information regarding physical-chemical properties for chemicals often varies widely between sources and the quality of data cannot be compared without specific review of the individual studies. This is demonstrated by the available data on the physical-chemical properties of Chlordecone presented in Table 1.1. The two values for the vapour pressure are rather uniform (0.3 and $0.4x10^5$ Pa) but the water solubility found in literature varies by an order of magnitude (0.35–3.0 and the lowest value is considered to be unreliable.[8]

The comparison of Chlordecone with already listed POPs is presented in Table 2.2. As a starting point for this comparison, the highest and lowest values for Chlordecone (Table 1.1) were used. For already listed POPs, information was sought on the UNEP-POPs homepage. Among the currently listed POPs, most of the relevant properties were available for aldrin, chlordane, dieldrin, DDT, hexachlorobenzene, mirex, toxaphene, endrin and heptachlor. Missing information (water solubility of mirex) was sought in US ATSDR (1995) and AMAP (2004). The US ATSDR (1995), quotes values of 0.2 and 0.6 mg/L, while the AMAP (2004), quotes Mackay for very low water solubility: $6.5x10^{-5}$ mg/L. In order to avoid introduction of what seems to be an outlier in the comparison, the value for water solubility of mirex from US ATSDR (1995) was used.

The water solubility and vapour pressure as well as Henry's Law Constants calculated from these values of the currently listed POPs are summarised in Table 2.2 together with information on Chlordecone from Table 1.1.

**Table 2.2 Water solubility (WS), vapour pressure (VP) and (calculated) Henry's Law Constant (HLC) (at 25°C) for Chlordecone and currently listed POPs.**

| Substance | WS mg/L | VP Pa | HLC Pa $m^3$/mol |
|---|---|---|---|
| Chlordecone-min | 1.0 | 0.00003 | 0.0049[1] |
| Chlordecone-max | 3.0 | 0.00004 | 0.02[2] |
| POP-min | 0.0012 (DDT) | 0.000025 (DDT) | 0.04 (endrin) |
| POP-max | 3.0 (toxaphene) | 27 (toxaphene) | 3726 (toxaphene) |
| POP-2nd max | 0.5 (dieldrin) | 0.04 (heptachlor) | 267 (heptachlor) |

1: Calculated from maximum water solubility and minimum vapour pressure
2: Calculated from minimum reliable water solubility and maximum vapour pressure

Table 2.2 shows that the water solubility of Chlordecone is at the level of the most water soluble among the currently listed POPs (toxaphene and dieldrin), while the vapour pressure is comparable to that of DDT. The highest of the two Henry's Law Constants that were calculated for Chlordecone is of the same order of magnitude as that of endrin. It should be noted that in presenting the data in table 2.2 it is not inferred that a chemical (in this case Chlordecone) is considered to meet the long range environmental transport criterion just because it fits within the range of values of currently listed POPs.

Further to this, it should be mentioned that the latest AMAP report on POPs (AMAP, 2004) describes the possibilities of particle borne transport for substances, which have Henry's Law Constants (HLC) close to that of Chlordecone (HLC = 0.0049 or 0.056). Based on HLC-values from AMAP (2004), it is concluded that semi-volatile compounds such as lindane (γ-HCH) (HLC = 0.000149) and chlordane (HLC = 0.342) are distributed between airborne particles and the gaseous phase, depending on the temperature. These can be washed out *via* precipitation and temporarily deposited in seawater or soil and can absorb to water, plant and soil surfaces from the gaseous phase. During favourable warm weather conditions, these compounds evaporate again into the atmosphere and undergo further atmospheric transport. This remobilization is also called the 'grasshopper effect'. The role of stormy weather situations in remobilization of semivolatile compounds into the atmosphere is obvious but still scarcely investigated (AMAP, 2004).

Besides, certain physical-chemical properties of Chlordecone, such as the partition coefficients log $K_{ow}$ (octanol-water partition coefficient) and log Kaw (air-water partition coefficient), are similar to those of some toxaphene components which, added to its persistence in air and water, would mean that coupled long range transport in atmosphere and oceans may take place (*i. e.* the substance is exchanging between atmospheric gas phase and oceanic dissolved phase and can be

[8] Availability of high quality data regarding physical-chemical properties could support more firm conclusions.

transported in either phase). (Wania, F. 2006, personal communication). Chlordecone has a very low Henry's law constant and a high mass fraction is found in water, and therefore it can be inferred that transport with ocean currents contributes to the long-range transport of Chlordecone.

In a recent modeling study, Scheringer *et. al.*,(2006), investigated the persistence and long range transport potential of these potential POPs, including chlordecone and hexabromobiphenyl, using an OECD screening tool which based the evaluation of overall environmental persistence and transport potential on the results of several of the currently available multimedia environmental fate models (see also Klasmeier *et. al.*, 2006, and Fenner *et. al.*, 2005 for a more detailed explanation). They concluded that the four POP candidates have persistence and long range transport potential properties similar to those of several known POPs in this evaluation. Furthermore, they included the uncertainty regarding the data quality in an uncertainty analysis, which indicated that the result is valid although there are considerable uncertainties in the chemical properties of the four POP candidates. It should be noted that environmental fate modeling results strongly depend on the assumptions made, specifically when essential data such as environmental half-lives are not known. In addition, results for substances like Chlordecone, which are strongly bound to particles and are of very low volatility, are highly dependent on the medium to which they are emitted, i.e., to air, to water, or to soil. The emission to air scenario always yields the highest transfer efficiency, and that value is displayed in the Scheringer *et. al.*, (2006) plots. Transfer efficiency will likely differ by several orders of magnitude when evaluated under soil and water emission scenarios.

