

## CONTAMINATION OF WATER AND SEDIMENTS BY OBSOLETE PESTICIDES AT VIKUGE FARM, KIBAHA DISTRICT, TANZANIA

MJ Mihale and MA Kishimba

Chemistry Department, University of Dar es Salaam  
P.O Box 35061, Dar es Salaam, Tanzania.

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### ABSTRACT

*Sediments and water from Vikuge State Farm, Coast Region, Tanzania, where, in 1986, a "donation" of 170 m<sup>3</sup> of partially expired pesticides were stored in an open shed- which eventually collapsed, were analysed for 80 different pesticide residues and metabolites. DDT and HCH, two of the most persistent organochlorines pesticides, were the most dominant pesticides found in both matrices. In sediments the levels were up to 99,620 mg/kg dry weight of  $\Sigma$ DDT and up to 7,400 mg/kg dry weight of  $\Sigma$ HCH. The mean total HCH concentrations were 0.95  $\mu$ g/l in tap water, 0.28 mg/l in surface water and 3.09  $\mu$ g/l in ground water. Of the four common HCH isomers,  $\alpha$ -HCH and  $\beta$ -HCH had the highest on average concentrations in the water. Mean total DDT concentrations were 1.76  $\mu$ g/l in tap water, 0.18  $\mu$ g/l in surface water and 9  $\mu$ g/l in ground water. The only other pesticides detected were in sediments and water are azinphos-methyl, an organophosphorous insecticide and thiabendazol, a systemic fungicide, respectively. During the rainy season, the well-water, which is used for domestic purposes by the villagers, has levels of pesticides higher than those allowed by WHO for drinking water. It is thus recommended that immediate decontamination measures be undertaken. In the meantime, villagers should sediment the well-water and should not use it at all during the rainy seasons.*

### INTRODUCTION

Developing countries (including Tanzania) cannot afford to purchase environment friendly pesticides that are more expensive because of their low purchasing power, a factor which forces these countries to opt for cheaper pesticides many of which have pronounced adverse long-term environmental and public health effects. They are also tempted to receive aid, in form of pesticides, for use in agriculture (the mainstay of their economies) and in public health programmes to combat vectors of diseases such as malaria, sleeping sickness, etc.

A "donation" of 170 m<sup>3</sup> of partly expired pesticides from Greece in 1986 was received by the Tanzania Government through the Ministry of Agriculture without any proper prior arrangements (Kishimba and Kalemera

1994). They were thus "temporarily" stored in an open shed with an earthen floor, which eventually collapsed with obvious dire environmental and public health consequences. Analyses of surface soil samples from the old storage at the farm revealed that the total pesticide content was up to almost 40% by dry mass (Kishimba and Mihale 2002). This paper reports on the analyses of surface, ground and drinking water, together with sediments, collected from and around the old storage site and the village, which is situated on lower ground, with respect to the former.

