICA

MARCH 2003

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EXECUTIVE SUMMARY

- This review examines 92 pesticides used in the Caribbean categorized in terms of field of use (fungicide, herbicide, insecticide, nematicide, plant growth regulators).
- Toxicity, quantities imported, environmental fate, social factors, priorities for monitoring and recommendations are discussed, taking into consideration quantities imported, toxicity, persistence in the environment and social factors.
- The fates of pesticides have been determined from kinetic and thermodynamic data in temperate and tropical ecosystems. It has been recognized that there is usually a big difference between kinetic and thermodynamic data for tropical ecosystems and temperate ecosystems. Hence there is an urgent need to collect kinetic data on pesticide behaviour in tropical ecosystems to determine the pesticide fate in tropical ecosystems.
- There are clear indications that herbicides constitute a very large proportion of pesticides imported into Caribbean each year, and this trend is likely to continue. Given the persistence of some of these pesticides in the environment it is necessary that their use and residues be monitored.
- Since Organochlorides (OC's) are almost banned throughout the Caribbean, there is less urgency to monitor OC pollution of the environment. However, we believe there are sites throughout the Caribbean that have heavy OC pollution loads, for example, the Kingston Harbour and Hunt's Bay, both in Jamaica.
- The importance of a general education programme to sensitise the public to both acute and chronic toxic effects of pesticides cannot be overstated. Training and certification of Extension Officers, farmers and pesticide applicators must accompany this.
- The importation and use of pesticides in the Caribbean can be reduced if the public and farmers in particular, are educated about the environmental fate of pesticides and their adverse health effects, as well as trained in proper application techniques.
- Integrated Management of Pests and Pesticides (IMPP) should be implemented regionally on a phased basis, incorporating biological controls and prudent cultural practices.
- We have no knowledge of any systematic study of fertilizer load having been done in the Caribbean, although the quantities of imports are increasing. Systematic study is required to establish the level of risk between the actual load of fertilizers in the Caribbean and danger to the ecosystem and human health.

ACRONYMS

a.i.	active ingredient
CREAMS	Chemicals, Run-off and Erosion from Agricultural Management Systems
EHBA	extrahepatic biliary atresia
IPM	Integrated Pest Management
IMPP	Integrated Management of Pests and Pesticides
OC	organochloride
OP	organophosphate
PAHO	Pan American Health Organization
PANNUPS	Pesticide Action Network Updates Service
PCA	Pesticide Control Authority, Jamaica
PCB	Pesticide Control Board
PRG	Pesticide Research Group, Chemistry Department, University of the West Indies, Mona
RADA	Rural Agricultural Development Authority
RSCIS	Royal Society of Chemistry Information Services
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
USNLM	U.S. National Library of Medicine
UWI	University of the West Indies
WSSA	Weed Science Society of America
WHO	World Health Organization

BACKGROUND

This report contributes to a three-year research project *Impact and amelioration of sedimentation and agro-chemicals in Caribbean coastal waters* which is funded by DFID's NRSP LWI programme (R7668). It follows on from an earlier LWI project *Review of the impacts of pollution by sediments and agro-chemicals of tropical coastal waters with reference to the Caribbean region* (R7111). The present project is managed and conducted by two organisations: the University of York, responsible for the sedimentation aspects of the project; and MRAG Ltd, responsible for agro-chemical components of the project. Agro-chemical related activities are undertaken in St Lucia and Jamaica. The project commenced in June 2000 and ends in July 2003, with the publication of guidelines for best management practices for agro-chemical management.

This toxicity review is the work of the Pesticides Research Group at the University of the West Indies, Mona, Jamaica. Comments and insight have been received from a number of project partners[1] and these have contributed to the final report. The document aims to provide an understanding of the toxicity of agro-chemicals[2] in use in St Lucia and Jamaica, and the wider Caribbean, and describes management practices for different pesticides. This information will contribute to the wider project objective of a broader technical understanding of the extent in which agro-chemicals are applied and impacting the environment in two representative countries in the Caribbean. St Lucia has been selected as a detailed case study, including thorough investigation of the fate of agro-chemicals in the environment. Jamaica has been chosen to provide a contrasting example of intensive agricultural land use on a large Caribbean Island. Agro-chemical inputs will be estimated there from importation data and literature review and differences in appropriate best management practices explored.

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[1] Jamaica: UWI, CARDI, CCAM; St Lucia: MAFF, CEHI, UK: MRAG Ltd.

[2] For the purpose of the current project, the term agro-chemical includes pesticides and fertilisers used in agriculture.

INTRODUCTION

Ninety-two (92) pesticide formulations used in the Caribbean, have been selected for review based on their toxicity and/or quantities imported. Throughout this review the pesticides are grouped according to field of use to facilitate easy comparison of the properties of the pesticides used for similar purposes. Each formulation, its active ingredients, WHO Toxicity Class, and the Priority which we think each of the region's governments should attach to monitoring the active ingredients and their effects on human health and the environment, is given in Table 1 below. We do not believe that there is sufficient high quality quantitative or qualitative data to assign the priorities according to strict mathematical functions. Hence, we have assigned the priorities based on the expert judgement of the Pesticide Research Group taking into consideration three of the more important quantitative factors – import quantity, toxicity, and half-life values (see Figures 1a and 1b) – as well as knowledge of the social conditions within which the agrochemicals are being used. This means that the priority settings should be adjusted periodically to reflect changes in the knowledge base of the agrochemicals' properties, importation and usage patterns, as well as social conditions.

Table 1: Select Active Ingredients and Formulations used in the Caribbean

Field of Use	Product Name	Active Ingredient(s)	WHO Toxicity Class[3]	Priority
FUNGICIDES	Benlate	Benomyl	3	L
	Bravo	Chlorothalonil	3	L
	Cuprosan	copper oxychloride	3	L
		Maneb		
		Zineb	5	
	Trimiltox-Forte	copper oxychloride	3	L
		Mancozeb	3	L
	Phyton 28	copper sulphate pentahydrate	2	L
	Fusilade	fluazifop-p-butyl	N/A	
	Aliette	Fosetyl	3	L
	Anvil	Hexaconazole	3	L
	Fungaflor 75 SP	Imazalil	3	L
	Ridomil MZ 72 WP	Mancozeb	3	L
		Metalaxyl	3	L
	Ridomil 2E	Metalaxyl	3	L
	Tilt 250 EC	propiconazole	3	M/H
HERBICIDES	2,4-D Amine	2,4-D	2	L
	Actril DS 60EC	2,4-D	2	L
	Ametrex 50 SC	ametryn	3	L
	Amigan 50 SC	ametryn	3	L
		terbutryn	3	L
	Atramet Combi 50 SC	ametryn	3	L
		atrazine	3	L
	Gesapax 80 WDG	ametryn	3	L

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[3] See Table 3 for WHO definition of toxicity classes.

Field of Use	Product Name	Active Ingredient(s)	WHO Toxicity Class[3]	Priority
	Talent	asulam	3	L
		paraquat	2	H
	Asulox 40C	asulam	3	L
	Atranex 50 SC	atrazine	3	L
	Aatrex nine-o 85sg	atrazine	3	L
	Hyvarx 80 WP	bromacil	3	L
	Krovar 1 DF	bromacil	3	L
		diuron	3	L
	Reglone	diquat	3	M
	Karmex DF	diuron	3	L
	Diuron 80 WP	diuron	3	L
	Diurex 80 WP	diuron	3	L
	Fusilade 2000 EC	fluazifop-butyl	3	L
	Glyphosate 41% SL	glyphosate	3	M
	Rival	glyphosate	3	M
	Roundup	glyphosate	3	M
	Glytex	glyphosate	3	M
	Folar 460 SC	glyphosate	3	M
		terbuthylazine	5	
	Touchdown	glyphosate-trimesium	3	M/H
	Velpar L	hexazinone	3	L
	Image	imazaquin	3	L
	Lorox	linuron	3	L
	Dual	metolachlor	3	L
	Sencor 75 Turf	metribuzin	3	L
	Ally 60 DF	metsulfuron	3	L
	Daconate 4 35 LQ	MSMA (sodium hydrogen methylarsonate)	3	L
	Gai-quat	paraquat	2	H
	Gramaquat Super	paraquat	2	H
	Gramocil	paraquat	2	H
		diuron	3	L
	Paraquat 24%	paraquat	2	H
	Gramoxone	paraquat	2	H
	Nabu-S	sethoxydim	3	L
	Igran	terbutryn	3	L
	Folar 460 SC	terbuthylazine	5	
	Terbutrex 50	terbutryn	3	L
	Treflan 45	trifluralin	3	M/H
INSECTICIDES	Avid	abamectin	1	L
	Agri-Mek	abamectin	1	L
	Triatix	amitraz	3	L
	Sevin 85 WP	carbaryl	3	L
	Dursban	chlorpyrifos	2	M
	Knock-out	chlorpyrifos	2	M

Field of Use	Product Name	Active Ingredient(s)	WHO Toxicity Class[3]	Priority
	Pestban	chlorpyrifos	2	M
	Prevail	cypermethrin	2	L
	Demon EC	cypermethrin	2	L
	Trigard	cyromazine	3	L
	Basamid	dazomet	3	L
	Basudin	diazinon	3	L
	Diazinon	diazinon	3	L
	Thiodan 50 WP	endosulfan	2	H
	Mocap	ethoprophos	2	H
	Danitol 10EC	fenpropathrin	3	M
	Actellic 10 EC	pirimiphos-methyl	3	L
	Admire 2 Flowable	imidacloprid	2	L
	Marathon	imidacloprid	2	L
	Demand CS	lambda cyhalothrin	2	L
	Karate	lambda cyhalothrin	2	L
	Malathion	malathion	3	L
	Lannate	methomyl	1	H
	Diverside K	n-octyl bicycloheptene dicarboximide	3	L
		piperonyl butoxide	N/A	
	Pounce	permethrin	2	L
	Prelude	permethrin	2	L
	Diverside K	piperonyl butoxide	N/A	
	Actellic 50 EC	pirimiphos-methyl	3	L
	Tambo 440 EC	profenofos	3	M
		cypermethrin	2	L
	Selecron	profenofos	3	M
NEMATICIDES	Rugby	Cadusafos	1	L/M
	Furadan 10G	Carbofuran	1	L
	Miral	Isazofos	1	L
	Vydate L	Oxamyl	1	H
PLANT GROWTH REGULATORS	Tamex	Butralin	3	L
	Cycocel	chlormequat chloride	3	L
	Trecit	ethyl 1-naphthalene acetate	3	L
	Pix	mepiquat chloride	3	L
	Mepichlor	mepiquat chloride	3	L
	Bonzi	Paclobutrazol	3	L

These pesticides have been reviewed with particular attention given to their toxicity with regards to the marine environment. Fertilizers used in the Caribbean are also reviewed. Many of the situations and examples presented are drawn from the reviewers' experience of the Eastern Caribbean and Jamaican contexts, and take into consideration the particular needs, frameworks and capabilities of the different Caribbean territories. This review makes extensive use of published studies done on these pesticides in temperate countries, but also draws on the literature available for tropical regions and unpublished laboratory studies done by the Pesticide Research Group of the Chemistry Department, UWI, Mona. Most of the existing research that has been published was performed with temperate environments in mind, and apposite parameters chosen. Nevertheless, the data are included in this review as they contribute to our overall understanding of the behaviour of the pesticides and suggest in varying degrees what may pertain in the tropics. Also, wherever these studies have any particular relevance to the tropical environment the relevant data has been highlighted.

As is frequently the case when discussing pesticides, it is important to distinguish between active ingredients and formulations. Formulations can be identified by their tradenames, e.g. Trimiltox, which has two active ingredients, viz. copper oxychloride and mancozeb. (In this review the names of formulations are written with initial capitals, and active ingredients with common letters.) In addition to active ingredients, pesticide formulations may also include synergistic agents which increase the activity, that is, the effectiveness of the pesticide. In addition to the active ingredients, and the synergistic agents, there may also be inert ingredients. Hence, the quantity of pesticide formulations will usually be larger than the quantity of active ingredients. It is also important to note that the toxicity of active ingredients may be substantially different from the toxicity of the formulation. Most of the toxicity data in this review is with respect to active ingredients.

A cursory survey of the quantities of pesticides recently imported into two Caribbean territories, Jamaica and St Lucia, which we believe are representative of the Caribbean region, illustrates the urgency and importance of this Review at this time. The April 2000-March 2001 *Annual Report* from the Jamaican PCA presents data on the quantities of pesticide formulations imported into Jamaica for both agricultural and non-agricultural use. It indicates that 2,817,076.40 kg of pesticides was imported between April 2000 and March 2001 (PCA, 2001). The 1999 *Annual Report* from the PCA indicates that 3,115,271.2 kg of pesticides was imported for the calendar year 1999 (PCA, 1999). In that year, approximately 90% of the total pesticides went to the agricultural sector. Table 2, presents the data for several other Caribbean territories for the period 1996-1999.

Table 2: Total imports of pesticides into selected Caribbean countries for the years 1996 – 1999

	Total Pesticide Imports (kg)			
Country	1996	1997	1998	1999
Antigua and Barbuda[4]			272, 446	533, 680
Dominica[5]	574, 200	499, 323	410, 496	4, 783, 153
St. Vincent and the Grenadines[6]	679, 365	1, 480, 189	783, 703	2, 109, 236
Grenada[7]	114, 945	65, 891	150, 538	76, 475

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[4] Personal communication with Pesticides Control Board, Antigua and Barbuda, 2000.

[5] Personal communication with Pesticides Control Board, Dominica, 2000.

[6] Personal communication with Pesticides Control Board, St. Vincent and the Grenadines, 2000.

Suriname[8]	6, 271, 481	15, 946, 805	7, 894, 401

We have simplified the detailed active ingredients import data for Jamaica (Table 4 and Fig. 1) and St Lucia (Table 5 and Figures 3a and 3b) for the period 1998-2000, to provide an overview of the quantities and volumes imported, by assuming that one litre of active ingredient is equal to one kilogram. This simplification does not allow for rigorous statistical analysis, but the errors introduced are marginal enough to allow us to make simple conclusions. The Jamaican figures, 1998 (516, 294 kg), 1999 (1, 571, 433 kg) and 2000 (1, 361, 932 kg), as well as the St Lucian 1998 (50, 089 kg), 1999 (189, 685 kg), and 2000 (50, 605 kg) confirm the belief of many members of the agricultural industry and environmental watch groups throughout the Caribbean, that increasing quantities of pesticides, and agro-chemicals in general, are being used in the Caribbean.