**Conclusion**

In summary, the above discussion shows that the available data on Chlordecone are not conclusive when it comes to long-range atmospheric transport in gaseous form. However, atmospheric transport of particle-bound substances and transport of sediment particles in ocean currents, as well as biotic transport, could also contribute to long-range environmental transport of Chlordecone Coupled atmosphere-ocean transport also seems quite possible.

Due to a lack of monitoring data on Chlordecone the assessment of the potential for long-range transport of Chlordecone must be based on physico-chemical properties and modelling data. The modelling study of Scheringer *et. al.*, 2006, shows clearly that long range environmental transport is possible (and possibly more than actually estimated), even considering the uncertainties surrounding the physico-chemical properties.

In accordance with paragraph 7 (a) of Article 8 of the Convention, and taking into account that a lack of full scientific certainty should not prevent a proposal from proceeding, Chlordecone is likely, as a result of its long-range environmental transport, to lead to significant adverse human health and environmental effects such that global action is warranted.

## 2.3 Exposure

### 2.3.1 Environmental concentrations

The available information regarding environmental concentrations of Chlordecone is very limited and includes only areas in the vicinity of production (US) or use (Martinique).

The US ATSDR (1995), illustrates the presence of Chlordecone in the environment following production of the substance. In 1977, 12 years after production of Chlordecone began and 2 years after the production ceased, average concentrations of Chlordecone in estuarine water (dissolved) were <10 ng/L (ppt) (Nichols 1990). In October 1981, 6 years after production ceased, Chlordecone water concentrations ranged from not detectable to 0.02 µg/L (ppb) (Lunsford *et. al.*, 1987). Groundwater monitoring data are lacking, but because Chlordecone binds tightly to organic matter in soil, leaching into groundwater is not expected to occur extensively (Abbreviated from US ATSDR, 1995).

Recent monitoring data from the United States demonstrate the persistence of Chlordecone, known as Kepone in the United States. The substance is included in the U.S. EPA National Lake Fish Tissue Study to estimate the national distribution of selected residues in fish tissue from lakes and reservoirs in the lower 48 states. There were a total of 881 samples collected and analyzed between 2000 and 2005. For Chlordecone, there were 152 hits (17.25%), ranging from 12.3 and 2008 ppb. (Jensen, 2006).

In Martinique, the widespread use of Chlordecone until 1993 has resulted in contamination of soils and surface water in most of the island (Bocquené & Franco, 2005). These authors reported an investigation from 2002 of the presence of a series of pesticides in the water at the mouth of seven rivers. They measured Chlordecone in particulate matter or sediment of six of the seven rivers at concentrations up to 57 µg/kg in particulate matter, and up to 44 µg/kg in sediment.

Bocquené & Franco (2005), quoted other investigations in which concentrations of Chlordecone in the range 1.20 to 2.13 µg/L were measured in rivers of Martinique in 2002-2001. They also stated that Chlordecone was "ubiquitous" in river water used for drinking water.

Further to this, the report prepared for L'Assemblée Nationale (Beaugendre, June 2005), described the history of the use of Chlordecone in Guadeloupe and Martinique, and mentioned several monitoring programmes which are expected to result in reports at the end of 2005. However, these reports have not been available when drafting this document.

### 2.3.2 Human exposure

In the US ATSDR (1995), the experience from production of Chlordecone is summarised as follows: Chlordecone has not been detected in human adipose tissue or in blood samples from the general population, although historically it was detected in human milk samples collected in the south-eastern United States (EPA 1978c). Information is available regarding Chlordecone levels in blood of occupationally exposed workers and their families during 1974-1975 employed at the Hopewell, Virginia site. (Cannon *et. al.*, 1978; Epstein 1978; Knishkowy & Baker 1986; Taylor *et. al.*, 1978). (Quoted from US ATSDR, 1995) Further data on human exposure is quoted in section 2.4.1.

Information regarding human exposure resulting from direct use (application) of Chlordecone in the Caribbean Islands is not available. However, monitoring data in agricultural soils, crops, freshwater fish, littoral fish and shellfish indicates that human exposure more than 10 years after the use of chlordecone has ceased in Martinique and Guadeloupe, is still possible In soils having received Chlordecone, residues in crop are proportional to soil contamination and may exceed the recommended national residues limits (50 µg/kg to 200 µg/kg). This concerns mainly root vegetables such as radish (max. measured concentration: 0.055 µg/kg), sweet potatoes (max. measured concentration: 0.300 µg/kg), taro root (max. measured concentration: 0.230 µg/kg), but also aerial part of plants, such as sugar cane (max. measured concentration: 0.690 µg/kg), or pineapple (max. measured concentration: 0.160 µg/kg). In addition, workers are directly exposed to contaminated soils. Concentrations in fisheries products (freshwater and estuarine water) have also been found to exceed in some occasions national residues limits up by a factor of 100 (max. measured concentration: 20 mg/kg). National provisions have been taken in order to prohibit fisheries activities in contaminated area (Cabidoche *et. al.*, 2006).

