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Modelling of DDT dynamics in Lake Kariba, a tropical man-made lake, and its implications for the control of tsetse flies

Håkan Berg

Berg, H., Department of Systems Ecology, Stockholm University, S-106 91 Stockholm, Sweden

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DDT has been extensively used in western Zimbabwe for the control of tsetse flies and malaria mosquitoes, and in agriculture. The input between 1967 and 1990 in the Lake Kariba area was used to do a retrospective analysis of the distribution and dynamics of the insecticide in the environment. The results are compared with trends in available data and it is suggested that the turnover of DDT in the tropics is faster than in temperate areas. It is also suggested that the residence time of DDT in Lake Kariba is comparatively short due to the lake's characteristics of being man-made and tropical. These characteristics may also influence the accumulation of DDT in the aquatic biota. Potential environmental effects from ground spraying with DDT are compared with other technically and economically feasible methods to control tsetse flies in the area, such as targets treated with deltamethrin and aerial spraying with endosulfan.

1. Introduction

The use of pesticides in developing countries has increased during the last decades and is expected to increase even faster in the future (Alabaster 1981, Edwards 1985, Jain 1992). The reason for this is rather obvious. According to Owen (1973), in addition to man himself, insects are a main overall threat to economic development in tropical Africa. It is estimated that pests, including insects, fungi etc. destroy well over one- third of the annual world production of food. In Africa, losses are over 40% and the main part of pesticide used are insecticides (Alabaster 1981, Edwards 1985, Murti 1989). The creation of large water reservoirs is also often accompanied by an increase in endemic diseases. In the Zambezi River system, for example, malaria, sleeping sickness and bilharzia are important health problems (Jörgensen & Vollenweider 1989). The main deterrent against organisms transmitting these diseases is the use of different pesticides (Alabaster 1981, Calamari 1985), of which some have been partially or completely banned in developed countries for environmental reasons (Alabaster 1981). In Africa, for example, approximately 13 000 tons of DDT was used between 1960 and 1989 (Ngabe & Bidleman 1992).

In Zimbabwe, consumption of DDT probably reached about 800 tons per annum in the early 1970s, but was withdrawn from domestic and gardens use in 1973, and was restricted solely for the control of malaria mosquitoes and tsetse flies in 1985 (Mpofu 1987, Douthwaite & Tingle 1994). Ground spraying with DDT against tsetse flies was performed in Zimbabwe for the first time in 1967 (Barrett 1992). Between 1967 and 1976, the total area sprayed annually ranged from 8 000 to 11 000 km^2 and the amount of DDT applied increased from 10 to 20 kg active ingredient (a. i.) per sq. km (Barrett 1992). Operations from the late 1970s up until 1980 were limited due to the independence war, but recommenced in 1980, concentrating initially on the western region (Fig. 2). After 1981, tsetse control operations were conducted only on the Zambezi front. By that time, concern about the environmental impact of using DDT for tsetse control had increased and other tsetse control techniques became more important (Shereni 1990, Barrett 1992). From 1982 to 1988, large-scale aerial spraying operations using endosulfan were carried out, and in 1984 the use of odour-baited targets treated with deltamethrin was introduced (Barrett 1992). As a result, the total area sprayed with DDT declined, and in 1990 it was only some 200 km^2 (Barrett 1992).

It has been proposed that the tsetse control operations are mainly responsible for the high levels of DDT in the Lake Kariba ecosystem (Matthiessen 1985). One of the first and most comprehensive reports on the contamination of DDT in the environment around Lake Kariba was done by Matthiessen (1985). This study focused mainly on the drainage basin of Lake Kariba and included mostly terrestrial samples, although some samples of sediments, water, mussels (*Mutela dubia)* and fish (*Hydrocynus vittatus)* were analysed from river mouths entering the lake. Two other reports address the levels in crocodiles and crocodile eggs (Wessels et al. 1980, Phelps et al. 1989), and Mhlanga et al. (1986) analysed DDT residues in kapenta fish. After this, several reports considering the effects of DDT on the terrestrial fauna (Lambert 1993, Tingle 1995, Tingle & Grant 1995), and especially the avian fauna, have been published by Douthwaite (1992a–d), and Douthwaite et al. (1992).

The simulations performed in this study are a first approach trying to address the distribution and fluxes of DDT in the tropical ecosystem of Lake Kariba, relating concentrations in different compartments throughout the system to past and future loadings. There are reasons to believe that the behaviour of DDT and other persistent organochlorine compounds in the tropics are different compared with temperate areas, and because of the continued use of these compounds in the tropics it is important to develop a better understanding of their behaviour under tropical conditions (Bourdeau et al. 1989, cf. Iwata et al. 1994, Loganathan & Kannan 1994). A basic understanding of the mass balances that are established as DDT, originating from various sources, migrates between soil, air, water, sediment and biota, is necessary to evaluate potential hazards of DDT to the environment (Clark et al. 1988). In addition, if the flow of DDT can be tracked through the lake system, major sources can be identified. Thus, models provide systematic methods of considering the complex behaviour of chemicals in a tropical ecosystem, and may indicate where to concentrate future research.

Accordingly, the objective of this study was to perform a retrospective analysis of the distribution and dynamics of DDT between 1967 and 1990 in the Lake Kariba area. The results are compared with other studies considering the distribution and dynamics of DDT in tropical and temperate areas. Another aim was to evaluate how the comparatively high volatilisation rate in the tropics affects the behaviour of DDT and what environmental consequences this may have. Lastly, the potential environmental effects from ground spraying with DDT are compared with other technically and economically feasible methods to control tsetse flies in the area.

2. Methods

2.1. Study area

The Kariba Dam was built on the Zambezi River between 1955 and 1958 for the production of hydroelectric power (Beadle 1981). It is one of the biggest man-made lakes in the world with a surface area of 5 360 km^2 and a mean depth of 29 m (Fig. 1) (Ramberg et al. 1987). About 80 percent of the water comes from the Zambezi River, 14 percent from secondary rivers and 6 percent comes in the form of rain (Marshall 1984). The water exchange time is approximately 2.5 years (Table 1) (Marshall 1984). The lake is warm and

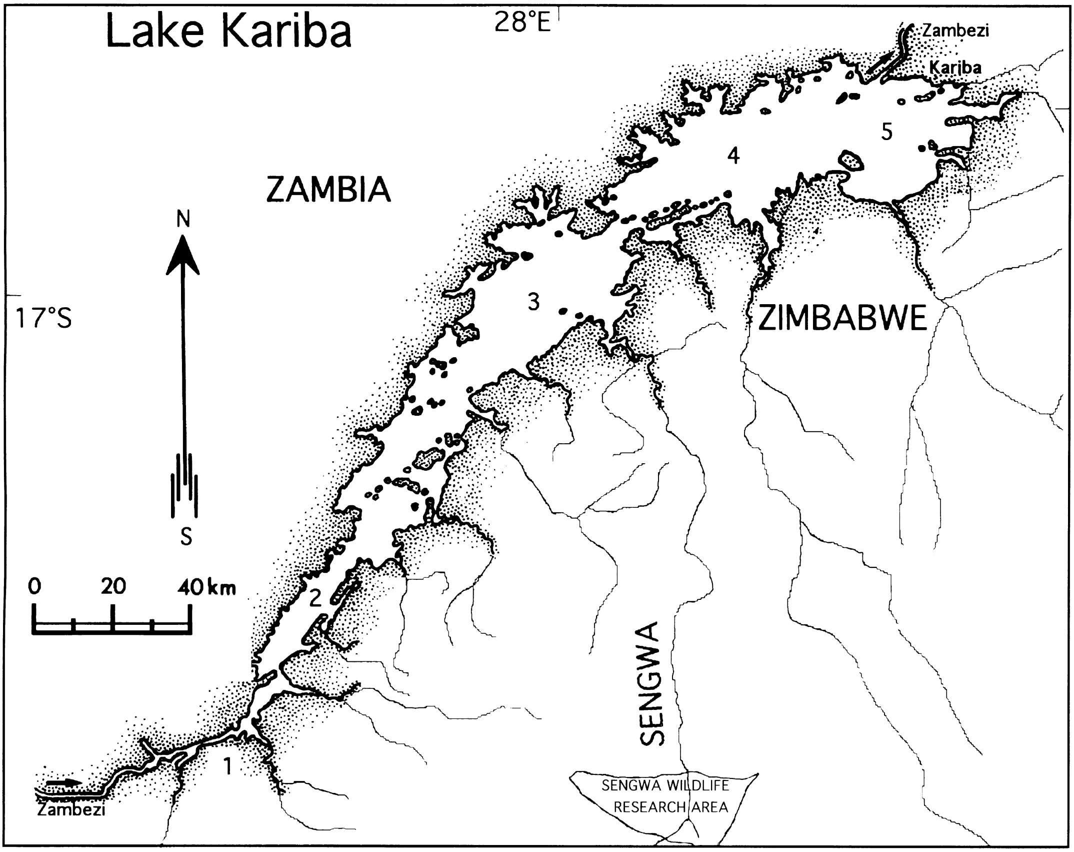

Fig. 1. Lake Kariba with its five main basins. The lake is 320 km long and has an area of 5 364 km^2 (Balon & Coche 1974). The catchment area on the Zimbabwean side covers the area where most of the DDT was applied between 1967 and 1990 (Matthiessen et al. 1984, Barrett 1992). The Sengwa Wildlife Research Area was used to estimate the distribution of pesticides in the local environment. The distribution and fluxes of DDT between air, soil, water, sediment and biota were calculated for the whole Lake Kariba area by estimating the volume of the different compartments. The water volume was set to 156.5 km^3 using an average depth of 29.2 m (Balon & Coche 1974). The sediment volume was estimated from a sediment depth of 5 cm. The area of soil 1 was calculated as the average area within the Lake Kariba catchment area where ground spraying with DDT for tsetse control had taken part between 1968 and 1987 (5 420 km^2) (Barrett 1992). The agriculture area (soil 3) was set to 700 km^2 (Matthiessen 1985), and the area of soil 2 was defined as the area left to give a total system area (i.e. lake and drainage area) of 40 000 km^2. The soil depth was estimated as 5 cm for soil 1 and 2, and 10 cm for soil 3 (cf. Matthiessen et al. 1984, Hutton 1991, Mackay 1991, Van de Meent 1993). The atmospheric height was set to 1000 m reflecting that region of the troposphere which is most effected by local emissions (Mackay et al. 1992). For other environmental parameters, see Table 1.

