



Distribution and health risk assessment of organochlorine pesticides (OCPs) in industrial site soils: A case study of urban renewal in Beijing, China

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Received 25 March 2008; revised 23 April 2008; accepted 10 June 2008

Abstract

A field survey was conducted in a contaminated industrial site of southern Beijing, China to investigate the contents and distribution of the organochlorine pesticides (α -, β -, γ -, δ -HCH, *p,p'*-DDT, *p,p'*-DDE, *p,p'*-DDD and *o,p'*-DDT) in the profiles of soil, and a health risk assessment was carried out with CalTOX multimedia exposure model. Results showed that mean concentrations of total hexachlorocyclohexane isomers (HCHs) and total dichlorodiphenyltrichloroethane isomers (DDXs) in soils were in the range of 13.20–148.71 mg/kg, and 3.02–67.43 mg/kg, respectively. Organochlorine pesticides (OCPs) content peaked in the surface and declined in soil profile with depth. The amounts of HCHs in three profiles of soil were larger than DDXs. Composition analysis indicated that there was a trend of degradation of OCPs in the site, but the mean of HCHs and DDXs concentration were over the state warning standard limit (HCHs, 0.50 mg/kg; DDXs, 0.50 mg/kg). According to current land use development, health risk assessment with CalTOX and Monte Carlo analysis showed that health risks mainly came from two exposure pathways: dermal uptake and inhalation, and the total risk values all exceeded the general acceptable health risk value (10^{-6}). The sensitivity analysis indicated that five parameters significantly contributed to total risk.

Key words: pollution; hexachlorocyclohexane isomers (HCHs); dichlorodiphenyltrichloroethane isomers (DDXs); soil profiles; Monte Carlo; sensitivity

DOI: 10.1016/S1001-0742(08)62278-0

Introduction

The usage of organochlorine pesticides (OCPs) has been banned in China more than 20 years because of their persistence (Dimond and Owen, 1996), bioaccumulation (Nakata *et al.*, 2002) and toxicity to human and other animals and plants (Jones and de Voogt, 1999). In May 2001, dichlorodiphenyltrichloroethane was listed out by the Stockholm Convention as one of 12 persistent organic pollutants. China historically produced large amounts of hexachlorocyclohexane (HCH) and dichlorodiphenyltrichloroethane (DDT) for agriculture. A total of 4.9 million tons of HCH and 0.4 million tons of DDT was produced from the 1950's through 1983 (Li *et al.*, 2006), and occupied 33% and 20% of the global production for HCH and DDT, respectively (Hua and Shan, 1996; Fu *et al.*, 2003). It was reported that about 14 million hm^2 of farmland in China were considered polluted by organochlorine pesticides in 1983 (Feng *et al.*, 2003). In last century, about 8000 tons of OCPs were used in agriculture in Beijing (Zhao and Ma, 2001), and from 1972 to 1974, 610 tons of HCH and 190 tons of DDT were sold in suburbs of Beijing (Zhu *et al.*, 2005).

China is experiencing rapid urbanization and industrialization as cities are undergoing urban renewal, restoration and development, especially in capital city, Beijing. However, industrial development has contributed significant stress to both urban and natural environments during the last two decades (Zeng *et al.*, 2001). A dozens of chemical plants located in southern Beijing until the end of last century, which caused serious air, water, and soil pollution. For the Beijing 2008 Olympic games, government commanded all chemical plants moved out of the city to improve environmental quality, but the residues left from the application of OCPs may still be harmful to the health of local residents. It is absolutely necessary to define the health risks of OCP's and quantify their distribution in the soils to help us prevent future impacts to human health.

Health risk assessment has been utilized for more than 30 years. In previous studies, most research has focused on the distribution of OCPs at regional scale. However, little research has exists to predict the risk to the residents' health based on OCPs residue distribution in contaminated industrial sites, particularly for the purpose of real estate development. This project aims to investigate the levels of OCPs in a typical contaminated industrial site soil and adopt the CalTOX multimedia exposure model to assess the human health risk. This research seeks to determine the

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OCPs residue distribution at the site and calculate the level of risk caused by OCPs contamination to human health. Assessing and modeling OCPs distribution may identify sensitive parameters by sensitivity analysis, which can provide the foundation for better environmental management and site exploitation to minimize health risks related to OCPs contaminated soil.