## 2.4 Hazard assessment for endpoints of concern

### 2.4.1 Toxicity

**Toxicokinetics in experimental animals and in man**

The US ATSDR (1995) and EHS 43 (IPCS, 1984) both record that Chlordecone is well absorbed following oral, dermal and inhalation exposure. Toxicokinetic data are mainly available from studies in experimental animals (*e. g.* Blanke *et. al.*, 1978; Boylan *et. al.*, 1979; Cohn *et. al.*, 1978; Egle *et. al.*, 1978; Fujimori *et. al.* 1982a; Guzelian *et. al.*, 1981; Hall *et. al.* 1988; Hewitt *et. al.*, 1986b; Kavlock *et. al.*,1980; Plaa *et. al.*, 1987; Richter *et. al.*, 1979; Shah *et. al.*, 1987; Skalsky *et. al.*, 1980; as reported in IPCS, 1984). Following absorption, it is widely distributed in the body, with accumulation in the liver and to a lesser extent in fat, brain and kidneys, both in experimental animal studies and in humans (as reported in US ATSDR (1995) and EHS 43 (IPCS, 1984). Following administration of a single oral dose to rats at 40 mg/kg body weight, the highest concentrations were found in the adrenal glands and liver, followed by the fat and lung (Egle *et. al.*, 1978, quoted from IPCS, 1984). Chlordecone has been reported to be slowly metabolised *via* reductive biotransformation to Chlordecone alcohol in the rat (Blanke *et. al.*, 1978, as reported in EHS 43). Elimination from the body is slow, with a half-life of the order of several months and Chlordecone disappears more slowly from the liver than from other tissues (Egle *et. al.*, 1978, quoted from IPCS, 1984). Elimination is mainly *via* the faeces, a total of 66% of the dose in the Egle study being removed in the faeces and 2% in the urine in the 84 days following administration (Egle *et. al.*, 1978, quoted from IPCS, 1984).

EHC 43 reports that Chlordecone was detected in high concentrations in the liver (range 13.3-173 mg/kg), whole blood (range 0.6-32 mg/litre), and subcutaneous fat (range 2.2-62 mg/kg) of 32 male workers (Cohn *et. al.*, 1976, adapted from IPCS (1984). In occupationally-exposed workers, serum Chlordecone concentrations ranged from 120 to 2109 µg/litre, and dropped to 37 - 486 µg/litre 6-7 months after exposure had ceased (Adir *et. al.*, 1978, reported in IPCS (1984). The half-life of Chlordecone in these workers was estimated to be 63-148 days. Reductive biotransformation to Chlordecone alcohol has also been reported in humans (Blanke *et. al.*, 1978, as reported in EHS 43). Chlordecone was eliminated, primarily in the faeces, at a mean daily rate of 0.075% of the estimated total store in the body (Cohn *et. al.*, 1976, quoted from IPCS, 1984).

**Toxicity of Chlordecone in animal studies**

Chlordecone is of high acute toxicity in experimental animal studies, with an $LD_{50}$ of approximately 100 mg/kg in the rat and ranging from 65 mg/kg in the rabbit to 250 mg/kg in the dog (taken from IPCS, 1984, Table 2). Acute toxicity effects include tremors indicative of a neurotoxic effect on the nervous and/or musculoskeletal systems, investigated by many authors as reported in US ATSDR (1995). The neurotoxic effects of Chlordecone have been reported in chickens (Naber & Ware, 1965), quail (McFarland & Lacy, 1969), fish (Couch *et. al.*, 1977), hamsters (Martinez *et. al.*, 1976), mice (End *et. al.*, 1979), rats (Epstein, 1978), and man (Martinez *et. al.*, 1978). Acute oral administration of Chlordecone is also associated with reproductive effects (Khera *et. al.*, 1976; Uzodinma *et. al.*, 1984a; Yarbrough *et. al.*, 1981) and hepatotoxicity in some studies (Fujimori *et. al.*, 1983; Mehendale 1977b, 1981b; Teo & Vore 1991) (quoted from US ATSDR (1995).

Repeated exposure to Chlordecone also causes reproductive, neurological, musculoskeletal and liver toxicity at doses as low as 10 mg/kg bw/day, although effects in other organs including kidney, thyroid, adrenals, and testes have also been reported (US ATSDR, 1995, IPCS, 1984). A Lowest-Observed-Adverse-Effect-Level (LOAEL) of 1.17 mg/kg bw/day was recorded in a 3 month feeding study in rats and signs of toxicity included focal necrosis in liver, enlargement of the adrenal gland, tremor, hyperactivity and exaggerated startle response (Cannon and Kimbrough, 1979, as quoted in US ATSDR, 1995). Histopathological changes in the liver, reduction in thyroid follicular size and colloid content and increase in epithelial cell height were reported in a 21-month gavage study in the rat, with a LOAEL of 0.07 mg/kg bw/day in males (Chu *et al*, 1981, as quoted in US ATSDR, 1995). Renal effects (proteinuria and increased severity of glomerulosclerosis) were seen in a 2-year feeding study in rats, with a NOAEL of 0.05 mg/kg/day (Larson *et. al.*, 1979b, as quoted in US ATSDR, 1995). Oral Chlordecone treatment caused decreased spleen and thymus weights, leukocyte counts, natural killer cell activity, and mitogenic responsiveness (EPA 1986c; Smialowicz *et. al.*, 1985; Swanson and Wooley, 1982); decreased natural killer cell activity (Smialowicz *et. al.*, 1985); and significant increase in plaque-forming cells (Chetty *et. al.*, 1993c) (as reported in ATSDR, 1995). The NOAEL was 5 mg/kg bw/day and the LOAEL was 10 mg/kg bw/day.

