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An Assessment Report on:
DDT-Aldrin-Dieldrin-Endrin-Chlordane
Heptachlor-Hexachlorobenzene
Mirex-Toxaphene
Polychlorinated Biphenyls
Dioxins and Furans

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For:

The International Programme on Chemical Safety (IPCS)
within the framework of the
Inter-Organization Programme for the Sound Management of Chemicals (IOMC)

The International Programme on Chemical Safety (IPCS) WHO ILO UNEP

IOMC INTER-ORGANIZATION PROGRAMME FOR THE SOUND MANAGEMENT OF CHEMICALS

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The International Programme on Chemical Safety (IPCS) is a joint venture of the United Nations Environment Programme, the International Labour Organisation, and the World Health Organization. The main objective of the IPCS is to carry out and disseminate evaluations of the effects of chemicals on human health and the quality of the environment. Supporting activities include the development of epidemiological, experimental laboratory, and risk-assessment methods that could produce internationally comparable results, and the development of human resources in the field of chemical

safety. Other activities carried out by the IPCS include the development of know-how for coping with chemical accidents, strengthening capabilities for prevention of an response to chemical accidents and their follow-up, coordination of laboratory testing and epidemiological studies, and promotion of research on the mechanisms of the biological action of chemicals.

The Inter-Organization Programme for the Sound Management of Chemicals (IOMC), was established in 1995 by UNEP, ILO, FAO, WHO, UNIDO, and OECD (Participating Institutions), following recommendations made by the 1992 UN Conference on Environment and Development to strengthen cooperation and increase international coordination in the field of chemical safety. The purpose of the IOMC is to promote coordination of the policies and activities pursued by the Participating Organizations, jointly or separately, to achieve the sound management of chemicals in relation to human health and the environment.

PREFACE

At its ninth meeting in May 1995, the UNEP Governing Council adopted Decision 18/32 concerning Persistent Organic Pollutants. The decision invites the Inter-Organization Programme on the Sound Management of Chemicals (IOMC), working with the International Programme on Chemical Safety (IPCS) and the Intergovernmental Forum on Chemical Safety (IFCS) to undertake an assessment process addressing persistent organic pollutants (POPs). This process is to initially begin with 12 specific compounds and should consolidate existing information on the relevant chemistry and toxicology, transport and disposition, as well as the availability and costs of substitutes to these substances. The effort will also assess realistic response strategies, policies, and mechanisms for reducing and/or eliminating emissions, discharges, and other losses of these substances. This information will serve as the basis for recommendations to be developed by the IFCS on potential international actions to be considered at the session of the UNEP Governing Council and the World Health Assembly in 1997.

IPCS, in consultation with the organizations participating in the IOMC, has proceeded with the initial phase of the work. The initial effort aims to compile the existing information on the chemistry, toxicology, relevant transport pathways and the origin, transport and disposition of the substances concerned and additionally, reference briefly what information is available on the costs and benefits associated with substitutes, and the socio-economic aspects of the issue. The effort builds on ongoing activities including the substantial work in progress under the Long-Range Transboundary Air Pollution Convention and the 1995 International Expert Meeting on POPs sponsored by Canada and the Philippines.

This assessment report is a shortened version of a companion document "A Review of the Persistent Organic Pollutants: DDT, Aldrin, Dieldrin, Endrin, Chlordane, Heptachlor, Hexachlorobenzene, Mirex, Toxaphene, Polychlorinated Biphenyls, Dioxins and Furans" (PCS 95.39). This assessment report presents a distillation of the critical issues and facts but, for ease of reading, references have been omitted. The reader who desires more information and references should consult the larger review document cited above which is available upon request.

A draft version of this assessment report was submitted as an information document to the Intergovernmental Conference to Adopt a Global Programme of Action for the Protection of the Marine Environment from Land-Based Activities, Washington, D.C., 23 October - 3 November 1995. This final version of the assessment report is being submitted as a background document for the second meeting of the Intersessional Group of the IFCS to be held in March 1996. This document will serve as a basis for development of a work plan to complete the assessment process called for in the UNEP Governing Council Decision.

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1. INTRODUCTION

Persistent organic pollutants (POPs) are organic compounds that, to a varying degree, resist photolytic, biological and chemical degradation. POPs are often halogenated and characterised by low water solubility and high lipid solubility, leading to their bioaccumulation in fatty tissues. They are also semi-volatile, enabling them to move long distances in the atmosphere before deposition occurs.

Although many different forms of POPs may exist, both natural and anthropogenic, POPs which are noted for their persistence and bioaccumulative characteristics include many of the first generation organochlorine insecticides such as dieldrin, DDT, toxaphene and chlordane and several industrial chemical products or byproducts including polychlorinated biphenyls (PCBs), dibenzo-p-dioxins (dioxins) and dibenzo-p-furans (furans). Many of these compounds have been or continue to be used in large quantities and, due to their environmental persistence, have the ability to bioaccumulate and biomagnify. Some of these compounds such as PCBs, may persist in the environment for periods of years and may bioconcentrate by factors of up to 70,000 fold.

POPs are also noted for their semi-volatility; that property of their physico-chemical characteristics that permit these compounds to occur either in the vapour phase or adsorbed on atmospheric particles, thereby facilitating their long range transport through the atmosphere.

These properties of unusual persistence and semi-volatility, coupled with other characteristics, have resulted in the presence of compounds such as PCBs all over the world, even in regions where they have never been used. POPs are ubiquitous. They have been measured on every continent, at sites

representing every major climatic zone and geographic sector throughout the world. These include remote regions such as the open oceans, the deserts, the Arctic and the Antarctic, where no significant local sources exist and the only reasonable explanation for their presence is long-range transport from other parts of the globe. PCBs have been reported in air, in all areas of the world, at concentrations up to 15ng/m3; in industrialized areas, concentrations may be several orders of magnitude greater. PCBs have also been reported in rain and snow.

POPs are represented by two important subgroups including both the polycyclic aromatic hydrocarbons and some halogenated hydrocarbons. This latter group includes several organochlorines which, historically, have proven to be most resistant to degradation and which have had wide production, use and release characteristics. These chlorinated derivatives are generally the most persistent of all the halogenated hydrocarbons. In general, it is known that the more highly chlorinated biphenyls tend to accumulate to a greater extent than the less chlorinated PCBs; similarly, metabolism and excretion is also more rapid for the less chlorinated PCBs than for the highly chlorinated biphenyls.

Humans can be exposed to POPs through diet, occupational accidents and the environment (including indoor). Exposure to POPs, either acute or chronic, can be associated with a wide range of adverse health effects, including illness and death.

Laboratory investigations and environmental impact studies in the wild have implicated POPs in endocrine disruption, reproductive and immune dysfunction, neurobehavioural and disorders and cancer. More recently some POPs have also been implicated in reduced immunity in infants and children, and the concomitant increase in infection, also with developmental abnormalities, neurobehavioural impairment and cancer and tumour induction or promotion. Some POPs are also being considered as a potentially important risk factor in the etiology of human breast cancer by some authors.

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2. PROPERTIES AND ENVIRONMENTAL BEHAVIOUR OF POPs

The behaviour and fate of chemicals in the environment is determined by their chemical and physical properties and by the nature of the environment. The chemical and physical properties are determined by the structure of the molecule and the nature of the atoms present in the molecule. Depending on the structure of the molecule, these physical and chemical properties span a large range of values. Compounds may be of very low persistence, of low toxicity and be immobile. These compounds are unlikely to present a risk to the environment or to human health. At the other end of the scale are those compounds that are persistent, mobile and toxic and it is this range of the distribution where the toxic and lipophilic POPs are found. Environmental behaviour and exposure are strongly related. Thus, the risk of exposure to a substance will be much lower if the substance is not persistent and the risk, if any, will be localized unless the substance has properties which allow its movement to distant locations.

It must be recognized that relatively few substances possess the necessary properties to make them POPs. In fact, if the range of these properties were presented as a distribution, only those compounds at the extreme ends of the distribution would express the degree of persistence, mobility and toxicity to rank them as POPs (Figure 2).

Some substances may be very persistent in the environment (i.e., with half-lives (t½) greater than 6 months). The nature of this persistence needs to be clarified - it is the length of time

the compound will remain in the environment before being broken down or degraded into other and less hazardous substances. Dissipation is the disappearance of a substance and is a combination of at least two processes, degradation and mobility. It is not an appropriate measure of persistence as mobility may merely result in the substance being transported to other locations where , if critical concentrations are achieved, harmful effects may occur.

One important property of POPs is that of semi-volatility. This property confers a degree of mobility through the atmosphere that is sufficient to allow relatively great amounts to enter the atmosphere and be transported over long distances. This moderate volatility does not result in the substance remaining permanently in the atmosphere where it would present little direct risk to humans and organisms in environment. Thus, these substances may volatilize from hot regions but will condense and tend to remain in colder regions. Substances with this property are usually highly halogenated, have a molecular weight of 200 to 500 and a vapour pressure lower than 1000 Pa.

In order to concentrate in organisms in the environment, POPs must also possess a property that results in their movement into organisms. This property is lipophilicity or a tendency to preferentially dissolve in fats and lipids, rather than water. High lipophilicity results in the substance bioconcentrating from the surrounding medium into the organism. Combined with environmental persistence and a resistance to biological degradation, lipophilicity also results in biomagnification through the food chain. Biomagnification results in much greater exposures in organisms at the top of the food chain.

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3. CHEMISTRY AND TOXICOLOGY.CHEMISTRY AND TOXICOLOGY

3.1 Chemstry

POPs are, by definition, organic compounds that are highly resistant to degradation by biological, photolytic or chemical means. POPs are often halogenated and most often chlorinated. The carbon-chlorine bond is very stable towards hydrolysis and, the greater the number of chlorine substitutions and/or functional groups, the greater the resistance to biological and photolytic degradation. Chlorine

attached to an aromatic (benzene) ring is more stable to hydrolysis than chlorine in aliphatic structures. As a result, chlorinated POPs are typically ring structures with a chain or branched chain framework. By virtue of their high degree of halogenation, POPs have very low water solubility and high lipid solubility leading to their propensity to pass readily through the phospholipid structure of biological membranes and accumulate in fat deposits.

Halogenated hydrocarbons are a major group of POPs and, of these, the organochlorines are by far the most important group. Included in this class of organohalogens are dioxins and furans, PCBs, hexachlorobenzene, mirex, toxaphene, heptachlor, chlordane and DDT. These substances are characterized by their low water solubility and high lipid solubility and, like many POPs, are noted for their environmental persistence, long half-lives and their potential to bioaccumulate and biomagnify in organisms once dispersed into the environment.

Although some natural sources of organochlorines are known to exist, most POPs originate almost entirely from anthropogenic sources associated largely with the manufacture, use and disposition of certain organic chemicals. In contrast, HCB, dioxins and furans are formed unintentionally in a wide range of manufacturing and combustion processes.

As pointed out above, POPs are typically semi-volatile compounds, a characteristic that favours the long-range transport of these chemicals. They can thus move over great distances through the atmosphere. Volatilisation may occur from plant and soil surfaces following application of POPs used as pesticides.

Halogenated, and particularly chlorinated organic compounds have become entrenched in contemporary society, being utilized by the chemical industry in the production of a broad array of products ranging from polyvinyl chloride (millions of tonnes per year) to solvents (several hundreds of thousands of tonnes) to pesticides (tens of thousands of tonnes) and speciality chemicals and pharmaceuticals (thousands of tonnes down to kilogram quantities). In addition, both anthropogenic and non-anthropogenic sources also lead to production of undesirable by-products and emissions often characterized by their persistence and resistance to breakdown (such as chlorinated dioxins). As noted above, organochlorine compounds have a range of physico-chemical properties. In the environment, organochlorines can be transformed by a variety of microbial, chemical and photochemical processes. The efficiency of these environmental processes are largely dependent on the physico-chemical properties of the specific compound and characteristics of the receiving environment.

Cyclic, aromatic, cyclodiene-type and cyclobornane type chlorinated hydrocarbon compounds, such as some chlorinated pesticides, with molecular weights greater than 236 g/mol have been noted for their ability to accumulate in biological tissues, and to particularly concentrate in organisms that occupy positions in the upper trophic levels; not surprisingly, these compounds are also known for their persistence in the environment. Compounds included in this class often share many physico-chemical characteristics and include some of the earliest organochlorine pesticides such as DDT, chlordane, lindane, heptachlor, dieldrin, aldrin, toxaphene, mirex and chlordecone. Conversely, the lower molecular weight chlorinated hydrocarbons (less than 236 g/mol) may include a number of alkanes and alkenes

(dichloromethane, chloropicrin, chloroform) and are often associated with little acute toxicity, reversible toxicological effects and relatively short environmental and biological half-lives. Bioavailability, that proportion of the total concentration of a chemical that is available for uptake by a particular organism, is controlled by a combination of chemical properties of the compound including the ambient environment and the morphological, biochemical and physiological attributes of the organism itself.

Generally, excretion of organic pollutants is facilitated through the metabolic conversion to more polar forms. Because of their resistance to degradation and breakdown, the POPs are not easily excreted and those pollutants (e.g. toxaphene, PCBs etc.) most resistant to metabolism and disposition tend to accumulate in organisms and through the food chain. Notably, some organic pollutants may also be converted to more persistent metabolites than the parent compound, as is the case with the metabolic conversion of DDT to DDE. Similarly, the rapid metabolic conversion of aldrin to its extremely environmentally persistent metabolite dieldrin, is also noteworthy.

3.2 Toxicology

3.2.1 Environment

If analysed for in tissues or environmental samples, some POPs will almost always be found. As is the case with many environmental pollutants, it is most difficult to establish causality of illness or disease that is directly attributable to exposure to a specific persistent organic pollutant or group of POPs. This difficulty is further underscored by the fact that POPs rarely occur as single compounds and, individual field studies are frequently insufficient to provide compelling evidence of cause and effect in their own right. More to the point, however, is the fact that the significant lipophilicity of these compounds means that POPs are likely to accumulate, persist and bioconcentrate and could, thus, achieve toxicologically relevant concentrations even though discrete exposure may appear limited.