These figures in addition to the data in Table 2 also show that in 1999, for five of the six countries for which 1999 import data is available there was a dramatic increase in the total quantities of pesticides imported. We cannot say definitively why this increase occurred. However, by looking closely at Fig. 1 and Table 4 which give the import statistics for Jamaica, we were able to isolate the pesticides which contributed most to the increase *in Jamaica*. Thus, for the three year period 1998-2000, the following pesticides were the largest contributors to the total quantity of pesticides imported: mancozeb, 2,4-D, diuron, glyphosate, paraquat, and terbutryn. The statistics do not suggest that mancozeb is important to explaining the dramatic increase in 1999 imports, since the quantity of mancozeb imported in 1999 did not increase significantly over that imported in 1998, nor did it decrease significantly in 2000 as was the case with the other pesticides named (except for terbutryn which increased further in 2000). For the other pesticides, the quantities imported increased over the previous year as follows: 2,4-D (91%), diuron (41%), glyphosate (253%), paraquat (106%), terbutryn (145%). These pesticides are all herbicides. An increase in acreage planted, replanting of cash crops, or heavy rains could all have created this demand for herbicides. In St Lucia, a large proportion of their pesticide imports is used to protect bananas (their main export crop) rather than sugarcane (Jamaica's main export crop). The largest increase by far as evident in Fig. 2-App. A was in the imports of the nematicide ethoprophos (1475%). The quantity of ethoprophos (102, 085 kg) imported in 1999 was greater than the total quantity of all pesticides imported into St Lucia in 1998 (50, 089 kg) and in 2000 (50, 605 kg). Imports of carbofuran, another nematicide, did not increase significantly over 1998 in 1999 (3%), but remained high, decreasing by 94% of the 1998 value in 2000. Imports of imazalil, used in fungicidal formulations, increased by 636% and was the fifth largest contributor to the spike. Imports of herbicides also increased significantly and contributed to the spike: diuron (216%), glyphosate-trimesium (271%), paraquat (157%). What this data suggests is that St Lucia, and perhaps the other banana-growing Caribbean territories (for which we have no detailed data), had to battle increased nematode infestation and weeds, in 1999. Generally speaking, there are several as yet unmentioned possible reasons for dramatic increases in agrochemical imports: dumped agrochemicals (as in 2000-2001, when fertilizers were dumped in Jamaica from the Dominican Republic), a concerted drive to eradicate a particular pest (such as that now underway in Jamaica to eradicate screw-worm). Additional data is needed in order to make more definitive explanations.

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[7] Personal communication with Pesticides Control Board, Grenada, 2000.

[8] Personal communication with Pesticides Control Board, Suriname, 2000.

Finally, before we get into the details of the toxicity, environmental fate and social factors with regards to these pesticides, let us emphasize how important it is to scrutinize the sources of information and opinion in any discussion of pesticides, their toxicity, environmental fate and impact on human society. Pesticide research is typically conducted or paid for by pesticide manufacturers, who have the resources to widely publicise these studies, and suppress unfavourable studies (Cox, 1998). The pesticide manufacturing industry, worth more than US$ 15 billion (Robinson, 1997), contributes large sums of money to the EPA and to American politicians. The approval process for pesticide registration requires the *pesticide manufacturers* to supply the relevant data to the EPA, which makes its decisions based on those findings. As early as 1983, the EPA publicly announced that it had discovered "routine falsification of data" by Industrial Biotest Laboratories (IBT), one of the largest laboratories performing tests in support of pesticide registration (Cox, 1998). In 1992, Craven Labs which performed studies for 262 pesticide companies, was fined 15.5 million dollars and ordered to pay 3.7 million dollars in restitution for "manually manipulating scientific equipment to produce false reports" and "falsifying laboratory notebook entries" (Cox, 1998). As late as 1998, Monsanto Co. was in negotiations with the New York attorney-general over "false and misleading claims" regarding its glyphosate product, Roundup (Cox, 1998). Thus, even pesticides approved by government agencies such as the EPA, may be very dangerous to humans as well as the environment. For this reason, all data regarding half-lives, toxicity and other effects of pesticides which are issued by manufacturers must be treated as suspect, and compared with independent studies carried out by academia, state agencies and NGO's.

Generally, whereas fertilizers are not thought to be dangerous to the health of applicators, it is generally accepted that many pesticides are detrimental to human health in a dose-dependent relationship. Opinions on pesticides and human health failure tend to fall into two general categories. On the one hand, generally there are those who are very ardent supporters of reduced pesticide use because of what they see as very detrimental results in the ecosystem and on human health. This includes NGO's such as the Northwest Coalition for Alternatives to Pesticides (NCAP) which publishes the *Journal for Pesticide Reform*, the worldwide organization Pesticide Action Network which publishes the HTML newsletter *PANNUPS*, individuals such as immunologist Wayne Sinclair, who presents his findings on pesticides used in Florida on his website (www.chem-tox.com), organic farmers and patrons of organic foods. On the other hand are those who advocate the use of pesticides as a very effective means of increasing and preserving the current levels of food production. This group includes many powerful agrochemical manufacturers such as Monsanto Co. and DuPont, farmers of large and small acreages, governments of poor countries and of the United States (which dominates world-production and export of agrochemicals), and the related industries and groups which benefit directly from the agrochemical industry. Our position is that we recognize the dangers posed by pesticide use to human health and the environment, but also the many social, economic and environmental benefits to be gained by farmers and the general populace from prudent and educated use of pesticides.

SECTION 1: TOXICITY

There are two types of toxicity: acute and chronic. A discussion of the relative importance of the two and their differences is presented in Section 3: Social Factors. Displayed in Tables 6 to 11 is the acute toxicity of the active ingredients with respect to rats, birds, aquatic organisms, and bees, in the hope of providing a good idea of the possible impact of the active ingredients on various animal species in the environment. Acute toxicity data for other species, including the small freshwater crustacean *Daphnia magna*, are included in the Detailed Notes for each a.i. Discussion of the chronic toxicity primarily with regards to human health is under Section 3: Social Factors.

Measures of acute toxicity include LC_{50}, LD_{50}, and EC_{50}. LC stands for lethal concentration. LC_{50} is the concentration of a material in air which causes the death of 50% of a group of test animals. The material is taken in over a set period of time, for our purposes, usually 96 or 48 hours. The LC_{50} helps determine the short-term poisoning potential of a material. LD stands for lethal dose. The LD_{50} is the amount of a material, given all at once, which causes the death of 50% of a group of test animals. The LD_{50} can be determined for any route of entry, but oral (given by mouth), dermal (applied to skin), and dietary LD_{50}'s are most common. The LD_{50} is another measure of the short-term poisoning potential of a material. EC stands for effect concentration. EC_{50} is the concentration that produces a sub-lethal response, that is, immobilization, loss of equilibrium, or possibly death, in 50% of the observed organisms.

As we are especially concerned with the effects of agrochemicals on the marine environment, several peer-reviewed guidelines for interpreting the data for aquatic organisms have been paraphrased (International Occupational Safety and Health Information Centre, 2001). Agrochemicals can be categorized as follows:

1) very toxic: 96 hr LC_{50} (for fish) <= 1 mg/l (OECD guideline) and/or 48 hr EC_{50} (for *Daphnia magna*) <= 1mg/l
2) toxic: 96 hr LC_{50} (for fish) 1mg/l < LC_{50} <=10mg/l or 48 hr EC_{50} (for *Daphnia magna*) 1mg/l < EC_{50} <=10mg/l
3) harmful: 96 hr LC_{50} (for fish) 10mg/l < LC_{50} <=100mg/l or 48 hr EC_{50} (for *Daphnia magna*) 10mg/l < EC_{50} <=100mg/l

A. Fungicides

Please refer to Table 6: Toxicity of Fungicides to rats, birds, aquatic organisms, and bees.

Detailed Notes on Fungicides

benomyl

Toxicity to aquatic organisms: Benomyl is highly to very highly toxic to fish. The order of susceptibility to benomyl for various fish species from least susceptible to most susceptible is catfish, bluegill, rainbow trout, and goldfish (USNLM, 1995). The main breakdown product, carbendazim, had the same order of toxicity as benomyl. Crayfish have an LC_{50} greater than 100 mg/L. The estimated bioconcentration factor (BCF) ranges from 159 in rainbow trout up to 460 in bluegill sunfish, indicating that benomyl does not tend to significantly concentrate in living tissue benomyl (USNLM, 1995).

Toxicity to other organisms: A single application of benomyl to turf grass can substantially reduce some soil dwelling organisms. The compound is very lethal to earthworms at low concentrations over a long time period. The 7-day LC_{50} in earthworms is 1.7 mg/L and the 14-day LC_{50} is 0.4 mg/L (USDA, 1984). Benomyl also decreases the mixing of soil and thatch. The effects last for up to 20 weeks (Potter *et al.*, 1990).

chlorothalonil

Toxicity to birds: Chlorothalonil is practically non-toxic to birds. Most avian wildlife are not significantly affected by this compound (USNLM, 1995).

Toxicity to aquatic organisms: Chlorothalonil and its metabolites are highly toxic to fish, aquatic invertebrates, and marine organisms. Fish, such as rainbow trout, bluegill, and channel catfish are noticeably affected even when chlorothalonil levels are low (less than 1 mg/L). Chlorothalonil does not store in fatty tissues and is rapidly excreted from the body. Its bioaccumulation factor is quite low (USNLM, 1995).

copper oxychloride

Toxicity to mammals: Low oral doses (3-5 g) usually result in symptoms of gastroenteritis without affecting absorption, after which the active substance is for the most part eliminated by vomiting. Moderate doses (5-8 g) cause damage to capillaries and digestive-tract mucous membranes, signs of heavy metal poisoning, and loss of water and electrolytes. Death may occur at high doses (8-12 g) (Kidd and James, 1994).

copper sulphate pentahydrate

Toxicity to mammals: Since oral intake leads to nausea, the acute oral LD_{50} is hard to determine. In feeding trials, rats receiving 500 mg/kg diet showed weight loss, whilst those receiving 1000 mg/kg exhibited damage to the liver, kidneys, and other organs (Kidd and James, 1994).

Toxicity to birds: Copper sulphate is practically non-toxic to birds. It poses less of a threat to birds than to other animals.

Toxicity to aquatic organisms: Copper sulphate is highly toxic to fish (Pimentel, 1971). Even at recommended rates of application, this material may be poisonous to trout and other fish, especially in soft or acid waters. Its toxicity to fish generally decreases as water hardness increases. Fish eggs are more resistant than young fish fry to the toxic effects of copper sulphate (Gangstad, 1986). Copper sulphate is toxic to aquatic invertebrates, such as crab, shrimp, and oysters.

Toxicity to other organisms: Bees are endangered by Bordeaux mixture (Kidd and James, 1991). Copper sulphate may be poisonous to sheep and chickens at normal application rates. Most animal life in soil, including large earthworms, have been eliminated by the extensive use of copper containing fungicides in orchards (Pimentel, 1971).

fluazifop-p-butyl

Toxicity to aquatic organisms: Fluazifop-p-butyl may be highly to moderately toxic to fish, but only slightly toxic to other aquatic species, such as invertebrates. The reported 96-hour LC_{50} values for the technical product in fish species are 0.53 mg/L in bluegill sunfish and 1.37 mg/L in rainbow trout, indicating very high to high toxicity (USNLM, 1995). The 48-hour LC_{50} in *Daphnia magna* (an aquatic invertebrate) is reported as greater than 10 mg/L, indicating only slight toxicity (USNLM, 1995).

hexaconazole

Toxicity to aquatic organisms: LC_{50} for *Daphnia* 2.9 mg/L.

imazalil

Toxicity to mammals: Causes irreversible eye damage.

mancozeb

Toxicity to aquatic organisms: Mancozeb is moderately to highly toxic to fish and aquatic organisms. Reported 48-hour LC_{50} are 9 mg/L in goldfish, 2.2 mg/L in rainbow trout, 5.2 mg/L in catfish, and 4.0 mg/L in carp (USNLM, 1995). The reported 72-hour LC_{50} for mancozeb in crayfish is greater than 40 mg/L; the 48-hour LC_{50} is 3.5 mg/L in tadpoles (E. I. DuPont, 1983).

metalaxyl

Toxicity to aquatic organisms: Metalaxyl is practically non-toxic to freshwater fish. Freshwater aquatic invertebrates are slightly more susceptible to metalaxyl. *Daphnia magna*, (a small freshwater crustacean) has an LC_{50} of 12.5 to 28 mg/L, depending on the product formulation (USEPA, 1988a). This indicates that metalaxyl is slightly toxic to this organism. There is little tendency for metalaxyl to accumulate in the edible portion of fish. Metalaxyl did not accumulate beyond seven times the background concentration and it was quickly eliminated after exposed fish were placed in fresh (metalaxyl-free) water (USEPA, 1988a).

B. Herbicides

Please refer to Table 7: Toxicity of Herbicides to rats, birds, aquatic organisms, and bees.

The use of herbicides has increased worldwide and the Caribbean is no exception. More than half a million kilograms of herbicides were imported into Jamaica in 1998 by the PCA and importation has increased since then. All herbicides imported in Jamaica are of toxicity category 2 or 3. Gramoxone, Gramocil and Paraquat 24% (active ingredient paraquat) and Glytex, Rival, and Roundup (active ingredient glyphosate) are the some of the most popular herbicides in the Caribbean. Glyphosate is comparatively less toxic than paraquat and hence of medium priority for immediate monitoring. Paraquat, although quite popular as a herbicide, is quite toxic, and there seems to be no antidote available in the Caribbean. Atrazine is quite popular throughout the Caribbean as a herbicide due to its low toxicity. However, we have investigated the degradation of atrazine and other triazine derivatives and found these herbicides to be extremely stable. So, their application should be properly regulated. It is interesting to note that the photochemical degradations of these herbicides are quite fast: half-life ranges from 3-6 minutes using 125W Hg lamp. Hence it is quite possible that the degradation of these herbicides in soils in tropical ecosystems take place much faster than in temperate climates. Still, one has to worry about the accumulation of these pesticides in sediments.

Detailed Notes on Herbicides

2,4-D

Toxicity to aquatic organisms: Some formulations of 2,4-D are highly toxic to fish while others are less so. For example, the LC_{50} ranges between 1.0 and 100 mg/L in cut throat trout, depending on the formulation used. Channel catfish had less than 10% mortality when exposed to 10 mg/L for 48 hours (Stevens and Sumner, 1991; USEPA, 1988b). Green sunfish, when exposed to 110 mg/L for 41 hours, showed no effect on swimming response. Limited studies indicate a half-life of less than 2 days in fish and oysters (Nat'l Research Council Canada, 1978). Concentrations of 10 mg/L for 85 days did not adversely affect the survival of adult dungeness crabs. For immature crabs, the 96-hour LC_{50} is greater than 10 mg/L, indicating that 2,4-D is only slightly toxic. Brown shrimp showed a small increase in mortality at exposures of 2 mg/L for 48 hours (USNLM, 1995; Howard, 1991).

Toxicity to other organisms: Moderate doses of 2,4-D severely impaired honeybees brood production. At lower levels of exposure, exposed bees lived significantly longer than the controls (Kidd and James, 1991; USNLM, 1995).

ametryn

Toxicity to aquatic organisms: Ametryn is highly toxic to crustaceans and moderately to highly toxic to molluscs (Briggs, 1992).

atrazine

Toxicity to aquatic organisms: Atrazine is slightly toxic to fish and other aquatic life. Atrazine has a low level of bioaccumulation in fish. In whitefish, atrazine accumulates in the brain, gall bladder, liver, and gut (USNLM, 1995).

bromacil

Toxicity to birds: 8-day dietary oral LC50 for bromacil is over 10,000 ppm in mallards and quail (Clayton and Clayton, 1981). This indicates that it is practically nontoxic to these species.

Toxicity to aquatic organisms: Bromacil is slightly to practically nontoxic to fish. It is not toxic to aquatic invertebrates (USNLM, 1995).