Hepatocarcinogenicity (hepatocellular carcinoma) of Chlordecone has been demonstrated in rats and mice (males and females) (NCI 1976, Reuber, 1978, 1979, as quoted in IPCS, 1984 and US ATSDR, 1995). Tumours have been observed at doses as low as 1 mg/kg bw/day in the rat and in mice at a dose of 2.6 mg/kg bw/day (NCI, 1976, as quoted in US ATSDR (1995). The International Agency for Research on Cancer (IARC) concluded in 1987 that there was sufficient evidence that Chlordecone is carcinogenic in mice and rats and possibly carcinogenic to humans (Group 2B). Chlordecone is not genotoxic in *in vitro* microbial and mammalian cell gene mutation assays, in a clastogenicity test and in the dominant lethal assay (Mortelmans *et. al.*, 1986; Probst *et. al.*, 1981; Schoeny *et. al.*, 1979, Tong *et. al.* 1981; Williams 1980, Khera *et. al.*, 1976; Simon *et. al.*, 1986, as reported in ATSDR (1995), although it has been reported to interfere with cell-to-cell communication (Tsushimoto *et. al.*, 1982, Caldwell and Loch-Caruso, 1992, as reported in US ATSDR (1995), suggests that it produces liver tumours by an epigenetic, tumour-promoting mechanism involving both hepatic toxicity and hypertrophy, including cytochrome P-450 induction.

Oral administration of Chlordecone to animals causes decreased fertility or fecundity and litter size, reduced sperm count and testicular atrophy (Khera *et. al.*, 1976; Linder *et. al.* 1983; Uzodinma *et. al.*, 1984a; Yarbrough *et. al.* 1981, as reported in US ATSDR (1995). A LOAEL of 0.83 mg/kg/day was recorded for sperm effects in a 90 day feeding study in rats, while effects on seminal vesicles and prostate were apparent at 1.67 mg/kg bw/day (Linder *et. al.*, 1983) (Quoted from US ATSDR (1995).

Chlordecone is also a developmental toxicant. As reported in US ATSDR (1995) and EHC 43 (IPCS, 1984), gestational exposure of rats and mice to low doses of Chlordecone resulted in increased stillbirths and decreased postnatal viability, reduced fetal or neonatal weight and/or skeletal ossification and a low incidence of malformations such as renal pelvis dilatation, undescended testes, enlarged cerebral ventricles, clubfoot, fused vertebrae or ribs, and encephalocele. Chlordecone administered at levels of 2, 6, and 10 mg/kg bw/day to rats and 2, 4, 8, and 12 mg/kg body weight per day to mice on days 7 - 16 of gestation caused 19% maternal mortality in rats at the highest dose and fetuses exhibited reduced weight, reduced degree of ossification, oedema, undescended testes, enlarged renal pelvis, and enlarged cerebral ventricles. (Chernoff & Rogers, 1976, as reported in IPCS, 1984). Lower dose levels induced reductions in fetal weight and degree of ossification. Male rats born to treated dams did not show any reproductive impairment. The reproductive performance of mice fed 0, 10, 30, or 37.5 mg Chlordecone/kg diet was impaired in terms of offspring and litter size (Huber, 1965, as reported in IPCS, 1984). No litters were produced by females fed 40 mg/kg, but litter production did resume within 7 weeks following withdrawal of the Chlordecone, although litters were still smaller than those of untreated controls (quoted from IPCS (1984)). Anovulation and persistent vaginal estrus were observed in female mice given Chlordecone at a dose level of 2 mg/kg bw/day) (Swartz *et. al.*, 1988, as quoted in US ATSDR, 1995), and similar changes were observed in female offspring of maternal rats given 15 mg/kg/day of Chlordecone on gestation days 14-20 (Gellert and Wilson, 1979, as

quoted in US ATSDR, 1995), although no effects on vaginal patency or fertility were observed in female offspring of maternal mice given 20 mg/kg/day during gestation days 8-12 or 14-18 (Gray and Kavlock 1984, as quoted in US ATSDR, 1995).

**Toxicity of Chlordecone in humans**

Available human data support the conclusion that Chlordecone has a similar toxicity profile in humans to that seen in experimental animal studies. As reported in US ATSDR (1995), a high incidence of nervous system toxicity was seen in a single group of workers exposed to Chlordecone during its manufacture (Cannon *et. al.*, 1978; Martinez *et. al.*, 1978; Sanbom *et. al.*, 1979; Taylor 1982, 1985; Taylor *et. al.*, 1978, taken from US ATSDR (1995)). Exposure of this population occurred by a combination of inhalation, oral, and dermal exposures, although the dermal route was suggested to be the predominant route. The toxicity was manifested as tremors, visual difficulties, muscle weakness, gait ataxia, in coordination, headache, and increased cerebrospinal fluid pressure (US ATSDR (1995). Prolonged exposure to high concentrations of Chlordecone in the workplace has been suggested to cause oligospermia and decreased sperm motility among male workers, although fertility was not impaired (Guzelian 1982a; Taylor 1982, 1985; Taylor *et. al.*, 1978, taken from US ATSDR (1995). A correlation between blood levels, atmospheric levels and sperm effects has however been difficult to prove conclusively (US ATSDR (1995). Epidemiological evidence for carcinogenicity of Chlordecone in exposed humans following inhalation exposure to Chlordecone is extremely limited (US ATSDR, 1995, IPCS, 1984). Liver biopsy samples taken from 12 workers with hepatomegaly resulting from intermediate- or chronic-duration exposures to high concentrations of Chlordecone showed no evidence of cancer (Guzelian *et. al.*, 1980, taken from US ATSDR (1995). However, conclusions from this study are limited by the very small number of workers sampled (US ATSDR, 1995)