1 Materials and methods

1.1 Site description

The contaminated study site is located between the third and the fourth ring road in southern Beijing and is surrounded by a residential area. The site measured roughly 40000 m² and was occupied by a OCPs (HCH and DDT) pesticides plant until 1983. Now, this area is replaced by a paint factory. Future restoration and residential development are currently planned for the site.

1.2 Sample collection

Soil samples (118) was collected randomly from three soil layers: the surface (0–20 cm), the root zone (20–200 cm) and the vadose zone (200–400 cm). Soils consisted of brown and sandy loam and silt loam. Soil samples were air-dried at room temperature and ground to pass through a 60-mesh sieve. All samples were kept frozen at –4°C until analysis and were subject for measurement of HCHs (α -HCH, β -HCH, γ -HCH, and δ -HCH) and DDXs (p,p' -DDT, p,p' -DDE, p,p' -DDD, and o,p' -DDT).

1.3 Extraction and cleanup

Soil sample (10 g) was weighed, put into centrifuge tubes and mixed with 30 mL acetone/petroleum ether solvent (1:1, V/V) before the mixture was extracted for 20 min with ultrasonic shaking apparatus (HH-500DB, Kunshan Hechuang ultrasonic instruments Corp., China). The mixture was centrifuged and the extracts were collected. The same extraction was repeated twice to the filter residue. All extracted solutions were concentrated to 1 mL by a rotary evaporator (R201, BUCHI Laboratory Equipment, Switzerland), and further purified with a glass column (12 mm i.d.) loaded with 5 g of activated florisil. The elution was subsequently carried out using 20 mL of hexane, followed by 100 mL of hexane containing 5% ethyl acetate (V/V). The eluent was filtrated with anhydrous sodium sulfate. Finally, the solution was concentrated to 50 mL under a gentle steam of pure nitrogen.

1.4 Analysis procedure and quality assurance

The extracted samples were analyzed for OCPs using a PE Autosystem XL gas chromatograph (GC) (Perkin Elmer Auto System XL, USA) equipped with a HP-5 silica capillary column (25 m \times 0.32 mm i.d. \times 0.25 μ m film thickness) and a ⁶³Ni electron capture detector (μ -ECD). The oven temperature was maintained at 150°C, raised to 180°C at a rate of 8°C/min, then programmed to 250°C at 20°C/min and held for 5 min. The injector and the detector were maintained at 250 and 300°C, respectively. Nitrogen (purity > 99.999%) was employed as the carrier gas at

flow rate of 2.8 mL/min. GC peaks were identified based on the retention time of individual authentic standards. Quantification of OCPs was carried out by peak area measurements based on the external standards calibration curves.

The detection limits for soil samples are 0.0001 mg/kg for α -HCH, 0.0002 mg/kg for β -HCH, 0.0001 mg/kg for γ -HCH, 0.0005 mg/kg for δ -HCH, 0.0005 mg/kg for p,p' -DDE, 0.0003 mg/kg for o,p' -DDT, 0.0002 mg/kg for p,p' -DDD, and 0.0005 mg/kg for p,p' -DDT, respectively. Recoveries α -HCH, β -HCH, γ -HCH, δ -HCH, p,p' -DDE, o,p' -DDT, p,p' -DDD, and p,p' -DDT are 83.5%, 85.0%, 87.6%, 91.2%, 93.8%, 94.2%, 90.1%, and 88.1%, respectively. The analysis of three blanks covering the entire analytical procedure (from the extraction to the GC analysis) was subjected to assess the interference from the reagents and glassware. GC analysis was repeated twice for each replicate sample and the relative standard deviation (RSD) of replicate samples were less than 15%.

1.5 Multimedia risk assessment model

CalTOX is a multimedia, multiple pathway risk assessment model constructed by Californian Environmental Protection Agency used extensively to estimate chemical fate and human exposure to contaminants from polluted soil, air, ground water, surface water and sediment in the vicinity of hazardous waste sites (McKone, 1993; Bonnard, 2006). The model consists with the multimedia environmental fate model and multiple pathway exposure models. Eight environmental compartments were considered in the model: ambient air, groundwater, plant leaves, plant leaf surfaces, root-zone soil, the vadose zone soil below the root zone, surface water and sediments. Twenty-three exposure pathways were also included (Chang *et al.*, 2004).