Toxicity to other organisms: Bromacil is not toxic to honeybees (Kidd and James, 1991).

fluazifop-p-butyl

Toxicity to aquatic organisms: Fluazifop-p-butyl may be highly to moderately toxic to fish, but only slightly toxic to other aquatic species, such as invertebrates. The reported 96-hour LC_{50} values for the technical product in fish species are 0.53 mg/L in bluegill sunfish and 1.37 mg/L in rainbow trout, indicating very high to high toxicity (USNLM, 1995). The 48-hour LC_{50} in *Daphnia magna* (an aquatic invertebrate) is reported as greater than 10 mg/L, indicating only slight toxicity (USNLM, 1995).

glyphosate

Toxicity to aquatic organisms: Technical glyphosate acid is practically nontoxic to fish and may be slightly toxic to aquatic invertebrates. The 96-hour LC50 is 120 mg/L in bluegill sunfish, 168 mg/L in harlequin, and 86 mg/L in rainbow trout (WSSA, 1994). The reported 96-hour LC50 values for other aquatic species include greater than 10 mg/L in Atlantic oysters, 934 mg/L in fiddler crab, and 281 mg/L in shrimp (WSSA, 1994). The 48-hour LC50 for glyphosate in *Daphnia* (water flea), an important food source for freshwater fish, is 780 mg/L (WSSA, 1994). Some formulations may be more toxic to fish and aquatic species due to differences in toxicity between the salts and the parent acid or to surfactants used in the formulation (WSSA, 1994; Monsanto, 1985). There is a very low potential for the compound to build up in the tissues of aquatic invertebrates or other aquatic organisms (Monsanto, 1985).

Toxicity to other organisms: The reported contact LC_{50} values for earthworms in soil are greater than 5000 ppm for both the glyphosate trimethylsulfonium salt and Roundup (WSSA, 1994).

hexazinone

Toxicity to other organisms: The herbicide is toxic to larch trees (Larix spp.), and should not be used for weed control in forested areas (Kidd and James, 1991).

imazaquin

Toxicity to birds: Imazaquin is practically non-toxic to birds when used as recommended (USEPA, 1986a; Meister, 1995).

Toxicity to aquatic organisms: Imazaquin is practically non-toxic to fish when used as recommended (USEPA, 1986a; Meister, 1995).

Toxicity to other organisms: imazaquin is non-toxic to honeybees at 100 µg/bee. Imazaquin is practically non-toxic to wildlife when used as recommended (USEPA, 1986a; Meister, 1995).

linuron

Toxicity to aquatic organisms: Linuron is slightly toxic to fish and aquatic invertebrate species. The reported LC_{50} for linuron in trout and bluegill is 16 mg/L (USNLM, 1995; USEPA, 1985a). The median threshold levels, i.e. levels at which adverse, sublethal effects were apparent in 50% of the test animals, are greater than 40 mg/L in crawfish and tadpoles exposed over a 48-hour period (WSSA, 1994).

metolachlor

Toxicity to aquatic organisms: Metolachlor is moderately toxic to both cold- and warmwater fish, including rainbow trout, carp, and bluegill sunfish. Studies on algae and fish exposed to metolachlor in water indicate that very little is accumulated and that any accumulated material is excreted rapidly when the organisms are placed in clean water (USEPA, 1987a).

Toxicity to other organisms: Contact LC_{50} in earthworms is 140 ppm (U.S. Dept of Health, 1993).

metribuzin

Toxicity to aquatic organisms: Metribuzin is slightly toxic to fish (Kidd and James, 1991; WSSA, 1994). The reported 48-hour LC_{50} in *Daphnia magna* is 4.5 mg/L, indicating similar toxicity (WSSA, 1994). The 96-hour LC_{50} in marine/estuarine shrimp is 48.3 mg/L (USEPA, 1985b).

Toxicity to other organims: Metribuzin may be phytotoxic to non-target plant species (Kidd and James, 1991; WSSA, 1994).

metsulfuron

Toxicity to aquatic organisms: 48-hour LC_{50} for *Daphnia magna* >150 mg/L.

paraquat

Toxicity to aquatic organisms: The LC_{50} for the aquatic invertebrate *Daphnia pulex* is 1.2 to 4.0 mg/L (USNLM, 1995). In rainbow trout exposed for 7 days to paraquat, the chemical was detected in the gut and liver, but not in the meat of the fish. Aquatic weeds may bioaccumulate the compound. In one study, 4 days after paraquat was applied as an aquatic herbicide, weeds sampled showed significant residue levels (Stevens and Sumner, 1991). At high levels, paraquat inhibits the photosynthesis of some algae in stream waters (Stevens and Sumner, 1991).

sethoxydim

Toxicity to aquatic organisms: A 3-hour LC_{50} of 1.5 mg/L is reported in *Daphnia* (Kidd and James, 1991).

terbutryn

Toxicity to aquatic organisms: Terbutryn is moderately toxic to fish (Meister, 1992). However, except when applied directly to 6 inches of water, residues have been calculated to be insignificant even to the most sensitive aquatic animal species (Kidd and James, 1994).

trifluralin

Toxicity to aquatic organisms: Trifluralin is very highly toxic to fish and other aquatic organisms. Variables such as temperature, pH, life stage, or size may affect the toxicity of the compound. Trifluralin is highly toxic to *Daphnia*, with a 48-hour LC_{50} of 0.5 to 0.6 mg/L (Mayer and Ellersieck, 1986). The compound shows a moderate tendency to accumulate in aquatic organisms.

Toxicity to other organisms: At exposure levels well above permissible application rates (100 mg/kg), trifluralin has been shown to be toxic to earthworms. However, permitted application rates will result in soil residues of approximately 1 ppm trifluralin, a level that had no adverse effects on earthworms (Mayer and Ellersieck, 1986).

C. Insecticides

Please refer to Table 8: Toxicity of Insecticides to rats, birds, aquatic organisms, and bees.

Almost all chlorinated pesticides are banned in the Caribbean except endosulfan. Endosulfan was used heavily during the last two decades by coffee farmers in Jamaica because of its effectiveness in eradicating coffee berry borer. Recently it has been found that the pests are becoming increasingly resistant to endosulfan. One must emphasize at this point that the farmers should adopt proper agricultural practices to avoid this calamity. They are now trying to substitute chlorpyrifos for endosulfan, but again, if we are not careful, pests will grow immune to chlorpyrifos.

Since chlorinated pesticides are banned, more and more OP and carbamate pesticides are used. These pesticides are more eco-friendly since their degradation rates are faster than organochlorines and hence, they are less persistent in the ecosystem. It should be pointed out here that although the chlorinated pesticides viz. DDT, dieldrin, lindane etc. have been banned for quite some time, these pesticides are still found in the ecosystem either because of their exceedingly long half life or because farmers are importing these pesticides illegally from North America. Chlorpyrifos has been found to be a very effective insecticide and hence is widely used in the island. This has created problems. Although the degradation rate is somewhat faster than that of chlorinated pesticides, it is still regarded as a persistent pesticide since it takes about six months to reach the level of tolerance. Without knowledge of its fate, farmers have been using this pesticide with various agricultural crops. In recent months, we at the Pesticide Research Laboratory have found ppm levels of chlorpyrifos in cans of callaloo which were ready for exportation, although the USFDA has declared zero tolerance for chlorpyrifos in canned callaloo. But, most of the farmers had no choice but to use this pesticide since the most effective pesticide for vegetable crops, Lannate (a.i. methomyl) has been categorised by the PCA as a restricted pesticide because of its high oral toxicity (LD_{50} 17 mg/kg). Lannate, a carbamate, has two very important properties for which it has been used worldwide for a vast number of vegetable crops. One property is that is has a quick knock-down ability, and the other, is its very rapid degradation rate. The crops can be harvested within two days after spraying since the pesticide completely degrades to water and CO_2 within that time. Hence restricting importation of a pesticide only on the basis of its oral toxicity can be economically and environmentally disastrous for the country. All of the insecticides now imported in Jamaica and St Lucia (see Table 1) should be monitored regularly.

Detailed Notes on Insecticides

abamectin

Toxicity to aquatic organisms: Abamectin is highly toxic to fish and extremely toxic to aquatic invertebrates (USEPA, 1990a). Its 48-hour LC_{50} in *Daphnia magna,* is 0.003 mg/L. The 96-hour LC_{50} for abamectin is 0.0016 mg/L in pink shrimp, 430 mg/L in eastern oysters, and 153 mg/L in blue crab (Wislocki, Grosso and Dybas, 1989). While highly toxic to aquatic organisms, actual concentrations of abamectin in surface waters adjacent to treated areas are expected to be low. Abamectin did not bioaccumulate in bluegill sunfish exposed to 0.099 µg/L for 28 days in a flow-through tank. The levels in fish were from 52 to 69 times the ambient water concentration, indicating that abamectin does not accumulate or persist in fish (Wislocki, Grosso and Dybas, 1989).

amitraz

Toxicity to birds: Amitraz may affect reproduction in birds. The avian reproduction NOEL is less than 40 ppm (USEPA, 1987b).

Toxicity to aquatic organisms: *Daphnia*, a fresh water invertebrate, exhibited toxic effects at 35 ppb of amitraz in water (Meister, 1994).

carbaryl

Toxicity to birds: Carbaryl is practically non-toxic to wild bird species (Kidd and James, 1991).

Toxicity to aquatic organisms: Ukeles (1962) studying marine phytoplankton and toxicants, found that carbaryl was toxic to all five species tested. However, one species, *Monochrysis lutheri*, could withstand a sublethal dose for as long as six months. Christie's (1969) results of experiments with *Chlorella pyrenidosa* and carbaryl agree with Ukeles: they found concentrations of 0.1 mg/liter to be toxic. In experimenting with algae (*Scenedesmus quadricaudata*), Stadnyk, Campbell and Johnson (1971) found that cell growth was stimulated by carbaryl, the cell biomass increasing 44 to 57 percent in six days. The stimulation of cell growth was attributed to an increased nitrogen source from degradation of carbaryl. Hence, Stadnyk at al. concluded, carbamate insecticide could lead to increased algal blooms. Some accumulation of carbaryl can occur in catfish, crawfish, and snails, as well as in duckweed. Residue levels in fish were 140-fold greater than the concentration of carbaryl in water. Under conditions below neutrality, the bioaccumulation risk may be significant (Kidd and James, 1991).

Toxicity to other organisms: Carbaryl is lethal to many non-target insects, including bees and beneficial insects (Kidd and James, 1991).

chlorpyrifos

Toxicity to aquatic organisms: Chlorpyrifos is very highly toxic to freshwater fish, aquatic invertebrates and estuarine and marine organisms (USEPA, 1989a). Cholinesterase inhibition was observed in acute toxicity tests of fish exposed to very low concentrations of this insecticide. Application of concentrations as low as 0.01 pounds of active ingredient per acre may cause fish and aquatic invertebrate deaths (USEPA, 1989a). Chlorpyrifos toxicity to fish may be related to water temperature. The 96-hour LC_{50} for chlorpyrifos is 0.009 mg/L in mature rainbow trout, 0.098 mg/L in lake trout, 0.806 mg/L in goldfish, 0.01 mg/L in bluegill, and 0.331 mg/L in fathead minnow (USEPA, 1986b). When fathead minnows were exposed to Dursban for a 200-day period during which they reproduced, the first generation of offspring had decreased survival and growth, as well as a significant number of deformities. This occurred at approximately 0.002 mg/L exposure for a 30-day period (U.S. Public Health Service, 1995). Chlorpyrifos accumulates in the tissues of aquatic organisms. Studies involving continuous exposure of fish during the embryonic through fry stages have shown bioconcentration values of 58 to 5100 (Racke, 1992). Due to its high acute toxicity and its persistence in sediments, chlorpyrifos may represent a hazard to sea bottom dwellers (Schimmel, S.C. *et al.*, 1983). Smaller organisms appear to be more sensitive than larger ones (Racke, 1992).

Toxicity to other organisms: Aquatic and general agricultural uses of chlorpyrifos pose a serious hazard to wildlife and honeybees (Kidd and James, 1991; USEPA, 1984a).

cypermethrin

Toxicity to aquatic organisms: Cypermethrin is very highly toxic to fish and aquatic invertebrates. Its acute LC_{50} in *Daphnia magna*, a small freshwater crustacean, is 0.0002 mg/L Bradbury and Coats, 1989). Cypermethrin is metabolized and eliminated significantly more slowly by fish than by mammals or birds, which may explain this compound's higher toxicity in fish compared to other organisms (Bradbury and Coats, 1989). The half-lives for elimination of several pyrethroids by trout are all greater than 48 hours, while elimination half-lives in birds and mammals range from 6 to 12 hours (USNLM, 1995; Bradbury and Coats, 1989). The bioconcentration factor for cypermethrin in rainbow trout was 1200 times the ambient water concentration, indicating that there is a moderate potential to accumulate in aquatic organisms (USEPA, 1989c). Elimination of half of the accumulated amount of the compound took nearly eight days. After 14 days 70 to 80% of the material had been eliminated from the organisms (USEPA, 1989c).

diazinon

Toxicity to birds: Birds are quite susceptible to diazinon poisoning. In 1988, the EPA concluded that the use of diazinon in open areas poses a "widespread and continuous hazard" to birds. Bird kills associated with diazinon use have been reported in every area of the country and at all times of the year. Canadian geese and mallard ducks may be exposed to LC_{50} concentrations in very short periods of time after application (from 15 to 80 minutes depending on the application rate of the pesticide). Birds are significantly more susceptible to diazinon than other wildlife (U.S. Public Health Service, 1995).

Toxicity to aquatic organisms: Diazinon is highly toxic to fish. In rainbow trout, the diazinon LC_{50} is 2.6 to 3.2 mg/L (Kidd and James, 1991). In hard water, lake trout and cutthroat trout are somewhat more resistant. Warm water fish such as fathead minnows and goldfish are even more resistant with diazinon LC_{50} values ranging up to 15 mg/L (U.S. Public Health Service, 1995). There is some evidence that saltwater fish are more susceptible than freshwater fish. Bioconcentration ratios range from 200 in minnows to 17.5 for guppies. These studies show that diazinon does not bioconcentrate significantly in fish (Howard, 1991).

endosulfan

Toxicity to birds: Endosulfan is highly to moderately toxic to bird species (Kidd and James, 1991). Male mallards from 3 to 4 months old exhibited wings crossed high over their back, tremors, falling, and other symptoms as soon as 10 minutes after an acute, oral dose. The symptoms persisted for up to a month in a few animals (Hudson, Tucker and Haegele, 1984).

Toxicity to aquatic organisms: Endosulfan is very highly toxic to four fish species and both of the aquatic invertebrates studied; in fish species, the reported 96-hour LC_{50} values were (in µg/L): rainbow trout, 1.5; fathead minnow, 1.4; channel catfish, 1.5; and bluegill sunfish, 1.2. In two aquatic invertebrates, scuds (*G. lacustris*) and stoneflies (*Pteronarcys*), the reported 96-hour LC_{50} values were, respectively, 5.8 µg/L and 3.3 µg/L (Johnson and Finley, 1980). The 24-hour LC_{50} for the tropical fish, red hybrid Tilapia is 0.031 mg/L (Robinson, 1997). The bioaccumulation for the compound may be significant; in the mussel (*Mytilus edulis*) the compound accumulated to 600 times the ambient water concentration (USNLM, 1995). Bioaccumulation in the tropical fish, red hybrid Tilapia, after 1 hour of exposure to endosulphan at 0.0075 mg/L was 0.55 µg/g (94%) of α-endosulphan and 0.44 µg/g (100%) of β-endosulphan. Only 26 percent of α-endosulphan was and 39 percent of β-endosulphan were eliminated in 72 hours (Robinson, 1997).

ethoprophos

Toxicity to aquatic organisms: The 24-hour LC_{50} for ethoprophos for the tropical fish, red hybrid Tilapia, was 8.41 mg/L (Robinson, 1997). Bioaccumulation after 1 hour of exposure to ethoprophos at 1 mg/L in the surrounding water was 2.6 µg/g (74%) (Robinson, 1997).

imidacloprid

Toxicity to birds: Imidacloprid is toxic to upland game birds. The LD_{50} is 152 mg/kg for bobwhite quail, and 31 mg/kg in Japanese quail (Kidd and James, 1991; Meister, 1994). In studies with red-winged blackbirds and brown-headed cowbirds, it was observed that birds learned to avoid imidacloprid treated seeds after experiencing transitory gastrointestinal distress (retching) and ataxia (loss of coordination). It was concluded that the risk of dietary exposure to birds via treated seeds was minimal. Based on these studies, imidacloprid appears to have potential as a bird repellent seed treatment (Avery, Decker and Fischer, 1994; Avery, Decker, Fischer and Stafford, 1993).