monomictic with temperatures ranging from about 20°C in winter to over 30°C in summer (Balon & Coche 1974). The climate is characterised by a short erratic rainy period in December–March, and a dry winter in May–September (Balon & Coche 1974). The temperature ranges from 10°C to 35°C (Balon & Coche 1974) and rainfall varies between 650 mm near the lake (540 m. a.s.l.) to 1 000 mm in the Zambezi escarpments (800 m. a.s.l.) (Hutton 1991). The catchment area on the Zimbabwean side (where tsetse control has taken part) is traversed by frequent small streams. Most soils in the area are shallow, less than 0.25 m deep, lithosoils which are unsuitable for agriculture (Hutton 1991, Douthwaite & Tingle 1994). The texture is mainly loamy sand and sandy loam (Hutton 1991). The erosion hazard is above average in the area (Hutton 1991), and it is estimated that approximately 5 000 tons of soil per sq. km cropland are lost each year (Grohs 1992). However, only a small fraction of this ends up in rivers and reservoirs (Grohs 1992). Different estimates of the sediment yields of the drainage basin downstream from the Victoria Falls lie in the range of 4–400 tons per sq. km.

(Balon & Coch 1974, Piney 1988). For more characteristics of the environment, see Fig. 1 and Table 1.

After the lake was formed, it underwent changes in its physicochemical characteristics and a marked succession in plant and animal development occurred (Coche 1968, McLachlan & McLachlan 1971, Balon & Coche 1974, Balon 1978, Marshall et al. 1982, Ramberg et al. 1987, Machena & Kautsky 1988, Machena 1989). Today, the lake constitutes an important resource to the area, and beside the generation of hydroelectric power to Zambia and Zimbabwe, tourism and fish production are viewed as major economic benefits (Oldhof & Shapiro 1989). In the mid seventies, the fish catches declined, which presumably was due to natural changes in the Lake Kariba ecosystem (Ramberg et al. 1987), but there has also been speculations that they could be associated to environmental degradation (Mubamba 1990). Unfortunately, very little work has been done studying the impact from pesticides on the fish stocks (Mubamba 1990). However, considering the importance of the lake's fisheries resources, the future use of persistent pesticides in the catchment area should be as small as possible.

2.2. Predicted environmental concentrations

Environmental concentrations of the insecticides, used to control tsetse flies, are estimated for two spatial scales. On the regional scale, emissions of DDT are regarded as dif-

Table 1. Environmental characteristics.

Lake Kariba			
Water and sediment areas	5364	km^2	(1)
Water volume	156.5	km^3	(1)
Mean water depth	29.2	m	(1)
Inflow			
Zambezi R.	51.7	$km^3 \times yr^{-1}$	(1)
Secondary rivers	9.2	$km^3 \times yr^{-1}$	(1)
Replacement time	2.4	yrs.	(2)
Active sediment depth	0.05	m	
Biomass	1.5	mg dry wt. $\times$ l^{-1}	(3)
Suspended matter	30	mg dry wt. $\times$ l^{-1}	(1)
Temperature	25	°C	(1)
Sengwa (downstream Sengwa WRA)			
Length	165	km	
Volume	0.1	km^3	
Flow	0.3	$km^3 \times yr^{-1}$	
Catchment area			
Soil 2 (unsprayed)	28516	km^2	
Soil 1 (sprayed)	5420	km^2	(4)
Soil 3 (agriculture)	700	km^2	(5)
Soil 1 & 2 depth	0.05	m	
Soil 3 depth	0.1	m	
Runoff	220	$mm \times yr^{-1}$	
Erosion	200	$ton \times km^{-2} \times yr^{-1}$	
Rain	730	$mm \times yr^{-1}$	(1)
Temperature	10–35	°C	(1)
Compositions			
Org. carbon soil 1 & 2	1	%	(6)
Org. carbon soil 3	3	%	(7)
Water soil	10	%	
Org. carbon sediment	3.5	%	(8)
Water sediment	90	%	
Org. carbon susp. matter	8	%	(8)
Water susp. matter	90	%	
Fat biota	2	% (vol.)	(5)
Water biota	90	%	
Density solid matter	2.5	$ton \times m^{-3}$	(9)

(1) Balon & Coch 1974, (2) Marshall 1984, (3) Machena et al. 1993), (4) Barrett 1992, (5) Matthiessen et al. 1984, (6) Tingle & Grant 1995, (7) Bonazountas & Wagner 1982, (8) Troell & Berg in manus, (9) van de Meent 1993.

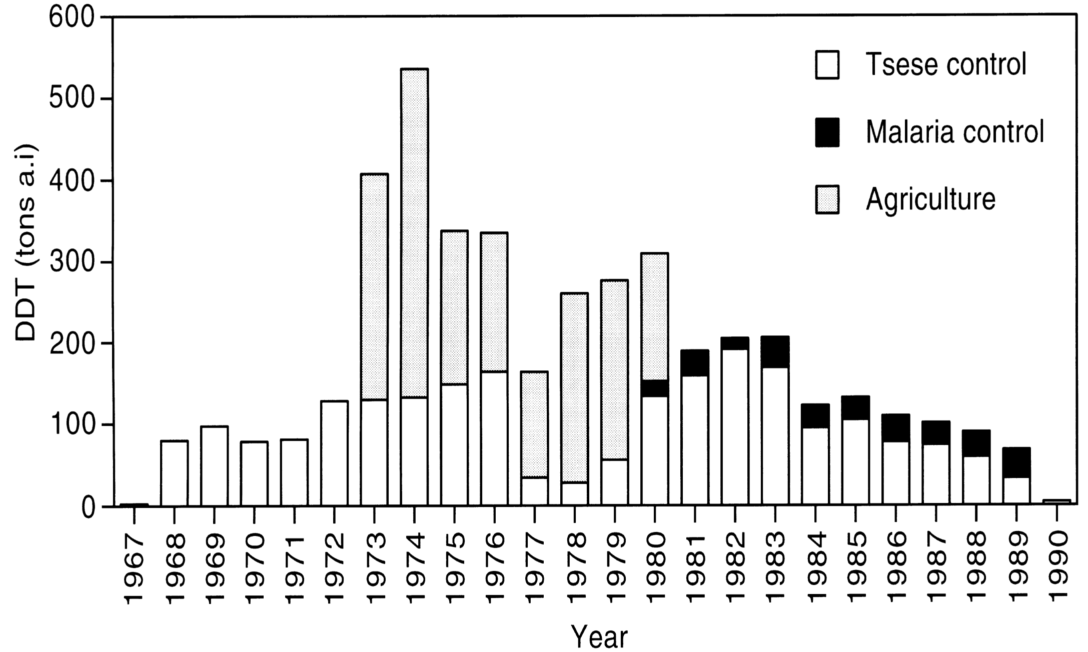

Fig. 2. The main input of DDT in the area between 1968 and 1989 was from tsetse control operations, with an average input of 108 ton per year (Barrett 1992). Mosquito control operations were estimated to contribute approximately 30 tons annually since 1980, which is about 10–15% of the total use in Zimbabwe (Anon. 1991). The mean agriculture input between 1973 and 1980 was about 220 tons (Matthiessen 1985). Although much of the drainage area of Lake Kariba is situated in Zambia, Mathiessen et al. (1984) found that more than 95% of the DDT sprayed in the immediate catchment area of the lake occurred in Zimbabwe, and therefore the input from Zambia was not taken into account. However, an import of 0.02 $\mu g \times l^{-1}$ DDT in water from the Zambezi River and 2 $ng \times m^{-3}$ in air were included, between 1972 and 2000, as a non point source of DDT from areas outside Lake Kariba.

fuse and continuous, and long-term exposure levels are estimated for the whole Lake Kariba area. On the local scale, emissions are allowed to vary in time, and levels are estimated for a specific area to account for time- and space-varying processes. Predicted levels in the local environment are based on three basic scenarios which are estimated to be equally efficient for the control of tsetse flies in the area (Barrett 1992). The methods considered are ground spraying with DDT, targets treated with deltamethrin and aerial spraying with endosulfan. The predicted environmental concentrations are used to make a first order comparison of the potential environmental hazard associated to these three methods, and should be regarded more as relative than absolute concentrations expected in the local environment of Lake Kariba. Physicochemical properties of DDT, deltamethrin and endosulfan are found in Table 2.

2.2.1. Regional distribution of DDT

The regional distribution of DDT was calculated with SimpleBox, version 1.0 (van de Meent 1993). SimpleBox is a multimedia "Mackay-type" model (Mackay 1991) in the sense that it is a fate model in which the environmental compartments are represented by homogenous boxes. Eight different compartments were considered; air, water, sediment, suspended matter, aquatic biota, soil sprayed with DDT for tsetse control (soil 1), soil contaminated from mosquito control operations (soil 2) and agriculture soil (soil 3) sprayed with DDT. Characteristics of the different environmental compartments are presented in Fig. 1 and Table 1, and transportation of DDT between the compartments is described in Fig. 3.

Four simulations were performed to calculate the fluxes of DDT at steady state, representing periods with different input of DDT to the Lake Kariba area between 1970 and 2000 (Fig. 4a–d). These are nonequilibrium, steady-state computations, or "level III" models (Mackay 1991). In the first simulation, 108 tons of DTT from tsetse control operations were applied annually to soil 1 and 30 tons from malaria control to soil 2 (Fig. 4a). In the second run, an import of 1 ton of DDT with water and 44.2 tons with air were added to the first scenario (Fig. 4c). In the third simulation, the input to soil 2 was changed to an input of 223 tons of

DDT to soil 3 (Fig. 4b). In the fourth scenario, DDT was only imported through water and air (Fig. 4d).