### MATERIALS AND METHODS

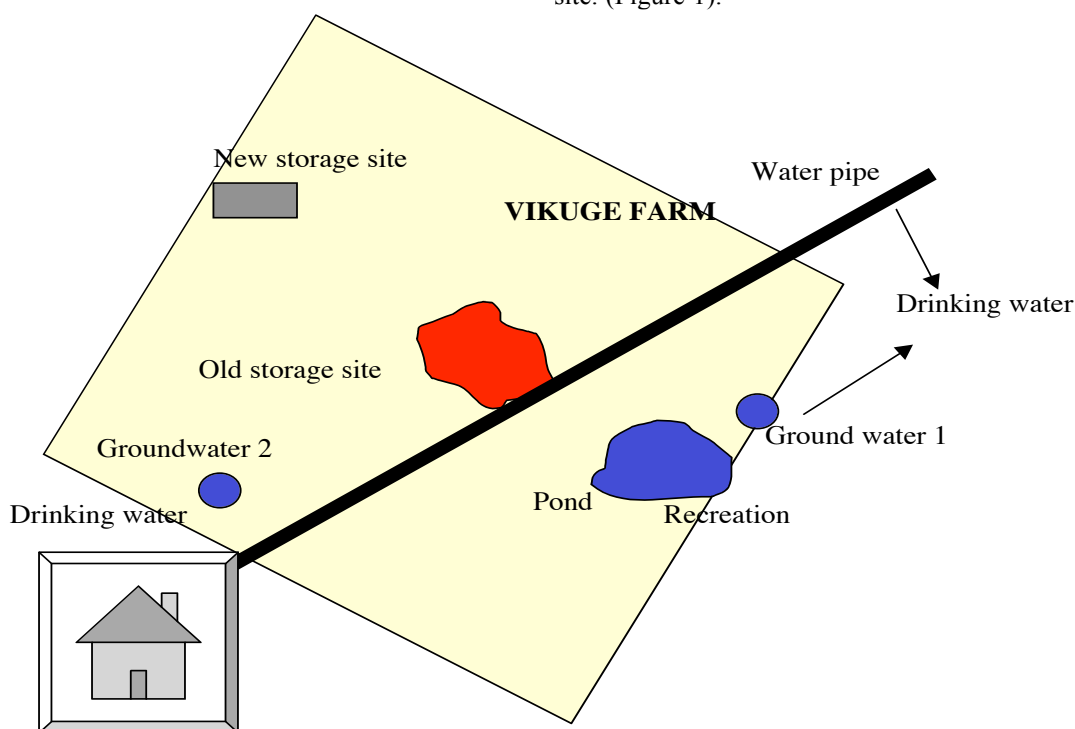
#### Materials

All the reagents and solvents used were of analytical grade and included eighty (80) different pesticide standards ordered from Dr. Ehrenstorfer GmbH (Ausburg,

Germany). Working standard solutions were made by dilution of the stock standards and mixtures of standards of different concentrations were used in most cases for the screening of the pesticide residues and metabolites. All volumetric glassware used was teflon stoppered. Varian Star 3400 and Hewlett Packard 5890A gas chromatographs equipped with  $^{63}\text{Ni}$  Electron Capture, (EC) and Nitrogen-Phosphorous, (NP) detectors were used for analysis.

#### Sampling

Sediment samples were collected along the drainage ditch in May 2001 using spatula. Samples were taken from different points along the ditch uphill and downhill sides, and only up to 5 cm depth. All samples were wrapped in aluminium foil and put into plastic bags. Water samples were collected in August 2000, March and May 2001 in one litre teflon stoppered glass sampling bottles. Sodium chloride (100 g) was added as a preservative. Tap-water samples were taken from a tap downhill after the pipe has passed through the old storage site. (Figure 1).



**Figure 1:** Overview of Water Sampling Points

Groundwater samples were taken from two different wells from which the villagers draw their water for domestic purposes. Surface water samples were taken from a pond in the nearby surroundings. Water samples were

transported to the laboratory and kept in a refrigerator at  $< 4$  oC until extraction.

#### Extraction

Sediment samples were extracted by using the Solid Dispersion Extraction method (Akerblom 1995). The sediment sample was

mixed thoroughly and two sub-samples (10g each) were taken simultaneously. One sub-sample was put in a petri dish and dried at 105°C to constant weight for determination of the water content. The other sub-sample is for analysis. The sub-sample for analysis was grounded with sodium sulphate (30g) until a free-flowing powder was formed. The powder was then extracted by shaking successively with Acetone/Cyclohexane 1:1 v/v (50, 3 x 20 ml). The extract was then filtered through a glasswool into a separating funnel and then shaken with sodium chloride solution (200ml) and the phases were separated. The organic phase was collected in an E-flask through a funnel containing oven dried anhydrous sodium sulphate (15g), while the aqueous phase was further extracted with ethyl acetate/cyclohexane (15:85 v/v) (50ml). The extracts were drained through the same funnel containing anhydrous sodium sulphate for drying. The funnel was then rinsed ethylacetate/ cyclohexane (15:85 v/v) ( 3 x 10 ml). Finally the combined dried extract was concentrated *in vacuo* at 35 °C to less than 2 ml.

Water samples were extracted using the Liquid-Liquid Extraction (LLE) method with ethyl acetate (Åkerblom 1995). The unfiltered sample (1000 ml) was quantitatively transferred into a separating funnel, and the sampling bottle was rinsed with ethyl acetate (120 ml) which was then mixed with a sample in a separating funnel. The combined contents were then shaken vigorously for about 2 minutes and later allowed to stand to let the phases separate. The organic phase was drawn into an E-flask that contained oven dried anhydrous sodium sulphate (18 g) for drying. The aqueous phase was repeatedly extracted as above, with ethyl acetate (2 x 60 ml) and the extracts were combined in one flask. The

extract was treated in the same way as the sediment extracts.