Toxicity to aquatic organisms: In tests with the aquatic invertebrate *Daphnia magna*, the 48-hour EC_{50} (effective concentration to cause toxicity in 50% of the test organisms) was 85 mg/l (1). Products containing imidacloprid may be very toxic to aquatic invertebrates.

Toxicity to other organisms: Imidacloprid is highly toxic to bees if used as a foliar application, especially during flowering, but is not considered a hazard to bees when used as a seed treatment (Kidd and James, 1991).

lambda cyhalothrin

Toxicity to aquatic organisms: Lambda cyhalothrin is very highly toxic to many fish and aquatic invertebrate species. Reported LC_{50}s in these species are as follows: bluegill sunfish, 0.21 µg/L; rainbow trout, 0.24 µg/L; *Daphnia magna*, 0.36 µg/L; mysid shrimp, 4.9 ng/L; sheepshead minnow, 0.807 ng/L (Kidd and James, 1991; USEPA, 1988c). A median effect concentration, EC_{50} (i.e. the concentration at which the effect occurs in 50% of the test population), for the eastern oyster of 0.59 ng/L has been reported (USEPA, 1988c). Bioconcentration is possible in aquatic species, but bioaccumulation is not likely. Bioconcentration in channel catfish has been reported as minimal, with rapid depuration (elimination) (USEPA, 1992a). A bioconcentration factor of 858 has been reported in fish (4, species unspecified), but concentration was confined to non-edible tissues and rapid depuration was observed (USEPA, 1988c).

malathion

Toxicity to birds: Malathion is moderately toxic to birds. The reported 5- to 8-day dietary LC_{50} is over 3000 ppm in Japanese quail, mallard, and northern bobwhite, and is 2639 ppm in ring-neck pheasants (Smith, 1993). Furthermore, 90% of the dose to birds was metabolised and excreted in 24 hours via urine (Menzer, 1987).

Toxicity to aquatic organisms: Malathion has a wide range of toxicities in fish, extending from very highly toxic in the walleye (96-hour LC_{50} of 0.06 mg/L) to highly toxic in brown trout (0.1 mg/L) and the cutthroat trout (0.28 mg/L), moderately toxic in fathead minnows (8.6 mg/L) and slightly toxic in goldfish (10.7 mg/L) (Kidd and James, 1991; U.S. Public Health Service, 1995; Smith, 1993). Various aquatic invertebrates are extremely sensitive, with EC_{50} values from 1 µg/L to 1 mg/L (Menzie, 1980). Malathion is highly toxic to aquatic invertebrates and to the aquatic stages of amphibians. Because of its very short half-life, malathion is not expected to bioconcentrate in aquatic organisms. However, brown shrimp showed an average concentration of 869 and 959 times the ambient water concentration in two separate samples (Howard, 1991).

methomyl

Toxicity to birds: Methomyl is highly toxic to birds. The acute oral LD_{50} in bobwhite quail is 24.2 mg/kg (USEPA, 1987f). The oral LD_{50} of methomyl is 28 mg/kg in hens. All deaths occurred within ten minutes of dosing. The clinical signs of toxicity included tearing of the eyes, salivation, occasional convulsions, and respiratory disorders. In Japanese quail, the LD_{50} is 34 mg/kg (USEPA, 1987f). The LD_{50} of a 90% pure formulation is 15.9 mg/kg in eight-month-old mallards, and 15.4 mg/kg in three- to four-month-old male pheasants (Tucker, 1970). The LD_{50} for starlings is 42 mg/kg and for redwinged blackbirds is 10 mg/kg (USEPA, 1987f).

Toxicity to aquatic organisms: Methomyl is moderately to highly toxic to fish and highly toxic to aquatic invertebrates. The 96-hour LC_{50} in rainbow trout for a liquid formulation of methomyl is 3.4 mg/L and for bluegill sunfish is 0.8 mg/L (Kidd and James, 1991). The 48-hour LC_{50} for *Daphnia magna* is 0.0287 mg/L (Howard, 1991). A 28-day fish residue study indicated that methomyl did not accumulate in fish tissue (USEPA, 1987f). Methomyl is unlikely to bioconcentrate in aquatic systems (Howard, 1991).

Toxicity to other organisms: Methomyl is highly toxic to bees both by direct contact and through ingestion (Howard, 1991).

permethrin

Toxicity to aquatic organisms: Aquatic ecosystems are particularly vulnerable to the impact of permethrin. A fragile balance exists between the quality and quantity of insects and other invertebrates that serve as fish food (Penick Corp, 1979). The 48-hour LC_{50} for rainbow trout is 0.0125 mg/L for 24 hours, and 0.0054 mg/L for 48 hours (Kidd and James, 1991). The 48-hour LC_{50} in bluegill sunfish and salmon is 0.0018 mg/L (Kidd and James, 1991). As a group, synthetic pyrethroids were toxic to all estuarine species tested. They had a 96-hour LC_{50} of less than or equal to 0.0078 mg/L for these species (Schimmel, 1983). The bioconcentration factor for permethrin in bluefish is 715 times the concentrations in water and is 703 in catfish. This indicates that the compound has a low to moderate potential to accumulate in these organisms.

Toxicity to other organisms: Permethrin is extremely toxic to bees. Severe losses may be expected if bees are present at treatment time, or within a day thereafter (Ray, 1991; Morse, 1987). Permethrin is also toxic to wildlife (Hallenbeck and Cunningham-Burns, 1985). It should not be applied, or allowed to drift, to crops or weeds in which active foraging takes place (Kidd and James, 1991).

D. Nematicides

Please refer to Table 9: Toxicity of Nematicides to rats, birds, aquatic organisms, bees.

Among the nematicides imported in the island, Furadan 10G (a.i. carbofuran) and Vydate L (a.i. oxamyl) are extremely toxic. Furadan 10G decomposes quickly and hence the detection of this pesticide in the ecosystem is extremely difficult. Vydate L is quite stable in water. Because of their highly toxic nature and Vydate’s stability in water, monitoring of these pesticides should be given priority.

Detailed Notes on Nematicides

carbofuran

Toxicity to birds: Carbofuran is highly toxic to birds. One granule is sufficient to kill a small bird. Bird kills have occurred when birds ingested carbofuran granules, which resemble grain seeds in size and shape, or when predatory or scavenging birds have ingested small birds or mammals that have eaten carbofuran pellets (USEPA, 1991). Red-shouldered hawks have been poisoned after eating prey from carbofuran-treated fields (Smith, 1993). The LD_{50} is 25-39 mg/kg in chickens consuming carbofuran as a powder (Kidd and James, 1991).

Toxicity to aquatic organisms: Carbofuran is highly toxic to many fish. The compound has a low potential to accumulate in aquatic organisms. The bioconcentration factor ranges from 10 in snails to over 100 in fish (Howard, 1991).

Toxicity to other organisms: Carbofuran is toxic to bees except in the granular formulation (Kidd and James, 1991).

oxamyl

Toxicity to birds: Oxamyl is very highly toxic to birds (Kidd and James, 1991). The acute oral LD_{50} for oxamyl in quail is 4.18 mg/kg (Kidd and James, 1991). Hens given single oral doses of oxamyl at 20 to 40 mg/kg of body weight followed by intramuscular injections of 0.5 mg of atropine, an antidote, exhibited early symptoms of cholinesterase inhibition with full recovery after 12 hours. No signs of delayed neurotoxicity were observed in these same hens (USEPA, 1987g).

Toxicity to aquatic organisms: Concentrations of 0.5 to 5.0 mg/L may have an effect on *Daphnia magna*, an aquatic invertebrate (Mayer and Ellersieck, 1986).

E. Plant Growth Regulators

Please refer to Table 10: Toxicity of Plant Growth Regulators to rats, birds, aquatic organisms, and bees.

Detailed Notes on Plant Growth Regulators

paclobutrazol

Toxicity to aquatic organisms: 48-hour LC50 for *Daphnia magna* 33.2 mg/L.

SECTION 2: ENVIRONMENTAL FATE

In this section we will discuss the means by which agrochemicals enter the soil and aquatic environments, and factors affecting pesticide persistence in these environments. By examining the means by which pesticides enter the environments we may be able to identify ways to minimise their deleterious effects by minimizing the quantities unnecessarily or improperly introduced. Persistence in the environment has direct implications for environmental impact, since the longer the agrochemical stays in the environment the greater the potential that it will harm non-target and/or beneficial organisms.

Agrochemicals enter the soil and water environments through the actions of farmworkers in four primary ways (Robinson, 1997):

1. directly by application to control soil and water-inhabiting pests
2. through fall-put of upward drift from aerial, spraying and dusting applications
3. from run-off of spray droplets from plant surfaces, either by themselves or along with precipitation to the soil, and erosion and run-off from agricultural lands to water
4. from the remains of plants and animals

Of the pesticides sprayed on foliage, it is estimated that up to 80% may eventually reach the soil (Edwards, 1973). When spraying, up to one-third of the droplets may fall onto the ground between the plants. Contaminated soil particles may drift into the air and end up on earth or water. Usually over 50% of pesticides drifts into the atmosphere during application and later settles on soil or water on their own or along with precipitation (Matthews, 1985). Residues in air may be carried long distances away by wind, and high levels of pesticides in the atmosphere have been reported since 1964 (Lloyd and Tweedle, 1964). Droplet size is the most important factor determining the extent to which gravitational, meteorological, and electrostatic charges regulate the movement of droplets (Matthews, 1985). Matthews found that a 1µm diameter droplet took 61,309 times more time (28.1 hours) to fall from three meters in still air than a 1µm diameter droplet (1.65 seconds). The quantity of residues added to soil environments from the leftover matter of animals or plants was considered by Edwards to be of academic interest only, since they remove more pesticides from soil than they add to them (Edwards, 1973). On the other hand, containers of liquid pesticides have 2-3 percent of the concentrate remaining after normal emptying, which could contribute significantly to the contamination of soil and aquatic environment if not suitably disposed (Stojanovic, Kennedy and Shuman, 1972).

Of the four types of pesticides used, organochlorines (OCs), organophosphates (OPs), carbamates, and pyrethroids, (Robinson, 1997) to control soil and aquatic pests, the most common residues found in soils are OCs, OPs and carbamates are less persistent in soil than OCs. The fate of a pesticide in soil depends upon its physical removal and degradation which involves erosion or run-off, leaching, volatilisation, biological removal, and microbial, chemical or photochemical degradation (Robinson, 1997).

A reasonably good correlation exists between the persistence of pesticides in the soil and their solubility (Edwards, 1966). DDT, one of the most persistent pesticides, was stated to be the least water soluble organic substance known (0.0002 ppm). Pesticides are removed from agricultural lands by running water both in solution (run-off) and adsorbed to soil particles lost in erosion processes (wash-off). Soil surface movement of pesticides is usually accelerated by:

a. steep topography
b. low soil permeability
c. considerable rainfall
d. strong adsorption of pesticides to soil particles (Robinson, 1997).

Soluble pesticides also tend to be rapidly leached, especially moderately polar solids with high melting points (Robinson, 1997). Leaching is the downward movement of pesticides in solution through the soil resulting from rainfall or flooding of land. For most pesticides, the degree to which it will adsorb to soil particles is a very useful indicator of its leaching potential. Polar pesticides such as some nematicides are weakly adsorbed and consequently have the greatest potential for leaching to groundwater. In contrast, lipophilic chemicals such as permethrin and trifluralin, are strongly adsorbed to soil and therefore little potential for leaching (Nicholls, 1988). Their water solubility, though usually low, is usually more than is required to dissolve unadsorbed residues. The mobility of weak acids depends on soil pH. Permanent anions and weak acids can be very weakly adsorbed and hence might easily reach groundwater (Nicholls, 1988). Pesticides in soils may be degraded into various compounds more polar than the parent compounds making them more prone to leaching. Robinson (1997) found in his laboratory studies, that the leaching potential of pesticides increased as follows: OC < OP < carbamates.

Pesticides adsorb to organic content and clay, so pesticides applied to soils rich in organic matter especially, but also clay, will persist much longer than in soils with low organic content and/or clay content. Leaching, even of mobile pesticides can be impeded by strong adsorption to ash residues from stubble burning which lie close to the surface. Soil structure will also affect the loss of a pesticide by leaching, with pesticides leaching to great depths in sandy soils but spreading more laterally in clay soils. Rainfall and rate of evaporation of water from soil surface also influences leaching. Nicholls (1988) found that when rainfall exceeded evaporation there was net downward movement of water and pesticides and when evaporation exceeded rainfall there was net upward movement of water and pesticides.

Temperature and precipitation have great effect on persistence, especially here in the tropics. Higher temperatures increase water solubility and volatilisation, the latter being a very important route of pesticide loss. Increased rainfall also results in less pesticide being adsorbed to soil, and favours microbial activity which would result in increased breakdown of pesticides.

Microbial degradation plays a very important role in decomposition of pesticides in soil (Hill and Wright, 1978). Fungi and actinomycetes have much greater effect on pesticides in the soil than bacteria do (Alexander, 1965). When pesticides are added to the soil there may be an interval called a lag period before substantial breakdown occurs (Hague and Freed, 1974). This is the time required for microorganisms to build up a population at the site of the pesticide. In potato fields, levels of aldicarb, oxamyl, and ethoprophos, decreased more rapidly in the soil of annually treated plots when compared with frequently-treated plots and sterilized soil (Smelt et al., 1987), suggesting a large microbial population in annually treated soil which reduced lag time. Printz et al. (1995) reported a fivefold increase in the degradation of methabenzthiazuron when maize straw was added to the soil which they attributed to increased microbial activity in the soil. Microbial degradation is thought to be the major route of pesticide loss for OPs.

The discussion up to this point makes possible certain deductions. First, pesticide persistence will be affected by the method of application (Anderson, 1987), since if the pesticide is incorporated into the soil, as in root dips, injections and seed dressings, it will be less exposed to volatilisation, as well as photochemical degradation, which for some pesticides (e.g. imazalil) is a very important factor. Residues on the surface of the soil will volatilise faster than the same amount incorporated into the soil. If incorporated into the soil it may also be more readily leached to great depths where it will be less exposed to removal by animals, plants and microbes. Second, the method of tillage will affect persistence, since conservative tillage systems such as chisel, ridge plant and no-till, reduce run-off (Sauer and Daniel, 1987). Sauer and Daniel found that conservative tillage systems reduced run-off of atrazine, alachlor and chlorpyrifos (Sauer and Daniel, 1987). Less run-off means more pesticide on and/or in the soil, and therefore a chance for removal by more environmentally-friendly mechanisms. Less run-off also means the farmer can use smaller quantities of agrochemicals. Long-term pesticide residues from agricultural fields were estimated by Weber *et al.* to range from 0.5 to 2.2 percent, the lowest being from pastures, grassed areas, and fields where conservative tillage practices were utilized (Weber, Shea and Strek, 1980). Finally, formulation will also influence persistence, since the rate of pesticide adsorption onto soil fractions is inversely related to the particle size of the pesticide in the formulation (Edwards, 1975). Lichtenstein *et al.* (1962) found that persistence increased in the following order: emulsion left on soil surface < granules left on soil surface < emulsion incorporated into the soil < granules incorporated into the soil. Wauchope reported that the loss of residues from agricultural land ranged from 5 to 16 percent, wettable powder being most prone, particularly after rainfall (Wauchope *et al.*, 1992). On the question of the effect of pesticide concentration on persistence however, scientists remain divided.