Experimentally, POPs have been associated with significant environmental impact in a wide range of species and at virtually all trophic levels. While acute effects of POPs intoxication have been well documented, adverse effects associated with chronic low level exposure in the environment is of particular concern. Noteworthy in this context is the long biological half life of POPs in biological organisms thereby facilitating accumulation of seemingly small unit concentrations over extended periods of time. For some POPs, there is some experimental evidence that such cumulative low level exposures may be associated with chronic non-lethal effects including potential immunotoxicity, dermal effects, impairment of reproductive performance and frank carcinogenicity.

Immunotoxicity in association with exposure to different POPs has been reported by several authors. Investigators have demonstrated immune dysfunction as a plausible cause for increased mortality among marine mammals and have also demonstrated that consumption of persistent organic pollutant contaminated diets in seals may lead to vitamin and thyroid deficiencies and concomitant susceptibility to microbial infections and reproductive disorders. Investigators have also noted that immunodeficiency has been induced in a variety of wildlife species by a number of prevalent POPs, including TCDD's,

PCBs, chlordane, HCB, toxaphene and DDT.

Exposure to POPs has been correlated with population declines in a number of marine mammals including the common seal the harbour porpoise, bottle-nosed dolphins and beluga whales from the St. Lawrence River. More notably, a clear cause and effect relationship has been established between reproductive failure in mink and exposure to some POPs.

The scientific literature has demonstrated a direct cause and effect relationship in mink and ferrets between PCB exposure and immune dysfunction, reproductive failure, increased kit mortality, deformations and adult mortality. Similarly, investigators have also demonstrated a convincing correlation between environmental concentrations of PCBs and dioxins with reduced viability of larvae in several species of fish. Noteworthy as well is a report suggesting significant reproductive impairment in a number of Great Lakes species described as top level predators dependent on the Great Lakes aquatic food chain. Supporting this is the observation that wildlife, including stranded carcasses of St. Lawrence beluga whales, with reported high incidence of tumours have contained significantly elevated concentrations of PCBs mirex, chlordane and toxaphene. A 100% incidence of thyroid lesions in coho, pink and chinook salmon sampled in the Great Lakes over the last two decades has also been reported to be associated with increased body burdens of POPs.

3.2.2 Human health

As noted for environmental effects, it is also most difficult to establish cause and effect relationships for human exposure of POPs and incident disease. As with wildlife species, humans encounter a broad range of environmental exposures and frequently to a mixture of chemicals at any one time. Much work remains to be done on the study of the human health impact of exposure to POPs, particularly in view of the broad range of concomitant exposing experienced by humans.

The weight of scientific evidence suggests that some POPs have the potential to cause significant adverse effects to human health, at the local level, and at the regional and global levels through long-range transport.

For some POPs, occupational and accidental high-level exposure is of concern for both acute and chronic worker exposure. The risk is greatest in developing countries where the use of POPs in tropical agriculture has resulted in a large number of deaths and injuries. In addition to other exposure routes, worker exposure to POPs during waste management is a significant source of occupational risk in many countries. Short-term exposure to high concentrations of certain POPs has been shown to result in illness and death. For example, a study in the Philippines showed that in 1990, endosulfan became the number one cause of pesticide-related acute poisoning among subsistence rice farmers and mango sprayers. Occupational, bystander and near-field exposure to toxic chemicals is often difficult to minimize in developing countries. Obstacles in managing workplace exposure are in part due to poor or non-existent training, lack of safety equipment, and substandard working conditions. As well, concerns resulting from near-field and bystander exposure are difficult to identify due to inadequacies in monitoring of the

ambient environment and inconsistencies in medical monitoring, diagnosis, reporting and treatment. These factors contribute to a lack of epidemiological data. Earliest reports of exposure to POPs related to human health impact include an episode of HCB poisoning of food in south-east Turkey, resulting in the death of 90% of those affected and in other exposure related incidences of hepatic cirrhosis, porphyria and urinary, arthritic and neurological disorders. In another acute incident in Italy in 1976, release of 2,3,7,8-TCDD to the environment resulted in an increase of chloracne. The US EPA is currently reviewing dioxin related health effects especially for the non-carcinogenic endpoints such as immunotoxicity, reproductive disorders and neurotoxicity.

Such frank expressions of effects are not as common in the case of exposure to lower concentrations derived from the environment and the food chain. Laboratory and field observations on animals, as well as clinical and epidemiological studies in humans, and studies on cell cultures collectively demonstrate that overexposure to certain POPs may be associated with a wide range of biological effects. These adverse effects may include immune dysfunction, neurological deficits, reproductive anomalies, behavioural abnormalities and carcinogenesis. The scientific evidence demonstrating a link between chronic exposure to sublethal concentrations of POPs (such as that which could occur as a result of long-range transport) and human health impacts is more difficult to establish, but gives cause for serious concern. Swedish investigations have reported that dietary intake of PCBs, dioxins and furans may be linked to important reductions in the population of natural killer cells (lymphoytes), while other reports have suggested that children with high organochlorine dietary intake may experience rates of infection some 10-15 times higher than comparable children with much lower intake levels. The developing fetus and neonate are particularly vulnerable to POPs exposure due to transplacental and lactational transfer of maternal burdens at critical periods of development. It has also been reported that residents of the Canadian Arctic, and who exist at the highest trophic level of the Arctic aquatic food chain, have PCB intake levels in excess of the acceptable daily intake, and that may place this population at special risk for reproductive and developmental effects. In another report, children in the northern Quebec region of Canada who have had significant exposure to PCBs, dioxins and furans through breast milk also had a higher incidence of middle ear infections than children who had been bottle fed. Most authors, however, conclude that the benefits of breast feeding outweighs the risks.

Studies of carcinogenesis associated with occupational exposure to 2.3.7.8-TCDD also seem to indicate that extremely high-level exposures of human populations do elevate overall cancer incidence. Laboratory studies provide convincing supporting evidence that selected organochlorine chemicals (dioxins and furans) may have carcinogenic effects and act as strong tumour promoters.

More recently, literature has been accumulating in which some researchers have suggested a possible relationship between exposure to some POPs and human disease and reproductive dysfunction. Researchers have suggested that the increasing incidence of reproductive abnormalities in the human male may be related to increased estrogen (or estrogenic type) compound exposure *in vitro*, and further suggest that a single maternal exposure during pregnancy of minute amounts of TCDD may increase the frequency of cryptorchidism in male offspring, with no apparent sign of intoxication in the mother. Associations have been made between human exposure to certain chlorinated organic contaminants and cancers in human populations. Preliminary evidence suggests a possible association between breast

cancer and elevated concentrations of DDE. While the role of phytoestrogens and alterations in lifestyle cannot be dismissed as important risk factors in the dramatic increase in estrogen dependent breast cancer incidence, correlative evidence suggesting a role for POPs continues to mount. This latter theory has been supported in a report that noted that levels of DDE and PCBs were higher for breast cancer case patients than for control subjects, noting that statistical significance was achieved only for DDE. While a causal relationship between organochlorine exposure and malignant breast disease remains far from proven, the possibility that chronic low level exposure, when coupled with the known bioaccumulative properties of POPs, may even contribute in some small way to overall breast cancer risk has extraordinary implications for the reduction and prevention of this very important disease.

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4. ENVIRONMENTAL FATE AND TRANSPORT OF POPs

By definition, POPs are likely to be more persistent, mobile, and bioavailable than other substances. These properties are conferred by the structural makeup of the molecules and are often associated with greater degrees of halogenation. Included in this group of substances are some older chlorinated pesticides like DDT and the chlordanes, polychlorinated biphenyls, polychlorinated benzenes, and polychlorinated dioxins and furans. The physico-chemical properties of these compounds are such that they favour sufficiently high atmospheric concentrations that result in global redistribution by evaporation and atmospheric transport.

4.1 Physiochemical Properties and Environmental Partitioning

The physical properties of greatest importance are water solubility, vapour pressure, Henry's law constant (H), octanolwater partition coefficient (KOW), and the organic carbonwater partition coefficient (KOC). Persistence in the environment is the other important property of a substance since transport can extend the range of exposure to persistent substances far beyond the immediate area of use and/or release.

4.2 Environmental Influences on Persistence, Movement and Deposition

Persistence can be reduced by environmental transformation processes. These are: biotransformation; abiotic oxidation and hydrolysis; and photolysis. The relative importance of these processes depends on the rates at which they occur under natural environmental conditions. These rates are, in turn, dependent on the chemical structure and properties of the substance and its distribution in the various compartments of the environment. As would be expected, environmental factors have little effect on the breakdown and transformation of POPs. In addition, those that might have some effect are less effective in polar regions. Given the continued use and release of POPs in other parts of the globe, the result of this is a net accumulation of POPs in the polar regions.

Some of the above physical properties are strongly dependent on environmental conditions. For example, temperature strongly affects vapour pressure, water solubility, and, therefore, Henry's law constant. The net exchange direction for substances in the open ocean also reflects differences in surface water temperature and atmospheric concentration. For example, net movement of POPs in the Bay of Bengal in the Indian Ocean is from the ocean to the atmosphere while that in polar regions is the reverse. Temperature may also affect deposition in other locations. The distribution of POPs is inversely related to vapour pressure, and thus to temperature. Lower temperatures favour greater partitioning of these compounds from the vapour phase to particles suspended in the atmosphere. This increases the likelihood of their removal and transport to the surface of the earth by rain and snow (Figure 3).

Countries in the tropics experience higher year-round temperatures than countries in the temperate and polar regions of the globe. The practice of using some pesticides in tropical agriculture during the warmer, wetter growing season may facilitate the rapid dissipation of POPs through air and water.

These and other observations suggest that inputs of POPs to tropical coastal water bodies through river discharge are less significant than in temperate zones. The residence time in the tropical aquatic environment is quite short and transfer to the atmosphere is greater in these areas. The relatively short residence time of POPs in the tropical water bodies might be viewed as favourable for local organisms. However, it does have more far-reaching implications for the global environment because these volatilized residues from the tropics then disperse through the global atmosphere.

The present-day distribution of POPs in the oceans is consistent with a major change in distribution pattern during the last decades. Until the early 1980s, there were higher concentrations of POPs (such as DDT, and PCBs) in the midlatitude oceans of the northern hemisphere, probably reflecting the large usage in developed countries such as Japan, Europe, and North America. This distribution has not been seen in the most recent samples.

Atmospheric transport and accumulation of POPs (PCBs, DDT, HCHs, and chlordanes) in the polar regions has been extensively documented. Accumulation in polar regions is partly the result of global distillation followed by cold condensation of compounds within the volatility range of PCBs and pesticides. These contaminants are continually deposited and reevaporated and fractionate according to their volatilities (Figure 3). The result is relatively rapid transport and deposition of POPs having intermediate volatility, such as HCB, and slower migration of less volatile substances such as DDT

(Figure 4).

The characteristics of polar ecosystems intensify the problems of contamination with POPs. The colder climate, reduced biological activity and relatively small incidence of sunlight would be expected to increase the persistence of the POPs.

4.3 Deposition

Considerable data on concentrations of POPs in samples from the Arctic and the Antarctic are available and are summarized in the companion document to this assessment. Most of these data are published in summary form as means or means with ranges. It was not possible to access the raw data from which these means were calculated, however, the range of concentrations are presented in Table 4-1 for information. Inspection of this data showed indications of declines in concentrations since some of these POPs were banned or restricted. The maintenance of a central database of all analytical data on the POPs would greatly aid in determining spatial and temporal trends in the data and linking these to changes in use pattern of these substances.

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5. USES, SOURCES, ALTERNATIVES

5.1 Introduction

The twelve POPs which are the subject of this report, are used in or arise from industry, agriculture and disease vector control; nine are pesticides used on agricultural crops and/or for public health vector control. By the late 1970 s, all of the nine pesticides and PCBs had been either banned or subjected to severe use restrictions in many countries. Current information indicates that some of these POPs are still in use in parts of the world where they are considered as essential for ensuring public health. In an effort to further reduce their use in these countries, it is important to understand what countries are using these POPs, and how they are applied. It was found that there is considerable information that describes the aggregate volume of POPs produced and used in the world, however, there is very little reliable data about the specific uses in each country. Although this lack of specific data makes it difficult to evaluate the rationale for the continued use of the nine pesticides, the available information still allows one to discuss the use patterns and barriers to adoption of alternatives in a generic fashion.

5.2 Uses and Sources of POPs

Most, if not all, of the nine pesticides in question are still in use or existing in many countries. However, the actual quantity that specific countries may be currently using is unknown. There are no central registers of individual country use, although some organizations, like the FAO, United Nations

Economic Commission for Europe, and the World Bank have begun to assemble aggregate use data. The cumulative production of most of the compounds, as of approximately 1987, is outlined in Table 5-1. Thus, while country specific data was not found, the cumulative global (sometimes only US or "other" countries not defined) were identified. While this does not tell enough about usage to know specifically where and how much of these compounds are being used it does show that the compounds are in fact still in use and aids in forming a general picture of use patterns.

5.3 Alternatives to POPs

A variety of chemical and non-chemical alternatives are available for the POPs. Lists of alternative pesticides have been cited for use in developed countries and are described in Table 5-1. It is important to note that not all developing countries use POPs, and those countries that allow the use of certain POPs do not do so to the exclusion of alternatives. For example, in Honduras integrated pest management (IPM) systems are used in some areas that rely on the judicious use of newer and pest specific pesticides and biological control methods. In these same areas, there exists a well developed distribution network for both pest control technologies and information. In other areas of Honduras, where there are fewer producers operating smaller farms, the use of older compounds, including some POPs, is common for a variety of reasons, including:

* common social attitudes that foster the continued use of older products,

* poor dissemination of both alternatives and information,

* relatively high degree of illiteracy that constrains the dissemination of any information, and

* other production related factors that limit the practical adoption of alternatives.