#### **Clean up**

Dilutions of up to 100,000 times was used as a clean up procedure for sediment extracts. Because of low levels of the pesticide residues and metabolites in water, the extracts which needed dilution were diluted only up to 100 times. Acid/alkali treatment techniques were also used in the clean up of sediment and water samples as described by Åkerblom (1995).

#### **Analysis and quantification**

Analysis for the residues and metabolites was done as described by Åkerblom (1995). Varian Star 3400 and Hewlett Packard 5890A gas chromatographs equipped with <sup>63</sup>Ni Electron Capture (EC) and Nitrogen-Phosphorous (NP) detectors were used for the analysis. SE-30 and OV-1701 megabore columns (30 m x 0.32 mm x 0.5 µm) were used in each detector. Nitrogen was used as both a carrier and make up gas in the ECD at a flow rate of 30 ± 1 ml/min. In the NPD, helium was used as a carrier gas at a flow rate of 0.5 - 1 ml/min and nitrogen, at a flow rate of 29 ± 1 ml/min, was used as the make up gas. The temperature programme was 90 °C held for 1 min, 30 °C/min to 180 °C, 4 °C/min to 260 °C held for 12 minutes. The injector and detector temperatures were 250 °C and 300 °C, respectively. Identification of residues was effected by running samples and external reference standards in GC and then comparing the chromatograms. A peak was not considered relevant unless it appeared in both columns in a given detector. The method average detection limits are given in Table 1.

Recoveries of residues were in the range 59 – 80 % for HCH and 70 – 99 % for DDT which is within the acceptable range (Åkerblom 1995).

**Table 1:** Average method detection limits

Analyte	Sediments [(mg/kg dry weight) x10 <sup>-2</sup> ]	Water (µg/l)
α-HCH	3	0.4
β-HCH	2	0.4
γ-HCH	0.1	0.1
δ-HCH	0.1	0.2
<i>p,p'</i> -DDE	0.06	0.1
<i>p,p'</i> -DDD	0.7	0.3
<i>o,p'</i> -DDT	2	0.6
<i>p,p'</i> -DDT	10	0.6

### RESULTS AND DISCUSSION

Sediments in the drainage ditch at Vikuge Farm were analysed for organic matter content. The organic matter content in these sediments was variable ranging from 0.9% to 21.5%, with a mean of 8.58%.

#### Levels of pesticide residues in sediments

All the HCH and DDT residues found to be abundant in the soil from the old storage site (Kishimba and Mihale 2002) were also detected in the sediments. These include α-HCH, β-HCH, γ-HCH δ-HCH, *p,p'*-DDE, *p,p'*-DDD, *o,p'*-DDT and *p,p'*-DDT.

#### Levels of HCH residues in sediments

The concentration of total HCH in sediments were in the range 280-7,400 mg/kg dry weight (Table 1), with higher concentrations in downhill samples. The β-isomer had overall higher concentrations as compared to α-HCH. Other isomers' concentrations were below the method detection limits. There was an overall

increase in concentrations when moving from uphill to downhill.

Pesticide residues in sediments can be used to provide an indication of pesticide dynamics in the environment (Kammerbauer and Moncada 1998). The high concentration of ΣHCH implies the presence of a constant supply of HCH especially for downhill samples (Table 2). Detecting only γ-HCH and δ-HCH would indicate the storage of lindane on the site (Bacci 1994, Menone et al. 2001). The detection of only α- and β-isomers suggests that technical HCH (which usually contains a high proportion of α-isomer) was in the consignment stored at Vikuge. The non-detection of γ- and δ-isomers can be accounted for by their relatively small proportions in the technical HCH and the photochemical isomerization of the isomers and the rarity of detecting δ-isomer (Fu et al. 2001, Falandsyz et al. 2001).