**Effects on endocrine systems**

The effects of Chlordecone on reproduction indicate that this pesticide has effects on endocrine systems. It has been evaluated under the EU-Strategy for Endocrine Disrupters[9] and has been placed in category 1 (evidence of endocrine-disrupting activity in at least one species using intact animals), in the priority list of chemicals established under the EU-Strategy. This categorisation is based on evidence of ED activity in a number of experimental systems including the mouse uterotropic assay, increased uterine weight in rats given multiple injections of Chlordecone postnatally and receptor binding assays, indicative of an oestrogenic effect (as reported in BKH report, 2000, US ATSDR, 1995).

**Conclusion on effects assessment and toxicity of Chlordecone**

Chlordecone is readily absorbed into the body and accumulates following prolonged exposure. The pesticide is both acutely and chronically toxic, producing neurotoxicity, immunotoxicity, reproductive, musculoskeletal and liver toxicity at doses between 1 - 10 mg/kg bw/day in experimental animal studies. Liver cancer was induced in rats at a dose of 1 mg/kg body weight per day and in mice at a dose of 2.6 mg/kg bw/day, and reproductive effects are seen at similar dose levels. The International Agency for Research on Cancer has classified Chlordecone as a possible human carcinogen (IARC group 2B).

Table 2.3 summarises the outcomes of key toxicological studies on Chlordecone, including the NOAEL/LOAEL derived in each study. The studies included in this Table have been selected from the very large database on toxicological studies on Chlordecone, on the basis of the importance of the endpoint investigated (*e. g.* reproductive toxicity, carcinogenicity, other key target organ toxicity), robustness of the reported studies and the dose level (NOAEL/LOAEL) at which effects were reported. These studies were considered to be particularly relevant for characterisation of the toxicological risks of these compounds, and some of these studies have been used by US ATSDR to define Minimal Risk Levels (MRLs) for Chlordecone (US ATSDR, 1995).

**Table 2.3 Summary of key toxicological studies on Chlordecone.**

| Species | Study type | Effect | LOAEL/NOAEL (mg/kg bw/day) | Reference |
|---|---|---|---|---|
| Rat Fischer 344 | Short-term/acute toxicity 10 day repeat dose gavage study | 65% loss in body weight, changes in clinical chemistry parameters | 10 mg/kg bw/day (LOAEL)<br>5 mg/kg bw/day (NOAEL) | EPA, 1986 (as quoted in US ATSDR, 1995). |
| Rat Fischer 344 | Short-term/acute toxicity 10 day repeat dose | Reductions in spleen and thymus weights, numbers of neutrophils, and natural killer cell activity, secondary to generalized toxicity | 10 mg/kg bw/day (LOAEL)<br>5 mg/kg bw/day | EPA, 1986· Smialowicz *et. al.*, 1985, (as quoted in US ATSDR, 1995). |