To calculate the exposure dose and human risk, the model can be described by Eq. (1) (USEPA, 1989):

$$H = \frac{\sum_k (1 - \exp(-ADI_k \times CSF_k))}{\sum_k ADI_k \times CSF_k} \approx \quad (1)$$

where, H is the human health risk, ADI_k (mg/(kg·d)) is the average daily intake of the contaminants by receptors through exposure route k (inhalation, ingestion, or dermal absorption), CSF_k ((kg·d)/mg) is the cancer slope factor of exposure route k , which is derived from dose-response studies (McKone, 1993).

When an environmental concentration is assumed constant over the exposure duration (ED), the population-averaged potential dose (for ingestion or inhalation routes) or absorbed dose (for dermal contact) is the average daily dose rate (ADI_{ijk}).

$$ADI_{ijk} = C_i \times \frac{C_j}{C_i} \times \frac{IU_{jk}}{BW} \times \frac{EF \times ED}{AT} \quad (2)$$

where, ADI_{ijk} (mg/(kg·d)) is the average daily intake from environmental media i (air, soil or groundwater), exposure media j (drinking water, food, etc.) and exposure route k (inhalation, ingestion, or dermal absorption), C_i (mg/kg)

is the contaminant concentration in environmental medium i , C_j (mg/kg) is the pollution concentration in exposure media j . In this expression C_j/C_i is the intermedia-transfer ratio, which expresses the ratio of contaminant concentration in the exposure medium j to the concentration in an environmental medium i , IU_{jk} is the contact rate of exposure media j through exposure route k , and IU_{jk}/BW (kg^{-1}) is the intake or uptake factor per unit body weight associated with the exposure medium j , and route k . EF (d/year) is the exposure frequency, ED (year) is the exposure duration, AT (d) is the averaging time, and BW (kg) is the body weights of receptors.

Ideally, all types of exposure pathways should be considered, but time and cost constraints prevented the inclusion of all parameters in the model. Site conceptual modeling can then be conducted to determine the significant exposure pathways by considering the chemical and landscape properties, as well as the fate/transport of chemicals (American Society for Testing and Materials, 2000). In Beijing, residents do not drink or directly use the ground and surface water, and most of the food is imported from outside of the city. Furthermore, the study site has no plant cover. Considering these factors, likely exposure pathways included dermal uptake, inhalation and direct soil ingestion.

CalTOX input parameters are included in Table 1, the HCHs (α -, β -, γ -HCH) and DDXs (p,p' -DDT, p,p' -DDE, p,p' -DDD) parameters are CalTOX default parameters. Parts of landscape parameters were collected from the contaminated site.

Table 1 Landscape properties and exposure factors used in CalTOX model

Landscape property	Value	Exposure factors	Value
Site area (m^2)	40000	Body weight (kg)	65
Ambient temperature (K)	286	Inhalation rate (m^3/d)	20
Water content in root zone (%)	15	Lifetime (year)	70
Wind velocity (cm/s)	312.5	Exposure duration (year)	20
Soil organic carbon content (g/g)	0.0155	Surface area (m^2/kg)	0.026
		Soil ingestion (mg/d)	0.35

Landscape properties are collected from environmental monitoring data of the contaminated site; exposure factors are default values of the CalTOX model.

A Monte Carlo technique was used to calculate the uncertainty of estimated risk by repeating a random sample 5000 times, which has been examined sufficient to avoid unacceptable variance in different realizations. Sensitivity analysis method was used to rank the input parameters on the basis of their contribution to variance in the output because information is a driving factor in the overall uncertainty of risk estimates for populations (Chen and Ma, 2006).

2 Results and discussion

2.1 Content of HCHs and DDXs in the site

Table 2 shows the concentration, arithmetic means, standard error (SE), minimum and maximum of OCPs in the soil samples. Because the production plants were distributed irregularly in the site, the concentration of ΣHCHs and ΣDDXs was also unevenly distributed. The concentration mean of ΣOCPs in the contaminated site soil in the range 16.22–216.14 mg/kg, 13.20–148.71 mg/kg for ΣHCHs , and 3.02–67.43 mg/kg for ΣDDXs in different layers. There were significant differences among the soil layers in total OCPs, with the highest concentrations in surface soil layer, decreasing with depth.