Several mathematical models including the Agricultural Run-off Management (ARM), Continuous Pesticide Simulation (CPS), and the Chemicals, Run-off and Erosion for Agricultural Management Systems (CREAMS) have been developed to predict the run-off of pesticides from agricultural systems (Singh, 1985). These predictive models, accurate within 10% when field data on toxaphene and atrazine were tested, consider degradation, transport between vertical spatial zones in the soil column and partitioning between water and particulate phases as factors.

In the aquatic environment, volatilisation is more rapid than from soil (Lichtenstein and Schultz, 1970), and is the primary process by which pesticides sprayed directly on ponds are lost (Spencer *et al.*, 1973). Agrochemicals present in water may adsorb to both living and non-living particles (Crosby, 1973). Agrochemicals may be adsorbed by aquatic plankton, invertebrates, vegetation and fish (Wojtalik et al., 1971; Edwards, 1974). Aquatic unicellular organisms show high and rapid sorption capacities. Agrochemicals taken up by aquatic flora and fauna may be degraded or accumulated in their tissues, which may result in bioaccumulation. Bioaccumualtion is the the result of bioconcentration and biomagnification. Bioconcentration is the preferential uptake of the pesticide via direct partitioning of the chemical between the organisms and the immediate environment whereas biomagnification is the uptake of the pesticide from food and the selective transfer of the chemical from organisms of lower trophic levels to organisms of higher trophic levels (Huckle and Milburn, 1990). Aquatic plants can influence the location and concentration of residues through absorption by floating parts and subsequent release by submerged parts (Frank, 1970). Degradation microbes in the aquatic environment appears to be more significant in the biochemical transformation of pesticides but may be slower than in soil (Hill and Wright, 1978).

In Table 11, we list the known degradation products of the pesticides, called metabolites. Degradation products are important since they may be more toxic than the a.i., and more persistent. Hence the degradation products and their characteristics must be considered when deciding what pesticides to use and where. The "Potential for Uptake" indicates whether some plants might take up the pesticide and store it, or its metabolites. This has great implications for the health of consumers, whether *homo sapiens* or some other species. Half-life data for both soil and aquatic environment is provided where available. This data can only be approximate given the wide variety of situations possible.

As we are especially concerned with the persistence of agrochemicals in aquatic environments, guidelines for interpreting the relative importance of agrochemical longevity in aquatic environments are provided here. "The substance which degrades at the rate of 10%/10days is considered to have considerable sustainability in aquatic environment and needs to be monitored. When 70% of the substance degrades within 28 days, then the substance is considered to be degrading within a reasonable time frame. None of these substances are readily degradable" (International Occupational Safety and Health Information Centre, 2001).

Please refer to Table 11: Environmental Fate of Pesticides.

A. FUNGICIDES

chlorothalonil

Metabolites: In mammals, chlorothalonil is excreted largely unchanged but there is some metabolism to 4-hydroxy-2,5,6-trichloroisophtalonitrile which is more toxic than the parent compound (Kidd and James, 1994).
Half-life in soil and groundwater (temperate conditions): Chlorothalonil is moderately persistent. Increased soil moisture or temperature increases chlorothalonil degradation. It is not degraded by sunlight on the soil surface (USNLM, 1995). Chlorothalonil has high binding and low mobility in silty loam and silty clay loam soils, and has low binding and moderate mobility in sand (USEPA, 1987h). Chlorothalonil was not found in any of 560 groundwater samples collected from 556 U.S. sites (USEPA, 1987h).
Half-life in water (temperate conditions): In very basic water (pH 9.0), about 65% of the chlorothalonil was degraded into two major metabolites after 10 weeks (USEPA, 1987h).
Effect on vegetation: Chlorothalonil's residues may remain on above-ground crops at harvest, but will dissipate over time. Chlorothalonil is a fairly persistent fungicide on plants, depending on the rate of application. Small amounts of one metabolite may be found in harvested crops (Vettorazzi, 1979).

copper oxychloride

In/On vegetation: Nelson's studies of copper exychloride in Jamaica provided a half-life of 40.1 days on coffee berries (Nelson, 1993).

hexaconazole

In soil and groundwater: Rapidly degraded in soil (Kidd and James, 1994).

imazalil

In soil and groundwater: Imazalil is highly persistent in the soil environment (Wauchope *et al.*, 1992). It is soluble in water, but strongly bound to soils (Wauchope *et al.*, 1992), and thus unlikely to pose a risk to groundwater. In a plot where seven applications were made at 14-day intervals, leaching was practically nonexistent and accumulation did not appear to be a problem (FAO, 1985).

In water: In acid to neutral aqueous solutions, imazalil is stable for at least 8 weeks at 40 ºF. Decomposition occurs at elevated temperatures and under the influence of light (FAO, 1977).

In vegetation: One week after treated barley seed was sown in soil, about 76% of the imazalil was in the adjacent soil and about 29% was in the seedcoat. After 3 weeks, only 6% was in the green plant parts. Under normal storage conditions, oranges dipped in 2000 mg active ingredient/L and stored, have residues (89%) present as the parent compound. Only a small amount of imazalil was present in the pulp, and part of this may have resulted from handling during peeling (FAO, 1977). Studies with apples gave similar results.

mancozeb

In soil and groundwater: Mancozeb is of low soil persistence, with a reported field half-life of 1 to 7 days (Wauchope *et al.*, 1992). Mancozeb rapidly and spontaneously degrades to ETU (ethylenethiourea) in the presence of water and oxygen (USEPA, 1988e). ETU may persist for longer, on the order of 5 to 10 weeks (Wauchope *et al.*, 1992). Because mancozeb is practically insoluble in water, it is unlikely to infiltrate groundwater (Kidd and James, 1991). Studies do indicate that ETU, a metabolite of mancozeb, has the potential to be mobile in soils (USEPA, 1987m). However, ETU has been detected (at 0.016 mg/L) in only 1 out of 1295 drinking water wells tested (USEPA, 1988e).

metalaxyl

In soil and groundwater: Under field conditions, metalaxyl has a half-life of 7 to 170 days in the soil environment (Wauchope *et al.*, 1992). A representative half-life in moist soils is about 70 days (Wauchope *et al.*, 1992). Increased sunlight may increase the rate of breakdown in the soil. It is poorly sorbed by soils and highly soluble in water (Wauchope *et al.*, 1992); these properties in combination with its long persistence pose a threat of contamination to groundwater. It readily leaches in sandy soil, although increased organic matter may decrease the rate of leaching (Kimmel, Casida and Ruzo, 1986). In a large-scale, national survey (USA), metalaxyl was detected in the groundwater of several states at concentrations of 0.27 µg/L to 2.3 mg/L (Williams, Holden, Parsons and Lorber, 1988).

In water: At pH levels of 5 to 9 and temperatures of 20 to 30 ºC, the half-life in water was greater than 4 weeks (USEPA, 1998a). However, exposure to sunlight reduced the half-life to 1 week (USEPA, 1988a).

In vegetation: Plants absorb foliar applications through the leaves and stems, and can translocate the compound throughout the plant. Metalaxyl is not absorbed directly from the soil by plants (FAO, 1983).

B. HERBICIDES

2,4-D

Metabolites: Breakdown in plants is by a variety of biological and chemical pathways (USEPA, 1987e). Absorption of 2,4-D is almost complete in mammals after ingestion and nearly all of the dose is excreted in the urine. Men given 5 mg/kg excreted about 82% of the dose as unchanged 2,4-D. The half-life is between 10 and 20 hours in living organisms. There is no evidence that 2,4-D accumulates to significant level in mammals or in other organisms (Howard, 1991).

In soil and groundwater: 2,4-D has low soil persistence (Wauchope *et al.*, 1992). Soil microbes are primarily responsible for its disappearance (Howard, 1991). Steenson and Walker in their studies with soil isolates, found that aerobic conditions were essential for bacterial decomposition of the acid. Despite its short half-life in soil and in aquatic environments, the compound has been detected in groundwater supplies in at least five States and in Canada (Wauchope *et al.*, 1992). Very low concentrations have also been detected in surface waters throughout the U.S. (USEPA, 1992b).

In water: In aquatic environments, microorganisms readily degrade 2,4-D. Rates of breakdown increase with increased nutrients, sediment load, and dissolved organic carbon (Howard, 1991). However, even more important are the size and nature of the microbial population. DeMarco also found that microbial decomposition in river water was inhibited by a decrease in temperature. Gerking (1948) and Walsh, Keltner and Matthews (1969) reported growth stimulation of algae by 2,4-D. Srinivasan and Chacko (1952) found that the populations of both phytoplankton and zooplankton increase increased in the presence of 2,4-D.

In/On plants: 2,4-D interferes with normal plant growth processes and is 2,4-D is toxic to most broad leaf crops, especially cotton, tomatoes, beets, and fruit trees (USNLM, 1995; USEPA, 1987e). Moisture on plant foliage does not strongly influence plant response to 2,4-D vapour; however, water as a carrier seems to increase the volatility of 2,4-D (McLane, 1963).

ametryn

Metabolites: Excretion of ametryn is rapid. In rats, all but 2 to 7% is eliminated in the urine and faeces within 72 hours (USEPA, 1994).

In soil and groundwater: Loss from the soil is principally by microbial degradation (USEPA, 1989b; Kidd and James, 1994). Ametryn moves both vertically and laterally in soil due to its high water solubility (Thomson, 1982). Because it is persistent, it may leach as a result of high rainfall, floods, and furrow irrigation (USEPA, 1989b). In a study of surface and groundwater contaminants in the U.S, ametryn was found in six states, in very few surface water samples and in 4% of the groundwater samples. The maximum concentration found was 0.1 µg/L in surface water and 450 µg/L in groundwater (USEPA, 1987d).

In/On vegetation: Ametryn, is a herbicide which inhibits photosynthesis and other enzymatic processes. Ametryn is broken down into non-toxic substances by tolerant plants and, to a lesser extent, by sensitive plants (Kidd and James, 1994).

atrazine

Metabolites: Atrazine is readily absorbed through the gastrointestinal tract. When a single dose of 0.53 mg atrazine was administered to rats by gavage, 20% of the dose was excreted in the feces within 72 hours. The other 80% was absorbed across the lining of the gastrointestinal tract into the bloodstream. After 72 hours, 65% was eliminated in the urine and 15% was retained in body tissues, mainly in the liver, kidneys, and lungs (Stevens and Sumner, 1991).

In soil and groundwater: Atrazine is highly persistent in soil. Chemical hydrolysis, followed by degradation by soil microorganisms, accounts for most of the breakdown of atrazine.

Hydrolysis is rapid in acidic or basic environments, but is slower at neutral pHs. Addition of organic material increases the rate of hydrolysis. Atrazine applied to soil detoxifies in sunlight. Bioassays show that losses of 47 percent and 60 percent occur during exposure to spring sunlight for 25 and 60 days, respectively (Comes and Timmons, 1965). Starch-encapsulation significantly reduced the volatisation of atrazine from 14% to < 1% (Winehold, Sadegi and Gish, 1993). Atrazine can persist for longer than 1 year under dry or cold conditions (Howard, 1991). Atrazine is moderately to highly mobile in soils with low clay or organic matter content. Guo et al. found that the more organic content there was in soil, the less mobile atrazine was (Guo, Bicki, Hinesly an Felsot, 1991). Because it does not adsorb strongly to soil particles and has a lengthy half-life (60 to >100 days), it has a high potential for groundwater contamination despite its moderate solubility in water (Wauchope *et al.*, 1992). Atrazine is the second most common pesticide found in private wells and in community wells (USNLM, 1995). Trace amounts have been found in drinking water samples and in groundwater samples in a number of states in the USA (Howard, 1991; USEPA, 1990b). A 5-year survey of drinking water wells detected atrazine in an estimated 1.7% of community water systems and 0.7% of rural domestic wells nationwide. Levels detected in rural domestic wells sometimes exceeded the MCL (USEPA, 1990b). The recently completed **National Survey of Pesticides in Drinking Water** (USA) found atrazine in nearly 1% of all of the wells tested (USEPA, 1990b).

In water: Chemical hydrolysis, followed by biodegradation, may be the most important route of disappearance from aquatic environments. Only slight detoxicification in aqueous solution occurs in the absence of soil at 50 ºC or 95 ºC (Harris, 1966). In a study with both aerobic and anaerobic lake sediment samples, Hance and Chesters (1969) found that hydroxyatrazine, atrazine's major degradation product, was degraded more rapidly in the aerobic than the anaerobic sample. However, the converse was true for soil samples. Bioconcentration and volatilization of atrazine are not environmentally important (Howard, 1991).

In/On vegetation: In susceptible plant species, atrazine inhibits photosynthesis. In tolerant plants, it is metabolized (Kidd and James, 1991). Most crops can be planted 1 year after application of atrazine. Atrazine increases the uptake of arsenic by treated plants (USNLM, 1995).

bromacil

Metabolites: A number of studies show that uracils, the class of compounds to which bromacil belongs, are absorbed into the body from the gut and excreted primarily in the urine (USNLM, 1995; USEPA, 1988d). Small amounts of bromacil were detected in the milk of lactating cows that were given 5 mg/kg in their food (Gosselin, Smith and Hodge, 1984). No bromacil was found in the urine or faeces of these cows (USNLM, 1995).

In soil and groundwater: Bromacil is moderately to highly persistent in soil. Its half-life is about 60 days, but may be as much as 8 months under some conditions (Wauchope *et al.*, 1992). Soil persistence is correlated to the organic content of the soil (USNLM, 1995; USEPA, 1975). At 18 months after 22.4 kg of bromacil were sprayed on abandoned field sites, residues of the herbicide were detectable, in decreasing amounts, in loamy sand, silt loam, silty clay loam, and light silty clay loam soils. Organic matter content, cation exchange capacity, total nitrogen, and soluble salt concentrations were significantly correlated with residue persistence (USNLM, 1995). Bromacil is expected to leach quite readily through the soil and contaminate groundwater. The amount of leaching is dependent on the soil type and the amount of rainfall or irrigation water. The potential for bromacil to leach and contaminate groundwater is greatest in sandy soils. In normal soils, it can be expected to leach to a depth of 2 to 3 feet (USNLM, 1995). Tests show that at increased temperatures and long exposures to sunlight, there is very little loss of bromacil from dry soil. It does not readily volatilize, nor does it break down in sunlight (USNLM, 1995). Laboratory studies show that 5 to 30% of bromacil is lost 6 to 9 weeks after application to the soil, as carbon dioxide (USNLM, 1995).

In/On vegetation: Improper application of bromacil will destroy shade trees and other desirable vegetation (VanDriesche, 1985).

fluazifop-p-butyl

In water: Fluazifop-p-butyl is rapidly hydrolized (cleaved apart by water) under most conditions to the fluazifop acid (Jordan and Cudney, 1987; Ecobichon, 1991). It is relatively stable to breakdown by UV or sunlight, and nonvolatile (Kidd and James, 1991; WSSA, 1994).