[9] http://europa.eu.int/comm/environment/endocrine/strategy/substances_en.htm

| Species | Study type | Effect | LOAEL/NOAEL (mg/kg bw/day) | Reference |
|---|---|---|---|---|
| | gavage study | | (NOAEL) | |
| Rat Fischer 344 | Short-term/acute toxicity 10 day repeat dose gavage study | Increased startle response | 2.5 mg/kg bw/day (LOAEL) 1.25 mg/kg bw/day (NOAEL) | EPA, 1986c (as quoted in US ATSDR, 1995). |
| Rat (Sherman) | 3 month feeding study | Focal necrosis in liver, enlargement of the adrenal gland, hyperplasia and hypertrophy of cortical cells, tremor. hyperactivity, exaggerated startle response | 1.17 mg/kg bw/day (LOAEL) | Cannon and Kimbrough 1979 (as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Rat, Wistar | 2 year feeding study | Renal effects (proteinuria and increased severity of glomerulosclerosis) | 0.25 mg/kg bw/day. (LOAEL) 0.05 mg/kg bw/day (NOAEL) | Larson *et. al.*, 1979b (as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Rat Sprague-Dawley | 21 month gavage study | Histopathological changes in liver, reduction in follicular size and colloid content and increase in epithelial cell height in thyroid | 0.07 mg/kg bw/day (LOAEL), in males | Chu *et. al.*, 1981(as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Rat, Wistar | 3 month feeding study | Testicular atrophy | 0.5 mg/kg bw/day. (LOAEL) 0.25 mg/kg bw/day (NOAEL) | Larson *et. al.* , 1979b (as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Rat (Osborne-Mendel) and mouse (B3C6F1) | 80 week feeding study | Hepatocellular adenoma and carcinoma | 1.2 mg/kg bw/day. (LOAEL, rat) and 2.6 mg/kg bw/day (LOAEL, mouse) | NCI, 1976, Reuber, 1978, 1979(as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Rat | Multiple injections of Chlordecone to neonatal rats | Uterotrophic response - uterine weights increased in a dose-related manner | 10 mg/kg bw/day (LOAEL, Gellert, 1978) $\leq$6 mg/kg bw/day (LOAEL, Hammond *et. al.*, 1979 [1]) | Gellert 1978 Hammond *et. al.*, 1979 (as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Rat, Hotzman strain, ovarectomized immature females | Rats injected x 3 with 0 - 45 mg/kg bw/day Chlordecone $\pm$ 0.01, 0.1, 1 or 10 mg/kg bw/day estradiol benzoate | Uterotrophic response. Effect was additive to that of estradiol benzoate over the dose range studied | Dose of 20 mg/kg bw/day Chlordecone appeared to be threshold for embryo implantation functions | Johnson, 1996 |
| Rat | 90-day feeding study | Decrease in sperm motility and viability, decreased sperm, decrease in the weight of seminal vesicles and prostate | 0.83 mg/kg bw/day LOAEL for sperm effects 1.67 mg/kg bw/day LOAEL for effects on seminal vesicles and prostate | Linder *et. al.*, 1983 (as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Mouse, Balbc | 130 day feeding study | 8% decrease in litter size and 19% increase in pair-days to litter (constant oestrus) | 1.3 mg/kg bw/day. (LOAEL) | Huber, 1965 (as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Rats and mice | 2, 6, and 10 mg/kg bw/day by gavage to rats and 2, 4, 8, and 12 mg/kg bw/day to mice on days 7 - 16 of gestation. | Reduced foetal weight, reduced degree of ossification, oedema. undescended testes, enlarged renal pelvis, and enlarged cerebral ventricles. Reductions in fetal weight and degree of ossification at lower dose levels. Maternal mortality at top dose. In the mouse, fetotoxicity was observed only at the highest dose level and consisted of increased fetal mortality and clubfoot. | 2 mg/kg bw/day. (LOAEL, rat) | Chernoff & Rogers, 1976). (as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Balbc mice | 160 day feeding study | Increased ovulation, persistent oestrus | 2 mg/kg bw/day. (LOAEL) | Swartz *et. al.*, 1988 (as quoted in IPCS, 1984 and US ATSDR, 1995). |
| Rat | Reproductive toxicity | Increased ovulation, persistent oestrus in female offspring of maternal rats given Chlordecone on gestation days 14-20 | 15 mg/kg/day (LOAEL) | Gellert and Wilson, 1979, as quoted in US ATSDR, 1995) |

| Species | Study type | Effect | LOAEL/NOAEL (mg/kg bw/day) | Reference |
|---|---|---|---|---|
| Humans | Occupational exposure | Histories of tremors, unfounded nervousness or anxiety, and visual difficulties. Also skin rashes | Mean blood levels of Chlordecone in workers reporting adverse effects were 2.53 ppm<br>Skin rashes reported in workers with blood Chlordecone levels in excess of 2 µg/L | Cannon *et. al.*, 1978 (as quoted in IPCS, 1984 and US ATSDR, 1995). |

### 2.4.2 Ecotoxicity

A summary of results of aquatic ecotoxicity tests with Chlordecone from the Ecotox database (US EPA, 2006) is given in Table 2.4.

In addition to this, the EHC 43 (IPCS, 1984), summarised a series of experiments investigating the bioavailability of Chlordecone, noting that it is strongly adsorbed on sediment. Exposure of aquatic organisms is therefore partly *via* the water phase and partly *via* sediment. D'Asaro & Wilkes (1982) examined the effects of sediments previously exposed to Chlordecone at a known concentration, and of James River sediments contaminated with Chlordecone, on an estuarine community established in aquaria supplied with non-filtered sea water. Mysid shrimps showed a dose-related mortality rate, when exposed to sediments previously equilibrated at 0.1, 1.0, or 10 µg Chlordecone/L. Mysids were not affected by James River sediment. Put concentration in sediments, if available Oysters showed dose-dependent reduced shell growth when exposed to Chlordecone-equilibrated sediments, and also responded adversely to river sediment. Lugworms *Arenicola cristata* died after 28 days of treatment with sediment exposed to 10 µg Chlordecone/L, though numbers were not affected by lower doses. Both lugworms and oysters concentrated Chlordecone from the sediment. (Quoted from EHC 43, (IPCS, 1984)).