5.4 Constraints to Adoption of Alternative Technologies

Why the alternatives that are available are not being used is an important issue. There are many barriers to the adaptation of these alternatives and to the adaptation of technologies in general especially in developing countries. Some of the alternatives are simply more costly both in price and in other resources required to apply them compared to the older more hazardous compounds. Some alternatives are believed to be more acutely toxic to the applicator than the POPs and therefore more hazardous to the individual, thus adding a human health cost dimension.

Other barriers to adoption include education and training. Education and training on both the older compounds as well as the possible alternatives is necessary for everyone in the production chain including the individual users and vendors. It may be that many individuals do not realize how hazardous the older chemicals are, what alternatives are available, and how to use these alternatives effectively.

The infrastructure and regulations that are needed to manage the use of pesticides, as well as educate and train individuals in the use of possible alternatives is not fully developed in all countries. Not all countries have the necessary infrastructure to implement effective management programs, nor do they have the infrastructure for the types of training that is described above.

The regulatory structure that some developing countries have adopted is based on the developed countries regulatory structure. This structure is often not adaptable or appropriate to the particular situation in the developing country. In addition, both financial and human resources needed to make such structures function effectively are often insufficient. Once a regulatory system is in place that is compatible with the resources available then, influence on the gradual elimination of older and hazardous compounds can be initiated.

The first initiative that is necessary to investigate these issues further is an in-depth inventory of the 12 compounds in individual countries, including a close examination of the amount used, the reasons for use, the alternatives available for the specific uses and the barriers that exist to the adaptation of alternatives specific to the country. Possibly a few case studies could be performed that would give a general idea of the answers to these questions. Once more quantitative data is available, then more meaningful work can be done in evaluating different alternatives and aiding in the implementation of these alternatives.

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6. SUBSTANCE PROFILES FOR THE PERSISTENT ORGANIC POLLUTANTS

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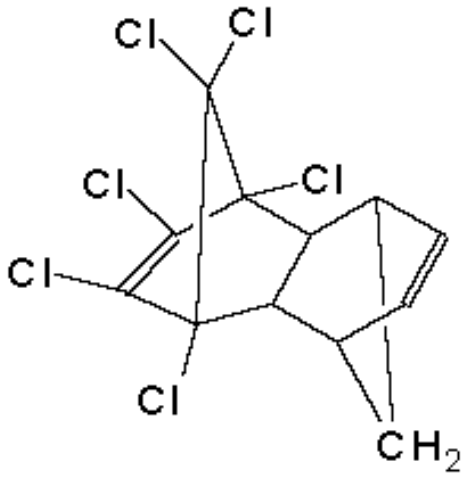
6. SUBSTANCE PROFILES FOR THE POPs

Information on countries that have taken action to ban or severely restrict compounds is derived from multiple sources dating back to 1987. This information needs to be verified and updated.

6.1 *ALDRIN*

Chemical properties



CAS chemical name: 1,2,3,4,10,10-Hexachloro-1,4,4a,5,8,8a-hexahydro-1,4:5,8-dimethanonaphthalene.

Synonyms and Trade Names (partial list): Aldrec, Aldrex, Aldrex 30, Aldrite, Aldrosol, Altox, Compound 118, Drinox, Octalene, Seedrin.

CAS No.: 309-00-2; molecular formula: C12H8Cl6; formula weight: 364.92

Appearance: White, odourless crystals when pure; technical grades are tan to dark brown with a mild chemical odour.

Properties: Melting point: 104 C(pure), 49-60 C(technical); boiling point: 145 C at 2 mm Hg; KH: 4.96 x 10-4 atm m3/mol at 25 C; log KOC: 2.61, 4.69; log KOW: 5.17-7.4; solubility in water: 17-180 µg/L at 25 C; vapour pressure: 2.31 x 10-5 mm Hg at 20 C.

Aldrin is a pesticide used to control soil insects such as termites, corn rootworm, wireworms, rice water weevil, and grasshoppers. It has been widely used to protect crops such as corn and potatoes, and has been effective to protect wooden structures from termites. Aldrin is readily metabolized to dieldrin by both plants and animals. As a result, aldrin residues are rarely found in foods and animals, and then only in small amounts. It binds strongly to soil particles and is very resistant to leaching into groundwater. Volatilization is an important mechanism of loss from the soil. Due to its persistent nature and hydrophobicity, aldrin is known to bioconcentrate, mainly as its conversion products. Aldrin is banned in many countries, including Bulgaria, Ecuador, Finland, Hungary, Israel, Singapore, Switzerland and Turkey. Its use is severely restricted in many countries, including Argentina, Austria, Canada, Chile, the EU, Japan, New Zealand, the Philippines, USA, and Venezuela.

Aldrin is toxic to humans; the lethal dose of aldrin for an adult man has been estimated to be about 5g, equivalent to 83 mg/kg body weight. Signs and symptoms of aldrin intoxication may include headache, dizziness, nausea, general malaise, and vomiting, followed by muscle twitchings, myoclonic jerks, and convulsions. Occupational exposure to aldrin, in conjunction with dieldrin and endrin, was associated with a significant increase in liver and biliary cancer, although the study did have some limitations, including a lack of quantitative exposure information. There is limited information that cyclodienes, such as aldrin, may affect immune responses.

The acute oral LD50 for aldrin in laboratory animals is in the range of 33 mg/kg body weight for guinea pigs to 320 mg/kg body weight for hamsters. Reproductive effects in rats were observed when pregnant females were dosed with 1.0 mg/kg aldrin subcutaneously. Offspring experienced a decrease in the median effective time for incisor teeth eruption and increase in the median effective time for testes descent. There is, as yet, no evidence of a teratogenic potential for aldrin. IARC has concluded that there is inadequate evidence for the carcinogenicity of aldrin in humans, and there is only limited evidence in experimental animals. Aldrin is therefore not classifiable as to its carcinogenicity in humans (IARC, Group 3).

Aldrin has low phytotoxicity, with plants affected only by extremely high application rates. The toxicity of aldrin to aquatic organisms is quite variable, with aquatic insects being the most sensitive group of invertebrates. The 96-h LC50 values range from 1-200 µg/L for insects, and from 2.2-53 µg/L for fish. Long term and bioconcentration studies are performed primarily using dieldrin, the primary conversion product of aldrin. In a model ecosystem study, only 0.5% of the original radioactive aldrin was stored as aldrin in the mosquitofish (*Gambusia affinis*), the organism at the top of the model food chain.

The acute toxicity of aldrin to avian species varies in the range of 6.6 mg/kg for bobwhite quail to 520 mg/kg for mallard ducks. Aldrin treated rice is thought to have been the cause of deaths of waterfowl, shorebirds and passerines along the Texas Gulf Coast, both by direct poisoning by ingestion of aldrin treated rice and indirectly by consuming organisms contaminated with aldrin. Residues of aldrin were detected in all samples of bird casualties, eggs, scavengers, predators, fish, frogs, invertebrates and soil.

As aldrin is readily and rapidly converted to dieldrin in the environment its, fate is closely linked to that of dieldrin. Aldrin is readily metabolised to dieldrin in both animals and plants, and therefore aldrin residues are rarely present in animals and then only in very small amounts. Residues of aldrin have been detected in fish in Egypt, the average concentration was 8.8 µg/kg, and a maximum concentration of 54.27 µg/kg.

The average daily intake of aldrin and dieldrin was calculated to be 19µg/person in India, and 0.55 µg/person in Vietnam. Dairy products, such as milk and butter, and animal meats are the primary sources of exposure.

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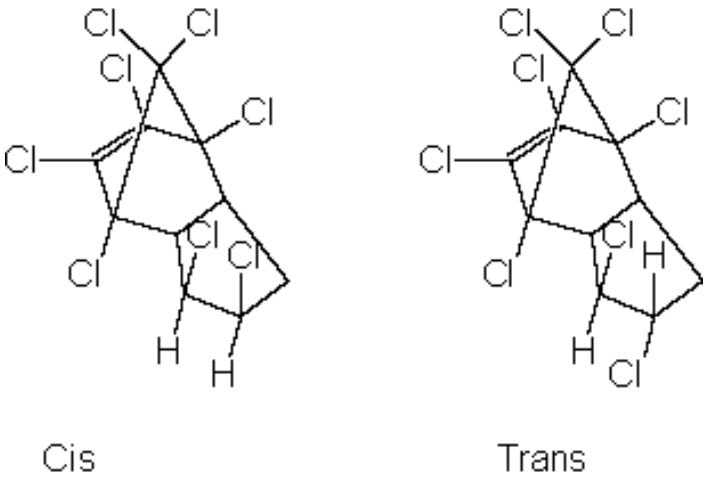
6.2 *CHLORDANE*

Chemical properties



CAS Chemical Name: 1,2,4,5,6,7,8,8-octachloro-2,3,3a,4,7,7a-hexahydro-4,7-methano-1H-indene

Trade names: (partial list): Aspon, Belt, Chloriandin, Chlorkil, Chlordane, Corodan, Cortilan-neu, Dowchlor, HCS 3260, Kypchlor, M140, Niran, Octachlor, Octaterr, Ortho-Klor, Synklor, Tat chlor 4, Topichlor, Toxichlor, Veliscol-1068.

CAS No.: 57-74-9; molecular formula: C10H6Cl8; formula weight: 409.78

Appearance: colourless to yellowish-brown viscous liquid with an aromatic, pungent odour similar to chlorine;

Properties: Melting point: <25 C; boiling point: 165 C at 2 mm Hg; KH: 4.8 x 10-5 atm m3/mol at 25 C; log KOC: 4.58-5.57; log KOW: 6.00; solubility in water: 56 ppb at 25 C; vapour pressure: 10-6 mm Hg at 20 C.

Chlordane is a broad spectrum contact insecticide that has been used on agricultural crops including vegetables, small grains, maize, other oilseeds, potatoes, sugarcane, sugar beets, fruits, nuts, cotton and jute. It has also been used extensively in the control of termites. Chlordane is highly insoluble in water, and is soluble in organic solvents. It is semi-volatile and can be expected to partition into the atmosphere as a result. It binds readily to aquatic sediments and bioconcentrates in the fat of organisms as a result of its high partition coefficient (log KOW = 6.00). Action to ban the use of chlordane has been taken in Austria, Belgium, Bolivia, Brazil, Chile, Columbia, Costa, Rica, Denmark, Dominican Republic, EU, Kenya, Korea, Lebanon, Liechtenstein, Mozambique, Netherlands, Norway, Panama, Paraguay, Philippines, Poland, Portugal, Santa Lucia, Singapore, Spain, Sweden, Switzerland, Tonga, Turkey, United Kingdom, Yemen and Yugoslavia. Its use is severely restricted or limited to non-agricultural uses in Argentina, Belize, Bulgaria, Canada, China, Cyprus, Dominica, Egypt, Honduras, Indonesia, Israel, Mexico, New Zealand, South Africa, Sri Lanka, USA and Venezuela.

Early studies on occupational exposure found no toxic effects in workers involved in the production of chlordane with up to 15 years of exposure. In a survey of 1105 workers associated with pest control, most of whom used chlordane, however, only three attributed illness to it (mild dizziness, headache, weakness). Chlordane exposure has not been associated with increased risk of mortality from cancer. Significant changes in the immune system were reported in individuals who complained of health effects which they associated with chlordane exposure.

Acute oral toxicity for chlordane in laboratory animals ranges from 83 mg/kg for pure cis-chlordane in rats to 1720 mg/kg for hamsters. Subchronic (90 day) inhalation exposure in rats and monkeys at doses up to 10 mg/m3 resulted in increases in the concentration of cytochrome P-450 and microsomal protein in rats. The results of this study provide a no-effect level in the rat of approximately 0.1 mg/m3 and in excess of in 10 mg/m3 the monkey.

Mice were fed diets containing chlordane for 6 generations. At 100 mg/kg, viability was decreased in the first and second generation, and no offspring were produced in the third generation. At 50 mg/kg, viability was decreased in the third and fourth generation, and at 25 mg/kg no statistically significant effects were observed after 6 generations. Offspring of rabbits administered chlordane orally on the 5th - 18th days of gestation did not exhibit changes in behaviour, appearance or body weight were observed, and no teratogenic effects were reported. IARC has concluded that, while there is inadequate evidence for the carcinogenicity of chlordane in humans, there is sufficient evidence in experimental animals. IARC has classified chlordane as a possible human carcinogen (Group 2B).

The acute toxicity of chlordane to aquatic organisms is quite variable, with 96-hour LC50 values as low as 0.4 µg/L for pink shrimp. The acute oral LD50 to 4-5 month old mallard ducklings was 1200 mg/kg body weight. The LC50 for bobwhite quail fed chlordane in their diet for 10 weeks was 10 mg/kg diet.

The half-life of chlordane in soil has been reported to be approximately one year. This persistence, combined with a high partition coefficient, provides the necessary conditions for chlordane to bioconcentrate in organisms. Bioconcentration factors of 37,800 for fathead minnows and 16,000 for sheepshead minnow have been reported. Data suggest that chlordane is bioconcentrated (taken up directly from the water) as opposed to being bioaccumulated (taken up by water and in food). The chemical properties of chlordane (low water solubility, high stability, and semi-volatility) favour its long range transport, and chlordane has been detected in arctic air, water and organisms.