Figure 1 presents the percentages of HCHs ($\Sigma\text{HCHs} = \alpha\text{-HCH} + \beta\text{-HCH} + \gamma\text{-HCH} + \delta\text{-HCH}$) and DDXs ($\Sigma\text{DDXs} = p,p'\text{-DDD} + p,p'\text{-DDE} + p,p'\text{-DDT} + o,p'\text{-DDT}$) isomers in vertical soil layers and it clearly shows that the ΣHCHs were higher than ΣDDXs . In different soil layers, the ΣHCHs ranged from 68.81% to 81.31% while the ΣDDXs ranged from 18.69% to 31.19%. ΣHCHs were increasingly prevalent with depth. HCHs and DDXs

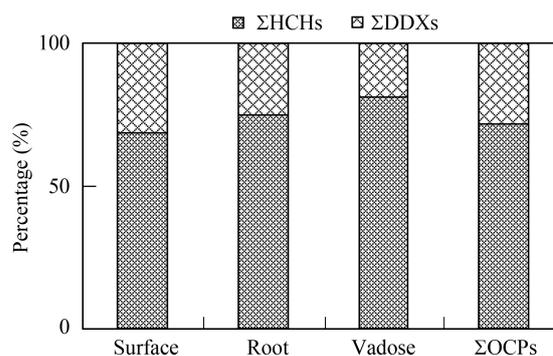


Fig. 1 Content of HCHs and DDXs in the industrial site.

Table 2 Concentration of organochlorine pesticides in soil profiles from contaminated site (mg/kg dw)

	Surface				Root zone				Vadose zone			
	Mean	SE	Max.	Min.	Mean	SE	Max.	Min.	Mean	SE	Max.	Min.
α -HCH	43.97	19.07	957.31	0.02	42.29	22.69	880.98	0.01	5.41	4.49	161.67	0.01
β -HCH	93.21	27.03	948.32	0.32	54.98	17.71	490.08	0.09	2.34	1.33	48.48	0.02
γ -HCH	6.07	3.81	83.52	0.03	11.55	5.33	60.18	0.12	5.00	4.82	58.05	0.01
δ -HCH	5.46	2.15	61.03	0.02	3.04	1.04	15.68	0.06	0.45	0.19	2.04	0.04
p,p' -DDE	9.58	1.61	48.26	0.17	6.53	2.40	92.82	0.01	0.30	0.08	1.85	0.02
p,p' -DDD	23.84	11.14	446.08	0.08	12.89	7.81	235.01	0.02	1.31	0.83	15.24	0.02
p,p' -DDT	25.28	6.14	219.69	0.44	13.77	4.90	155.4	0.03	1.05	0.35	7.23	0.02
o,p' -DDT	8.73	2.31	83.43	0.13	4.11	1.48	48.57	0.02	0.36	0.09	0.95	0.05
ΣOCPs	216.14				149.16				16.22			

occupied 71.76% and 28.24% at the site, respectively. Differences between the incidence of these compounds can be explained by differences in their speed of metabolic degradation in the environment (Willett *et al.*, 1998), and the production yield differences before the plant closed.

2.2 Characteristics of OCPs in soil profiles and accumulation

The concentration and distribution of OCPs at this site are subject to physical, chemical and biological processes such as biodegradation, chemical degradation, photolysis, leaching, adsorption, volatilization, etc. (Wang *et al.*, 2006). HCHs and DDXs in soil may transfer to lower layers or stay in the surface soil, and both can cause long term negative effect on ecosystem and human health (Sun *et al.*, 1997).

Figure 2a illustrates the vertical distribution of HCH isomers in three soil layers. The content of β -HCH in the upper two layers is higher than other isomers, and in the contaminated site, the total β -HCH is also higher than others. The content of different HCH isomers could be shown as β -HCH > α -HCH > γ -HCH > δ -HCH. But in the deep layer, there is little difference in HCH isomers distribution from the upper two layers, in that γ -HCH and α -HCH have stronger penetration ability than β -HCH and δ -HCH (Cooperation group, 1980), and they move downwards to more than β -HCH and δ -HCH. The typical HCH generally contains 60%–70% of α -HCH, 5%–12% of β -HCH, 10%–15% of γ -HCH and 6%–10% of δ -HCH (Walker *et al.*, 1999). The physicochemical properties of these HCH isomers are different, β -HCH has the lowest water solubility and vapor pressure, making it the most stable and relatively resistant to microbial

degradation (Ramesh *et al.*, 1991; Willett *et al.*, 1998; Walker *et al.*, 1999; Manz *et al.*, 2001). However, it is the most persistent and least volatile isomer (Buser and Muller, 1995). The α -HCH isomer can be converted to β -HCH in the environment (Wu *et al.*, 1997; Walker *et al.*, 1999) such that β -HCH tends to accumulate in the soil (Li, 1999). Figure 2b represents the content of different isomers of HCH at the contaminated site, it clearly shows that β -HCH was the dominant at the site, and α -HCH, β -HCH, γ -HCH, and δ -HCH occupied 34%, 55%, 8%, and 3%, respectively. Great changes occurred as compared to original components of HCHs because of its metabolic degradation, and the components of HCH isomers indicate that much α -HCH is metabolized to the other isomers.