**Table 2:** Concentrations of HCH Residues in Sediment Samples.

Sampling Point		Concentration (in mg/kg dry weight) x 10 <sup>3</sup>				
		α-HCH	β-HCH	γ-HCH	δ-HCH	SHCH
Uphill points	i	< 0.04	0.28	< 0.05	< 0.09	0.28
	ii	< 0.04	0.4	< 0.05	< 0.09	0.4
	iii	< 0.04	1.55	< 0.05	< 0.09	1.58
	iv	0.2	3.8	< 0.05	< 0.09	4
Downhill points	v	1.4	1.1	< 0.05	< 0.09	2.5
	vi	0.28	1.8	< 0.05	< 0.09	2.08
	vii	0.85	6.55	< 0.05	< 0.09	7.4
	viii	0.2	0.92	< 0.05	< 0.09	1.12

The spatial distribution of  $\Sigma$ HCH showed an irregular pattern in sediment samples. This fact indicates a significant contamination by technical HCH and an accumulative property of the HCH isomers in sediments due to their dispersible nature.  $\beta$ -HCH was present in all the sediment samples. The concentration of  $\beta$ -HCH in technical HCH ranges from 5-12%. However, the  $\beta$ - isomer was found in a large proportion in sediment, almost in all uphill samples (Table 2). The relatively higher percentage of  $\beta$ -isomer in sediments even though it has a lower percentage in technical HCH implies that  $\beta$ -HCH is the most persistent isomer among the HCH isomers (Lee *et al.* 2001). This is supported by the low Henry's law constant and low water solubility of the isomer (Tomlin 2000). The behaviour of HCH isomers in the environment may be explained by volatility and water solubility rather than adsorption to organic matter. Other transport processes such as run-off and input through ground water (the leaching process) might also affect the final composition of HCH isomers in sediments (Lee *et al.* 2001).

#### Levels of DDT residues and metabolites in sediments

The concentration of SDDT ranged from 3,490 mg/kg to 99,620 mg/kg dry weight and relatively high levels were found in the

downhill samples (Table 3). The trend in SDDT concentrations showed that downhill samples had higher concentrations of almost all the DDT residues and metabolites (*o,p'*-DDT, *p,p'*-DDT, *p,p'*-DDE and *p,p'*-DDD), than the uphill samples.

A pattern of distribution almost similar to that of HCH residues in sediments was observed for the DDT residues and metabolites in sediments (Table 3). The DDT residues in the sediments had higher concentrations than their metabolites. In general, the concentrations of the residues increased along the ditch when moving from uphill to downhill. Comparison of the *p,p'*-DDE and *p,p'*-DDD in all the samples shows that *p,p'*-DDD concentrations are relatively higher than those of *p,p'*-DDE suggesting the presence of TDE (DDD) as a pesticide on its own) in the donated consignment. Because DDT is expected to degrade to its fairly stable degradation products, DDD and DDE in soil, the DDT/SDDT ratio is used as an indicator of the duration of time that the DDT residues have been in the environment (Nowell *et al.* 1999). In a situation like this where one of its metabolites, DDD, is used as a pesticides on its own, this ratio is inaccurate. The only ratio that can accurately predict the degradation rate of DDTs in sediment is the DDE/DDT.

**Table 3:** Concentrations of DDT residues and metabolites in sediment samples

Sampling Point	Concentration (in mg/kg dry weight) x 10 <sup>3</sup>						
	<i>p,p'</i> -DDE	<i>p,p'</i> -DDD	<i>o,p'</i> -DDT	<i>p,p'</i> -DDT	SDDT	DDE/DDT	
Uphill points	i	0.1	0.42	0.69	2.18	3.49	0.03
	ii	0.27	1.03	1.69	2.67	5.66	0.06
	iii	0.27	2.05	4.61	3.57	10.91	0.03
	iv	1.63	27.8	7.95	37.9	75.28	0.04
Downhill points	v	1.2	5.74	6.35	36.42	49.71	0.03
	vi	0.89	3.8	74.28	20.65	99.62	0.01
	vii	2.8	11	16.68	28.4	58.88	0.2
	viii	0.5	2.07	3.35	15.4	21.32	0.03

In all the sediment samples analysed, the DDE/DDT ratios were less than one (Table 3), which shows that there is delayed degradation of DDT isomers. This can be accounted for by the absence of favourable conditions for DDT degradation such as Na-bentonite clay surfaces, neutral to alkaline pH environment and adapted microorganisms which may be either sparse or absent altogether as they would have been eliminated by the high concentrations of the pesticides. It has been reported that aerobic degradation of DDT in sediments is prohibited or significantly decreased in presence of high concentration of DDT (Wu *et al.* 1999).