Chlordane exposure may occur through food but, due to its highly restricted uses, this route does not appear to be a major pathway of exposure. The isomer gamma-chlordane was detected in only 2 (8.00 and 36.17 µg/kg wet weight) of 92 samples of Egyptian fish and in 2 of 9 samples (2.70 and 0.48 ppb) of food products imported into Hawaii from western Pacific rim countries. Chlordane has been detected in indoor air of residences of both Japan and the US. Exposure to chlordane in the air may be an important source of exposure to the US population. Mean levels detected in the living areas of 12 homes in New Jersey prior to and after treatment for termites ranged from 0.14 to 0.22 µg/m3, respectively . Mean levels in non-living areas (crawl spaces and unfinished basements) were higher; 0.97 µg/m3 before treatment and 0.91 µg/m3 after treatment. Levels detected in New Jersey homes before and after regulations restricting chlordane use fell from 2.6 to 0.9 µg/m3.

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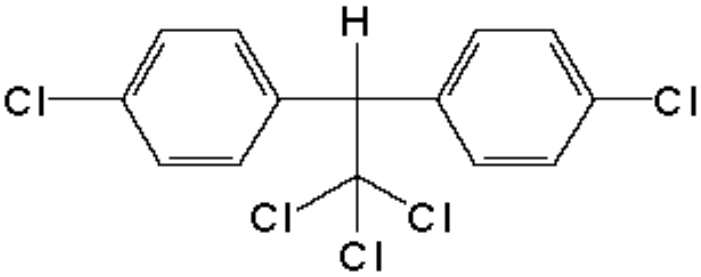
6.3 *DDT*

Chemical properties



CAS Chemical Name: 1,1'-(2,2,2-Trichloroethylidene)bis(4-chlorobenzene)

Synonyms and Trade Names (partial list): Agritan, Anofex, Arkotine, Azotox, Bosan Supra, Bovidermol, Chlorophenothan, Chloropenothane, Clorophenotoxum, Citox, Clofenotane, Dedelo, Deoval, Detox, Detoxan, Dibovan, Dicophane, Didigam, Didimac, Dodat, Dykol, Estonate, Genitox, Gesafid, Gesapon, Gesarex, Gesarol, Guesapon, Gyron, Havero-extra, Ivotan, Ixodex, Kopsol, Mutoxin, Neocid, Parachlorocidum, Pentachlorin, Pentech, PPzeidan, Rudseam, Santobane, Zeidane, Zerdane.

CAS No.: 50-29-3; molecular formula: C14H9Cl5; formula weight: 354.49.

Appearance: Odourless to slightly fragrant colourless crystals or white powder.

Properties: Melting point: 108.5 C; boiling point: 185 C at 0.05 mm Hg (decomposes); KH: 1.29 x 10-5 atm·m3/mol at 23 C; log KOC: 5.146-6.26; log KOW: 4.89-6.914; solubility in water: 1.2-5.5 µg/L at 25 C.

DDT was widely used during the Second World War to protect the troops and civilians from the spread of malaria, typhus and other vector borne diseases. After the war, DDT was widely used on a variety of agricultural crops and for the control of disease vectors as well. It is still being produced and used for vector control. Growing concern about adverse environmental effects, especially on wild birds, led to severe restrictions and bans in many developed countries in the early 1970s. The largest agricultural use of DDT has been on cotton, which accounted for more than 80% of the US use before its ban there in 1972. DDT is still used to control mosquito vectors of malaria in numerous countries.

DDT is highly insoluble in water and is soluble in most organic solvents. It is semi-volatile and can be expected to partition into the atmosphere as a result. Its presence is ubiquitous in the environment and residues have even been detected in the arctic. It is lipophilic and partitions readily into the fat of all living organisms and has been demonstrated to bioconcentrate and biomagnify. The breakdown products of DDT, 1,1-dichloro-2,2-bis(4-chlorophenyl)ethane (DDD or TDE) and 1,1-dichloro-2,2bis(4-chlorophenyl)ethylene) (DDE), are also present virtually everywhere in the environment and are more persistent than the parent compound.

The use of DDT has been banned in 34 countries and severely restricted in 34 other countries. The countries that have banned DDT include Argentina, Australia, Bulgaria, Canada, Colombia,

Cyprus, Ethiopia, Finland, Hong Kong, Japan, Lebanon, Mozambique, Norway, Switzerland, and the USA. Countries that have severely restricted its use include Belize, Ecuador, the EU, India, Israel, Kenya, Mexico, Panama, and Thailand.

DDT has been widely used in large numbers of people who were sprayed directly in programs to combat typhus, and in tropical countries to combat malaria. Dermal exposure to DDT has not been associated with illness or irritation in a number of studies. Studies involving human volunteers who ingested DDT for up to 21 months did not result in any observed adverse effects. A non-significant increase in mortality from liver and biliary cancer and a significant increase in mortality from cerebrovascular disease has been observed in workers involved in the production of DDT. There is some evidence to

suggest that DDT may be suppressive to the immune system, possibly by depressing humoral immune responses. Perinatal administration of weakly estrogenic pesticides such as DDT produces estrogen-like alterations of reproductive development, and there is also limited data that suggest a possible association between organochlorines, such as DDT and its metabolite DDE, and risk of breast cancer.

DDT is not highly acutely toxic to laboratory animals, with acute oral LD50 values in the range of 100 mg/kg body weight for rats to 1,770 mg/kg for rabbits. In a six generation reproduction study in mice, no effect on fertility, gestation, viability, lactation or survival were observed at a dietary level of 25 ppm . A level of 100 ppm produced a slight reduction in lactation and survival in some generations, but not all, and the effect was not progressive. A level of 250 ppm produced clear adverse reproductive effects. In both these and other studies, no evidence of teratogenicity has been observed.

IARC has concluded that while there is inadequate evidence for the carcinogenicity of DDT in humans, there is sufficient evidence in experimental animals. IARC has classified DDT as a possible human carcinogen (Group 2B).

DDT is highly toxic to fish, with 96-hour LC50 values in the range of 0.4 µg/L in shrimp to 42 µg/L in rainbow trout. It also affects fish behaviour. Atlantic salmon exposed to DDT as eggs experienced impaired balance and delayed appearance of normal behaviour patterns. DDT also affects temperature selection in fish.

DDT is acutely toxic to birds with acute oral LD50 values in the range of 595 mg/kg body weight in quail to 1,334 mg/kg in pheasant, however it is best known for its adverse effects on reproduction, especially DDE, which causes egg shell thinning in birds with associated significant adverse impact on reproductive success. There is considerable variation in the sensitivity of bird species to this effect, with birds of prey being the most susceptible and showing extensive egg shell thinning in the wild. American kestrels were fed day old cockerels injected with DDE. Residues of DDE in the eggs correlated closely with the dietary DDE concentration and there was a linear relationship between degree of egg shell thinning and the logarithm of the DDE residue in the egg. Data collected in the field has confirmed this trend. DDT (in conjunction with other halogenated aromatic hydrocarbons) has been linked with feminization and altered sex-ratios of Western Gull populations off the coast of southern California, and Herring Gull populations in the Great Lakes.

DDT and related compounds are very persistent in the environment, as much as 50% can remain in the soil 10-15 years after application. This persistence, combined with a high partition coefficient (log KOW = 4.89-6.91) provides the necessary conditions for DDT to bioconcentrate in organisms. Bioconcentration factors of 154,100 and 51,335 have been recorded for fathead minnows and rainbow trout, respectively. It has been suggested that higher accumulations of DDT at higher trophic levels in aquatic systems results from a tendency for organisms to accumulate more DDT directly from the water, rather than by biomagnification. The chemical properties of DDT (low water solubility, high stability and semi-volatility) favour its long range transport and DDT and its metabolites have been detected in arctic air, water and organisms. DDT has also been detected in virtually all organochlorine monitoring programs and is generally believed to be ubiquitous throughout the global environment.

DDT and its metabolites have been detected in food from all over the world and this route is likely the greatest source of exposure for the general population . DDE was the second most frequently found residue (21%) in a recent survey of domestic animal fats and eggs in Ontario, Canada, with a maximum residue of 0.410 mg/kg. Residues in domestic animals, however, have declined steadily over the past 20 years. In a survey of Spanish meat and meat products, 83% of lamb samples tested contained at least one ofthe DDT metabolites investigated, with a mean level of 25 ppb. An average of 76.25 ppb *p,p'*-DDE was detected in fish samples from Egypt. DDT was the most common organochlorine detected in foodstuffs in Vietnam with mean residue concentrations of 3.2 and 2.0 µg/g fat in meat and fish, respectively. The estimated daily intake of DDT and its metabolites in Vietnam was 19 µg/person/day. Average residues detected in meat and fish in India were 1.0 and 1.1 µg/g fat respectively, with an estimated daily intake of 48 µg/person/day for DDT and its metabolites.

DDT has also been detected in human breast milk. In a general survey of 16 separate compounds in the breast milk of

lactating mothers in four remote villages in Papua, New Guinea, DDT was detected in 100% of samples (41), and was one of only two organochlorines detected. DDT has also been detected in the breast milk of Egyptian women, with an average total DDT detected of 57.59 ppb and an estimated daily intake of total DDT for breast feeding infants of 6.90 µg/kg body weight /day. While lower than the acceptable daily intake of 20.0 µg/kg body weight recommended by the Joing FAO/ WHO Meeting on Pesticide Residues (JMPR), its continuing presence raises serious concerns regarding potential effects on developing infants.

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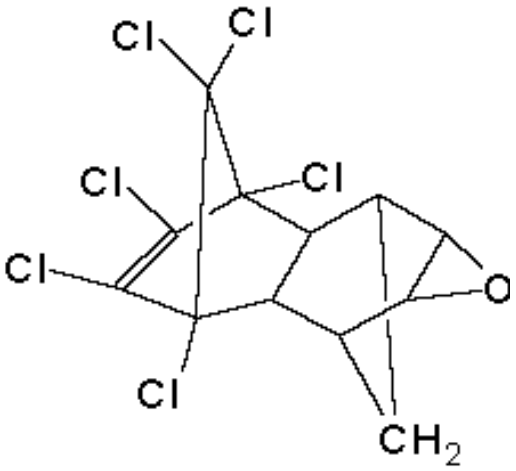
6.4 *DIELDRIN*

Chemical properties



CAS Chemical Name: 3,4,5,6,9,9-Hexachloro-1a,2,2a,3,6,6a,7,7a-octahydro-2,7:3,6-dimetanonapth[2,3-*b*]oxirene.

Synonyms and Trade Names (partial list):Alvit, Dieldrite, Dieldrix, Illoxol, Panoram D-31, Quintox.

CAS No.:60-57-1; molecular formula: C12H8Cl6O; formula weight: 380.91.

Appearance: A stereo-isomer of endrin, dieldrin may be present as white crystals or pale tan flakes, odourless to mild chemical odour.

Properties: Melting point: 175-176 C; boiling point: decomposes; KH: 5.8 x 10-5 atm·m3/mol at 25 C; log KOC: 4.08-4.55; log KOW: 3.692-6.2; solubility in water: 140 µg/L at 20 C; vapour pressure: 1.78 x 10-7 mm Hg at 20 C.

Dieldrin has been used in agriculture for the control of soil insects and several insect vectors of disease but this latter use has been banned in a number of countries due to environmental and human health concerns. Principle contemporary uses are restricted to control termites and wood borers and against textile pests (WHO, 1989). Dieldrin binds strongly to soil particles and hence is very resistant to leaching into groundwater. Volatilization is an important mechanism of loss from the soil and, because of its persistent nature and hydrophobicity, dieldrin is known to bioconcentrate.

Action to ban dieldrin has been taken in many countries, including Bulgaria, Ecuador, the EU, Hungary, Israel, Portugal, Singapore, Sweden, and Turkey. Its use is severely restricted in numerous countries, including Argentina, Austria, Canada, Colombia, Cyprus, India, Japan, New Zealand, Pakistan, USA and Venezuela.

In a study using human volunteers, the subjects received dieldrin daily for 2 years. All the volunteers continued in excellent health, and clinical, physiological and laboratory findings remained essentially unchanged through the exposure period and an 8 month follow up. In a study of workers from a plant involved in the manufacture of aldrin, dieldrin and endrin, a statistically significant increase in liver and biliary tract cancers was observed, although the study did have some limitations, including lack of quantitative exposure information.

In laboratory studies, acute oral LD50 values in the range of 37 mg/kg body weight in rats to 330 mg/kg in hamsters have been found for dieldrin. As with other organochlorine compounds, the liver is the major target organ in rats, with effects that included increased liver/body weight ratio, hypertrophy and histopathological changes. The no observed adverse effect level (NOAEL) in rats is 0.5 mg/kg diet, equal to 0.025 mg/kg body weight/day. When rats were fed dieldrin in their diet over three generations, no changes in reproductive capacity were observed at any dose level tested. A NOAEL of 2 mg dieldrin /kg diet has been set for reproduction in rats. There was no evidence for teratogenic potential in studies in rats, mice or rabbits using oral doses of up to 6 mg/kg body weight. Abnormal development and fetotoxicity were observed in hamsters and mice, however, these results are unlikely to be of significance in view of the maternal toxicity noted at the high dose levels. There is limited evidence that cyclodienes such as dieldrin may affect immune responses. IARC has concluded that there is inadequate evidence for the carcinogenicity of dieldrin in humans, and limited evidence in experimental animals and has been classified by IARC in Group 3.

Dieldrin has low phytotoxicity. Plants are affected only by application rates much higher than suggested use rates. The acute toxicity of dieldrin is quite variable for aquatic invertebrates, with insects being the most sensitive group (values range from 0.2-40 µg/L). It is highly toxic to most species of fish tested in the laboratory (values range from 1.1-41 µg/L). Acute toxicity of dieldrin in frogs (96-h LC50) ranged from 8.7 µg/L for *Rana catesbeiana* tadpoles to 71.3µg/L for the tadpoles of *Rana pipiens*. Spinal deformities in embryo-larval tests were observed at concentrations as low as 1.3 µg/L for *Xenopus laevis* after a 10 day exposure.