In/On vegetation: After uptake by the leaves of plants, fluazifop-p-butyl is rapidly broken down in the presence of water to fluazifop-p, which is translocated throughout the plant (Kidd and James, 1991; WSSA, 1994). The compound accumulates in the actively growing regions of the plant (meristems of roots and shoots, root rhizomes and stolons of grass), where it interferes with energy (ATP) production and cell metabolism in susceptible species (Kidd and James, 1991; WSSA, 1994).

glyphosate

Metabolites in animals: In mammals, following oral administration, glyphosate is very rapidly excreted unchanged (Kidd and James, 1994).

hexazinone

In soil and groundwater: Hexazinone is broken down by soil microbes, which release carbon dioxide in the process (WSSA, 1994). Sunlight may also break down the compound via photodegradation (U.S. Dept of Agriculture, 1984). The rate of breakdown under natural field conditions will depend on many site-specific variables, including sunlight, rainfall, soil type, and rate of application. Hexazinone does not evaporate to any appreciable extent from soil (U.S. Dept of Agriculture, 1984). Hexazinone is very poorly adsorbed to soil particles, very soluble in water, and slowly degraded, so it is likely to be mobile in most soils and has the potential to contaminant groundwater.

In water: Photodecomposition, biodegradation, and dilution are the prime mechanisms for loss of hexazinone activity in aquatic systems (WSSA, 1994).

In/On vegetation: Hexazinone is readily absorbed in the root zone and translocated throughout the plant. It is less mobile following uptake from the foliage. It is converted in non-susceptible plants to less phytotoxic compounds. In susceptible plants, it is more persistent and can result in disruption of photosynthesis and chloroplast damage (WSSA, 1994).

imazaquin

In Soil and Groundwater: The movement of imazaquin in the soil is limited. The compound is nonvolatile. Loss from photodecomposition is minor. Thus, the resultant average persistance is season long or not more than 4-6 months when used at recommended rates (USEPA, 1986a, WSSA, 1989). Imazaquin readily breaks down via microbial breakdown in the soil. It is decarboxylated slowly to CO_2, as well as degraded to the major metabolite CL 266,066 and at least six minor metabolites (USEPa, 1986a).

In Surface Water: Imazaquin is stable to hydrolysis at pH 3 and 5 (USEPA, 1986a).

linuron

In soil and groundwater: Linuron is moderately persistent in soils, with a field half-life of 30 to 150 days in various soils and under various conditions (Wauchope *et al.*, 1992; Rao and Davidson, 1980). A representative field half-life is estimated to be approximately 60 days (USEPA, 1984b). Microbial degradation is the major process by which linuron is lost from soils; photodegradation and volatilization are not important contributors to its breakdown (WSSA, 1994). The metabolites of linuron (3,4-dichloroaniline and carbon dioxide) are less toxic than linuron (E. I. DuPont, not dated). Linuron is moderately bound to soil, and is soluble in water (Wauchope *et al.*, 1992). Losses may occur through transport of linuron in runoff water and on suspended colloidal matter. Linuron has been found at very low concentrations in well and groundwater samples in Georgia, Missouri, Virginia, and Wisconsin (USNLM, 1995).

In water: Linuron is slightly to moderately soluble in water, and is not readily broken down in water (USNLM, 1995).

metolachlor

In soil and groundwater: Metolachlor is moderately persistent in the soil environment. Soils with significant soil water content may show more rapid breakdown. Very little metolachlor volatilizes from the soil, and photodegradation will be a significant pathway for loss only in the top few inches (Zimdahl and Clark, 1982). Breakdown is mainly dependent upon microbial activity, and thus will be temperature-dependent (Zimdahl and Clark, 1982). Microorganism metabolism occurs by both aerobic and anaerobic processes, and is affected by temperature, moisture, amount of leaching, soil type, nitrification, oxygen concentrations, and sunlight (WSSA, 1994; Zimdahl and Clark, 1982). Metolachlor is moderately well sorbed by most soils (WSSA, 1994; Zimdahl and Clark, 1982). Soils with higher organic matter and clay content may sorb it better. It is slightly soluble in water (WSSA, 1994). Extensive leaching is reported to occur, especially in soils with low organic content (USEPA, 1987a). Metolachlor was one of four pesticides that were extensively studied throughout the nation in the National Alachlor Well Water Survey (USA). This several-year project analyzed the contents of over 6 million private and domestic wells. Metolachlor was detected in about 1% of the wells (about 60,000 wells) at concentrations ranging from 0.1 to 1.0 µg/L. It has also been found in a number of surface water samples in 14 states, at a maximum concentration of 0.138 mg/L (Howard, 1991). These levels may result from runoff during spring and summer applications to fields (USEPA, 1987i).

In water: Metolachlor is also relatively stable in water under natural sunlight. About 6.6 % was degraded by sunlight in 30 days, a slow and minimal rate (USEPA, 1987a).

metribuzin

In soil and groundwater: Metribuzin is of moderate persistence in the soil environment (Wauchope *et al.*, 1992). The half-life of metribuzin varies according to soil type and climatic conditions. Soil half-lives of 30 to 120 days have been reported; a representative value may be approximately 60 days (Wauchope *et al.*, 1992). Metribuzin is poorly bound to most soils and soluble in water, giving it a potential for leaching in many soil types (Wauchope *et al.*, 1992). Soil mobility is affected by many site-specific variables, including the amount of soil organic matter, particle size distribution, porosity, rainfall, and application rates. The major mechanism by which metribuzin is lost from soil is microbial degradation. Losses due to volatilization or photodegradation are not significant under field conditions (Kidd and James, 1991; Stevens and Sumner, 1991).

In water: If present, metribuzin would most likely be found in the water column rather than the sediment, due to its low binding affinity and high water solubility (Kidd and James, 1991).

metsulfuron

In Soil and Groundwater: The breakdown of metsulfuron-methyl in soils is largely dependant on soil temperature, moisture content, and pH. The chemical will degrade faster under acidic conditions, and in soils with higher moisture content and higher temperature (Smith, 1986). The chemical has a higher mobility potential in alkaline soils than in acidic soils, as it is more soluble under alkaline conditions. Metsulfuron-methyl is stable to photolysis, but will break down in ultraviolet light. Half-life estimates for metsulfuron-methyl in soil are wide ranging from 14 - 180 days, with an overall average of reported values of 30 days (Wauchope *et al.*, 1992). Reported half-life values (in days) for soil include: clay – 178 (Smith, 1986); sandy loam – 102 (Smith, 1986); clay loam - 70 (Wauchope *et al.*, 1992), 14-28 (Wauchope *et al.*, 1992), 14-105 (Wauchope *et al.*, 1992); silty loam - 120-180 (Thompson, MacDonald, Staznik, 1992).

In Surface Water: The dissipation time for metsulfuron-methyl was investigated in a mixed wood/boreal forest lake. The DT50 or length of time required for half of the material to dissipate in water was >84 days when high concentrations of metsulfuron-methyl were applied, and 29.1 days at concentrations that might be expected if the chemical is applied for forestry uses (Thompson, MacDonald, Staznik, 1992). The chemical is stable to hydrolysis at neutral and alkaline pHs, and has a half-life of 3 weeks at pH 5.0, 25 ºC and >30 days at 15 ºC (USEPA, 1989d).

paraquat

In soil and groundwater: Paraquat is highly persistent in the soil environment, with reported field half-lives of greater than 1000 days (WSSA, 1994; Wauchope *et al.*, 1992). The reported half-life for paraquat in one study ranged from 16 months (aerobic laboratory conditions) to 13 years (field study) (Rao and Davidson, 1980). Ultraviolet light, sunlight, and soil microorganisms can degrade paraquat to products which are less toxic than the parent compound. The strong affinity for adsorption by soil particles and organic matter may limit the bioavailability of paraquat to plants, earthworms, and microorganisms (WSSA, 1994; Wauchope *et al.*, 1992). The bound residues may persist indefinitely and can be transported in runoff with the sediment. Paraquat is not significantly mobile in most soils (USNLM, 1995). That which does not become associated with soil particles can be decomposed to a nontoxic end product by soil bacteria (Wagner, 1981). Thus, paraquat does not present a high risk of groundwater contamination. Of 721 groundwater samples analyzed, only one contained paraquat, at a concentration of 20 mg/L (USEPA, 1987j).

In water: Paraquat will be bound to suspended or precipitated sediment in the aquatic environment, and may be even more highly persistent than on land due to limited availability of oxygen. It had a half-life in a laboratory stream water column of 13.1 hours (Kosinski and Merkle, 1984). In another study, paraquat dichloride was stable for up to 30 days (USEPA, 1987j). In a third study of low levels in water, paraquat had a half-life of 23 weeks (USEPA, 1987j).

In vegetation: Paraquat dichloride droplets decompose when exposed to light after being applied to maize, tomato, and broad-bean plants. Small amounts of residues were found in potatoes treated with paraquat as a desiccant, and boiling the potatoes did not reduce the residue (USNLM, 1995).

sethoxydim

In soil and groundwater: Sethoxydim is of low soil persistence. Reported field half-lives are 5 to 25 days (WSSA, 1994; Wauchope *et al.*, 1992). It has a weak tendency to adsorb to soil particles. Laboratory leaching tests have suggested that sethoxydim could leach in soil.

However, in field tests, sethoxydim did not leach below the top 4 inches of soil, and it did not persist (USEPA, 1989e). On soil, photodegradation of sethoxydim takes less than 4 hours (WSSA, 1994). The product Poast photodegrades on soil surfaces with a half-life of approximately 3.7 hours (USEPA, 1989e). Disappearance of sethoxydim is primarily due to action by soil microbes.

In vegetation: Sethoxydim is absorbed rapidly by roots and foliage, and moves both upward and downward in plants from the point of absorption (WSSA, 1994). Sethoxydim is rapidly detoxified in most tolerant plants (WSSA, 1994). The product Poast accumulates in the tissues of crops planted in fields after harvest of treated crops. Measured residues were all below 0.066 ppm (Wauchope *et al.*, 1992).

terbutryn

In Soil and Groundwater: Terbutryn is readily adsorbed in soils with high organic or clay content (Menzie, 1980). The half-life in soil is 14-28 days (USEPA, 1987k). Depending on the application rate, the residual activity of terbutryn in soil is 3 to10 weeks. It is slightly mobile to immobile in soils. Data indicate that it will not leach in agricultural soils. However, its major breakdown product, hydroxy terbutryn, is more mobile and persistent and has potential to leach to groundwater (Menzie, 1980).

In Water: In water, terbutryn is not volatile. It will adsorb to sediment and suspended particulate matter. Half-lives of 180-240 days have been reported for degradation of terbutryn in pond and river sediment. It may be subject to very slow hydrolysis and biodegradation in water (WSSA, 1994).

trifluralin

In soil and groundwater: Trifluralin is of moderate to high persistence in the soil environment, depending on conditions. Trifluralin is subject to degradation by soil microorganisms. Trifluralin remaining on the soil surface after application may be decomposed by UV light or may volatilize. Reported half-lives of trifluralin in the soil vary from 45 to 60 days (Wauchope *et al.*, 1992) to 6 to 8 months (Kidd and James, 1991). After 6 months to 1 year, 80 to 90% of its activity will be gone (USNLM, 1995). It is strongly adsorbed on soils and nearly insoluble in water (Wauchope *et al.*, 1992). Because adsorption is highest in soils high in organic matter or clay content and adsorbed herbicide is inactive, higher application rates may be required for effective weed control on such soils (USNLM, 1995; WSSA, 1994). Trifluralin has been detected in nearly 1% of the 5590 wells tested. However, it has been detected at very low concentrations, typically ranging from 0.002 µg/L to 15 µg/L (USNLM, 1995).

In water: Trifluralin is nearly insoluble in water (Kidd and James, 1991). It will probably be found adsorbed to soil sediments and particulates in the water column.

C. INSECTICIDES

abamectin

Metabolites: Tests with laboratory animals show that ingested avermectin B1a (the main constituent (80%) of abamectin) is not readily absorbed into the bloodstream by mammals and that it is rapidly eliminated from the body within 2 days via the faeces (USEPA, 1990a). Rats given single oral doses of avermectin B1a excreted 69 to 82% of the dose unchanged in the faeces. The average half-life of avermectin B1a in rat tissue is 1.2 days (Thongsinthusak, 1990). Lactating goats given daily oral doses for 10 days excreted 89% of the administered avermectin, mainly in the faeces. Less than 1% was recovered in 1the urine (Thongsinthusak, 1990).

In soil and groundwater: Abamectin is rapidly degraded in soil. At the soil surface, it is subject to rapid photodegradation (USEPA, 1990a; Wislocki, Grosso and Dybas, 1989). Wislocki, Grosso and Dybas' studies provided several useful results. They found that loss of abamectin from soils is primarily due to microbial degradation. The rate of degradation was significantly decreased under anaerobic conditions. Also, because abamectin is nearly insoluble in water and has a strong tendency to bind to soil particles, it is immobile in soil and unlikely to leach or contaminate groundwater. Compounds produced by the degradation of abamectin are also immobile and unlikely to contaminate groundwater.

In water: When tested at pH levels common to surface and groundwater (pH 5, 7, and 9), abamectin did not hydrolyze (Wislocki, Grosso and Dybas, 1989).

In/On vegetation:: Plants do not absorb abamectin from the soil. Abamectin is subject to rapid degradation when present as a thin film, as on treated leaf surfaces. Under laboratory conditions and in the presence of light, its half-life as a thin film was 4 to 6 hours (Wislocki, Grosso and Dybas, 1989).

amitraz

Metabolites: Available data suggest that amitraz, following absorption into the blood, is not readily absorbed into tissues, and is mostly excreted unchanged via the urine (Kidd and James, 1991; USEPA, 1987b; Hayes and Laws, 1991).

In/On vegetation: Reports indicate that amitraz may cause crop injury to young peppers and pears during high temperature conditions (Thomson, 1993).

carbaryl

Metabolites: The metabolites of carbaryl have lower toxicity to humans than carbaryl itself. The breakdown of this substance is strongly dependent on acidity and temperature (USNLM, 1995). Most animals, including humans, readily break down carbaryl and rapidly excrete it in the urine and feces. Workers occupationally exposed by inhalation to carbaryl dust excreted 74% of the inhaled dose in the urine in the form of a breakdown product (USEPA, 1987c). The metabolism of up to 85% of carbaryl occurs within 24 hours after administration (USEPA, 1987c).

In soil and groundwater (temperate conditions): Karinen *et al.* reported a temperature effect on carbaryl decomposition. Beginnning with a concentration of ten mg/liter, 93 percent of the carbaryl was hydrolyzed after eight days at 28 ºC; during the same time at 3.5 ºC, only nine percent was hydrolyzed (Karinen et al., 1967). Fluorescent light slightly accelerated the hydrolysis of carbaryl to 1-naphthol in sea water at 20 ºC (Karinen *et al.*, 1967). Crosby identified the products of ultraviolet irradiation of carbaryl as 1-naphthol and methyl isocyanate (Crosby, 1970), both cholinesterase inhibitors (Mendoza *et al.*, 1969; Crosby, 1969).

In water: In studies by Karinen *et al.*, 43 percent of carbaryl dissolved in sea water (pH 7.5 to 8.1) was converted to 1-napthol in 17 days. In the presence of mud, less than ten percent of the original carbaryl concentration (in both hydrolyzed and unhydrolyzed form) remained after ten days. Although decomposition and adsorption on the soil particles were both responsible for the disappearance of free carbaryl, decomposition is slower in the presence of mud than in the absence of mud (Karinen *et. al.*, 1967).

In/On vegetation: Carbaryl decomposes on most crops with a half-life of three to four days (Back, 1965).

chlorpyrifos

Chlorpyrifos has been extensively studied by the Pesticide Research Group of the Chemistry Department, University of the West Indies. Relevant findings are summarized below.