The next simulation was a nonequilibrium, non steady-state computation, equivalent to a "level IV" model (Mackay 1991) (Fig. 4, large graph). In this scenario, all loadings were specified each year between 1967 and 1990 in accordance with Fig. 2. DDT used in tsetse control was applied to soil 1 throughout the period, DDT used in agriculture to soil 3 for 8 years (1973–1980) and DDT used in malaria control to soil 2 for 10 years (1980–1989) (Fig. 2). In addition to this, there was an annual import of 1 ton of DDT with water and 44.2 tons with air between 1972 and 2000. In the last ten years (1991–2000), it was assumed that no more DDT was used in the area, however the import of DDT with water and air was allowed to continue.

In the last run, the fluxes of DDT were calculated at different volatilisation/degradation- and erosion-rates (Fig. 5a–d). The sediment yield was set to 4 tons (Fig. 5a and b) and 200 tons (Fig. 5c and d) per sq. km, respectively. In addition to the high volatilisation/degradation rate used in the earlier simulations (Fig. 5b and d) (similar to tropical conditions), a lower dissipation rate, similar to what is expected in a temperate area, was used (Fig. 5a and c). For simplicity, it was assumed that all degradation rates were half of the rates in the earlier simulations. It was also assumed that a lower proportion DDT (60%) dissiputed throught volatilisation, resulting in a half-life of DDT in soil of 3 years. This is similar to estimated half-lives of DDT in temperate soils (Edwards 1966, 1975).

2.2.2. *Local distribution of pesticides*

Two models were used in this simulation. To estimate the fate of the DDT applied to soil, the Seasonal Soil Compartment Model (Seasoil) (Bonazountas & Wagner 1982) was used. The fate of pesticides in water was estimated with the Exposure Analysis Modelling System (EXAMS II) (Burns 1990), which was programmed to simulate the characteristics of a hypothetical river in the area.

To confirm the relevance of the predicted environmental concentrations calculated by the models, focus was on the Sengwa Wildlife Research Area, which earlier had been studied by Matthiessen (1985). This area is situated along the Sengwa River, some 110 km from Lake Kariba (Fig. 1). According to Matthiessen (1985), this area had been treated annually with 200 g a.i DDT per hectare for at least 10 years when he collected his samples in 1983. That year, it was estimated that approximately 900 km^2 within the catchment area of Sengwa had been sprayed with DDT. This was used as the average area sprayed in August each year with 18 tons a.i. DDT. The atmospheric deposition of DDT was estimated from the steady-state simulations. Between 1968 and 1972 and 1981 and 1989 it was set to 0.3 kg $\times$ km^{-2} $\times$ yr^{-1} and between 1973 and 1980 to 0.5 kg $\times$ km^{-2} $\times$ yr^{-1}. This was assumed to be deposited during the rainy season (November–April). DDT in run-off was estimated from the Seasoil model. Because of the range in reported erosion rates, two simulations were performed with annual sediment yields of 4 and 300 tons per sq. km. As very few data were available on the Sengwa River, general characteristics were estimated from other rivers in the area, data on Lake Kariba, and by consulting maps over the area (Balon & Coche 1974). Characteristics of the Sengwa River and the catchment area of Lake Kariba are found in Table 1.

To get a rough idea of the highest expected levels of deltamethrin in the aquatic environment, when using targets to control tsetse flies, a basic scenario was used (Barrett 1992). In this scenario, four targets are deployed per sq. km. These targets consist of black cotton cloths (1.8 $\times$ 1 m), which together are treated each year with approximately 0.75 g active ingredient (a.i) of deltamethrin initially and re-treated with 0.4 g a.i every three months (Barrett 1992). According to Hill and Johnson (1987), the dissipation is probably biphasic, with an initial high dissipation rate due to weathering processes (i.e. volatilisation, photodecomposition, hydrolysis etc.), followed by a slower degradation of residues that have penetrated the cloth. As a first rough estimate, it was assumed that approximately 50% of the total dose applied to the four targets (~2 g a.i.)

Table 2. Chemical characteristics.

	DDT		Deltamethrin		Endosulfan	
Molecular mass (g $\times$ mol^{-1})	354.5	(1)	505.2	(5)	406.9	(6)
Vapour pressure (Pa)	2.3×10^{-5}	(1)	2.0×10^{-6}	(5)	6.4×10^{-3}	(6)
Solubility (g $\times$ m^{-3})	3.1×10^{-3}	(1)	2.0×10^{-3}	(5)	0.33	(6)
Log K_{ow}	6.0	(1)	5.4	(5)	4.7	(6)
Degradation rate (day^{-1})						
Air	4.1×10^{-3}	(2)	4.1×10^{-3}	(2)	4.1×10^{-3}	(2)
Water	2.5×10^{-4}	(3)	0.02	(5)	0.03	(6)
Sediment	2.1×10^{-4}	(4)	–		–	
Soil	4.8×10^{-4}	(4)	5.3×10^{-2}	(5)	0.02	(6)

(1) Landner & Walterson 1989, (2) van de Meent 1993, (3) Wolfe et al. 1977, (4) See text for Fig. 3, (5) Torstensson 1989, (6) Andersson & Hanze 1992.

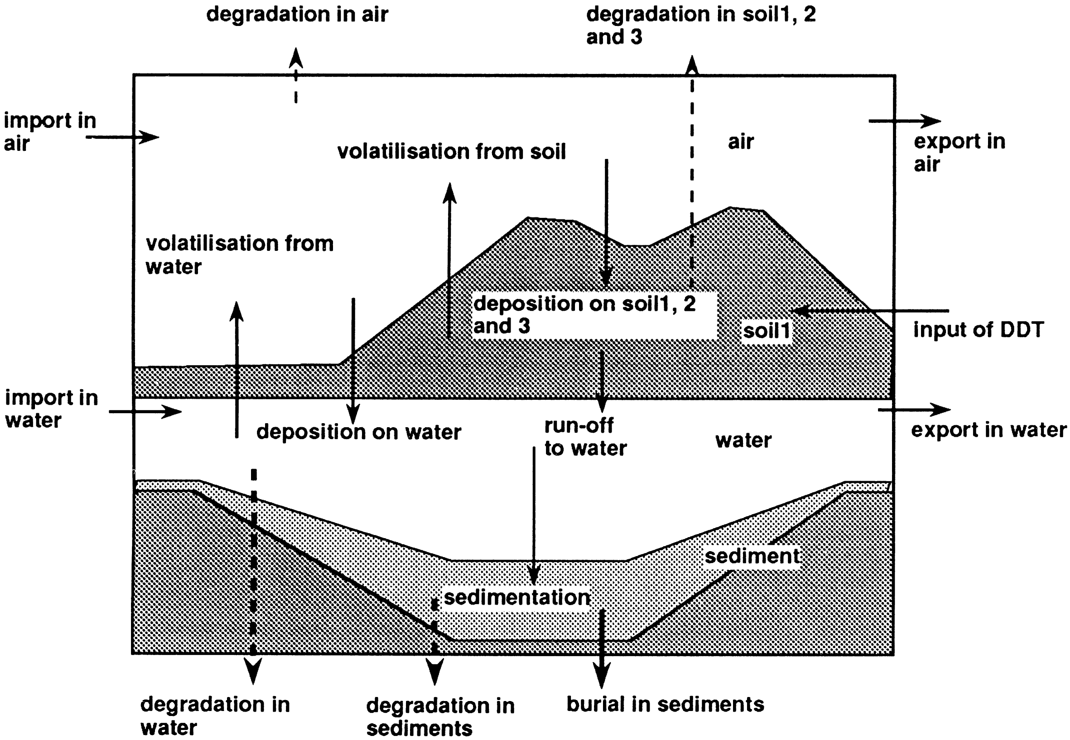

Fig. 3. Estimations of different environmental processes governing the dynamics of DDT in the Lake Kariba area. — Dissipation from soil was estimated by using the average ratio (0.4) between the amount of unchanged DDT in post-rain samples (DDT_1) to pre rain samples (DDT_0) (Matthiessen et al. 1984) in: $DDT_1/DDT_0 = e^{-(k_1+k_2)t}$, where t was the time between the pre- and post-samples equal to 180 days, k_1 the rate constant for dissipation through degradation, and k_2 the rate constant for dissipation through volatilisation (Sleicher & Hopcraft 1984). From above, k_1 and k_2 were calculated as 0.0005 and 0.0045 day^{-1},respectively, which corresponds to half-lives of about 4 years and 150 days. As a result the total half-life is 140 days, which conforms with field estimations of the dissipation rate of DDT from tropical soils (Yeadon & Perfect 1981, Nash 1983a, Sleicher & Hopcraft 1984, Samuel & Pillai 1989, Tingle 1995). — Run-off. The sediment yield was estimated as 200 tons per km^2 (cf. Piney 1988). — Dry deposition. The mass transfer coefficient for dry deposition was estimated by multiplying the deposition velocity of aerosol particles ($0.2\ cm \times s^{-1}$) (cf. Mackay & Patersson 1986, Bidleman 1988, Rapaport & Eisenreich 1988) with the fraction of DDT in air associated with these particles (5%) (Ngabe & Bidleman 1992, van de Meent 1993).— Wet deposition. The mass transfer coefficient for wet deposition was estimated by multiplying the washout ratio of DDT (50 000) (cf. Eisenreich et al. 1981, Bidleman & Leonard 1982, Mackay & Paterson 1986, Bidleman 1988) with the rainfall rate ($730\ mm \times yr^{-1}$) (cf. van de Meent 1993).— Degradation in air. A maximum half-life of 170 days was assumed (van de Meent 1993). — Import in air and water. The import of DDT with the Zambezi River was estimated from Mathiessen et al.'s results ($2\ \mu g \times l^{-1}$) (1984) but the air levels were rough estimates (cf. Ngabe & Bildeman 1992) and has probably varied over the years. — Degradation in water. The half-life in water was set to 8 years (Wolf et al. 1977). — Sedimentation rate of $0.8\ mm \times yr^{-1}$ was estimated from sedimentation values measured in basin 5 between 1991 and 1994 (Troell & Berg in manus). — Volatilisation from water. The liquid- and gas-phase mass transfer coefficients were set to 3.6×10^{-5} and $2.85 \times 10^{-3}\ m \times s^{-1}$, respectively (Eisenreich et al. 1981), and were used to estimate the volatilisation from water (van de Meent 1993). — Burial in sediments at a rate of $0.2\ mm \times yr^{-1}$ was estimated from the ratio between sedimentation and burial used by Mackay (1991). — Degradation in sediments. Oliver et al. (1989) estimated the half-life of DDT in three sediment cores from Lake Ontario as 14, 15 and 21 years. From this, the half-life in sediments from Lake Kariba was set to nine years, as the temperature in Lake Kariba is high all year around. Measurements of oxygen consumption in sediment cores also indicated a slightly higher activity in sediments from Kariba compared with temperate areas (Troell & Berg in manus).