The acute toxicity of dieldrin to avian species varies widely, with acute oral LD50 values in the range of 26.6 in pigeons to 381 mg/kg in mallard ducks. Mallard ducklings were exposed to dieldrin in the diet for 24 days. A 24 d NOAEL of 0.3µg dieldrin/g diet, based on growth impairment, was determined. Reproduction success in birds has not been consistently affected in the absence of maternal toxicity.

The acute LD50 of dieldrin to four species of voles range from 100 to 210 mg/kg body weight, suggesting that these microtine rodents are less susceptible than laboratory rodents to dieldrin. In another study, white tailed deer (*Odocoileus virginianus*) were fed diet containing dieldrin for up to 3 years. Adult survival was not affected, and fertility and *in utero* mortality was comparable for all groups. Fawns from treated does were smaller at birth, experienced greater postpartum mortality and weight gain was reduced. Blesbuck (*Damaliscuc dorcas phillipsi*) were fed diets containing dieldrin for 90 days. None of the animals fed 5 or 15 mg/kg diet died during the study period, but all animals at the higher dose levels died within 24 days.

The half life of dieldrin in temperate soils is approximately 5 years. This persistence, combined with high lipid solubility, provides the necessary conditions for dieldrin to bioconcentrate and biomagnify in organisms. Bioconcentration factors of 12,500 and 13,300 have been reported for guppies and sculpins, respectively. It is likely that dieldrin is bioconcentrated by aquatic organisms rather than bioaccumulated. Dieldrin's chemical properties (low water solubility, high stability, and semi-volatility) favour its long range transport, and dieldrin has been detected in arctic air, water and organisms.

Dieldrin residues have been detected in air, water, soil, fish, birds and mammals, including humans and human breast milk. As aldrin is readily and rapidly converted to dieldrin in the environment and in organisms, the levels of dieldrin detected likely reflect the total concentrations of both compounds. In Egypt, the estimated dietary intake of dieldrin by breast fed infants of 1.22 µg/kg body weight/ day. Diet is the main source of exposure to the general public. Dieldrin was the second most common pesticide detected in a survey of US pasteurized milk, detected in 172 of the 806 composite samples tested,

with a maximum level of 0.003 ppm. Dieldrin residues were detected in 9 of 602 (1.5%) samples of domestic animal fats and eggs in Canada, with a maximum of 0.050 mg/kg. Dieldrin was also detected in Spanish meat, residues of 20 to 40 ppb were detected in the fat of 8 to 15% of pork products (meat, cured sausage, pork bologna) and in 28% fresh poultry sausage. Dieldrin residues were detected in Oriental party beans at 3.45 ppb. The average daily intake of aldrin and dieldrin in India was calculated to be 19 µg/person, exceeding the acceptable daily intake of 6.0 µg/60 kg of body weight recommended by the Joint FAO/WHO Meeting on Pesticide Residues (JMPR). Dairy products, such as milk and butter, and animal meats were the primary sources of exposure. Exposure through food intake has been estimated at 0.55 µg/person in Vietnam.

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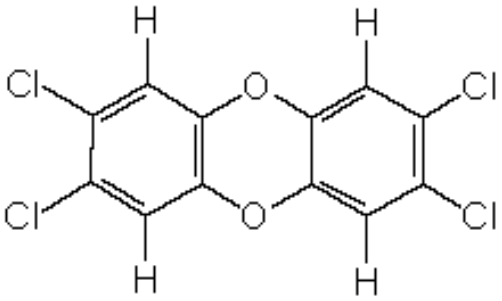
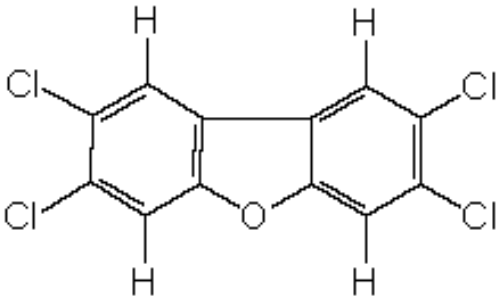
6.5 *POLYCHLORINATED DIBENZO - p - DIOXINS AND FURANS*

Chemical properties



2,3,7,8-TCDD



2,3,7,8-TCDF

Dioxins

Congener Group	Molecular weight (g/molecular)	Vapour Pressure (Pa X 10-3)	Water Solubility (mg/m3)	Log KOW
M1CDD	218.5	73-75	295-417	4.75-5.00
D2CDD	253.0	2.47-9.24	3.75-16.7	5.60-5.75
T3CDD	287.5	1.07	8.41	6.35
T4CDD	322.0	0.00284-0.275	0.0193-0.55	6.60-7.10

P5CDD	356.4	0.00423	0.118
7.40			
H6CDD	391.0	0.00145	0.00442
7.80			
H7CDD	425.2	0.000177	0.0024
8.00			
O8CDD	460.0	0.000953	0.000074
8.20			

Polychlorinated dibenzo-*para*-dioxins (dioxins) and polychlorinated dibenzofurans (furans) are two groups of planar tricyclic compounds that have very similar chemical structures and properties. They may contain between 1 and 8 chlorine atoms; dioxins have 75 possible positional isomers and furans have 135 positional isomers. They are generally very insoluble in water, are lipophilic and are very persistent. The chemical properties of each of the isomers has not been elucidated, further complicating a discussion of their properties which vary with the number of chlorine atoms present. Neither dioxins nor furans are produced commercially, and they have no known use. They are by-products resulting from the production of other chemicals. Dioxins may be released into the environment through the production of pesticides and other chlorinated substances. Furans are a major contaminant of PCBs. Both dioxins and furans are related to a variety of incineration reactions, and the synthesis and use of a variety of chemical products. Dioxins and furans have been detected in emissions from the incineration of hospital waste, municipal waste, hazardous waste, car emissions, and the incineration of coal, peat and wood. Of the 210 dioxins and furans, 17 contribute most significantly to the toxicity of complex of mixtures. In order to facilitate a comparison mixtures, International Toxicity Equivalency Factors (TEFs) have been assigned to individual dioxins and furans basedon a comparison of toxicity to 2,3,7,8-tetrachlorodibenzodioxin (2,3,7,8-TCDD). For example, 2,3,7,8-TCDF has been shown to be approximately one-tenth as toxic as 2,3,7,8-TCDD in animal tests, and its toxic equivalent value is 0.1. TEFs are regarded as risk management tools and they do not necessarily represent actual toxicity with respect to all endpoints. Rather, they tend to overestimate the toxicity of mixtures.

At the present time, the only persistent effect associated with dioxin exposure in humans is chloracne. Other health effects that have been reported include peripheral neuropathies, fatigue, depression, personality changes, hepatitis, enlarged liver, abnormal enzyme levels and porphyria cutanea tarda though no causal relationships were established in every case. Results of a study on 1,520 workers known to have been exposed to 2,3,7,8-TCDD for a period of at least one year, and with a latency of at least twenty years between exposure and diagnosis of disease, revealed a slightly, but significantly elevated mortality from soft tissue sarcoma and cancers of the respiratory system. As with other studies, interpretation of results was limited by the small number of deaths and by possible confounders including smoking and other occupational exposures. Two recent studies followed a young population from the area of the Seveso, Italy industrial accident. The first, a cancer study, examined a cohort of people aged 0-19 years living in the accident area at the time of the accident, for the period 1977-1986. While a consistent tendency toward increased risk was apparent, none of the relative risks were significantly elevated. Non-significant increases in thyroid cancer and myeloid leukemia were also observed. The study is limited, however, by the relatively short latency periods, the definition of exposure based on place of residence and the limited number of events. The second study examined the mortality of the same cohort of people for the same time period. Among the exposed, mortality owing to all causes did not deviate from expectations, however, as noted above, this study provides only limited evidence. Direct exposure of humans to furans has been reported in two incidents of rice oil contamination by PCBs contaminated with PCDFs, in Japan (Yusho) and Taiwan (Yucheng). While it is possible that the effects observed in these incidents may be due to the presence of furans, the similarity of structure, effects and mode of action of PCBs and PCDFs precludes a definite conclusion on the causative agent.

The acute oral toxicity in laboratory animals is highly variable, with LD50 values ranging from 0.6 µg/kg body weight in guinea pigs to 1,157 µg/kg in hamsters. Effects of dioxin exposure that are common to most, and sometimes all, species include wasting, lymphoid involution, hepatotoxicity, chloracne and epidermal changes, and gastric lesions. Other characteristic reponses include edema, ascites and hypopericardium in chickens; fetal death and resorption in rats and fetal wastage, embryotoxicity and malformations in mice. A three-generation study was conducted in which rats were fed diets containing 2,3,7,8-TCDD. Significant decreases in fertility and neonatal survival were observed in the f0 group receiving 0.1 µg TCDD/kg/day. At 0.01 µg TCDD/kg/day, fertility was significantly reduced in the f1 and f2 generations. Decreases in litter size, gestation survival and neonatal survival and growth were also observed at this dose level. No effect on fertility, litter size at birth or post natal body weight was observed in any generation of the 0.001 µg TCDD/kg/day group. Some teratogenic effects have been observed in mice in association with dioxin and furan exposure, including hydronephrosis and cleft palate The most teratogenic isomer was 2,3,4,7,8-pentachlorodibenzofuran, with an ED50 of 36 µg/kg for cleft palate and 7 µg/kg for hydronephrosis. Teratogenic responses observed are similar to those seen with TCDD, but theses compounds are only 1/10 to 1/100 as potent.

Dioxins, specifically 2,3,7,8-TCDD, are associated with a variety of adverse effects on the reproductive systems of both male and female rats. Male reproductive toxicity has included altered regulation of luteinizing hormone secretion, reduced testicular steroidogenesis, reduced plasma androgen concentrations, reduced testis and accessory sex organ weights, abnormal testis morphology, decreased spermatogenesis, and reduced fertility. Signs of female reproductive toxicity included hormonal irregularities in the oestrous cycle, reduced litter size and reduced fertility. A review of recent literature concerning 2,3,7,8-TCDD effects on immunocompetence suggests that 2,3,7,8-TCDD either indirectly (in the case of T-cells) or directly (in the case of B-cells) affects the maturational or differentiative processes of immunocompetent cells. Studies in exposed human populations and in non-human primates have shown that halogenated aromatic hydrocarbons produce measurable alterations in both innate and acquired immunity, although significant deficits in immunocompetence have not been conclusively associated with these changes. IARC has concluded that while there is inadequate evidence for the carcinogenicity of 2,3,7,8-TCDD in humans, there is sufficient evidence in experimental animals. IARC has classified 2,3,7,8-TCDD as a possible human carcinogen (Group 2B). Other chlorinated dibenzodioxins (other than 2,3,7,8-TCDD) are deemed not classifiable as to their carcinogenicity in humans.

Exposure of fish to dioxins and furans results in a delayed mortality that can continue many days post-exposure. Rainbow trout exposed to 2,3,7,8-TCDD and to 2,3,7,8-TCDF for 28 days, followed by a 28 day depuration period had a 56-day LC50 of 46 pg/L for TCDD, and a NOEC for TCDD based on growth and mortality below the lowest exposure concentration of 38 pg/L. The 56-day NOEC for TCDF was calculated to be 1.79 ng/L for mortality and 0.41 ng/L for growth. Mortality and behavioural changes such as lethargic swimming, feeding inhibition and lack of response to external stimuli continued after the 28 day exposure period ended. Early life stages of fish are very sensitive to the effects of dioxins, furans, and PCBs. Parts per trillion concentrations of these structurally related chemicals in lake trout and rainbow trout eggs exhibit toxicity through sac fry mortality associated with yolk sac edema and hemorrhages.

Great blue heron eggs collected from sites of low, intermediate and high contamination had levels of 2,3,7,8-TCDD in eggs of 10 ng/kg (wet weight), 135 ng/kg and 211 ng/kg, respectively. Although there was no effect on mortality of chicks, effects of contamination included decreased growth with increased TCDD level, depression of skeletal growth with increased TCDD levels and subcutaneous edema which increased with increasing PCDD and PCDF contamination. Also observed were shortened beaks and a scarcity of down follicles in the chicks from the more contaminated sites. Mink administered TCDD experienced the wasting syndrome associated with TCDD intoxication and gastric lesions at higher dosages. A 28 day oral LD50 was calculated to be 4.2 µg TCDD/kg body weight.

Dioxins and furans are considered to be very stable and persistent, as illustrated by the half life of TCDD in soil of 10-12 years. This persistence, combined with high partition coefficients (up to 8.20 for OCDD) provides the necessary conditions for these compunds to bioconcentrate in organisms. Bioconcentration factors of 26 707 has been reported in rainbow trout (*Salmo gairdneri*) exposed to 2,3,7,8-TCDD. The chemical properties of dioxins and furans (low water solubility, high stability and semi-volatility) favour their long range transport and these compounds have been detected in arctic organisms.

As with most other organochlorines, food is a major source of dioxins and furans in the general population, with food of animal origin contributing the most to human body burdens. In a survey of dioxins in US food, total PCDD/Fs ranged from 0.42 ppt to 61.8 ppt (wet weight) (total TEq range: 0.02 to 1.5 ppt). The estimated daily intake for adults ranged from 0.3 to 3.0 pg TEqs/kg body weight, and for breast fed infants the range was 35.3 to 52.6 pg TEqs/kg body weight. Recent estimates of adult average daily intake for Canada, Germany and the Netherlands are 1.52, 2 and 1 pg TEQ/kg bodyweight, respectively. These are below the TDI of 10 pg/kg body weight for lifetime exposure estimated by WHO.