Chemical degradation: Photochemical studies under sunlight provided half-life values of 1.81 to 5.29 days depending on the area of exposure and intensity of sunlight. Under high intensity ultraviolet radiation the half-lives were much shorter, ranged between 1.50 to 2.57 hours. Leaching studies of the emulsfiable concentrate (dursban formulation) by 170 cm^3 showed 80.7 percent and 85.6 percent of the chlorpyrifos remaining in the top 0-2 cm layer for marvelly sandy loam and Linstead clay loam respectively. Leaching by 340 cm^3 showed 70.2 and 78.8 percent of the chlorpyrifos remaining in the top 0-2 cm layer. Leaching of chlorpyrifos from the 'suSCon' granules was much less than for the emulsifiable concentrate. Encapsulated granules were also found to release chlorpyrifos very slowly in aqueous media (Morris, 1991).

In soil and groundwater: In the Pesticide Research Laboratory, the half-life of chlorpyrifos applied as dursban in marvelly sandy loam was less than a week. In Linstead clay loam in the field, the half-life of chlorpyrifos aplied as dursban was 1 month (Morris, 1991).

In water: A one-time rapid sampling of 17 rivers, 7 natural springs and 13 wells in Jamaica revealed average residues of 18.3 ng/g chlorpyrifos in 9 sediment samples, and 0.001 – 0.002 μg/L in 2 water samples (Robinson, 1997).

In/On vegetation: Despite the zero ppm level policy of the USFDA towards chlorpyrifos, canned callaloo ready for export from Jamaica are frequently found to contain 2-3 ppm levels of chlorpyrifos. Chlorpyrifos is banned for use on callaloos in Jamaica but is used because of the unavailability of a more suitable pesticide.

cypermethrin

Metabolites: For biotransformation of cypermethrin in animal tissues see *Pestic. Sci.* 1987, 21, 1 and ibid. 1990, **30**, 159 (Kidd and James, 1994).

In soil and groundwater: Under laboratory conditions, cypermethrin degrades more rapidly on sandy clay and sandy loam soils than on clay soils, and more rapidly in soils low in organic material (USEPA, 1989c). In aerobic conditions, its soil half-life is 4 days to 8 weeks (Kidd and James, 1991; Wauchope *et al.*, 1992; USEPA, 1989c). When applied to a sandy soil under laboratory conditions, its half-life was 2.5 weeks (Harris, 1981). Cypermethrin is more persistent under anaerobic conditions (USEPA, 1989c). It photodegrades rapidly with a half-life of 8 to 16 days. Cypermethrin is also subject to microbial degradation under aerobic conditions (USEPA, 1989c). Cypermethrin is not soluble in water and has a strong tendency to adsorb to soil particles. It is therefore unlikely to cause groundwater contamination (Kidd and James, 1991).

In water: In neutral or acid aqueous solution, cypermethrin hydrolyzes slowly, with hydrolysis being more rapid at pH 9 (basic solution). Under normal environmental temperatures and pH, cypermethrin is stable to hydrolysis with a half-life of greater than 50 days and to photodegradation with a half-life of greater than 100 days (USEPA, 1989c). In pond waters and in laboratory degradation studies, pyrethroid concentrations decrease rapidly due to sorption to sediment, suspended particles and plants. Microbial degradation and photodegradation also occur (Muir *et al.*, 1985; Agnihotri; 1986).

diazinon

In soil and groundwater: Diazinon has a low persistence in soil. Bacterial enzymes can speed the breakdown of diazinon and have been used in treating emergency situations such as spills. Arthrobacter and Streptomyces microbes are the predominant successful microflora in soil treated with the insecticide (Gunner and Zuckerman, 1968). They work synergistically to degrade the compound. Studies on the effects of diazonon on soil microflora show an increase in actinomycetes after application of diazinon to the soil (Sethunaathan and MacRae, 1969; Gunner and Zuckerman, 1968). Such an increase in one portion of the soil microflora might be expected to bring about significant changes in those soil properties influenced by microflora which are of direct importance to crop production, e.g. mineralization of soil nitrogen. Diazinon seldom migrates below the top half inch in soil, but in some instances it may contaminate groundwater (Howard, 1991). A one-time rapid sampling of 17 rivers, 7 natural springs and 13 wells in Jamaica revealed diazinon in the sediment of 2 rivers (Robinson, 1997).

endosulphan

In soil and groundwater: Degradation of endosulphan is much faster in normal soil than sterile soil. Robinson found that the half-life of endosulphan in sterile sandy loam soil and clay loam soil is 33.0 days, and in normal sandy loam soil and clay loam soil 24.5 days (Robinson, 1997). The degradation of endosulphan in soil and water matrices is faster in sunlight than in the dark. Half-life of α-endosulphan applied to soil exposed to sunlight was 25.2 days, in soil in the dark it was 215.8 days, while the half-life in water exposed to sunlight was 22.2 days and in water in the dark it was 35.1 days. Half-life of β-endosulphan applied to soil exposed to sunlight was 8.7 days, in soil in the dark it was 181.3 days, while the half-life in water exposed to sunlight was 45.3 days and in water in the dark it was 50.4 days. On Blue Mountain coffee plantations where endosulphan is extensively used only 1 to 3 percent leached to 10 to 15 cm depth.

In water: Large amounts of endosulfan can be found in surface water near areas of application (U.S. Agency for Toxic Substances and Disease Registry, 1990). It has also been found in surface water throughout the USA at very low concentrations (Howard, 1991). Robinson's studies in Jamaica produced degradation rates ($t_{1/2}$) of α-endosulphan in river, open sea and "closed sea" waters of 260.3 days, 303.2 days, and 104.9 days respectively (Robinson, 1997). Degradation rates ($t_{1/2}$) of β-endosulphan in river, open sea and "closed sea" waters were 547.5days, 151.5 days, and 86.9 days respectively. A 1990-91 monthly survey in Jamaica revealed endosulphan contamination of the Swift and Spanish Rivers and their coastal waters. A one-time rapid sampling of 17 rivers, 7 natural springs and 13 wells in Jamaica revealed residues of endosulphan in all but 3 rivers (Robinson, 1997).

ethoprophos

In soil and groundwater: Robinson (1997) found that the degradation of ethoprophos is much faster in normal soil than sterile soil. The half-life of ethoprophos in sterile sandy loam soil and clay loam soil is 28.8 days, and in normal sandy loam soil and clay loam soil 10.9 days. The degradation of ethoprophos in soil and water matrices is faster in sunlight than in the dark. Half-life of ethoprophos applied to soil exposed to sunlight was 4.7 days, in soil in the dark it was 12.3 days, while the half-life in water exposed to sunlight was 14.4 days and in water in the dark it was 24.7 days. On Blue Mountain coffee plantations where ethoprophos is extensively used only 2 to 3 percent of ethoprophos leached to 10 to 15 cm depth.

In water: Degradation rates ($t_{1/2}$) of ethoprophos in river, open sea and "closed sea" waters were 132.8 days, 81.2 days, and 64.9 days respectively. A one-time rapid sampling of 17 rivers, 7 natural springs and 13 wells in Jamaica revealed residues of ethoprophos in the sediment of only one river (Robinson, 1997).

fenpropathrin

Metabolites: listed in *Pestic. Sci.* **1985**, 16, 119.

imidacloprid

In Soil and Groundwater: The half-life of imidacloprid in soil depends on the amount of ground cover (it breaks down faster in soils with plant ground cover than in fallow soils) (Scholz and Spiteller, 1992). Organic material aging may also affect the breakdown rate of imidacloprid. Plots treated with cow manure and allowed to age before sowing showed longer persistence of imidacloprid in soils than in plots where the manure was more recently applied, and not allowed to age (Rouchard, Gustin and Waters, 1994). There is generally not a high risk of groundwater contamination with imidacloprid if used as directed. The chemical is moderately soluble, and has moderate binding affinity to organic materials in soils. However, there is a potential for the compound to move through sensitive soil types including porous, gravelly, or cobbly soils, depending on irrigation practices (Jenkins, 1994).

lambda cyhalothrin

Metabolism in animals: In rats, following oral adminstration, rapidly excreted in urine and faeces (Kidd and James, 1994).

In Soil and Groundwater: Lambda cyhalothrin is moderately persistent in the soil environment. Reported field half-lives range from four to 12 weeks (Wauchope *et al.*, 1992; USEPA, 1988c, 1992). Its field half-life is probably close to 30 days in most soils (Wauchope *et al.*, 1992). It shows a high affinity for soil; the reported Koc is 180,000 (Wauchope *et al.*, 1992). Lambda cyhalothrin is not expected to be appreciably mobile in most soils. There is little potential for groundwater contamination. Soils with high sand content or with very low organic matter content may tend to retain the compound to a lesser degree. In field studies of Karate, leaching of lambda cyhalothrin and its degradates from the soil were minimal (Kidd and James, 1991; USEPA, 1988c). Breakdown products formed in the soil environment are similar to those formed in mammalian systems, via the hydrolysis of the central ester bond and oxidation (Kidd and James, 1991). Breakdown rates of both the technical product and Karate were similar under aerobic and anaerobic conditions (USEPA, 1992a).

malathion

In soil and groundwater: Malathion is of low persistence in soil with reported field half-lives of 1 to 25 days (Wauchope *et al.*, 1992). Degradation in soil is rapid and related to the degree of soil binding (Howard, 1991). Breakdown occurs by a combination of biological degradation and nonbiological reaction with water (Howard, 1991). If released to the atmosphere, malathion will break down rapidly in sunlight, with a reported half-life in air of about 1.5 days (Howard, 1991). It is moderately bound to soils, and is soluble in water, so it may pose a risk of groundwater or surface water contamination in situations which may be less conducive to breakdown.

In water: Applied at 1 to 6 lb/acre in log ponds for mosquito control, it was effective for 2.5 to 6 weeks (Howard, 1991). The stability of malathion in solution is a function of pH as reported by Spiller (1961). Malathion is hydrolyzed instantaneously at pH 12.0 , but does not hydrolyze at pH below 7.0 for prolonged periods. The pesticide may therefore persist in neutral and acidic waters long enough to be taken up by aquatic organisms. In sterile seawater, the degradation increases with increased salinity. The breakdown products in water are mono- and dicarboxylic acids (Howard, 1991). In basic solutions, the rate of hydrolysis increases four-fold with a 10 ºC increase in temperature (Muhlmann and Schrader, 1957).

permethrin

In soil and groundwater: Permethrin is of low to moderate persistence in the soil environment, with reported half-lives of 30 to 38 days (Kidd and James, 1991) (Wauchope *et al.,* 1992). Permethrin is readily broken down, or degraded, in most soils except organic types. Soil microorganisms play a large role in the degradation of permethrin in the soil. The addition of nutrients to soil may increase the degradation of permethrin. It has been observed that the availability of sodium and phosphorous decreases when permethrin is added to the soil (WHO, 1990). Permethrin is tightly bound by soils, especially by organic matter. Very little leaching of permethrin has been reported (Wagenet, 1985). It is not very mobile in a wide range of soil types (WHO, 1990). Because permethrin binds very strongly to soil particles and is nearly insoluble in water, it is not expected to leach or to contaminate groundwater.

In water: The results of one study near estuarine areas showed that permethrin had a half-life of less than 2.5 days. When exposed to sunlight, the half-life was 4.6 days (WHO, 1990). Permethrin degrades rapidly in water, although it can persist in sediments (Thomson, 1985; Wagenet, 1985). There was a gradual loss of toxicity after permethrin aged for 48 hours in sunlight at 0.05 mg/L in water (Wagenet, 1985).

D. NEMATICIDES

carbofuran

In soil and groundwater: Carbofuran is soluble in water and is moderately persistent in soil. Its half-life is 30 to 120 days. In soil, carbofuran is degraded by chemical hydrolysis and microbial processes. Hydrolysis occurs more rapidly in alkaline soils. Carbofuran has a high potential for groundwater contamination (Howard, 1991). Small amounts have been detected (1 to 5 ppb) in water table aquifers beneath sandy soils in New York and Wisconsin (Howard, 1991). It is mobile to very mobile in sandy loam, silty clay, and silty loam soils; moderately mobile in silty clay loam soils; and only slightly mobile in muck soils. In the Sacramento valley, a rice-growing region of California, Nicosia *et al.* found that 1.72, 5.40 and 11.03 percent of applied carbofuran were discharged in run-off from three separate fields with 2.2 to 2.8 percent organic matter content (Nicosia *et al.*, 1991). Carbofuran breaks down in sunlight.

In/On vegetation: The half-life of carbofuran is about 4 days when applied to roots, and longer than 4 days if applied to leaves (USNLM, 1995).

oxamyl

In soil and groundwater: Oxamyl is of low persistence in soil with reported field half-lives of 4 to 20 days (Wauchope *et al.*, 1992). Loss is due to decomposition by aerobic and anaerobic bacteria (Wagenet, 1985). Oxamyl is hydrolyzed rapidly in neutral and alkaline soils and more slowly in acid soils (USEPA, 1987g). It does not readily bind, or adsorb, to soil or sediments and it has been shown to leach in soil (Wauchope *et al.*, 1992; USEPA, 1989c). Its adsorption is strongest in soils of high organic matter, and on sandy loam is fairly weak. An increase in temperature causes a decrease in adsorption (Wagenet, 1985). Since oxamyl degrades relatively quickly in the presence of bacteria, it is more likely to be found in groundwater than in surface water. It has been found in very small amounts in the states of

New York (1 to 60 µg/L) and Rhode Island (1 µg/L) (Cohen, 1986). Wherever conditions favour very rapid movement of leachate, oxamyl may reach the groundwater.

copper hydroxide

In/On vegetation: Nelson's studies of copper hydroxide (as Kocide) in Jamaica provided a half-life of 32.9 days on coffee berries (Nelson, 1993).

maneb

In soil and groundwater: Maneb is similar in its environmental fate to mancozeb (Wauchope *et al.*, 1992). Like mancozeb, maneb is of low persistence (with a reported field half-life of 12 to 36 days) but it is readily transformed into ETU, which is more persistent (Wauchope *et al.*, 1992). Since it is strongly bound by most soils and is not highly soluble in water (Wauchope *et al.*, 1992), it should not be very mobile. It therefore does not represent a significant threat to groundwater. Its breakdown product, ETU, may however be more highly mobile. Maneb breaks down under both aerobic and anaerobic soil conditions (USNLM, 1995). In one study, residues of maneb did not leach below the top 5 inches of soil (USNLM, 1995).

In water: Maneb degraded completely within 1 hour under anaerobic aquatic conditions (USNLM, 1995).

In/On vegetation: The main metabolite of maneb in plants is ethylenethiourea (ETU); this is then rapidly metabolized further. Significant amounts of ETU were formed in cooking vegetables that had been experimentally treated with maneb (USNLM, 1995).

SECTION 3: SOCIAL FACTORS AND POVERTY

In this section we will discuss the social context of agriculture and agrochemical use in the Caribbean. This discussion will include the effects of agrochemicals on human health, the relationship between agricultural employment and socio-economic status, and the relationship between agrochemical use and farmworkers' socio-economic status.

The Caribbean's historic development was based on Europe's desire for sucrose sugar (and its by-products) which was extracted from sugarcane which had been brought to the Caribbean islands from Madeira. The islands were developed as plantations growing this single non-native crop for export. The vast majority of the Caribbean population is here because their foreparents were imported as enslaved or indentured labour to work on those island plantations. Both the end of slavery and the political "independence" of the Caribbean island states were connected to the decline in profitability of Caribbean agriculture. So that, today, agriculture remains irretrievably connected in the collective Caribbean memory, to exploitation through slavery, colonialisation, and backwardness and unprofitability. Even though there were some positive developments in Caribbean agriculture in the twentieth century, such as the work of T. P. Lecky in producing the Jamaica Hope (a breed of cattle especially suited to the tropics) and the development of Blue Mountain coffee, agriculture is not a particularly attractive industry for young, bright Caribbean minds, investors or entrepreneurs. Politically and economically, agriculture's significance in the Caribbean has significantly waned. Unfortunately, the Atlantic Alliance and Japan through organs such as the World Trade Organization have pushed from the minds of Caribbean politicians notions such as food sufficiency, and convinced them almost wholly through other organs as well, such as the World Bank and the International Monetary Fund, that the only route forward is by embracing the information age, and shunning agriculture. It's no surprise then that the majority of Caribbean people involved in agriculture suffer the highest levels of poverty. In Jamaica for example, the parish of Westmoreland, which has the largest sugar factory in the English-speaking Caribbean, suffers the highest levels of poverty in the island. Regional neglect of agriculture also explains why some states such as Jamaica import continually larger quantities of food, even as the GDP per capita decreases. It also explains why although we use as much pesticides per hectare as the United States of America - the world's largest producer of pesticides - we have no significant production facilities and so little in the way of basic research into the effects of agrochemicals on human health and our habitat.