Sediments act as a sink or trap for hydrophobic compounds, (such as DDT, HCH, PCBs) since they have high organic matter. Once an organic compound enters the sediments, it may find itself in a highly reducing environment. Then, depending on its molecular characteristics, it may either be preserved or rapidly reduced. Thus the possibility of DDT to undergo reductive dechlorination in this situation is highly favoured. Chemicals with high octanol-water coefficients,  $K_{ow}$ , are readily adsorbed by natural sediments. Since DDT has a high  $K_{ow}$  ( $pK_{ow} = 6.4$ ), (Larson *et al.* 1996), it would be adsorbed and thus accumulate. But, because of its chemical nature, the reduction process would be very slow. Hence the amount of DDT in sediments was found to be high, as expected.

#### **Levels of other pesticide residues in sediments**

Besides HCHs and DDTs, other residue detected in sediment samples was azinphos-methyl an organophosphorous insecticide whose concentration was 360 mg/kg dry weight. Though azinphos-methyl is known to have a half-life of 1-3 days (Vogue *et al.* 2001), it remains unchanged in the soil for varying lengths of time depending on soil

organic matter and texture. The detection of azinphos-methyl, though in only one downhill sediment sample, is a clear indication that the original consignment consisted of a large quantity of this pesticide, since, even under sterile conditions, its half-life is reported to be at least 355 days (Vogue *et al.* 2001).

#### **Levels of pesticide residues and metabolites in water**

Lower levels of the same residues and metabolites predominant in the sediments were found in most water samples.

#### **Levels of HCH residues in water**

The concentrations of  $\Sigma$ HCHs in water were 0.95 mg/l in tap water, up to 0.44  $\mu$ g/l in surface water and up to 6.15  $\mu$ g/l in ground water, with means of 0.28  $\mu$ g/l in surface water and 3.09 mg/l in ground water (Table 4).

The concentration trend of HCHs showed that ground water samples had higher levels than either tap or surface water samples. Between tap and surface water samples, tap water samples had high concentrations of the HCHs compared to surface water samples (Table 4). This raises serious concern on the quality of the water.

Table 4 shows that a high proportion of the  $\Sigma$ HCH in samples analysed is contributed by  $\alpha$ - and  $\beta$ - isomers. This further supports the hypothesis of the presence of technical HCH in the consignment. High  $\alpha$ -HCH contamination of groundwater indicates that there is a continuous supply of technical HCH from the point source especially during the rainy seasons. This also shows that the isomer does not undergo any degradation before entering the water source (Nowell *et al.* 1999).

**Table 4:** Concentrations of different HCH residues in water

Sampling Point		Concentration (in $\mu\text{g/L}$ of Sample) $\times 10^{-2}$				
		$\alpha$ -HCH	$\beta$ -HCH	$\gamma$ -HCH	$\delta$ -HCH	$\Sigma$ HCH
Tapwater	tap	37	54	2	2	95
	S1	< 0.4	< 0.4	< 0.1	< 0.2	-
Surface water	S2	1	< 0.4	< 0.1	< 0.2	1
	S3	17	24	1	2	44
	S4	12	26	1	1	40
Ground water	G1	3	< 0.4	< 0.1	< 0.2	3
	G2	4	< 0.4	< 0.1	< 0.2	4
	G3	1	< 0.4	< 0.1	< 0.2	1
	G4	1	< 0.4	< 0.1	< 0.2	1
	G5	470	141	0.2	4	615*
	G6	420	144	0.2	4	568*
	G7	2	< 0.4	< 0.1	< 0.2	2

\*Rainy season (March, 2001) samples

**Levels of DDT residues in water**

Total concentrations of DDT was 1.76 mg/l in tap water, up to 0.35 mg/l in surface

water and up to 32.71  $\mu\text{g/l}$  in ground water, with means of 0.18  $\mu\text{g/l}$  in surface water and 9 mg/l in ground water (Table 5).