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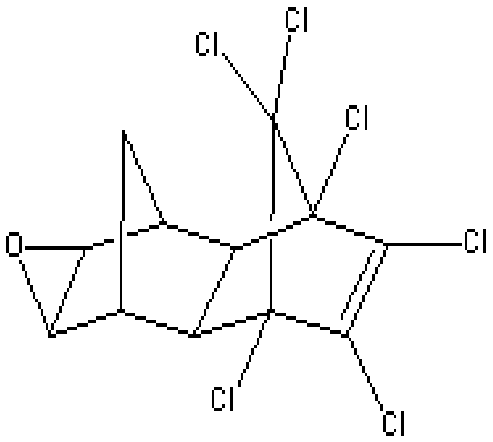
6..6 *ENDRIN*

Chemical properties



CAS Chemical Name: 3,4,5,6,9,9,-Hexachloro-1a,2,2a,3,6,6a,7,7a-octahydro-2,7:3,6-dimethanonaphth[2,3-*b*]oxirene.

Synonyms and Trade Names (partial list): Compound 269, Endrex, Hexadrin, Isodrin Epoxide, Mendrin, Nendrin.

CAS No.: 72-20-8; molecular formula: C12H8Cl6O; formula weight: 380.92.

Appearance: White, odourless, crystalline solid when pure; light tan colour with faint chemical odour for technical grade.

Properties: Melting point: 200 C; boiling point: 245 C (decomposes); KH: 5.0 x 10-7 atm·m3/molecular; log KOW: 3.209-5.339; solubility on water: 220-260 µg/L at 25 C; vapour pressure: 7 x 10-7 mm Hg at 25 C.

Endrin is a foliar insecticide used mainly on field crops such as cotton and grains. It has also been used as a rodenticide to control mice and voles. It is rapidly metabolised by animals and does not accumulate in fat to the same extent as other compounds with similar structures. It can enter the atmosphere by volatilization, and can contaminate surface water from soil run-off. Endrin is banned in many countries, including Belgium, Cyprus, Ecuador, Finland, Israel, Philippines, Singapore, Thailand and Togo. Its use is severely restricted in many countries, including Argentina, Canada, Chile, Colombia, the EU, India, Japan, New Zealand, Pakistan, USA, and Venezuela.

A study of workers involved in the production of aldrin, dieldrin and endrin did not find endrin in the blood of workers, except in cases of accidental, acute over-exposure. These findings are in agreement with results of a study of 71 workers in an endrin plant in the USA. Data on absenteeism, results of liver function tests, blood chemistry, blood morphology, urine

analysis, occurrence of sensitization, the incidence and pattern of diseases including the occurrence of malignant growth showed no difference between workers exposed to endrin and other chemical plant operators. A study of workers involved in the manufacture of aldrin, dieldrin and endrin found a statistically significant increase in liver and biliary tract cancers, although the study did have some limitations such as lack of quantitative exposure information. There is limited evidence that cyclodienes such as endrin may also depress immune responses.

The acute oral LD50 of endrin is in the range of 3 mg/kg body weight in monkeys to 36 mg/kg in guinea pigs. Male and female Long-Evans rats were fed endrin in the diet over three generations. No difference in appearance, behaviour, body weight, or number or size of litters was observed. The weights of liver, kidneys and brain were normal, and no histopathological abnormalities were observed in third generation weanlings. Significant increased mortality of pups in the second and third generations of rats fed 3 mg/kg was noted. Endrin was not teratogenic at levels that did not cause maternal toxicity. Endrin is metabolised rapidly by animals, and very little is accumulated in fat compared to compounds of similar structure (including its stereoisomer dieldrin). The formation of *anti*-12-hydroxyendrin is considered to be the major route of metabolism of endrin. IARC has concluded that there is inadequate evidence for the carcinogenicity of endrin in humans, and there is only limited evidence in experimental animals. Endrin is therefore not classifiable as to its carcinogenicity in humans (Group 3).

Endrin is highly toxic to fish, with most LC50 values below 1.0 µg/L. Sheepshead minnows embryos exposed for 23 weeks to 0.31 and 0.72 µg/L hatched early, and all those exposed to 0.72 µg/L died by the ninth day of their exposure, while those exposed at 0.31 µg/L were initially stunted and some died. The reproductive ability of the survivors of the 0.31 µg/L was impaired. No significant effects were observed at an exposure concentration of 0.12 µg/L. The lowest observed adverse effect level (LOAEL) for aquatic organisms was 30 ng/L over 20 days for reproduction in mysid shrimp. Reproduction in male and female mallard ducks was not impaired by diets containing 0, 0.5 or 3.0 mg/kg.

The half life of endrin in soil may be up to 12 years, depending on local conditions. This persistence, combined with a high partition coefficient (log KOW = 3.21-5.340), provides the necessary conditions for endrin to bioconcentrate in organisms. A bioconcentration factor of 6,400 was recorded for sheepshead minnows exposed to endrin from embryonic stage through adulthood. Bluegill sunfish exposed to water containing 14C-labelled endrin took up 91% of the radio-labelled endrin with in 48 hours, with a half life of loss from the tissues of approximately four weeks. *Leiostomus xantharus* exposed to 0.05 µg/L for 5 months had a tissue residue level of 78 µg/kg tissue. After 18 days in uncontaminated water, no residues were detected, suggesting that endrin disappears rapidly from this organism.

The chemical properties of endrin (low water solubility, high stability in the environment, and semi-volatility) favour its long range transport, and it has been detected in arctic freshwater. The main source of endrin exposure to the general population is residues in food however, contemporary intake is generally below the acceptable daily intake of 0.0002 mg/kg body weight recommended by the Joint FAO/WHO Meeting on Pesticide Residues (JMPR). Recent food surveys have generally not included endrin, and hence recent monitoring data are not available.

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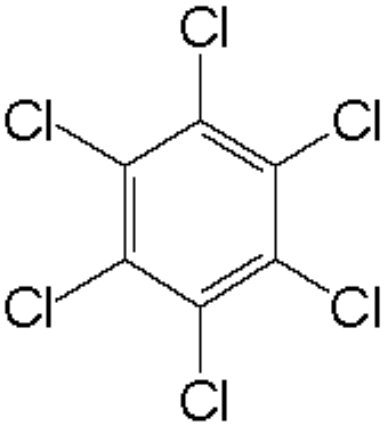
6.7 *HEXACHLOROBENZENE*

Chemical properties



CAS Chemical Name: hexachlorobenzene

Trade names: (partial list): Amaticin, Anticarie, Bunt-cure, Bunt-no-more, Co-op hexa, Granox, No bunt, Sanocide, Smut-go, Sniecotox

CAS No.: 118-74-1; molecular formula: C6Cl6; formula weight: 284.78;

Appearance: White monoclinic crystals or crystalline solid

Properties: Melting point: 227-230 C; boiling point: 323-326 C (sublimes); KH: 7.1 x 10-3 atm m3/mol at 20 C; log KOC: 2.56-4.54; log KOW: 3.03-6.42; Solubility in water: 40 µg/L at 20 C; vapour pressure: 1.089 x 10-5 mm Hg at 20 C.

Hexachlorobenzene (HCB) is a fungicide that was first introduced in 1945 for seed treatment, especially for control of bunt of wheat. HCB is also a byproduct of the manufacture of industrial chemicals including carbon tetrachloride, perchlorethylene, trichloroethylene and pentachlorbenzene. It is a known impurity in several pesticide formulations, including pentachlorophenol and dicloram and may be present as an impurity in others. HCB is highly insoluble in water, and is soluble in organic solvents. It is quite volatile and can be expected to partition into the atmosphere as a result. It is very resistant to breakdown and has a high partition coefficient (KOW=3.03-6.42), and is known to bioconcentrate in the fat of living organisms as a result. HCB is banned in Austria, Belgium, Czechoslovakia, Denmark, the EU, Germany, Hungary, Liechtenstein, Netherlands, Panama, Switzerland, Turkey, United Kingdom and the USSR. It is severely restricted or has been voluntarily withdrawn in Argentina, New Zealand, Norway and Sweden.

The most notable episode involving the effects of HCB on humans involves the ingestion of HCB treated seed grain in eastern Turkey between 1954 and 1959. The patients who ingested the treated seed experienced a range of symptoms including photosensitive skin lesions, hyperpigmentation, hirsutism, colic, severe weakness, porphyrinuria, and debilitation. Approximately 3,000-4,000 people developed porphyria turcica, a disorder of haem biosynthesis. Mortality was up to 14%. Mothers who ingested the seeds passed the HCB to their children by placental transfer and through maternal milk. Children born to these women developed "pembe yara" or pink sore, with a reported mortality rate of approximately 95%. A study of 32 individuals twenty years after the outbreak showed that porphyria can persist years after the ingestion of HCB. A small cross-sectional study of workers exposed to HCB did not find any evidence of cutaneous porphyria or any other adverse effects associated with exposure of 1 to 4 years.

The acute toxicity of HCB to laboratory animals is quite low, with acute oral LD50 values in the range of more than 2,600 mg/kg body weight in rabbits and 4,000 mg/kg in mice. Porphyria, skin lesions, hyperexcitability and changes in weight, enzyme activities and morphology of the liver have been reported in association with subchronic toxicity of HCB. HCB has also been reported to stimulate the immune system in rats, and suppress the immune system of mice. HCB has also been reported to produce adverse effects on reproduction and reproductive tissue. Female rats fed HCB in the diet experienced

offspring mortality, with a 21 day LD50 of 100 ppm. A four-generation reproduction study in rats fed HCB in the diet was conducted. HCB affected reproduction by reducing the number of litters whelped, litter size and the number of pups surviving to weaning. In a separate study, HCB at a concentration of 100 mg/kg body weight/day was associated with cleft palate and some kidney malformations in CD-1 mice. HCB exposure in several studies in cynomologous monkeys has resulted in degenerative changes in the ovarian surface epithelium, suppression of serum progesterone, atrophy of thymic cortex, a reduction in the number of lymphocytes, degenerative changes in the ovaries and kidney and degenerative changes in the liver compatible with porphyria tarda. IARC has concluded that while there is inadequate evidence for the carcinogenicity of HCB in humans, there is sufficient evidence in experimental animals. IARC has classified HCB as a possible human carcinogen (Group 2B).

HCB is unlikely to cause direct toxicological effects in aquatic animals at or below saturation concentrations (approximately 5 µg/L) in water. At an exposure concentration of 4.8 µg HCB/L for 32 days, there was no observed effect on embryonic through juvenile stages in developing fathead minnows (*Pimephales promelas*) giving a NOEC of 4.8 µg/L. The caldoceran *Daphnia magna*, the amphipods *Hylella azeteca*, and *Gammarus lacustris*, the annelid worm *Lumbricus variegatus*, and the fathead minnow *Pimephales promelas* were exposed to HCB at saturation concentration (5 µg/L) for 68 days. No effects on survival, growth or reproduction were observed. Adult Japanese quail (*Coturnix japonica*) were fed diets containing HCB for 90 days, resulting in increased mortality at 100 µg/g diet and significantly reduced hatchability at 20 µg/g. At 5 µg/g increased liver weight, slight liver damage and increased faecal excretion of coproporphyrin were observed. Experiments conducted in mink (*Mustela vison*) and European ferrets (*Mustela putorius furo*) with dietary HCB resulted in adult mortality are higher doses (125 and 625 mg HCB/kg diet) and decreased litter size, increased percentage of stillbirths, increased kit mortality and decreased kit growth. Mink were generally more susceptible than ferrets to the effects of HCB. Results from another study indicate that *in utero* exposure to HCB resulted in higher kit mortality than exposure via the mothers milk.

HCB is very persistent. Estimated half lives in soil from aerobic and anaerobic degradation range from 2.7 to 22.9 years. This persistence, combined with a high partition coefficient (log KOW = 3.03-6.42), provides the necessary conditions for HCB to bioconcentrate in organisms. Bioconcentration factors of 22,000 and 106,840 have been reported in fathead minnows and *Lumbricus variegatus* respectively. The chemical properties of HCB (low water solubility, high stability, and semi-volatility) favour its long range transport, and HCB has been detected in arctic air, water and organisms.

HCB is ubiquitous in the environment, and has been measured in foods of all types. HCB was one of two organochlorines detected in all samples of Spanish meat and meat products surveyed with mean levels ranging from 8 ppb (fat weight) in pork products (cured ham) to 49 ppb in lamb, with a maximum level of 178 ppb in lamb. HCB was detected in 13 of 241 serum samples from Colorado beef cattle in a monitoring program, with an average concentration of 3.1 ppb. A survey of US pasteurized milk detected HCB in 8 of 806 composite milk samples. A survey of foods from India found average concentrations of HCB ranging from 1.5 ng/g (fat weight) in both oils and milk to 9.1 ng/g in fish and prawns, with a maximum concentration of 28 ng/g in fish and prawns and an estimated daily intake of 0.13 µg/person. Average HCB residues in foods from Vietnam ranged from 0.28 ng/g (fat weight) in pulses to 27 ng/g in caviar, with an estimated daily intake of 0.10 µg/person.

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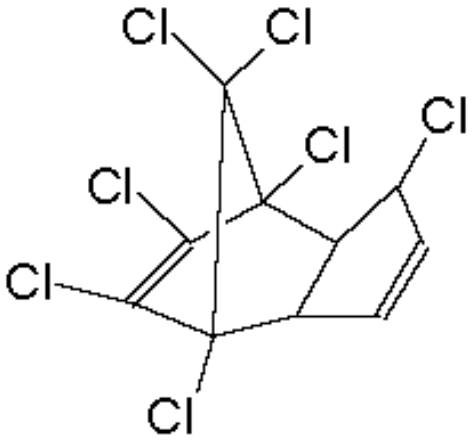
6.8 *HEPTACHLOR*

Chemical properties



CAS Chemical Name: 1,4,5,6,7,8,8-Heptachloro-3a,4,7,7a-tetrahydro-4,7-methanol-1*H*-indene.

Synonyms and Trade Names (partial list): Aahepta, Agroceres, Baskalor, Drinox, Drinox H-34, Heptachlorane, Heptagran, Heptagranox, Heptamak, Heptamul, Heptasol, Heptox, Soleptax, Rhodiachlor, Veliscol 104, Veliscol heptachlor.

CAS No.: 76-44-8; molecular formula: C10H5Cl7; formula weight: 373.32.