Figure 3a illustrates the DDXs' vertical distributions in the soil profiles. Similar to HCHs distribution patterns, the content of DDXs decreases with depth and in the order of p,p' -DDT > p,p' -DDD > p,p' -DDE > o,p' -DDT at the site. Technical DDT generally contains 75% p,p' -DDT, 15% o,p' -DDT, 5% p,p' -DDE, and < 5% others (Yang *et al.*, 2005). Figure 3b represents the content of different isomers of DDX at the site, which indicates that p,p' -DDT, o,p' -DDT, p,p' -DDE, and p,p' -DDD occupied 37%, 12%, 15%, and 36% respectively. As contrast to DDXs technical components, there are great changes because of its metabolic transformation. In soil, DDT can be transformed to the stable and toxic metabolites DDD and DDE, while under anaerobic conditions, the DDT may transfer to DDD. Otherwise, DDT may transfer to DDE under aerobic conditions (Heberer and Dnnbier, 1999; Kalantzi *et al.*, 2001). In our research the content of DDD was higher than DDE (Fig. 3b), indicating that the most probable degradation pathway transformed DDT to DDD.

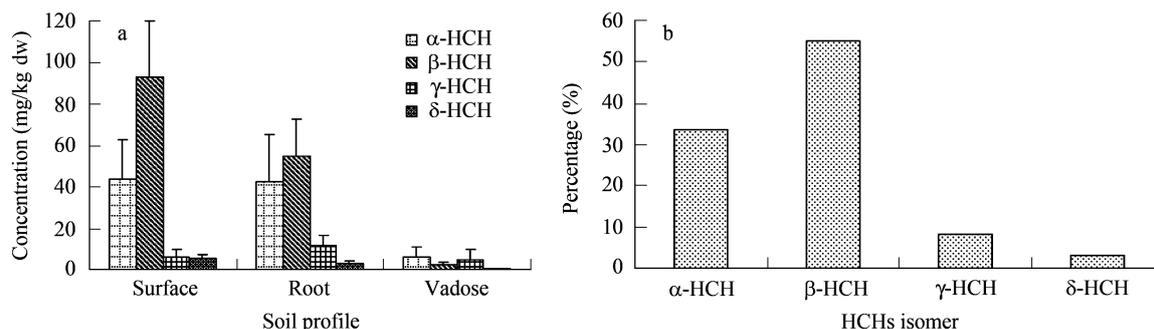


Fig. 2 Concentration of HCHs isomers in different soil profiles (a) and their percentage in the industrial site (b).

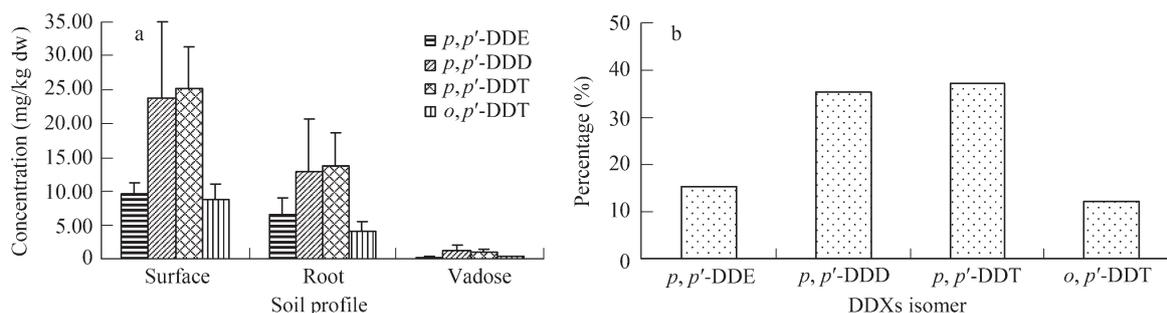


Fig. 3 Concentration of DDXs isomers in different soil profiles (a) and their percentage in the industrial site (b).