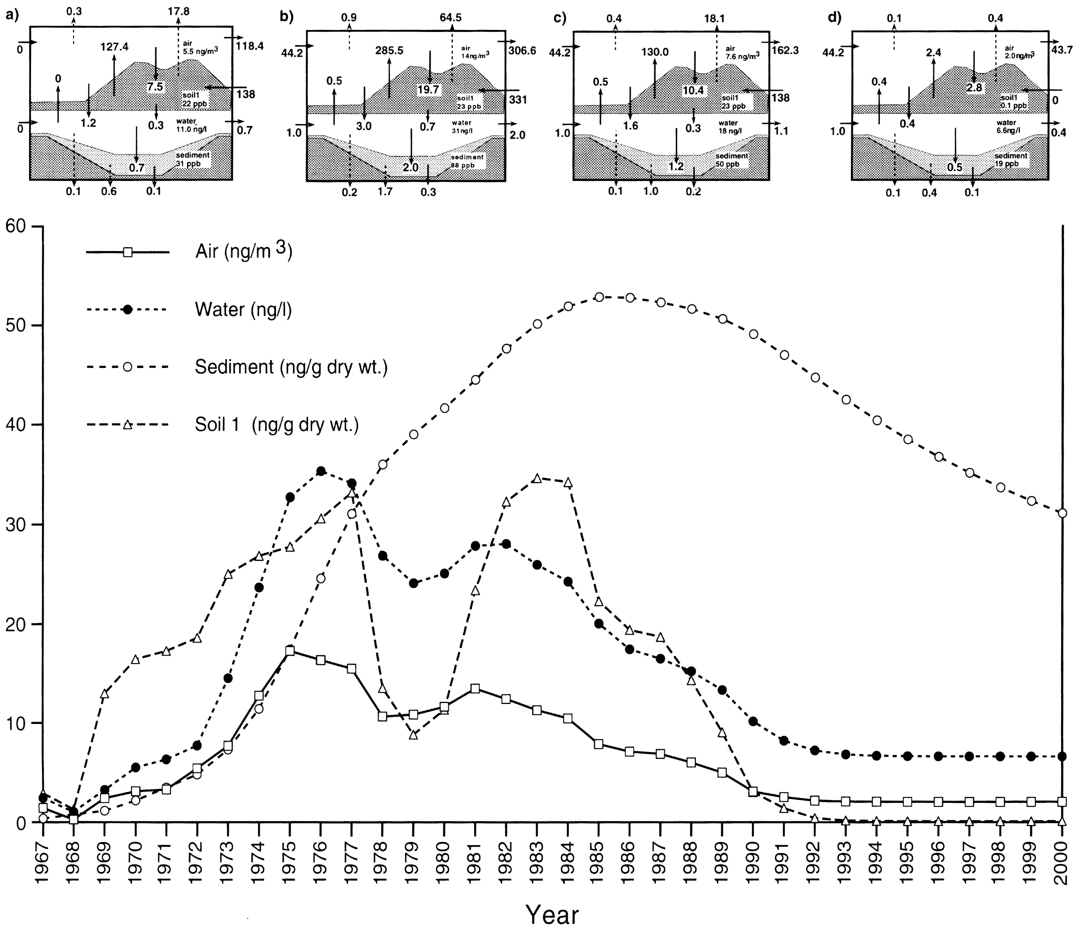

Fig. 4. Distribution and transportation of DDT (tons × yr^{-1}) between different environmental compartments in the Lake Kariba area. The fluxes are given at steady state and were calculated for four different periods as the load within the Lake Kariba system has varied over time (a–d). These are nonequilibrium, steady-state computations, or "level III" models (Mackay 1991). The time to reach steady state was about 6 to 10 years, and at the end of each period, the concentrations found at steady state were similar to those generated by the non-steady state nonequilibrium simulation (large graph). In this simulation, the load of DDT varied in accordance with Fig. 2.

is lost to the environment in the first month after application, and that all of this (i. e. 0.4 and 3 × 0.2 g a.i × km^{-2} $year^{-1}$) is deposited in the local environment, including rivers.

In aerial spraying, endosulfan is applied in five cycles from an aircraft flying at 15 m at 270 km × h^{-1} (Allsopp & Hursey 1986, Barrett 1992). The reported dose varies between 25 (Allsopp & Hursey, 1986) and 90 g a.i × $ha^{-1}yr^{-1}$ (Johnstone et al. 1990). Two simulations were performed to cover this range. The total amount was assumed to be applied as five pulse events between August and September, with 7 and 24 g a.i × ha^{-1} as the maximum loads in the first cycle (Allsopp & Hursey 1986, Johnstone et al. 1990). As the worst case, it was assumed that the total dose was applied to a river by mistake, which is likely to happen when riverbanks are sprayed.

2.3. Derivation of no-effect levels for the ecosystem

The no observable adverse effect level (NOEL) for the aquatic ecosystem was defined as the concentration of a pesticide at which (theoretically) 95% of the species in the ecosystem are unaffected. One percent of this level is considered to have negligible risk for the structure and functioning of the environment (Jager & Visser 1994). The derivation of the NOEL poses the problem that it should preferably be derived for all relevant species of the ecosystem considered (Guinée & Heijungs 1993). These data were not available, and to get a rough estimate of the no observable adverse effect level for the aquatic ecosystem, an extrapolation factor of 100 was applied to the lowest

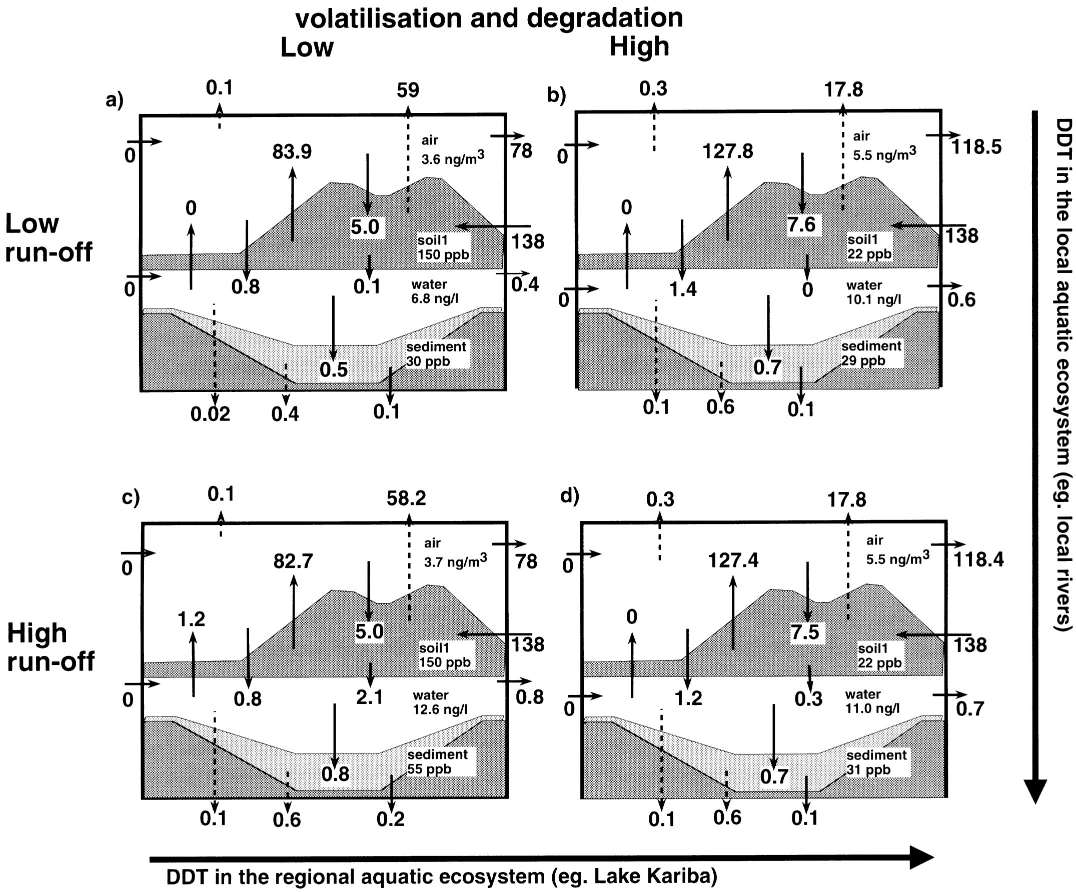

Fig. 5a–d. The influence of volatilisation/degradation and run-off on the distribution of DDT in the environment. DDT applied to soil is assumed to dissipate either through volatilisation, degradation or in surface run-off. — b, d: with a high volatilisation rate from soil, much DDT is exported to other areas and redistributed within the system through atmospheric deposition. Comparatively low DDT levels are accumulated in soil, but high levels are generated in air and water (regional scale). — a, c: at a decreased volatilisation rate, DDT in run-off increases and comparatively high levels are generated in nearby lakes and rivers (local scale).