Appearance: White to light tan, waxy solid or crystals with a camphor-like odour.

Properties: Melting point: 95-96 C (pure), 46-74 C (technical); boiling point: 135-145 C at 1-1.5 mm Hg, decomposes at 760 mm Hg; KH; 2.3 x 10 -3 atm·mm3/mol; log KOC: 4.38; log KOW; 4.40-5.5; solubility in water: 180 ppb at 25 C; vapor pressure: 3 x 10-4 mm Hg at 20 C.

Heptachlor is a non-systemic stomach and contact insecticide, used primarily against soil insects and termites. It has also been used against cotton insects, grasshoppers, some crop pests and to combat malaria. Heptachlor is highly insoluble in water, and is soluble in organic solvents. It is quite volatile and can be expected to partition into the atmosphere as a result. It binds readily to aquatic sediments and bioconcentrates in the fat of living organisms. Heptachlor is metabolised in animals to heptachlor epoxide, whose toxicity is similar to that of heptachlor, and which may also be stored in animal fat. The use of heptachlor has been banned in Cyprus, Ecuador, the EU, Portugal, Singapore, Sweden, Switzerland and Turkey. Its use is severely restricted in Argentina, Israel, Austria, Canada, Czechoslovakia, Denmark, Finland, Japan, New Zealand, Philippines, USA and USSR.

There is no information on accidental or suicidal intoxication by heptachlor in humans. Symptoms in animals include tremors and convulsions. A study of workers from a plant involved in the production of heptachlor and endrin found a significant increase in bladder cancer . This result was unexpected as no know bladder carcinogens were used at the plant, however, the small number of deaths (3) makes interpretation of these findings difficult. No deaths from liver or biliary tract cancer were observed, although mortality from cerebrovascular disease was higher than expected. There is limited evidence that cyclodienes such as heptachlor may affect immune responses.

The acute oral LD50 of heptachlor to laboratory animals is in the range of 40 mg/kg body weight in rats to 116 mg/kg in rabbits. Groups of male and female rats were administered daily doses of heptachlor orally beginning at 4 months of age, and continuing for 200 days. All the animals in the 50 and 100 mg/kg groups died by the 10th day of exposure. Three animals in the 5 mg/kg group and 1 in the control died before the end of the study. Beginning on the 50th day to the study, hyper-reflexia, dyspnoea and convulsions were observed in the rats exposed to 5 mg/kg. Histological examination revealed fatty degeneration of the liver cells and moderate fatty infiltration of the epithelium of the renal tubules in the 5 mg/kg exposed group.

In a reproduction study, rats were fed diets containing heptachlor in their diet throughout three generations. Mortality of pups in the 10 mg/kg group was slightly increased during the second and third weeks after birth in the second generation only. No adverse effects were observed in the lower dose levels. WHO has reported no evidence of teratogenicity of heptachlor in rats and rabbits. IARC has concluded that, while there is inadequate evidence for the carcinogenicity of heptachlor in humans, there is sufficient evidence in experimental animals. IARC has classified heptachlor as a possible human carcinogen (Group 2B).

Heptachlor has been strongly implicated in the decline of several wild bird populations including Canada geese and the American Kestrel in the Columbia Basin in the US. A population of Canada geese at the Umatilla National Wildlife Refuge in Oregon experienced lowered reproductive success, and adult mortality. Heptachlor epoxide residues were detected in the brains of dead birds and in the eggs of nests with low success. The reproductive success of American Kestrels in the same area was also reduced. Heptachlor epoxide residues in the eggs were associated with reduced productivity. The presence of residues in the eggs indicates that heptachlor is transferred through the food chain, as Kestrels are not seed eaters, which was the presumed route of exposure for the geese. Concentrations on the treated seeds were lower than the recommended usage level indicating that effects on wildlife may occur, even if heptachlor is used responsibly.

Mink were fed diets containing heptachlor for 28 days, followed by a 7 day recovery period to determine the subacute toxicity of heptachlor to mink. The NOEL for mortality was 50 mg/kg (5.67 mg/kg body weight/day). Signs of toxicity including reduced food consumption and loss of body weight were observed in mink fed the 25 mg/kg diet. In another study, adult male and female mink were fed diets containing heptachlor for 181 days (before and during the reproductive period) to determine effects on reproduction. All the mink fed diets containing 25 µg/g (male and female) died, within 88 and 55 days respectively. The LOAEL, based on reduced kit growth, was 6.25 µg/g.

The half life of heptachlor in temperate soil is up to 2 years. This persistence, combined with a high partition coefficient (KOW = 4.4-5.5), provides the necessary conditions for heptachlor to bioconcentrate in organisms. Bioconcentration factors of heptachlor and heptachlor epoxide in fathead minnows (*Pimephales promelas*) were 9,500 and 14,400, respectively. The chemical properties of heptachlor (low water solubility, high stability, and semi-volatility) favour its long range transport, and heptachlor and its epoxide have been detected in arctic air, water and organisms.

WHO suggests that food is the major source of exposure of heptachlor to the general population. Heptachlor has been detected in the blood of cattle from both the US and Australia. Heptachlor was detected in 30 of 241 samples in American cattle, and violations of the MRL for heptachlor were detected in 0.02 % of Australian cattle. In both instances, heptachlor was among the most frequently detected organochlorine. A daily intake of 0.25 µg/person/day (for heptachlor and heptachlor epoxide combined, based on a 60 kg person) was estimated for Vietnam, and of 0.07 µg/person/day (for heptachlor alone) for India.

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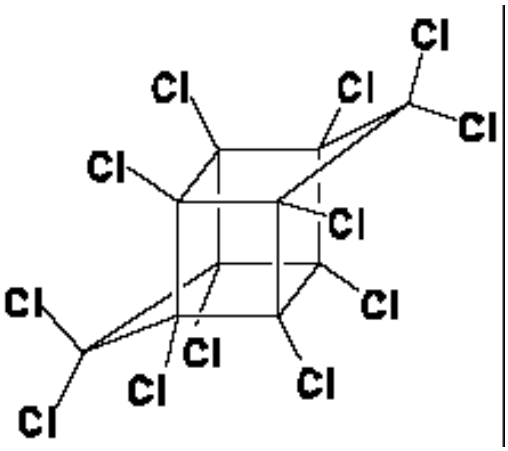
6.9 *MIREX*

Chemical properties



CAS chemical name: 1,1a,2,2,3,3a,4,5,5a,5b,6-dodecachloroacta-hydro-1,3,4-metheno-1H-cyclobuta[cd]pentalene

Synonyms and Trade Names (partial list): Dechlorane, Ferriamicide, GC 1283

CAS No.: 2385-85-5; molecular formula: C10Cl12; formula weight: 545.5

Appearance: White crystalline, odourless solid;

Properties: Melting point: 485 C; vapour pressure: 3 x 10-7 mm Hg at 25 C.

Mirex is a stomach insecticide with little contact activity. It's main use was against fire ants in the southeastern United States, but it has also been used to combat leaf cutters in South America, harvester termites in South Africa, Western harvester ants in the US, mealybug of pineapple in Hawaii and has been investigated for possible use against yellow jacket wasps in the US. It has also been used as a fire retardant in plastics, rubber, paint paper and electrical goods. Mirex is very resistant to breakdown, is very insoluble in water and has been shown to bioaccumulate and biomagnify. Due to its insolubility, mirex binds strongly to aquatic sediments.

There are no reports of injuries to humans resulting from exposure to mirex. Mirex residues in human adipose have been reported. A range of 0.16 - 5.94 ppm was reported in 6 of 1,400 samples collected in 1971-1972 in the southern US. Samples from 8 southeastern US states were collected, and residues detected in 10.2 percent of those tested, with a geometric mean of 0.286 ppm in lipid.

In acute studies, the oral LD50 of mirex to rats ranges from 600 to >3,000 mg/kg, depending on sex of the test animal and nature of the formulation tested. Short term effects included decreased body weight, hepatomegaly, induction of mixed function oxidases, and morphological changes in liver cells. Rats which were fed 5 ppm mirex in their diets for 30 days prior to mating and for 90 days after, showed reduced litter size and increased parental mortality. Reduced litter sizes, and viability of neonates, along with formation of cataracts were observed in rats fed 25 ppm mirex in the diet. IARC has concluded that while there is inadequate evidence for the carcinogenicity of mirex in humans, there is sufficient evidence in experimental animals. IARC has classified mirex as a possible human carcinogen (Group 2B).

A reduction in germination and emergence in several plant species was observed, which increased as the concentrations of mirex increased. Uptake, accumulation and translocation of mirex by a variety of plant species has also been seen. These results are questionable, however, as lipophilic compounds such as mirex are generally not known to be taken up and translocated by plants. Contamination of plants is primarily a surface phenomenon resulting from aerial deposition of emissions or deposition of compound that has volatilized from the surface of the soil.

Crustaceans are the most sensitive aquatic organisms, with larval and juvenile stages being the most sensitive. Delayed

mortality is typical of mirex poisoning in crustaceans. Larval crabs exposed to 0.1 and 10 µg/L did not exhibit any adverse effects on survival for 5 days after hatching. Delayed mortality then occurred at the 1 and 10 µg/L exposure levels. Mirex is also toxic to fish and can affect fish behaviour. Mirex has a low short term toxicity to birds with acute oral LD50 values in the range of 1,400 mg/kg body weight in pheasant to 10,000 mg/kg in quail.

Mirex is considered to be one of the most stable and persistent pesticides, with a half life of up to 10 years. This persistence, combined with lipophilicity, provides the conditions necessary for mirex to bioconcentrate in organisms. Bioconcentration factors of 2,600 and 51,400 have been observed in pink shrimp and fathead minnows, respectively. The chemical properties of mirex (low water solubility, high lipid solubility, high stability, and semi-volatility) favour its long range transport, and mirex has been detected in arctic freshwater and terrestrial organisms. The main route of exposure of mirex to the general population is through food, especially meat, fish and wild game, and intake is generally below established residue tolerances. Mirex residues were found in only one of 806 milk sample composites collected in a survey of US pasteurized milk. No residues of mirex were detected in any samples of fish in Egypt nor in any samples from the fat of domestic farm animals in Ontario, Canada.

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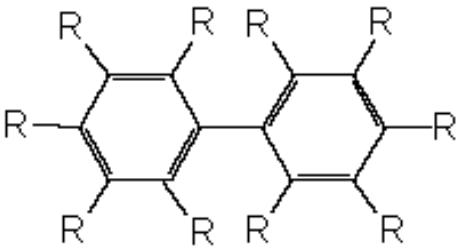
6.10 *POLYCHLORINATED BIPHENYLS*

Chemical properties



Trade Names for different mixtures (partial list): Aroclor, Pyranol, Pyroclor, Phenochlor, Pyralene, Clophen, Elaol, Kanechlor, Santotherm, Fenchlor, Apirolio, Sovol.

CAS No.: 1336-36-3

Congener Group	Molecular weight (g/molecular)	Vapour Pressure (Pa)	Water Solubility (g/m3)	log KOW
Monochlorobiphenyl	188.7	0.9-2.5	1.21-5.5	4.3-4.6
Dichlorobiphenyl	223.1	0.008-0.60	0.06-2.0	4.9-5.3
Trichlorobiphenyl	257.5	0.003-0.22	0.015-0.4	5.5-5.9

Tetrachlorobiphenyl	292.0	0.002	0.0043-0.010	5.6-6.5
Pentachlorobiphenyl	326.4	0.0023-0.051	0.004-0.02	6.2-6.5
Hexacholorbiphenyl	360.9	0.0007-0.012	0.0004-0.0007	6.7-7.3
Heptachlorobiphenyl	395.3	0.00025	0.000045-0.000	6.7-7
Octachlorobiphenyl	429.8	0.0006	0.0002-0.0003	7.1
Nonachlorobiphenyl	464.2	-	0.00018-0.0012	7.2-8.16
Decachlorobiphenyl	498.7	0.00003	0.000001-0.000	8.26

Polychlorinated biphenyls (PCBs) are mixtures of chlorinated hydrocarbons that have been used extensively since 1930 in a variety of industrial uses, including as dielectrics in transformers and large capacitors, as heat exchange fluids, as paint additives, in carbonless copy paper and in plastics. The value of PCBs for industrial applications is related to their chemical inertness, resistance to heat, non-flammability, low vapour pressure and high dielectric constant. There are 209 possible PCBs, from three monochlorinated isomers to the fully chlorinated decachlorobiphenyl isomer. Generally, the water solubility and vapour pressure decrease as the degree of substitution increases, and the lipid solubility increases with increasing chlorine substitution. PCBs in the environment may be expected to associate with the organic components of soils, sediments, and biological tissues, or with dissolved organic carbon in aquatic systems, rather than being in solution in water. PCBs volatilize from water surfaces in spite of their low vapour pressure, and partly as a result of their hydrophobicity; atmospheric transport may therefore be a significant pathway for the distribution of PCBs in the environment.

The toxicology of PCBs is affected by the number and position of the chlorine atoms, as substitution in the *ortho* position hinders the rotation of the rings. PCBs without *ortho* substitution are generally referred to as coplanar and all others as noncoplanar. Coplanar PCBs, like dioxins and furans, bind to the AL-receptor and may exert, thus, dioxin-like effects, in addition to AL-receptor independent effects which they share with non-coplanar PCBs (e.g. tumor promoter). Association between elevated exposure to PCB mixtures and alterations in liver enzymes, hepatomegaly, and dermatological effects such as rashes and acne has been reported. Adverse effects are predominantly associated with higher blood concentrations.