Anaerobic metabolism probably dominated the DDT's transformation because the contaminated site is located in lowland that, in the past, was used as a rice paddy, having anaerobic soils. Studies have shown that DDXs have likely been subjected to a long-term weathering when the value of $(DDE+DDD)/\sum DDXs$ exceeds 0.5 (Hitch and Day, 1992; Hong *et al.*, 1999). In the soil profiles at the contaminated site, the ratios of $(DDE+DDD)/\sum DDXs$ were 0.50, 0.52, and 0.53 for three layers, respectively, from top to bottom, suggesting that the DDX's have degraded gradually. In an earlier study where the ratio reached to 0.76 (Yang *et al.*, 2005). It was suggested that the ratio of $(DDE+DDD)/\sum DDXs$ at the site would be greater as time goes by.

2.3 Health risk assessment of OCPs in the site

We calculated the health risk in different exposure pathways according to the predefined residential exposure scenarios. The 95th percentile risk distribution is often be considered to be the starting point for risk management (Presidential/Congressional Commission on Risk Assessment and Risk Management, 1997), and 10^{-6} is the generally acceptable lifetime health risk.

Table 3 shows the results of health risk assessment for the contaminated site studied. Although a great deal of effort has been made by CalTOX's developers to define default values or relationships to calculate default values for all the input parameters (Bonnard, 2006), and the values are necessary to calculate the health risk, it is not provides the parameters of γ -HCH and o,p' -DDT. For this reason, we could not calculate the health risk from the two isomers. For different OCPs isomers, the human health risk values at 95th percentile by three exposure pathways are different. As β -HCH, p,p' -DDE, and p,p' -DDT are concerned, except for direct ingestion, they contribute a great part of health risk by dermal contact and inhalation intake. For α -HCH and γ -HCH, both of them contribute health risk by inhalation pathway. Moreover, p,p' -DDD contributes the health risk by dermal contact pathway. In all, the total risk from different pathway is ranked as: p,p' -DDT > β -HCH > p,p' -DDE > p,p' -DDD > α -HCH > γ -HCH. Results indicate that health risk mainly occurs from dermal uptake and inhalation. Direct ingestion

contributed least to the total risk, but is still a concern. This result contrasts with some previous studies that consider ingestion and inhalation as the predominant exposure pathways (Schuhmacher *et al.*, 2001; Chang *et al.*, 2004; Gloennec *et al.*, 2005; Chen and Ma, 2006), likely because most of these assessments considered food chains as the exposure pathway. In this study, we did not prioritize food chains as an exposure pathway when predefining the exposure conditions; because the study area is in Beijing, where most of the residents' food are transported from outside the city, thus little health risk came from ingestion intake.

2.4 Sensitivity analysis

In order to rank the importance of influential factors in CalTOX model, we conducted sensitivity analysis using Crystal Ball 7.2 software (Decisioneering Inc., Denver, Colorado, USA).

Table 4 lists the significant (> 1%) variables. For HCHs, the reaction half-life in root-zone soil (Thalf_s) was the most sensitive parameter, similar to results found for PAHs (Loranger and Courchesne, 1997). In CalTOX database (Mckone, 1993), the Thalf_s default values of α -HCH, β -HCH, and γ -HCH are 46.9, 105, and 105 d, respectively. In contrast, fraction dermal uptake from soil (dfct_sl) and exposure duration (ED) also played significant roles in health risk. For DDD and DDT, the soil adherence to skin (Slsk) and vapor pressure (VP) are also sensitive parameters. However, sensitivity analysis indicated that Thalf_s, dfct_sl, ED, EFsl, and Slsk are the most sensitive variables. Furthermore, the rank of these variables is different for different isomers due to the variability in soil characteristics and experimental conditions, under which Thalf_s were calculated (Jury *et al.*, 1987).

Because of the persistence of OCPs in the soil and their transportation through environmental media, the implication of sensitivity in this study indicates that health risk would stem from the characters of OCP's themselves and human exposure parameters in the model; the environmental parameters are comparatively less sensitive. However, it must be emphasized that these values were drawn from the publications and other experiments.