LC_{50} value found for either fish (96 hr), daphnia (48 hr), or algae (OECD 1990) (Table 4).

3. Results

3.1. Regional distribution of DDT

The input of DDT to the Lake Kariba area has varied over time (Fig. 2) and therefore steady-state levels were calculated for four different periods (Fig. 4a–d). The time to reach a steady state in the air, water and soil compartments is 6 to 10 years, but much longer for the sediments. Presumably, either burial or degradation in the sediments are overestimated in Fig. 4a–d, and there is a continuous net accumulation in the sediments. Thus, except for the sediments, the fluxes calculated in the steady-state simulations are probably reasonable estimates of the dynamics of DDT at the end of periods 4a–d, and the calculated levels are generally in good agreement with the levels calculated in the non steady-state simulation (Fig. 4).

In all simulations, the dominant fluxes are between air and soil (Fig. 4a–d). Fig. 4a may illustrate the situation before 1973, with a moderate use of DDT in the area and no import with water and air. Of the total amount of DDT used, 86% is exported to other areas, probably Zambia as the winds often come from the south (Balon & Coche 1974). Atmospheric fluxes are also the main local process to redistribute DDT to, for example, the lake. 1.2 tons of DDT enter the lake through deposition, while only 300 kg through run-off. About 13% of the DDT leaves the system through degradation in soil. Much less (< 1%) is degraded in air, water and sediments, although this is the main process to remove DDT from the sediments. About half of the DDT entering the lake ends up in the sediments and the other half is exported. About 125 kg is accumulated in the system through burial in sediments (104 kg) and leaching to ground water (21 kg).

The period between 1981 and 1989 is similar to the previous period except that there is assumed to be an import of DDT with water and air (Fig. 4c). The import of 45.2 tons of DDT is almost counteracted by an increased export of 44.3 tons. The missing 0.9 tons are mainly degraded within the system (800 kg) but are also buried in the sediments and lost to the ground water (100 kg).

The maximum input to the area is between 1973 and 1980 (Fig. 4b). The increased input to the soil compartment is counteracted by an increased flux to the atmosphere and export to other areas. However, the percentage of DDT exported decreases from 89% (cf. Fig. 4c) to 81%. Of the 67.3 tons of DDT (18% of the total output) that are degraded, 96% is degraded in soil. The increased degradation, compared with the other periods, is probably due to the high concentration of DDT used on agriculture soil (3 kg $\times$ ha^{-1} $\times$ yr^{-1}) (Matthiessen et al. 1984). The comparatively low soil levels seen in Fig. 4b are misleading, because in Fig. 4a–d only levels in soil 1 are shown and, in this scenario, most of the DDT is assumed to be applied to soil 3. However, the increased input of DDT to the system is clearly seen in the increased levels of DDT in air and water.

If the use of DDT in Zimbabwe is discontinued, much DDT may still be imported in air and water. However, in this case, almost all (98%) of the imported DDT is soon exported (Fig. 4d). The net fluxes between air, soil and water are comparatively small because most of the DDT entering soil

Table 3. Environmental concentrations of DDT in the Lake Kariba area in 1983, found by Matthiessen et al. (1984) and calculated in the simulations. The figures calculated with the Sesoil/EXAMII models are estimates of the levels expected in the Sengwa River, while the figures from the SimpleBox model are estimated for the whole Lake Kariba area.

Environmental compartment	Matthiessen et al. 1984 Lake Kariba[a]	Sengwa[b]	Seasoil/ EXAMII[c]	SimpleBox
Air	–	–	–	11.3
Water (µg $\times$ l^{-1})	(< 0.01–0.3)	< 0.02	*0.01*– 0.03	0.02
Sediment (µg $\times$ g^{-1}dry wt)				
pre-rain	(0.04–0.74)	0.12	*0.04*– 0.12	0.05
post-rain	(0.001–0.98)	0.02		
Mutela dubia (µg $\times$ g^{-1}fresh wt.)				
pre-rain	(0.02–1.0)	0.06	*0.01*– 0.03	0.15–
post-rain	(0.003–0.66)	0.13		0.20
Tigerfish (muscle, µg $\times$ g^{-1} fresh wt.)				
pre-rain	(0.04–0.45)	0.15		
post-rain	(0.04–0.08)	0.05		
Soil (µg $\times$ g^{-1}dry wt.)				
pre-rain	(0.01–2.6)	0.1	0.07–*0.08*	0.03
post-rain	(0.01–16.5)	0.07		

[a] The range is based on samples from different areas of Lake Kariba (sampled in 1983) (Matthiessen et al. 1984).
[b] The soil samples are from the Sengwa Wildlife Research Area and the other samples are from the outlet of the Sengwa River (sampled in 1983) (Matthiessen et al. 1984) (Fig. 1).
[c] The figures in italics are based on an erosion rate of 4 tons per sq. km and the other figures on an erosion rate of 300 tons per sq. km (see Methods).

and water through the atmosphere is volatilized or degraded. However, only half of the DDT imported in water is exported. The rest is transported to the sediments (500 kg), where it is degraded (80%) or buried (20%). In this situation, the input of DDT to the water is almost the same as in Fig. 4a, but originates from elsewhere.

In the last simulation, the input of DDT is changed each year, which is reflected in the soil, air and water concentrations (Figs. 2 and 4). The sediment levels, on the other hand, increase continuously and start to decline later and at a slower rate compared with the other compartments (Fig. 4). In 1983, the expected levels in air, soil, water and sediments are similar to those found by Matthiessen et al. (1984) (Table 3). However, the levels in the biota, estimated from the steady-state computations, are slightly higher compared with Matthiessen et al.'s (1984) data (Table 3).

In Fig. 5a–d it is seen that if less DDT volatilises, dissipation through degradation increases. As a result, less DDT is exported to other areas and the atmospheric deposition on the lake decreases. However, this is to some extent counteracted by an increased input of DDT in run-off, due to increased levels in soil. Consequently, if the volatilisation rate is low, a major part of the DDT may enter lakes and rivers in run-off (Fig. 5c). However, as it takes more than 40 years to reach a steady state in Fig. 5a and c, this is probably an extreme situation and the fluxes are probably often lower.

3.2. Local distribution of pesticides

Based on both high and low erosion rates, the predicted concentrations for 1983 are similar to the levels found by Matthiessen (1985) (Table 3). DDT levels in soil are stabilised after six to seven years. Also degradation, volatilisation and surface run-off are constant by that time. This corresponds well with the estimated time to reach a steady state in the SimpleBox simulations. Approximately 90% of the DDT dissipates through volatilisation and 10% through degradation. With a low erosion rate, the amount of DDT in surface run-off entering the river is almost 1 kg per year, which is about half of the atmospheric deposition of DDT on the river (2 kg). The highest expected DDT levels in water and sediments are found in the last parts of the river, close to the lake, and both the maximum and mean exposure concentrations are 0.01 $\mu g \times l^{-1}$. With a high erosion rate, approximately 23 kg of DDT enters the river in surface run-off, and the highest DDT levels are found in the upper part of the river, with maximum and mean exposure concentration of 0.06 and 0.04 $\mu g \times l^{-1}$, respectively (Table 4).

For endosulfan the mean water concentrations fluctuate between 0.01 and 0.06 $\mu g \times l^{-1}$ with maximum concentrations of 0.4 and 1.2 $\mu g \times l^{-1}$ at a low and high dose, respectively (Table 4). The deltamethrin levels are expected to be very low with a maximum concentration of 0.02 $ng \times l^{-1}$, which is much lower than the estimated no observable adverse effect levels (NOEL) (Table 4). The maximum predicted concentration of endosulfan is of

Table 4 Predicted environmental concentration in water versus toxic concentrations.

	DDT		Deltamethrin		Endosulfan	
Acute toxicity, LC_{50} ($\mu g \times l^{-1}$)						
Algae	–		–		150	(4)
Daphnia sp. (48 hr)	0.4	(1)	0.8	(3)	170	(4)
Fish (96 hr)	2	(1)	0.5	(5)	0.4	(4)
NOEL (aquatic ecosystem)	4×10^{-3}	(2)	5×10^{-3}	(2)	4×10^{-3}	(2)
Predicted Env. Conc. ($\mu g \times l^{-1}$)						
Mean	2×10^{-2}	(6)	5×10^{-6}		1×10^{-2}–6×10^{-2}	
Max.	6×10^{-2}	(7)	2×10^{-5}		0.4 –1.2	(8)

(1) Landner & Walterson 1989, (2) see Methods, (3) Torstensson 1989, (4) Andersson & Hanze 1992, (5) Smith & Strutton 1986, (6) mean of low and high erosion, see Methods, (7) high erosion, (8) low-high dose, see Methods.

the same magnitude as the acute toxic level for fish and the mean level is above the NOEL for the ecosystem (Table 4). The expected DDT concentrations are lower than the LC_{50}-values but above the estimated NOEL (Table 4).

4. Discussion

4.1. Predicted environmental concentrations

A major part of the DDT found in Lake Kariba probably originates from local use in the area. In tsetse control operations DDT is applied selectively at tsetse resting, refuge and breeding sites, which are primarily in valleys along river courses (Mpofu 1987). The uneven nature of the amount of DDT applied was not reflected in the models, although this is probably of minor importance for the redistribution of DDT within the system. According to Matthiessen (1985), DDT did not accumulate in the soil between different sprayings, but it was not clear how it dissipated. Applications are made in the dry season and, depending on how much DDT that remains in the soil at the onset of the rains, a certain amount will be washed out into rivers feeding Lake Kariba (Mpofu 1987). It has been suggested that a major factor in the dispersal dynamics of DDT in the area is related to the short, heavy rain storms which results in high erosive run-off (Zaranyika et al. 1994). This might be the case if large quantities of DDT are applied very close to the water (cf. Matthiessen et al. 1984), but in general the high evaporation rate in the area (Balon & Coche 1974, Hutton 1991) probably decreases the erosion hazard substantially.