**Table 5:** Concentrations of different DDT residues in water.

Sampling Point		Concentration (in $\mu\text{g/L}$ of Sample) $\times 10^{-2}$				
		<i>p,p'</i> -DDE	<i>p,p'</i> -DDD	<i>o,p'</i> -DDT	<i>p,p'</i> -DDT	$\Sigma$ DDT
Tap water	Tap	< 0.1	3	22	151	176
	S1	< 0.1	4	4	27	35
Surfacewater	S2	< 0.1	4	1	28	33
	S3	1	1	< 0.6	< 0.6	2
	S4	2	1	< 0.6	< 0.6	3
	G1	< 0.1	2	< 0.6	64	66
Groundwater	G2	< 0.1	6	< 0.6	64	70
	G3	< 0.2	3	1	18	22
	G4	< 0.1	1	1	25	27
	G5	13	112	790	1908	2822*
	G6	13	109	793	2366	3271*
	G7	< 0.1	4	2	22	28

\* Rainy season (March, 2001) samples

The distribution of individual DDT isomers and metabolites' concentrations in water were in descending order: *p,p'*-DDT > *p,p'*-DDD > *o,p'*-DDT in tapwater, *p,p'*-DDT > *o,p'*-DDT > *p,p'*-DDD > *p,p'*-DDE in surface water and *p,p'*-DDT > *o,p'*-DDT > *p,p'*-DDD > *p,p'*-DDE in ground water (Table 5). In all the water samples analysed,

*p,p'*-DDT had higher concentrations than the other residues followed by *o,p'*-DDT.

Of the DDT metabolites detected in water, *p,p'*-DDD had higher concentrations as compared to *p,p'*-DDE. This still is an indication of the presence of DDD (TDE, rothane) in the donated consignment.

Amongst the DDT compounds, the *p,p'*-isomer was more abundant than the *o,p'*-isomer probably because *o,p'*-isomer has a relative small percentage in the technical DDT. It is also known that *o,p'*-DDT undergoes a relatively rapid isomerisation to *p,p'*-DDT and thus its proportion diminishes (Nhan *et al.* 2001). Nevertheless, it is worth noting that *o,p'*-DDT residues were found in almost all water samples except two ground water samples (Table 5). It should be noted that although all DDTs are of known public health concern, the *o,p'*-DDT isomer is known to have endocrine disruptive effects by reducing sperm counts (Ulrich 2000).

From Table 5, one can calculate the non-metabolised fraction of DDT, (DDT/SDDT), which, in the water samples analysed range from 0.6 to 0.98. This high ratio of the parent compound indicates a continuous input from the point source and there is no appreciable degradation of the residues in soil before entering the aquatic environment. This argument can be supported by the fact that DDT appears to break down more quickly once it enters a river or any other waterway (Nowell *et al.* 1999). This can explain the fact that there is an almost constant supply of the residues from the contaminated site (point source), especially in downhill tap water and ground water.

The WHO tolerable daily intake (TDI) for DDT is set at 0.02 mg/kg body weight (WHO 1996). For a person who takes 2 litres of water per day, this implies that a 60 kg person is allowed to consume up to 0.6 mg of DDT in a litre of water. Comparing this with the highest DDT level of 32.71 µg/l (0.033 mg/l) detected in groundwater could mean that villagers at Vikuge who use the water for domestic purposes are not presently at any high health risk. However, in cases of heavy rains and heavy floods, the quality of these waters is questionable. This conclusion arises from the levels of the pesticide residues in the water after heavy

rains when water from the old storage area floods the lower areas, includes the wells.

Pesticide adsorption to soil or sediment is inversely related to its water solubility. The greater the water solubility the greater the mobility (Singh 2001), and the less the adsorption. Since the HCHs are more water soluble than the DDTs, a high proportion of them was expected to be in the water as compared to DDTs. When the concentrations of ΣHCH (Table 4) are compared to those of ΣDDT (Table 5), the concentrations of HCHs are lower than those of DDTs contrary to what was expected. This may not only be due to rapid HCHs' mineralization and formation of volatile organic compounds (Singh 2001), but also due to the lower content of HCHs as compared to DDTs in the soil at the point source (Kishimba and Mihale 2002).

Despite being widely invoked in literature, water solubility is rarely an important factor influencing deep leaching of pesticides. While the water solubility of most pesticides fall within the range of  $10^{-5}$ - $10^{-2}$  mg/l (Barbash *et al.* 1996), the pesticide concentrations measured in water ranges from  $10^{-1}$  –  $1 \times 10^5$  mg/l. The latter gives a better indication of the compounds mostly encountered in water are those that are less likely to be encountered, especially when appreciable time has elapsed after the release of the pesticides into the environment. Thus the majority of the pesticides detected are the chlorinated organic pesticide residues. Water solubility of a may sometimes give little information regarding the tendency of a pesticide to associate with the solid phase in the subsurface, where adsorption interactions are likely to be significant (Barbash *et al.* 1996). This study has shown that that despite their low water solubilities, HCH and DDT residues and metabolites and other organochlorine pesticides appreciably contaminate water at Vikuge.