Contamination of rice oil by PCBs in Japan (1968) and Taiwan (1979) has resulted in the exposure of a large number of people to PCBs and their contaminants PCDFs. Signs and symptoms of exposure from these incidents include enlargement and hyper secretion of the Meibomian glands of the eyes, swelling of the eyelids, and pigmentation of the nails and mucous membranes, occasionally associated with fatigue, nausea and vomiting. This was followed by hyperkeratosis and darkening of the skin with follicular enlargement and acneform eruptions, often with a secondary staphylococcal infection. Children born up to 7 years after maternal exposure in the Taiwan incident had hyperpigmentation, deformed nails and natal teeth, intrauterine growth delay, poorer cognitive development up to 7 years of age, behavioural problems and higher activity levels. The affected children appeared to "catch up" to controls at 12 years of age. Children born seven to twelve years after maternal exposure experienced mildly delayed development, but no differences in behaviour. Effects observed in these children is likely a result of the persistence of PCBs in the human body, resulting in prenatal exposure long after the exposure took place. These effects are consistent with the observations of poorer short term memory functioning in early childhood, in children exposed prenatally by mothers who had high consumption of Lake Michigan sports fish containing PCBs, amongst other POPs.

People exposed in the Yucheng incident had low resistance, and suffered from a variety of infections. Examination during the first year revealed decreased concentrations of IgM and IgA, decreased percentages of total T-cells, active T-cells and helper T-cells, but normal percentages of B-cells and suppressor T-cells; suppression of delayed type response to recalling antigens; enhancement of lymphocyte spontaneous proliferation and an enhancement in lymphoproliferation to certain mitogens. After three years, some, although not all, of the effects had disappeared. Cancer deaths in both male and female workers involved in the manufacture of electrical capacitors were significantly increased. A significant increase in haematological neoplasms and gastrointestinal cancers was observed in male workers. A non-significant increase in lung cancer was observed. The study was, however, limited by the small numbers of deaths.

PCBs have a low acute toxicity to laboratory animals, with acute oral LD50 values in rats in the range of 2 to 10 g/kg body weight. Effects are manifested primarily through chronic exposure. Effects on the liver, skin, immune system, reproductive system, gastrointestinal tract and thyroid gland have been observed associated with exposure to PCB mixtures or individual congeners. Adverse reproductive effects observed in several studies on monkeys exposed to PCBs include low birth weights, skin hyperpigmentation, behavioural disturbances, atrophy of the thymus and lymph nodes, bone marrow hypoplasia and hyperplasia of the gastric mucosa. Female rhesus monkeys fed diets containing Aroclor 1016 in the diet were bred after 7 months of dietary exposure. Neonatal weights in the 1.0 ppm group were significantly decreased. PCBs have not been observed to be teratogenic in studies involving rats and non-human primates when tested orally, during critical periods of organogenesis. A moderate but statistically significant inhibitory effect on the immune system of rhesus monkeys has been observed, resulting from chronic, low level exposure to Aroclor 1254 and that these effects may be due to altered T-cell and/or macrophage function. IARC has concluded that there is limited evidence for the carcinogenicity of PCBs in humans, and there is sufficient evidence in experimental animals. PCBs are therefore classified as probable humans carcinogens (Group 2A).

PCBs are toxic to aquatic organisms, with 96-hour LC50 values in the range of 0.015 mg/L in fathead minnows to 2.74 mg/L in bluegills. Fathead minnows were exposed to Aroclor 1242, 1248 or 1254 in a continuous flow bioassay for 9 months. Reproduction occurred at and below 5.4 µg Aroclor 1242/L, however, results were highly variable. A significant reduction in spawning was observed in fish exposed to 1.8 µg Aroclor 1254/L. Early life stages of fish are more sensitive to the effects of dioxins, furans, and PCBs. Parts per trillion concentrations of these structurally related chemicals in lake trout and rainbow trout eggs produce toxicity through sac fry mortality associated with yolk sac edema and haemorrhages.

PCBs have a low acute toxicity to birds, with 5-day dietary LC50 values in the range of 747 mg/kg diet in quail to >5,000 mg/kg in several species. Broiler breeder and leghorn hens who were fed diets Aroclor 1242 for one week experienced reduced hatchability and the effects continued after exposure was terminated.

There is growing evidence linking persistent halogenated aromatic hydrocarbons such as PCBs to reproductive and immunotoxic effects in wildlife. Two groups of 12 female seals (*Phoca vitulina*) were fed diets of fish from the western part of the Wadden Sea, or from the north-east Atlantic. Residue analysis showed statistically significant differences between the two diets for PCBs and DDE. The average daily intake for group 1 was 1.5 mg PCBs and 0.4 mg DDE, and 0.22 mg and 0.13 mg for group 2. Females were mated with undosed males and reproductive success was significantly lower in group 1. Mink fed Lake Michigan Coho salmon containing between 10 and 15 ppm PCBs as 30% of their diet for five months failed to whelp as did those fed a diet containing 5 ppm Aroclor 1254. The clinical signs and lesions observed in mink fed a diet containing Lake Michigan coho salmon included anorexia, bloody stools, fatty liver, kidney degeneration and gastric ulcers, and were similar to those fed a diet supplemented with PCBs.

The degradation of PCBs in the environment depends largely on the degree of chlorination of the biphenyl, with persistence increasing as the degree of chlorination increases. Half-lives for PCBs undergoing photodegradation range from approximately 10 days for a monochlorobiphenyl to 1.5 years for a heptachlorobiphenyl. The persistence of PCBs, combined with the high partition coefficients of various isomers (log KOW ranging from 4.3 to 8.26) provide the necessary conditions for PCBs to bioaccumulate in organisms. Bioconcentration factors of 120,000 and 270,000 have been reported

in fathead minnows. Concentration factors in fish exposed to PCBs in their diet were lower than those for fish exposed to PCBs in water, suggesting that PCBs are bioconcentrated (taken up directly from the water) as opposed to being bioaccumulated (taken up by water and in food). The chemical properties of PCBs (low water solubility, high stability, and semi-volatility) favour their long range transport, and PCBs have been detected in arctic air, water and organisms.

The main source of PCB exposure to the general population is through food, especially fish. PCB residues were detected in 8.5% of samples, with a maximum of 0.30 mg/kg fat, taken during a survey of the fat of domestic farm animals in Ontario, Canada between 1986 and 1988. In a survey of foods in Vietnam, the highest levels of PCBs were detected in fish and shellfish, with levels of 760 and 1,400 ng/g fat. The main sources of PCBs in the Vietnamese diets is cereals (including rice) and vegetables, and the daily intake of 3.7 µg/person/day is comparable to those of some industrialized countries . A survey of foods in India also found that the highest levels of PCBs were in fish, with an average of 330 ng/g fat. Again, the main source of PCB dietary intake (0.86 µg/person/day) was cereal and vegetable oil.

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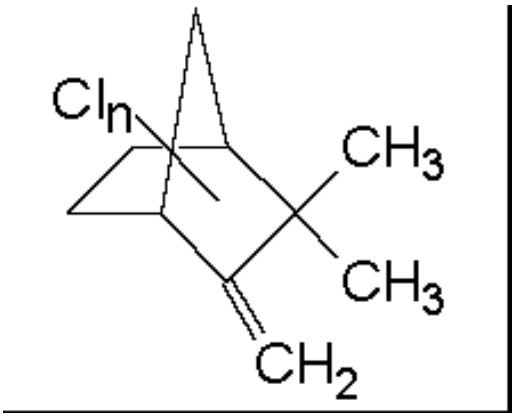
6.11 *TOXAPHENE*

Chemical properties



CAS Chemical Name: Toxaphene

Synonyms and Trade Names (parital list): Alltex, Alltox, Attac 4-2, Attac 4-4, Attac 6, Attac 6-3, Attac 8, Camphechlor, Camphochlor, Camphoclor, Chemphene M5055, chlorinated camphene, Chloro-camphene, Clor chem T-590, Compound 3956, Huilex, Kamfochlor, Melipax, Motox, Octachlorocamphene, Penphene, Phenacide, Phenatox, Phenphane, Polychlorocamphene, Strobane-T, Strobane T-90, Texadust, Toxakil, Toxon 63, Toxyphen, Vertac 90%.

CAS No.: 8001-35-2; molecular formula: C10H10Cl8; formula weight: 413.82

Appearance: Yellow, waxy solid with a chlorine/terpene-like odour.

Properties: Melting point: 65-90 C; boiling point: >120 C (decomposes); KH: 6.3 x 10-2 atm·m3/molecular; log KOC: 3.18 (calculated); log KOW: 3.23-5.50; solubility in water: 550 µg/L at 20 C; vapour pressure: 0.2-0.4 mm Hg at 25 C.

Toxaphene is a nonsystemic and contact insecticide that was used primarily on cotton, cereal grains fruits, nuts and vegetables. It has also been used to control ticks and mites in livestock. Toxaphene has been in use since 1949 and was the

most widely used insecticide in the USA in 1975. Toxaphene is highly insoluble in water, and has a half life in soil of up to 12 years. It has been shown to bioconcentrate in aquatic organisms and is known to undergo atmospheric transport. Toxaphene has been banned in 37 countries, including Austria, Belize, Brazil, Costa Rica, Dominican Republic, Egypt, the EU, India, Ireland, Kenya, Korea, Mexico, Panama, Singapore, Thailand and Tonga. Its use has been severely restricted in 11 other countries, including Argentina, Columbia, Dominica, Honduras, Nicaragua, Pakistan, South Africa, Turkey and Venezuela.

In a human volunteer study, twenty-five subjects were exposed to approximately 1 mg toxaphene/kg body weight/day in a closed chamber to an aerosol of toxaphene for a total of 13 days. Physical examination, blood and urine tests did not reveal any toxic effects. In a separate study, eight women working in an area that had been sprayed with toxaphene at a rate of 2 kg/ha had a higher incidence of chromosome aberrations (acentric fragments and chromatid exchanges) than in control individuals. Annual physical examination of 137 workers involved in the manufacture of toxaphene did not reveal adverse effects associated with the exposure . Similarly, a mortality survey of 199 employees who had worked with toxaphene found that none of the deaths appeared to be directly related to the exposure.

The acute oral toxicity of toxaphene is in the range of 49 mg/kg body weight in dogs to 365 mg/kg in guinea pigs. In a 13 week study, rats were fed diets containing toxaphene. Liver/body weight ratio and hepatic microsomal enzyme activities were increased in rats fed 500 ppm. Dose dependent histological changes were observed in the kidney, thyroid and liver. The NOAEL was determined to be 4.0 ppm (0.35 mg/kg). In another study, beagle dogs were fed toxaphene for 13 weeks. The liver/body weight ratio and serum alkaline phosphatase were increased in dogs fed 5.0 mg/kg. Mild to moderate dose dependent histological changes were observed in the liver and thyroid. The NOAEL for dogs was determined to be 0.2 mg/kg. Male and female rats were fed toxaphene in their diets for a total of 13 weeks in a one generation, two litter reproduction study. Effects in both the F0 and F1 adults at levels from 20 to 500 ppm included increased liver and kidney weight, and histological changes in the thyroid, liver and kidney. IARC has concluded that while there is inadequate evidence for the carcinogenicity of toxaphene in humans, there is sufficient evidence in experimental animals. IARC has classified toxaphene as a possible human carcinogen (Group 2B).

Toxaphene is essentially nontoxic to plants. In general, toxic effects have been observed only at levels much higher than the recommended usage level. Toxaphene is highly toxic, with 96-hour LC50 values in the range of 1.8 µg/L in rainbow trout to 22 µg/L in bluegill. Brook trout exposed to toxaphene for 90 days experienced a 46% reduction in weight at 0.039 µg/L, the lowest concentration tested. Egg viability in female trout was significantly reduced upon exposure to a concentration of 0.075 µg/L or more. Long term exposure to 0.5 µg/L reduced egg viability to zero. Female ring-necked pheasants exposed to 300 mg toxaphene/kg diet experienced reductions in egg laying and hatchability.

The half-life of toxaphene in soil ranges from 100 days up to 12 years, depending on the soil type and climate. This persistence, combined with a high partition coefficient (log KOW = 3.23-5.50) suggests that toxaphene is likely to bioconcentrate. Bioconcentration factors of 4,247 and 76,000 have been recorded in mosquito fish and brook trout, respectively. The chemical properties of toxaphene (low water solubility, high stability, and semi-volatility) favour its long range transport, and toxaphene has been detected in arctic air. Exposure of the general population is most likely through food, however levels detected are generally below maximum residue limits. Due to its being banned in many countries, recent food surveys have generally not included toxaphene and hence recent monitoring data are not available.

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7. CONCLUSIONS

The POPs described in this Assessment Report are characterized by their lipophilicity, persistence and semi-volatility. These characteristics pre-dispose these substances to long environmental persistence and to long range transport. These substances are also known for their ability to biomagnify and bioconcentrate under typical environmental conditions, thereby potentially achieving toxicological relevant concentrations. The semi-volatility of these substances facilitates their long range transport to and accumulation in the cooler and polar regions of the world, far removed from any source of use.

A number of the substances described in this report have been implicated in a broad range of adverse human health and environmental effects including impaired reproduction and endocrine dysfunction, immunosuppression and cancer. In many cases, the substances are considered as possible human carcinogens by the International Agency for Research on Cancer.

Many of the substances described in this report are still in use in at least some countries. The paucity of reliable data regarding use and disposal has meant that it has not been possible to accurately determine the quantities still in use, where they are used, the specific crops to which the pesticidal substances are being applied, and the direction and initiatives underway to eliminate these substances throughout the world. Where data does exist, it is plagued with a variety of limitations making it difficult to develop comprehensive and accurate use profiles.

While convincing substantive evidence exists for the actual and potential toxic impact of these substances to both human health and the environment, a comprehensive, accurate and reliable inventory of global manufacture, use and disposition, must be developed to allow the effective and efficient elimination of these substances throughout the world.

Several risk reduction strategies are available for the POPs. They involve greater use of alternatives to substances still in use and proper disposal of POPs in storage or in closed systems (e.g. PCBs). As this is a global problem, these strategies need to be coordinated on a global level and must be tailored to the socio-economic considerations of user nations.

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