More likely, most of the DDT applied to soil probably dissipate through degradation and volatilisation. When DDT is degraded to DDE the vapour pressure increases about eight times (Spencer 1975), followed by an increased volatilisation rate (Samuel & Pillai 1989). In sandy soils, with low organic content, the volatilisation is further enhanced by moisture, which displaces the pesticide from the surface particles (Spencer 1975). Accordingly, although Matthiessen et al. (1984) suggested that DDT volatilises quite slowly, this is probably the main dissipation route for DDT under tropical conditions (Spencer 1975, Yeadon & Perfect 1981, Nash 1983a, Sleicher & Hopcraft 1984, Samuel & Pillai 1989).

Sleicher & Hopcraft (1984) estimated that the rate of loss by sublimation of DDT from soils in Kenya was at least 10 times higher than the degradation rate. A similar ratio between volatilisation and degradation was used in the simulations, resulting in a volatilisation rate from soil 1 of approximately 0.5 $g \times ha^{-1} \times day^{-1}$ (Fig. 4a). This is about 30 times slower than measured at 30°C after 9 to 14 days (Spencer 1975, Nash 1983b). However, 0.5 $g \times ha^{-1} \times day^{-1}$ was estimated as an average volatilisation rate over the year in the area, and the initial volatilisation could have been much higher (cf. Edwards 1966, Tingle 1995). The predicted concentrations in soil were within the range reported for background levels by Matthiessen et al. (1984) and Tingle and Grant (1995) (Table 3), but lower than those levels recorded at specific "hot spots" in the area (Matthiessen et al. 1984, Tingle 1995) (Table 3). The soil levels were also similar to levels in soils from India (Nair & Pillai 1992), but low compared with levels from many temperate agriculture soils (Matthiessen et al. 1984).

Due to the high volatilisation rate, most of the DDT applied to soil is expected to be lost to the atmosphere and exported to other areas (Fig. 4a–d) (cf. Loganathan & Kannan 1994). Ngabe and Bidleman (1992) suggested that Africa may be an important source of DDT to the global atmosphere, and long range atmospheric export of DDT from the African continent was indicated by the comparatively high levels (70 pg p,p-DDT $\times m^{-3}$) found outside the Somalia coast in 1976 (Bidleman & Leonard 1982). Recently it was also suggested that DDT, originating from tropical areas, ultimately will be deposited in colder regions of the world (Bourdeau et al. 1989, Wania & Mackay 1993, Loganathan & Kannnan 1994). This phenomenon, referred to as global distillation (Goldberg 1991, Wania & Mackay 1993), implies that the continued use of DDT in tropical countries is a global problem, which may remain unsolved unless a total ban is enforced on its use throughout the world (Wania & Mackay 1993, Loganathan & Kannan 1994).

Furthermore, if the use of DDT in Zimbabwe is discontinued, airborne DDT may be a major input to Lake Kariba in the future (Fig. 4d). Atmospheric transportation and deposition have been stated as the main sources of DDT in North America (Eisenreich et al. 1981), originating from use in Central America and Mexico (Rapaport et al. 1985, Rapaport & Eisenreich 1988). With a mean DDT concentration in air of 0.03 ng × m^{-3}, Eisenreich et al. (1981) estimated that the atmospheric deposition of DDT on Lake Ontario was approximately 7. 2 g × km^{-2} × yr^{-1}, which is much lower than expected in Lake Kariba. However, the air levels were estimated to be much higher in Kariba (2 ng × m^{-3}), and the deposition of DDT is expected to increase with increasing DDT levels in air (Miller & Robinson 1989).

The predicted maximum air levels in 1975 (17 ng × m^{-3}) were similar to levels found in India between 1980 and 1982 (29 ng p,p-DDT × m^{-3}) (Kaushik et al. 1987), and to levels reported from the USA in the early seventies (Spencer 1975). In 1989, Ngabe and Bidleman (1989) found 1.2 ng p,p-DDT × m^{-3} in Brazzaville, in Congo, and similar levels were also found in India and Southeast Asia between 1989 and 1990 (Iwata et al. 1994). This indicates that the assumed concentration of 2 ng × m^{-3} is a reasonable background level of DDT in air, although it must have fluctuated over the years.

Besides deposition of airborne DDT, much DDT probably enters the lake through the Zambezi River (Matthiessen et al. 1984). Even with low concentrations of DDT in the water, this input may be of major importance for the future levels of DDT in the aquatic environment. As almost all waterborne DDT entering the lake is associated to particles (Matthiessen et al. 1984), which settle to the sediments, a large amount of the DDT could be expected to be retained within the lake. However, the short water replacement time in Lake Kariba probably has a substantial influence on the residence time of DDT in the lake, and about 40% of the DDT entering the lake each year is expected to be exported with the water. In Lake Michigan, which has a detention time of 85 years, hydraulic washout was estimated to be responsible for 2–4% of the total loss, while degradation and sedimentation were responsible for 3–35% and 61–95%, respectively (Bierman & Swain 1982).

Volatilisation could be another important mechanism for the removal of DDT from Lake Kariba (cf. Mackay & Paterson 1986). In Lake Ontario, it was estimated that the atmospheric deposition of PCB was almost counterbalanced by net volatilisation losses from the water column (Mackay & Diamond 1989). This is also expected to be the case in Lake Kariba when all the DDT entering the lake originates from sources outside the area (Fig. 4d). However, the volatilisation could be expected to decrease when the ratio between the concentration in water to air decreases (cf. Mackay 1991, Preston 1992), which seems to be the case when the main input of DDT is from local sources close to the lake. Accordingly, increased levels in air may not only increase the amount of DDT being deposited on the lake but also decrease the amount of DDT volatilizing from the lake.

The estimated water concentrations are similar to those found by Matthiessen et al. (1984) (Table 3), and of the same magnitude as reported from other studies in southern Africa (Greichus et al.1977, 1978a, Mhlanga & Madziva 1990). However, the expected levels between 1974 and 1984 (Fig. 4) are high compared with levels found in Asia in the late eighties (Nair & Pillai 1992, Iwata et al. 1994), and in the upper range of levels reported for freshwaters in USA during the sixties (Bevenue 1976).

Almost half of the DDT entering Lake Kariba each year is lost to the sediments, where more than 80% are degraded (Fig. 4a–d). The remaining 20% are buried in the sediments. Mackay and Diamond (1989) estimated that 82% of the PCB entering Lake Ontario were lost to the sediments and that almost 90% of this were buried. However, Oliver et al. (1989) stated that due to the very low sedimentation rate in Lake Ontario, accumulation and burial of contaminants are slow. Because of the slow sedimentation rate (0.3–4 mm/year), which is similar to the estimated rate in Lake Kariba (0.8 mm/year), the sediments can act as an important source for contaminants to the lake for many years. The estimated accumulation of DDT in Lake Kariba in the early seventies and nineties is similar to the accumulation in Lake Ontario in 1950 (13 ng × cm^{-2}) and 1970 (10 ng × cm^{-2}), but is approximately twice as high when the use was at its maximum in Zimbabwe, compared with when it peaked in North America (24 ng × cm^{-2}) (Eisenreich et al. 1989) (Oliver et al. 1989) (Fig. 4a–d) and Europe (19 ng × cm^{-2}) (Sanders et al. 1992).

The predicted levels of DDT in the sediment were within the same range as those reported by Matthiessen (1985) (Table 3), but slightly higher

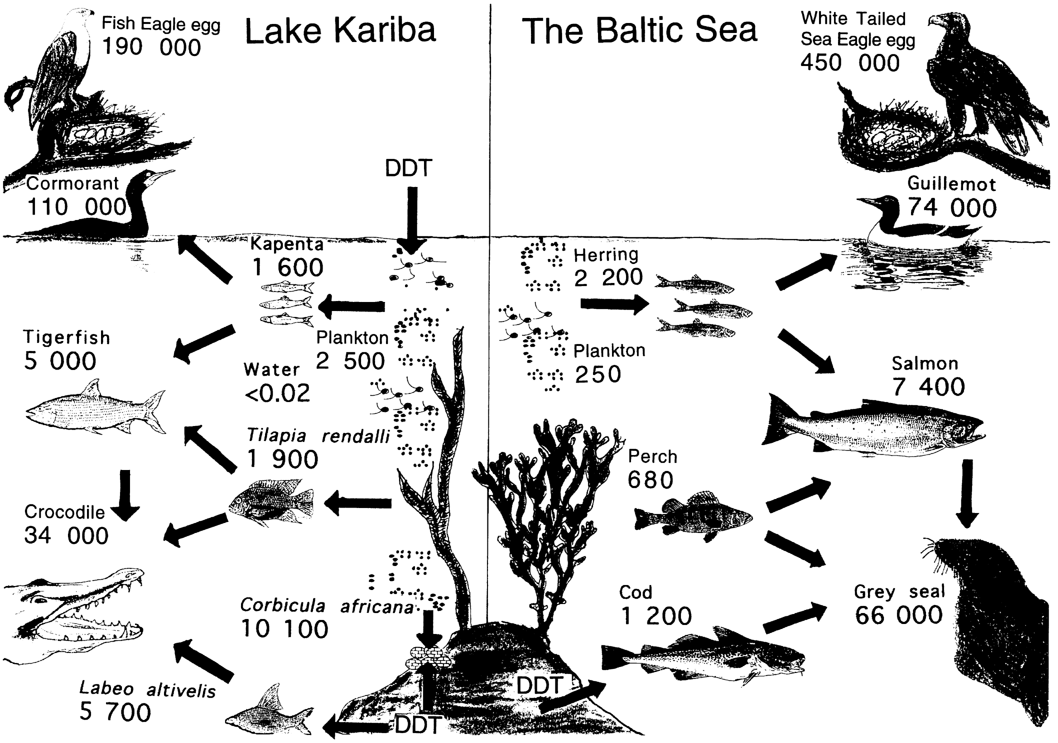

Fig. 6. Mean levels of sDDT (ppb in fat) in Lake Kariba (1987–1990), when DDT was still in use, compared with levels in the Baltic Sea (1979–1981). DDT was banned in many of the Baltic countries in the course of the 1970s. In Sweden it was banned at the end of 1969, but the forest industry was allowed to use small quantities for the next five years. The former states of the Soviet Union banned it in 1974, Poland in 1976 and Finland in 1977. West Germany abolished it in 1971, and East Germany in 1977. The use of DDT in Denmark was negligible at the end of the 1970s (Melvasalo 1981).

than levels from Lake McIlwaine in Zimbabwe, Lake Nakuru in Kenya and the dams Hartbeespoort and Voelvei in South Africa (0.057, < 0.001, 0.05 and 0.013 μg sDDT × g^{-1} dry wt., respectively) (Greichus et al. 1977, 1978a and b). However, these lakes were sampled between 1974 and 1975 when the use of DDT in South Africa had been severely restricted for 4 years (Van Dyk et al. 1982). During the early seventies, the use of DDT in Zimbabwe and Kenya had probably also been lower than in the beginning of the eighties (cf. Fig. 2). In 1987, levels in sediments from Lake McIlwain had increased to 0.076 μg sDDT × g^{-1} dry wt. (Mhlanga & Madziva 1990). This, together with the results from the simulations, suggests that the sediment levels may increase continuously (Fig. 4), although the use of DDT in Zimbabwe has decreased over the last years (Fig. 2). Thus, a decreased use of DDT may be reflected much later in the sediment concentrations as compared with the levels of DDT in soil, water and air.