Table 3 Risk values at 95th percentile for different exposure pathways in CalTOX model

Exposure pathway		Dermal	Inhalation	Ingestion	Total risk
α -HCH	Risk	8.72×10^{-7}	2.58×10^{-6}	1.13×10^{-9}	3.45×10^{-6}
	Contribution	25.27%	74.70%	0.03%	100.00%
β -HCH	Risk	4.76×10^{-6}	1.21×10^{-6}	7.07×10^{-9}	5.98×10^{-6}
	Contribution	79.59%	20.29%	0.12%	100.00%
γ -HCH	Risk	8.71×10^{-7}	2.20×10^{-6}	1.32×10^{-9}	3.07×10^{-6}
	Contribution	28.37%	71.59%	0.04%	100.00%
δ -HCH, o,p' -DDT ^a	/	/	/	/	/
p,p' -DDE	Risk	3.04×10^{-6}	2.59×10^{-6}	5.44×10^{-9}	5.64×10^{-6}
	Contribution	53.91%	45.99%	0.10%	100.00%
p,p' -DDD	Risk	5.03×10^{-6}	1.81×10^{-7}	9.39×10^{-9}	5.22×10^{-6}
	Contribution	96.35%	3.47%	0.18%	100.00%
p,p' -DDT	Risk	8.23×10^{-6}	3.18×10^{-6}	1.41×10^{-8}	1.14×10^{-5}
	Contribution	72.06%	27.82%	0.12%	100.00%

^a No given parameter value in the CalTOX model.

Table 4 Major parameters from the sensitivity analysis in the CalTOX model

Parameter	Contribution to total risk (%)					
	<i>p,p'</i> -DDE	<i>p,p'</i> -DDD	<i>p,p'</i> -DDT	α -HCH	β -HCH	γ -HCH
Thalf_s	38.23	27.48	6.15	93.99	82.01	88.85
dfct_sl	14.85	37.40	21.58	–	3.84	1.11
ED	13.34	7.43	39.11	–	1.98	2.60
foc_s	6.38	–	1.56	1.46	4.67	2.11
Efsl	6.17	14.58	9.05	–	1.22	–
S	3.76	–	2.05	–	–	–
Slsk	3.02	8.27	4.91	–	–	–
K_{oc}	2.91	–	–	–	1.46	–
H	2.31	–	1.16	–	–	–
BRa	2.09	–	1.24	–	–	–
VP	1.39	–	6.84	–	–	–

Thalf_s (d): reaction half-life in root-zone soil; dfct_sl: fraction dermal uptake from soil; ED (year): exposure duration; foc_s: organic carbon fraction in upper soil zone; Efsl (d/year): exposure frequency to soil on skin; S (mol/m³): solubility; Slsk (mg/cm²): soil adherence to skin; K_{oc} : organic carbon partition coefficient K_{oc} ; H ((Pa·m³)/mol): Henry's law constant; BRa (m³/(kg·h)): active breathing rate; VP (Pa): vapor pressure. –: <1% variable.

3 Conclusions

In the last fifty years, organochlorine pesticides were extensively produced and used in China. This project focused on the OCPs plant site in Beijing, analyzed the OCPs distributions in the contaminated soil and estimated their risk to human health from different exposure pathways. The results showed that after the government prohibited the OCPs' production and usage, the content of OCPs in an industrial site decreased with depth of soil layers and most of them degraded to more stable and toxic isomers. The trend will be increased with the passage of time.

Our CalTOX evaluation, combined with Monte Carlo and sensitivity analysis, shows that there is unacceptable health risk at the site if the site is used for real estate development. Under a realistic, predefined exposure scenario corresponding to the conditions and customs of Beijing's urban residents, we found that health risks can be attributed to dermal uptake and inhalation. Sensitivity analysis shows that the reaction half-life in root-zone soil, fraction dermal uptake from soil, exposure duration, exposure frequency to soil on skin and soil adherence to skin are the most sensitive variables to total health risk because the most sensitive factors result from the chemical nature of different pollutants and from human exposure parameters. These results will enable environmental managers to reduce health risks to local residents. Our results suggest that risk management should be introduced to the contaminated site, and that pollutants should be removed from soils at the site to achieve an acceptable level for future development. In the meantime, our research shows that environmental managers should protect residents from direct dermal exposure to the soil pollutants.

Acknowledgments

This work was supported by the National Natural Science Foundation of China (No. 70433001); the Knowledge Innovation Project of the Chinese Academy of Sciences (No. KZCX2-YW-422, 324). The authors are grateful to Dr. Robin Marushia and Prof. Larry Li for reviewing the manuscript, and thank Prof. Jizheng He for his suggestions.

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