**Table 2.4 Summary of key ecotoxicological studies on Chlordecone.**

| Taxonomic group and species | End point | Duration | Result mg/L | Reference [1] |
|---|---|---|---|---|
| Algae<br>*Chlorococcum sp., Dunaliella tertiolecta, Nitzschia sp., Thalassiosira pseudonana* | $EC_{50}$<br>growth inhibition | 7 days | 0.35 - 0.60<br>(formulation) | Walsh *et. al.*, 1977 |
| Algae<br>*Chlorococcum sp., Dunaliella tertiolecta, Nitzschia sp., Thalassiosira pseudonana* | $EC_{50}$<br>growth inhibition | 7 days | 350 – 600<br>(formulation) | Hansen *et. al.*, 1977 |
| Crustaceans<br>*Daphnia magna* | $EC_{50}$<br>immobility | 48 hours | 0.120 - 0.690 | Barera & Adams, 1983; Adams & Heidolph, 1985; Ziegenfuss *et. al.*, 1986 |
| Crustaceans<br>*Americamysis bahia, Callinectes sapidus, Palaemonetes pugio* | $LC_{50}$ | 96 hours | 0.01 - 0.210 | Nimmo *et. al.*, 1977, 1981; Hansen *et. al.*, 1977; Schimmel, 1977; US EPA, 1976 |
| Crustacean<br>*Daphnia magna* | NOEC<br>reproduction | 21 days | 0.0283 | McKee & Knowles, 1986 |
| Crustacean<br>*Daphnia magna* | NOEC<br>growth | 21 days | 0.025 | Adams & Heidolph, 1985 |
| Crustacean<br>*Americamysis bahia* | MATC<br>growth | 28 days | 0.000026 - 0.00034 | Nimmo *et. al.*, 1981 |
| Insect<br>*Chironomus tentans* | $LC_{50}$ | 48 hours | 0.17 - 2.3 | Adams *et. al.*, 1985; Ziegenfuss *et. al.*, 1986 |
| Fish<br>9 species | $LC_{50}$ | 96 hours, flow through | 0.0066 - 0.512 | Roberts & Bendl, 1982; Roberts & Fisher, 1985; Schimmel, 1977; Hansen *et. al.*, 1977; Mallat & Barron, 1988; Buckler *et. al.*, 1981 |
| Insect<br>*Chironomus tentans* | NOEC<br>development | 14 days | 17.9 mg/kg sediment | Adams *et. al.*, 1985 |

1: All are as quoted in Ecotox, US EPA 2006

In a publication from SETAC a collation of critical tissue residues (CTR) was presented and evaluated (Jarvinen *et. al.*, 1999). The database contains 32 entries for Chlordecone, with data originating from different studies (see Table 2.5). Some of the tissue residues were from studies where no effects were observed, so they may not represent the real CTR. Critical tissue residue values obtained in studies where effects were identified represent 15 CTR values for three fish species. For fathead minnow two studies are available with values of 1.7 and of 3.8-5.4 mg/kg ww. For sheepshead minnow 12 CTRs are available, ranging from 0.13 to 17 mg/kg ww with an average of 5.9 mg/kg ww. Furthermore, one CTR of 2.7 mg/kg ww for spot is available.

**Conclusion**

In summary, Chlordecone is very toxic to aquatic organisms. The most sensitive group is the invertebrates, which is not surprising for a substance with insecticidal properties. Even if the lowest effect concentration (0.000026 mg/L) was considered to be an outlier, the lowest effect concentrations would be well below 1 mg/L with the results of short term tests (mortality) in the range of 0.01 to 0.69 mg/L and those of long term tests (reproduction and growth) at 0.0025 and 0.0028 mg/L.

**Table 2.5 Collation of critical tissue residues (CTR)**

| Species | Life Stage | Exprte | Expo of Concentration | Results □g/g (wet) ) | effect |
|---|---|---|---|---|---|
| Cladoceran, Daphnia magna (Fw) | 1st instar | Water | 175 ng/L | 0.133 | Survival, Reproduction - No effect |
| Grass shrimp, Palaemonetes pugio (Sw) | 0.09g | Water; Diet | 0.04 µg/L; 0.118 µg/g (wet wt) | 0.147 | Growth - No effect |
| Blue crab, Callinectes sapidus (Sw) | Juvenile | Diet | 2.26 - 2.50 µg/g (wet wt) | 2.54 - 4.61 | Survival, Growth - No effect |
| Fathead minnow, Pimephales promelas (Fw) | Larvae-Adult | Water | 3.1 µg/L | 3.8 - 5.4 | Survival, Growth - Reduced |
| Fathead minnow, Pimephales promelas (Fw) | Larvae-Adult | Water | 1.2 µg/L | 2.6 | Survival, Growth - No effect |
| Fathead minnow, Pimephales promelas (Fw) | Embryo, 2nd generation | Water; Adult fish | 0.31 µg/L; 0.21-0.38 µg/g | 1.7 | Survival (hatchability) - Reduced |
| Fathead minnow, Pimephales promelas (Fw) | Embryo, 2nd generation | Water; Adult fish | 0.17 µg/L; 0.17-0.46 µg/g | 0.26 | Survival - No effect |
| Fathead minnow, Pimephales promelas (Fw) | Larvae, 2nd generation | Water; Adult fish | 0.31 µg/L; 0.21- 0.38 µg/g | 0.50 | Survival, Growth - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Adult | Water | 0.8 µg/L | 2.5 - 3.6 | Survival - Reduced 22% |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Adult | Water | 1.9 µg/L | 11 - 12 | Survival - Reduced 80% |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Adult | Water | 7.8 µg/L | 17 | Survival - Reduced 100% |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Adult | Water | 0.16 µg/L | 0.65 - 0.90 | Survival - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo | Adult fish | 11-12 µg/g | 11 | Survival - Reduced 25% |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo | Adult fish | 2.5 - 3.6 µg/g | 4.7 | Survival - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Larvae-Juvenile | Water; Adult fish | 1.9 µg/L; 11-12 µg/g | 8.4 | Survival - Reduced 63% |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Larvae-Juvenile | Water | 2.0 µg/L | 7.8 | Survival - Reduced 40% |