This is also indicated by the much slower decline of DDT in sediments, compared with fish, in both the Baltic Sea (Bignert et al. 1989, Nylund et al. 1992) and the Great Lakes (Bierman & Swain 1982). In combination with low burial rates, this implies that bottom living organisms will be exposed to higher levels of DDT than organisms living in pelagic waters, long after the input of DDT has decreased. However, the sediment levels in Lake Kariba will probably decrease faster than in the Baltic Sea (cf. Nylund et al. 1992) and the Great Lakes because of the shorter water retention time in Lake Kariba and the comparatively high degradation rate expected in the tropics.

4.2. DDT in the biota

So far, it can be concluded that the predicted levels in the abiotic environment are in accordance with concentrations found by Matthiessen et al. (1984) and comparable with levels found in areas with similar environmental conditions. This is important as the environmental concentrations determine the levels in the biota and the potential toxic hazard to organisms (cf. Klein 1989). However, in the steady-state simulations the concentrations in the biota were often overestimated, and it is possible that the different characteristics of tropical ecosystems may affect the accumulation of persistent pollutants in the foodweb.

This is, to some extent, supported when comparing levels in organisms from Lake Kariba with levels found in organisms from the Baltic Sea (Fig. 6). The highest levels are found in fish eagle (*Haliaeetes vocifer)* (Douthwaite 1992d), but also other top predators like crocodiles (*Crocodilus niloticus)* (Phelps et al. 1989), cormorants (*Phalacrocorax africanus)* (Douthwaite et al. 1992) and tigerfish (*Hydrocynus vittatus)* (Berg et al. 1992) have high levels compared with their prey. These levels are of the same magnitude or slightly lower compared with white-tailed sea eagle (*Haliaeetus albicilla)*, seals (*Haliochoerus grypus)* and salmon (*Salmo salar)* from the Swedish east coast in the Baltic Sea (Andersson et al. 1988, Odsjö & Olsson 1988, Reutergård 1988, Bignert et al. 1989). At lower trophic levels, the values are only slightly higher in Lake Kariba (Berg et al. 1992), despite the fact that organisms sampled in Lake Kariba had recently been exposed to DDT, while the corresponding samples from the Baltic Sea were measured some five to ten years after DDT was banned in the area (Odsjö & Olsson 1988, Reutergård 1988). When DDT was banned in Sweden, the levels in eggs from white-tailed sea eagle and breast muscle from guillmot (*Uria aalge)* was 900 and 400 μg sDDT $\times$ g^{-1} lipid wt., respectively (Reutergårdh 1988). In herring (*Clupea harengus*) the levels were 18 μg sDDT$\times$ g^{-1}lipid wt., which is about 8 times higher than 1981 (Olsson & Reutergårdh 1986).

The comparatively low levels in Lake Kariba may be related to the characteristics of the biotic compartment of the lake. It has been suggested that the equilibrium concentration of DDT at a trophic level is (Harrison et al. 1970); "(i) directly proportional to the total life span of its members, (ii) inversely proportional to the total biomass of the level, and (iii) proportional to the net retention of DDT at that level. The net retention at a level depends on the concentration of DDT in the lower levels, the rate at which organisms of the lower level are ingested, and the amount of DDT excreted".

In Lake Kariba, the pelagic food chain consist of phytoplankton, zooplankton, *Limnothrissa miodon*, a small-sized clupeid sardine called "kapenta" and, finally, the carnivorous tigerfish (Ramberg et al. 1987) (Fig. 6). The mortality rate of the kapenta is high and most die within one year (Marshall 1987, Hutton 1991). Little is known about the population dynamics of the zooplankton, but since the introduction of the kapenta they have decreased in size and can withstand high levels of predation, which indicates rapid growth and high turnover rates (Marshall 1991). Also the phytoplankton could be expected to have high turnover rates because of the high temperature and radiation in the region. Although Lake Kariba in general is regarded as oligotrophic for being a tropical lake (Marshall 1984), it is probably more productive than many temperate lakes and sustains a higher biomass over the year (Machena 1989).

Thus, according to the statements above, the levels of DDT could be expected to be low in the biota of Lake Kariba because; i) many species seem to have comparatively short life spans and high turnover rates; ii) the biomass is probably high compared with temperate lakes, and iii) only a minor part of, at least the primary, production seems to be consumed, although the consumption efficiency probably increases at higher trophic levels (cf. Machena et al. 1993). Furthermore, the high temperature suggests that organisms from tropical areas have a higher metabolism and may therefore degrade and excrete DDT faster than organisms from temperate areas. Another reason for the low levels in Lake Kariba could be the increased influence of UV radiation and temperature on the chemical and biological degradation of DDT, which may have been underestimated in the simulations.

The increased mobility of DDT expected under tropical conditions may also explain the comparatively low levels found in top predators from Lake Kariba. Harrison et al. (1970) suggested that each trophic level requires about four average life spans to reach an equilibrium in response to changes in DDT concentrations in the level below. This implies that although DDT had been used for more than 20 years around Lake Kariba, most of the top predators in the area may not yet have reached their maximum body burden of DDT. In the Baltic Sea, on the other hand, DDT had been used over a longer period of time, and the dissipation rate from the environment had probably also been slower than in Lake Kariba. Thus, top predators from the Baltic Sea may have had a longer time to reach an equilibrium with their environment, and have therefore accumulated higher levels than top predators from Lake Kariba. Furthermore, there have probably been more uncontaminated "refuge areas" in the vicinity of Lake Kariba than around the Baltic Sea, which in combination with the decreased use of DDT between 1977 and 1980 may have lowered the levels in the biota of Lake Kariba substantially.

In short-lived species, such as phytoplankton, zooplankton and even the kapenta, levels could be expected to be proportional to the levels in water, and show rapid responses to changes in the water column (Woodwell et al. 1971, Bierman & Swain 1982, Loganathan & Kannan 1991). In kapenta, sampled in 1990, the levels were approximately 70% of those recorded in 1985 (Mhlanga et al. 1986, Berg et al. 1992), which is in accordance with the predicted decrease of DDT in the water (Fig. 4).

In tigerfish, concentrations in muscle decreased between 1982 and 1990 from about 15 to 5 μg sDDT g^{-1} fat (Matthiessen et al. 1984, Berg et al. 1992), which is faster than in the kapenta. However, while equilibrium partitioning is the primary factor governing organochlorine levels in small fish, intake via food is of major importance in higher trophic predatory fish (Loganathan & Kannan 1994). Thus, during periods with high concentrations of DDT in the environment, the tigerfish probably accumulates comparatively high DDT levels through the food (cf. Connolly & Pedersen 1988, Clark et al. 1990). However, when levels in the environment decreases this input will probably be of less importance, and the levels in both kapenta and tigerfish will mainly be determined by the equilibrium between water and fish.

The biota in the benthic dietary pathway should, as pointed out earlier, be expected to show a much slower response than the pelagic biota because of the slower changes in sediment concentrations (Bierman & Swain 1982). In Lake Kariba, bottom living fish were found to contain higher levels of DDT than pelagic fish, although they were living in the same area and feeding at the same trophic level (Berg et al. 1992). These fish were sampled in 1990 when the sediment levels, according to the model, should be at its maximum, while the water levels had decreased significantly (Fig. 4).

In fish eagle, there were no significant decreases in levels between 1980 and 1990 and it is possible that these levels will continue to be high long after a discontinued use of DDT (Douthwaite et al. 1992). Thus, similar to temperate areas, top predators of especially aquatic food chains must be considered to be at high risk from persistent organic pollutants also in tropical regions (Loganathan & Kannan 1994).

In conclusion, there could be several reasons for the comparatively low levels of DDT in the biota of Lake Kariba. DDT entering the lake may be "diluted" in a comparatively large biomass, where only a minor amount of the DDT is accumulated in the food web. The high temperature probably increase the chemical and biological degradation of DDT, and due to the high volatilisation rate, much DDT is expected to be exported to other areas. Furthermore, due to the short retention time of Lake Kariba, much DDT is probably soon lost through the turbines, which may affect the levels in the aquatic biota substantially. Therefore, with the discontinued use of DDT in the area decreased levels in the biota would probably soon follow, although high levels may remain in top predators and organisms associated with the sediments.