In recent years globalisation has threatened the tenuous livelihood of many Caribbean farmers, both large and small. Local dairy farmers in Jamaica have dumped thousands of litres of fresh milk as dry powdered milk flooded into the market from overseas. Pig and beef cattle farmers have also been severely affected as cheap imported meat filled supermarket shelves. Sometime in the recent past chicken parts were allegedly dumped on the Jamaican market from the United States. Vegetable and fruit farmers have also had to compete with imported produce. At the same time local producers of refined foods have often been stymied in their efforts to export to the USA and Europe. For example, exporters of Walker's Wood products made extensive efforts supported by the Jamaican government to meet European standards for labelling and residue levels, but were last prevented from landing their goods in Europe because there was no proof that the products prior to shipping had been stored on shelves made from trees grown in sustainable forests – a standard the Jamaican government officials, academia and common man, all saw as a European protectionist concoction and gross hypocrisy. Thus Caribbean farmers see themselves as an embattled lot. One manifestation of this is clear – recently when the PCA (in Jamaica) restricted the use of Lannate (a.i. methomyl), callaloo farmers acquired and began using chorpyrifos which is banned for use on callaloo. The farmers will do what they see as necessary to survive, including using banned pesticides if they must. RADA has been proactive in its efforts to encourage IPM and reports moderate and encouraging success in the parishes of Trelawny and Portland. RADA also reports that the economic hardship facing some farmers has made it harder for farmers to acquire chemical pesticides and more willing to implement IPM, with *Bacillus thuringiensis* being the most frequently used biological agent. The PRG continues to investigate pesticides that may be effective, safe, and economic for use by local farmers.

As we stated in the Introduction, the source(s) of any information relating to agrochemicals and their effects on human health and the environment must be carefully scrutinised. This admonition goes double for farmers since it is they who are on the "frontlines" of pesticide use. Socio-economic class is the primary factor determining whom farmers get their information on pesticides from. Generally, there are two types of farm owners in the Caribbean, following directly from our colonial heritage. In Jamaica for example, there are large acreages of fertile, flat land used for growing cash crops for export. These farms are owned by wealthy, well-educated members of the upper socio-economic classes, and sometimes by foreign companies or foreigners. The majority of farmers however, are restricted to small plots of land on hillsides, and are poor, not well-educated, and from the lower socio-economic class. Large farm owners rarely come into direct contact with pesticides – their employees from the lower socio-economic classes do. Small farmers apply the pesticides themselves and are most interested in the most effective pesticides they can obtain. In the Caribbean therefore, it is the poor, little-educated, socio-economically disadvantaged who are at greatest risk from improper pesticide use. It is worth bearing in mind that while agrochemical use affects the health of agrochemical applicators and their families most, residents of the communities within which agrochemicals are used, and communities down-stream from areas of agrochemical application are also affected. So are consumers of produce grown in pesticide-treated areas.

According to Mr Bernard Goffe in a telephone interview on February 22, 2002, who as Parish Agricultural Manager for RADA in Trelawny, hosts the largest annual agricultural show in Jamaica, the pesticides salesmen are probably the greatest source of information for these farmers, along with the local farm stores. When farmers have problems relating to pesticide use they go to the community farm stores, which Mr Goffe laments, are operated by untrained personnel with the attendant risk of misinformation passing to the farmers. RADA Extension Officers do their best to advise the farmers about the best pesticides to use in terms of effectiveness, human health concerns and environmental friendliness, but must do so in competition with the very aggressive sales techniques of pesticide salesmen. "Lead farmers," that is, model farmers who are well-respected by their peers, are actively sought out by RADA and encouraged to adopt safe practices and evangelise these practices to their peers. The large farmers are served in Jamaica by several boards, including the Coffee Industry Board and the Banana Export Company Ltd (BECO). These boards have ongoing and continuous training sessions for their pesticide applicator employees.

Surveys and research studies in the USA suggest that farmworkers are exposed to pesticides, though we cannot assume that situations there obtain here, since most farms there are huge by Caribbean standards, while most farms here are small plots. A 1991 survey of farmworkers in Oregon, found that over a three month period, over 60 percent of the workers on conventional farms had been exposed to pesticides. Over 50 percent sought medical attention because of pesticide-related illnesses and over 40 percent took days off because of the same problems. In California, a 1993 review found that nearly one-quarter of pesticide applicators whose blood cholinesterase levels were being monitored had to be removed from work during the course of a year because levels had dropped below acceptable standards. (Cholinesterase is an enzyme involved in transmission of nerve impulses. Its activity is inhibited by several common classes of insecticides.) Surveys need to be done in the Caribbean to establish the status here. We conducted informal investigations in Jamaica but they did not yield substantial information. In the parish of Trelawny, RADA officials could remember no cases of accidental pesticide poisoning or illnesses resulting from pesticide exposure. Checks with a few sugar cane farmers in Westmoreland produced the same response: farm workers use proper protective gear and there are no accidental poisonings. However, Mr Goffe noted that small farmers are unwilling to use respirators "because they are uncomfortable."

The signal words on EPA-compliant pesticide labels ("Danger", "Warning", "Caution") are based on six acute toxicities: acute oral toxicity, acute inhalation toxicity, acute dermal toxicity, eye irritation, skin irritation, and skin allergies (Kemple, 2001). Many persons are mislead into believing that particular pesticides are safe for use because the toxicity levels presented by manufacturers or public officials may be high. These concentrations are not directly measured for humans, but extrapolated from laboratory studies on rats, birds, and other organisms, for humans. By definition, a human exposed to the calculated human LC_{50}/LD_{50} would stand only a 50% chance of survival. Few persons are ever exposed to such high levels, but this does not mean their health and lives are not endangered by exposure to these pesticides. Acute health effects may appear minutes, hours, or even days after exposure. Severe acute toxicity is death. Other observed types of acute toxicity involve racing of the heart, loss of feeling in the limbs, disequilibrium, choking and nausea. These effects are readily noticed and can be immediately brought to the attention of medical personnel. However, pesticide-related incidents and illnesses are rarely reported even in the USA (U.S. General Accounting Office, 1993) and there is no requirement for doctors to report such incidents in Jamaica and most likely throughout the entire English-speaking Caribbean. Of the fifty states comprising the USA, only one – California – has a 'well-developed' monitoring system, and researchers estimate that *only 20%* of incidents are reported there (Wilkinson, 1990). Of the 3 million annual severe poisonings, with 220,000 fatalities, (Jeyaratnam, 1985) estimated by the World Health Organization in 1990, 99% of all deaths occur in the developing world (Jeyaratnam, 1990).

The signal words on EPA-compliant pesticide labels do not consider the following chronic health effects: cancer, birth defects, reduced fertility, damage to the immune system, genetic damage, damage to organ systems, effects on hormone systems, or damage to the nervous system (Kemple, 2001). These signal words do not consider interaction with other chemicals either. Chronic toxicities are adverse health effects resulting from long-term exposure or persistent adverse health effects resulting from a short-term exposure. Long-term as used here means months or years. Because the health effect may not manifest immediately after exposure, the affected individual or medical personnel may not be able to identify the cause. A recent review of studies of farmworkers and their risk of cancer described seven studies in four states (California, Utah, Texas, and New Jersey) as well as one nationwide study, that identified significantly increased frequency of liver cancer, lung cancer, cancer of the pharynx, multiple myeloma, cancer of the stomach, cervical cancer, prostate cancer, and testicular cancer (Zahm and Blair, 1993). A survey of babies born in Imperial County, California, found that parents working in agriculture had an increased risk (almost double) of having children born with reduced limbs (Schwartz, Newsum and Heifetz, 1986). A study of children born with extrahepatic biliary atresia (EHBA: a liver defect which is lethal unless the child receives a liver transplant) found that fathers who worked on farms and were exposed to pesticides were twice as likely to have children with EHBA. Exposed mothers also had an increased risk of children with EHBA (Magee, Tockman and Lees, 1991). A survey of couples seeking treatment at an infertility clinic found that when couples sought treatment because of low sperm count, the men were ten times as likely to be agricultural workers as were men from couples seeking treatment for other reasons. The men with low sperm counts reported long-term (5-21 years) exposure to insecticides and other pesticides (Strohmer *et al.*, 1993). Unfortunately, there are no such studies on Jamaica or the Caribbean of which we are aware. Our checks with the public health authorities and the Cancer Registry (Jamaica) revealed that age and gender-specific data is collated for cancer cases, but no data has been gathered on the occupation or specific geographical location of cancer cases. Research in this area is urgently needed.

Specific procedures for handling and managing some of the pesticides being reviewed in this study are to be found in Appendix B. Much of that information can be found on the World Wide Web at http://www.ilo.org/public/english/protection/safework/cis/products/icsc/compguid.pdf.

SECTION 4:
CONCLUSIONS AND RECOMMENDATIONS

Clearly our investigations show that there are significant gaps in the data needed on how agrochemicals impact human health and the environment in the Caribbean. These gaps cannot be filled by ad hoc research since there is no one entity or group in whose interest it is to seek the answers to the questions which these gaps pose. A multidisciplinary approach involving environmental scientists, chemists, geographers, biologists, medical doctors, public health specialists, economists, international trade lawyers, educationalists and marketing experts is necessary. The regional governments, school systems, public health systems, agricultural extension services, environmental protection agencies and media companies all have to be involved.

Some of the questions for which we currently have incomplete answers at best and which demand urgent attention are:

1. What is the actual extent of agrochemical pollution in the Caribbean?
2. What effect is pesticide use having on indigenous and non-indigenous fish and wildlife?
3. For each of the agrochemicals being used, what is its degradation rate and products in **our environment**?
4. What mobility patterns do the agrochemicals exhibit in Caribbean soils, air and water bodies?
5. To what extent do accidental poisonings happen in the Caribbean?
6. What is the pesticide load being carried by Caribbean consumers of pesticide-treated foods grown in the region?
7. How is agrochemical usage affecting the reproductive and mental health of farmers, other agrochemical applicators, and their families?
8. What are the factors affecting agrochemical applicators' implementation of "best practices" in the use of agrochemicals?
9. What measures can we implement to guide purchasers/importers of agrochemicals in the Caribbean in making the most appropriate choices within the context of the aggressive marketing tactics of multinational producers of those products?
10. How do we effectively educate our population about the effects of agrochemicals on their health, biodiversity and the environment?

A comprehensive coordinated research programme which has the commitment and financial support of the region's governments needs to be carried out. This programme will necessarily be comprised of different projects with different disciplinary emphases. This proposal is admittedly very ambitious, and successful implementation would require the buy-in of many different stakeholders across the region. But this project has laid the groundwork, and could be expanded gradually in scope to embrace the various aims put forward.

Some steps that could be taken to advance the necessary research are:

1. The Chemistry Department, UWI, Mona, should be provided the funds necessary to begin field studies of agrochemical fate in the watershed areas and areas of heavy pesticide use in Jamaica.
2. The Chemistry Department, UWI, Mona, should be provided with the funds to collate the divers bodies of information relating to agrochemicals, their use and fate in the Caribbean (e.g. meteorological data, water sampling etc.)

3. The Chemistry Department in collaboration with the Department of Life Sciences should be provided with the funds and equipment necessary to extend studies into the fate of agrochemicals in marine biota.
4. The Chemistry Department, UWI, Mona, should be provided with the funds and technology to customize and utilize mathematical models such as CREAMS for the Caribbean.

In addition to this research, we think there are three areas that may be immediately tackled. We think that mechanisms should be put in place to allow for the monitoring of pesticide related illnesses. There already exist in Jamaica – which could serve as a pilot for regional implementation – a Cancer Registry Unit which collects cancer statistics in Montego Bay and at the University Hospital of the West Indies, in Kingston. The only problem with the Cancer Registry Unit's current procedures is that demographic data is not collected – making it impossible for us to trace which communities the affected persons are from and possible correlation with agrochemical use and/or exposure. Jamaicans (and perhaps most people in the Caribbean) enjoy better access to public health than most Americans and Europeans and we think that it would not place undue stress on the public health system to require that pesticide poisonings be reported to a central health authority.

The second area that may be immediately tackled is the design and implementation of an IMPP programme. A comprehensive regional program of Integrated Management of Pests and Pesticides (IMPP) such as that proposed by Mansingh (1987) should be designed by stakeholders, including pest operators, local and regional agrochemical importers and manufacturers, farmers, environmental groups, state bodies, the public, and the relevant university departments. IMPP places equal emphasis on integrated pest management (IPM), and integrated management of pesticides, including management of residue run-off, ecotoxicity, and environmental risk assessment. Robinson recommends that Mansingh's proposed IMPP is a more relevant framework for use in tropical island ecosystems than IPM as used in temperate countries, given that "pest control in developing countries is almost totally dependent upon pesticides," and "poor agronomic practices of hillside farmers encourage soil erosion and pesticide drift and run-off" (Robinson, 1997). Such a program would balance the interests of all affected groups and benefit all groups. For several years RADA, in Jamaica has had experimental plots. However anecdotal evidence suggests there has not been significant buy-in from farmers, and so as far as we know IMPP remains only an academic concern.

Ultimately, success of IMPP will depend to a large extent upon the political will of Caribbean governments, and their willingness to adjust their current line of thinking to safeguard the long-term health of their citizenry as well as biodiversity within the Caribbean, especially within the aquatic environment. Some of the adjustments we think necessary are:

1. to recognize the utmost importance of developing sophisticated procedures for selecting appropriate agrochemicals for our island ecosystems, that balance the effectiveness of the agrochemicals, their immediate economic cost, their threat to human health and life, their threat to the health of non-target and/or beneficial organisms, their persistence within the environment, and other effects on the environment, and their potential benefits to immediate users, and the wider society.
2. to recognize and protect the public against the immense financial and political clout of the agrochemical companies from which we import.

3. to acknowledge the influence of the farmers of cash crops such as coffee, banana, and citrus, and balance their well-being, the general public's well-being, and good environmental practices.

Given the large sums of foreign exchange spent to import agrochemicals into the region every year, it is in the region's interest for governments to develop sophisticated and accurate means of measuring the benefits and costs of agrochemicals to the individual states and the region as a whole, and to foster an indigenous agrochemical industry which does regional research, development and production.

The third area for immediate action is education. The relevant University departments and centres must play a greater role in educating the government and other important major decision-makers about the far-reaching effects of agrochemicals in addition to lobbying for more funds for research. In collaboration with the Pesticide Control Authorities/Boards of the different Caribbean territories, we should design and implement a programme to educate the general public and users of agrochemicals especially, of the many detrimental non-lethal effects of pesticides on the health of humans and other organisms, as well as of the undesirability of pesticide persistence in the environment. An educational programme with a more restricted set of educational goals (the "Mine-Yu-Cide" series in Jamaica) was implemented with some success, and we believe that an expanded programme now needs to be carried out throughout the region.

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