The high concentrations of organochlorine pesticides in well water indicate that affinity



for organic matter is an unreliable indicator whether an individual pesticide residue or a metabolite will reach groundwater, probably because of the effects of non-equilibrium transport or colloid transport (Barbash *et al.* 1996). Since most pesticides used today are organics and their chemical structures are more similar to those of natural organic matter, NOM, their degree of affinity to organic matter is highly pronounced. This 'like prefers like' rule of solution chemistry causes the pesticides to partition into soil organic matter, (sediments for example), from the aqueous solution. This tells us that detection of the residues in water implies a high concentration of the residues in deep sediments. This makes the sediments to be secondary contamination source by continuously releasing the residues in the aqueous media through sediment-water interface equilibration.

There is increasing evidence that Dissolved Organic Matter, DOM, in water can bind some pesticides. There have been reports that the apparent solubility of hydrophobic pesticides increase by the presence of DOM (Cheng 1990). Association of an organic chemical with dissolved organic matter can increase the solubility of organic molecules in water and a concomitant increase in subsurface mobility, a phenomenon referred to as facilitated transport (Larson *et al.* 1997). It has been found that significant fraction of the dissolved DDT and other residues found in natural waters may be bound to dissolved humic materials. Soluble humic substances can interact with slightly soluble pesticides, thereby increasing the concentration of the pesticide in the aqueous layer (Cheng 1990). Thus the high levels of DDT and other residues detected may be attributed by the presence of dissolved organic matter in water.

Surface-active substances (surfactants) may also influence the rates and pathways of transformation of pesticides in subsurface aqueous systems. The significance of this to pesticide behaviour and fate in ground water

arises from the fact that surfactants are commonly applied as adjuvants in pesticide formulations. Furthermore, natural organic matter exhibits surface-active properties because it typically possesses both hydrophobic and hydrophilic regions within its overall structure. As a result, the sequestration of pesticides within more hydrophobic regions of the natural organic matter may influence the accessibility of these molecules to other attacking chemical species. Because most of the functional groups within the natural organic matter are negatively charged, the association of a hydrophobic pesticide with the more hydrophobic region of NOM may reduce the rate of attack of the molecule by anionic reactants, but accelerates reactions with cationic species. The fact that the rate of dechlorination of DDT and DDD are accelerated by cationic surfactants suggests that these reactions would be slowed in presence of natural organic matter (Barbash *et al.* 1996).

#### **Levels of other pesticide residues in water**

Thiabendazole was detected in surface and groundwater at an appreciable amount at a concentration range of up to 0.17  $\mu\text{g/l}$  in surface-water and up to 0.59  $\mu\text{g/l}$  in groundwater samples. The pesticide residue was not detected in tap-water.

Groundwater contamination by pesticides is usually attributed to the magnitude of the sources, opportunities for interaction through the soil, chemical properties and geological conditions. A pesticide can leave the contaminated site either in its molecular form and enter surface or ground water, in a particle associated form in run-off or through plant uptake. The interaction of the chemical structure and environmental conditions controls the chemical's behaviour and ultimately its effect on the environment (Barbash *et al.* 1996).

#### **CONCLUSION**

The findings above illustrate once again the relationship between water contamination

and improper storage, spills and disposal of pesticides. The results show the presence of pesticides in tap, surface and groundwater wells from the residential area around a storage site. Although the concentrations detected appear to be below the levels of concern - over 15 years after the pesticides were improperly stored – one is left wondering what the situation was a the first years after the storage. Still the results show that pesticides, including the hydrophobic ones such as DDTs and HCHs, can contaminate the water including that intended for domestic uses.

It is thus highly recommended that water for domestic use should be subjected to sedimentation and filtration to remove NOM and DOM as an immediate and temporary measure in reducing the health risks and effects to come. Further government intervention is highly needed particularly on the supply of clean and safe drinking water. Decontamination by biological methods (bioremediation and phytoremediation) of the point source will be the only solution for preventing further pesticide residue spread through global distillation, run off and wind. Further studies are needed to determine the levels of these residues in air, organisms and in the studied media on a wider perspective in the course of assessing the extent of environmental pollution by obsolete pesticides.

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