4.3. Ecotoxicological implications

There are reasons to believe that the mobility of DDT in a tropical environment are different compared with temperate areas (Bourdeau et al. 1989, Murti 1989), because of the increased volatilisation rate under tropical conditions (Sleicher & Hopcraft 1984, cf. Mackay et al. 1985) (Fig. 5a–d). As the mobility determines the distribution of pesticides between different environmental compartments and governs their bioavailability to organisms, the mobility is a direct link between the use and input of pesticides and their ecotoxicological effects (Klein

1989). The increased volatilisation of DDT in the tropics is probably counterbalanced by increased export and deposition on land and water (Fig. 5b and d). Thus, on a regional scale, large quantities of DDT may rapidly be transferred into watersheds near nonpoint sources in these regions (Iwata et al. 1993). In colder areas, with lower volatilisation rates, the fluxes to the atmosphere are expected to be slower and more extended in time (cf. Wania & Mackay 1993), which would result in lower concentrations in air, slower deposition rates and decreased export of DDT to other regions (Fig. 5a and c).

However, with a slow volatilisation rate, much DDT is accumulated in soil and large quantities may be washed out into nearby watersheds. Although run-off in general seems to be a process of minor importance for the redistribution of DDT on a regional scale, it may be an important way that acute toxic levels are generated in the local environment (cf. Preston 1992). In temperate regions, this could be a comparatively large problem, due to the slow volatilisation rates expected in these areas (Fig. 5a–d). Using a very high soil-loss rate, Mackay et al. (1985) suggested that about 30% of DDT applied to soil was lost in run-off to water, while only 1.5% escaped to air. The rest was degraded. However, under normal conditions most DDT probably dissipates through volatilisation also in temperate areas, but not to the same extent as in the tropics (Edwards 1966, Spencer 1975) (Fig. 5a–d).

To make a first order estimate of potential acute toxic exposure levels of DDT to the aquatic fauna, the dynamics of DDT were simulated in the local environment. In general, the predicted DDT levels in the Sengwa River were in accordance with Matthiessen's results (1985), but a very high value (0.3 $\mu g \times l^{-1}$) in water indicates that much higher levels may be generated in run-off during, for example, heavy rainstorms (Matthiessen et al. 1984, Zaranyika et al. 1994). Unfortunately, such situations seem to be difficult to predict from models that more or less are built from a generalised environment. However, the predicted concentrations of DDT can still be used as a relative comparison of the estimated levels of deltamethrin and endosulfan in the aquatic environment (Table 4).

The very low amount deltamethrin applied to targets and the low tendency of deltamethrin to accumulate in the aquatic environment (Muir et al. 1985, Maguire et al. 1989, Caquet 1992) probably makes the use of targets the most environmentally sound method of controlling tsetse flies. The expected maximum concentration of deltamethrin was well below the estimated no adverse effect levels (NOEL) on the aquatic ecosystem (Table 4). However, if higher levels are used in, for example, ground- or aerial-spraying (cf. Barrett 1991, Lambert et al. 1991) it is likely that this would cause more severe acute toxic effects on the aquatic fauna than ground spraying with DDT (Lambert et al. 1991).

Also, aerial spraying with endosulfan may cause acute toxic effects on the aquatic fauna, especially with high doses in small water bodies (Cockbill 1979, Fox & Matthiessen 1982). Six to nine hours after aerial application of 9.6 g endosulfan $\times$ ha^{-1} Fox and Matthiessen (1982) found values of 0.2 and 4.2 $\mu g \times l^{-1}$, which were within the range of 24 hours LC_{50} values reported for fish in the area. These concentrations are similar to the estimated maximum levels, which were much higher than the estimated NOEL on the aquatic ecosystem (Table 4). However, the hazard to the aquatic fauna can probably be minimised by applying low levels under controlled conditions (Russel-Smith & Ruckert 1981, Johnstone et al. 1990).

For ground spraying with DDT, both the predicted mean and maximum levels of DDT in water were higher than the estimated no adverse effect levels (NOEL) on the aquatic ecosystem (Fig. 4 and Table 4). This is probably due to the comparatively high persistence of DDT, and high levels of DDT may prevail in the aquatic system during comparatively long periods. Accordingly, there could be reasons to believe that the use of DDT may have caused environmental effects in the Lake Kariba ecosystem, especially between 1974 and 1984 when levels in the environment were expected to be at their maximum (Fig. 4). In the Baltic Sea and the Great Lakes, the elevated concentrations of DDT during the late sixties were high enough to cause harmful effects on individuals and populations of top predators (Olsson & Reutergård 1986, Liroff 1989). Eggshell-thinning in birds, leading to reduced hatching, is a well-known effect from temperate areas (Ratcliff 1967, Anderson & Hickey 1974), and such eggshell-thinning was recently reported from fish eagle in Lake Kariba (Douthwaite 1992d) and could be of considerable risk to the cormorants (Douthwaite et al. 1992). High levels of DDT in tigerfish eggs indicate that the reproduction of fish could have been at risk, especially for those species migrating upstream to spawn (Matthiessen et al. 1984). It has also been suggested that DDT-spraying for tsetse fly control has reduced populations of songbirds over wide areas (Douthwaite 1992a–c). Furthermore, there seem to be lower amounts of insects and insect larvae in Lake Kariba, than in the earlier stages of the lake's history (cf. Machena & Kautsky 1988), which might be due to increased uses of insecticides. The decline in the fisheries in 1974 corresponds (Karenge 1992) with the increased levels of DDT in water by that time and the biomass of zooplankton has decreased significantly, followed by an increase in smaller species (Marshall 1991), which may indicate a stressed ecosystem (cf. Havens & Hanazato 1993). It is very hard to say if this stress is due to natural changes in the lake system or if it is imposed by human activities. However, as there are indications of ecological disturbance in the environment, it is necessary to continue developing new methods to control tsetse flies in order to decrease the use of persistent pesticides in the drainage area and to sustain natural resources, like the fisheries, clean water and wildlife in Lake Kariba.

5. Conclusions

In the 1970s, restrictions were imposed on the use of DDT in most developed nations. At the same time, developing countries in the tropics continued or even increased their use of DDT, both for agriculture and public health problems (Goldberg 1991, Loganathan & Kannan 1994). The high efficiency and low cost of DDT (Mellanby 1992) made the introduction of second-generation pesticides, such as synthetic pyrethroids, and the development of alternative control methods less attractive. Furthermore, it has been suggested that the negative side effects of DDT may be smaller in tropical environments because of decreased persistence and increased mobility of DDT under tropical conditions (cf. Hamsagar 1981). Thus, there may have been both practical and economical arguments for the continued use of DDT in tropical areas, although it had been banned in most developed countries.

According to the results from this study, the dynamics of DDT in the tropics is most likely different compared with temperate areas (cf. Murti 1989), but how this will affect the negative side effects on non-target organisms probably varies depending on which spatial and temporal scales that are considered.

On the regional scale, the increased volatilisation rate expected in Lake Kariba means that large quantities of the DDT applied in the catchment area are rapidly transferred to the lake during the earlier process of atmospheric transport (cf. Iwata et al. 1993). This is followed by increased levels of DDT in air and water, these levels being higher than those expected in temperate areas. However, the high input of DDT to the lake is counterbalanced by an increased export and comparatively little DDT is accumulated in the aquatic environment (cf. Bourdeau et al. 1989, Wania & Mackay 1993). Thus, organisms living in Lake Kariba could be expected to be exposed to high levels during comparatively short periods, while organisms in temperate lakes would be exposed to lower levels but over a longer period of time. This implies that, on a regional scale, accumulation along the food chain is the main hazard in temperate areas, while the main problems with DDT in Lake Kariba, in addition to elevated levels in top predators, may be the acute toxic effects on vulnerable species.

On the local scale, short periods with increased soil run-off could generate high toxic levels in rivers running through sprayed areas (Zaranyika et al. 1994). In temperate regions, this may be a comparatively large problem because much DDT is accumulated in the soil (Fig. 5a and c) and, especially during snowmelt, these storedup pollutants may suddenly be released (Wania & Mackay 1993). However, such "shock waves" of DDT in run-off could also be expected in the tropics because of the very intensive rainstorms that occur in these areas.

Thus, direct negative effects from DDT on the aquatic environment in Lake Kariba are probably at least as severe as those expected in a similar lake from temperate areas, both on a local and regional scale. Potential environmental hazards to the terrestrial ecosystem, on the other hand, could be expected to be lower in the tropics because of the high dissipation rate from the abiotic environment. However, it seems that much of the DDT is retained in the terrestrial food chain (Douthwaite 1992b, Lambert 1993) and that high levels are accumulated in insectivorous birds (Douthwaite 1992a and b). This has caused the decline of bird populations over large areas, which is a very well-known side effect from the use of DDT in temperate areas (Carson 1962).

Considering the temporal scale, the higher mobility of DDT in the tropics suggests that DDT might be a less severe problem in these areas compared with temperate areas. Less DDT is expected to be accumulated in the ecosystem and levels should decrease comparatively fast when the use of DDT is discontinued. In Lake Kariba, this process should be further enhanced by the riverine characteristics of the lake (Loganathan & Kannan 1991). However, if the levels are to decrease in the future it is necessary that the use of DDT not only decreases in Zimbabwe, but over the whole continent.

In conclusion, there are no environmental reasons why the use of DDT should continue in Zimbabwe when it was banned for of environmental reasons in temperate areas. Especially not in the Lake Kariba area, where the lake is the main resource and is heavily utilised both by man and wildlife. Comparatively large quantities of DDT, used in the drainage area, are expected to be transported to the lake, which may have deleterious side effects on the ecosystem of Lake Kariba. There are also no longer any practical or economical arguments for the continued use of DDT. New methods using second-generation pesticides have been developed, which have proved to be as efficient as DDT in controlling tsetse flies (Barrett 1992). The environmental drawbacks of these methods are probably less than those for DDT, and potential hazards to non-target organisms in Lake Kariba can probably be substantially reduced in the future. Methods based on more fundamental knowledge of insect ecology and biochemistry must be further developed and combined with new "third-generation pesticides", whose mode of action is better known. The extra costs associated with the development of such new methods, are probably minor compared with the potential costs of a degraded environment. As it has been suggested that tropical areas may act as major sources for organochlorine compounds in the global atmosphere (Wania & Mackay 1993), it should be of the direct interest of developed countries to provide financial and technical assistance.

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