| Species | Life Stage | Exprte | Expo of Concentration | Results □g/g (wet)_) | effect |
|---|---|---|---|---|---|
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Larvae-Juvenile | Water | 0.8 µg/L | 2.0 | Survival - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Larvae-Juvenile | Adult fish | 11-12 µg/g | 0.13 | Growth - Reduced |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Larvae-Juvenile | Water | 0.08 µg/L | 1.1 | Growth – Reduced |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo-Adult | Water | 0.78 µg/L | 5, 6.8* | Survival - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo-Adult | Water | 0.39 µg/L | 2.2, 3* | Growth – Reduced |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo-Adult | Water | 0.12 µg/L | 0.86, 1.2* | Growth - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo-Adult | Water | 0.78 µg/L | 5, 6.8* | Reproduction – Reduced |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo-Adult | Water | 0.39 µg/L | 2.2, 3* | Reproduction - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo, 2nd generation | Adult Fish + Water | 0.78 µg/L | 2.3 | Survival – Reduced |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Embryo, 2nd generation | Adult Fish + Water | 0.39 µg/L | 1.3 | Survival - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Fry, 2nd generation | Adult Fish + Water | 0.78 µg/L | 2.3 | Survival - No effect |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Fry, 2nd generation | Adult Fish + Water | 0.12 µg/L | 0.41 | Growth – Reduced |
| Sheepshead minnow, Cyprinodon variegatus (Sw) | Fry, 2nd generation | Adult Fish + Water | 0.074 µg/L | 0.30 | Growth - No effect |
| Spot, Leiostomus xanthurus (Sw) | Juvenile | Diet | 3.3 µg/g (wet wt) | 2.7 | Survival – Reduced |
| Spot, Leiostomus xanthurus (Sw) | Juvenile | Diet | 3.3 µg/g (wet wt) | 0.7 | Survival - No effect |
| Spot, Leiostomus xanthurus (Sw) | Juvenile | Water; Diet | 0.04 µg/L; 0.101 µg/g (wet wt) | 0.144 | Growth, No effect |

## 3 Synthesis of the information

Chlordecone is a synthetic chlorinated organic compound, which has mainly been used as an agricultural pesticide. It is closely related chemically to Mirex, a pesticide which is already listed in Annex A of the Stockholm Convention. Chlordecone is already listed in Annex I of the UNECE Protocol on POPs.

According to available data, Chlordecone can be considered to be highly persistent in the environment. Chlordecone is not expected to hydrolyse or biodegrade in aquatic environments, nor in soil. Direct photodegradation is not significant. Chlordecone does not volatilise to any significant extent.

With BCF-values in algae up to 6,000, in invertebrates up to 21,600 and in fish up to 60,200 and documented examples of biomagnification, Chlordecone is considered to have a high potential for bioaccumulation and biomagnification.

Concerning the potential for causing adverse effects, there is a convincing set of data. Chlordecone is readily absorbed into the body and accumulates following prolonged exposure. It is both acutely and chronically toxic, producing neurotoxicity, immunotoxicity, reproductive, musculoskeletal and liver toxicity at doses between 1 - 10 mg/kg bw/day in experimental animal studies. Liver cancer was induced in rats at a dose of 1 mg/kg body weight per day, and reproductive effects are seen at similar dose levels. The International Agency for Research on Cancer has classified Chlordecone as a possible human carcinogen (IARC group 2B). Moreover, Chlordecone is very toxic to aquatic organisms, most sensitive group being the invertebrates.

The available data on Chlordecone are not fully conclusive when it comes to long-range atmospheric transport in gaseous form. It should be noted that atmospheric transport of particle-bound substances and transport of sediment particles in ocean currents as well as biotic transport could also contribute to long-range environmental transport of Chlordecone.

Due to lack of monitoring data on Chlordecone, the assessment of the potential for long-range transport of Chlordecone is based on physico-chemical properties and especially, on modelling data. While the first of these two approaches may seem somehow insufficient, the modelling data state clearly Chlordecone's LRET potential.

Based on the available data, Chlordecone should be considered as a POP warranting global action.

Production and use of Chlordecone has ceased over the last decades in developed countries, but it is assumed that it can still be produced or used as an agricultural pesticide in some developing countries. If it is still used as pesticide, it will be directly released to the environment. Moreover, the high persistency of the substance has caused high contamination of soil and waters in the areas where it has been used and these contaminated sites can serve as a source of pollution for long times.

## 4 Concluding statement

It has been demonstrated that Chlordecone meets all the criteria laid down in Annex D of the Stockholm Convention. Moreover, it is chemically very similar to Mirex, an organochlorine pesticide which is already listed in the Stockholm Convention. It is very persistent in the environment and has a great potential for bioaccumulation and in addition there is clear evidence of its biomagnification. While there is no monitoring data from areas remote from sources, the physical and chemical properties, as well as the modelling results, suggest that Chlordecone can be transported long distances bound to particles in air and water, and possibly through coupled transport between these two compartments. Chlordecone is associated with a wide range of harmful effects on both mammals and aquatic organisms.

As Chlordecone can travel in the atmosphere far from its sources, neither a single country nor group of countries alone can abate the pollution caused by this substance. Regional action has already been considered necessary and Chlordecone is totally banned under the UNECE Convention on Long-range Transboundary Air Pollution Protocol on Persistent Organic Pollutants. Although the production and use of Chlordecone seems to be ceased in most countries, its reintroduction remains possible. This could lead to increased releases and levels in the environment.

Based on the available evidence, Chlordecone is likely as a result of its long-range environmental transport to lead to significant adverse human health and environmental effects such that global action is warranted.

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