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EXPLORATORY REPORT CHLORINATED PARAFFINS

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SUMMARY

This report contains general information on chlorinated paraffins (CPs) concerning the existing standards, emissions, exposure levels and effect levels. The document is to be considered as a start for the national discussion during an exploratory meeting on integrated criteria documents and serves internationally as a background document for the working group on diffuse sources of the Paris commission (PARCOM).

In general data on CPs are very scarce and incomplete, hampering a sound risk evaluation.

In the Netherlands no production of CPs takes place. The major application in the Netherlands is their use as secundary plasticizer for PVC. Other applications are plasticizer, lubricants and flame retardants applied in paints, coatings, rubber, oils, etc. However, no emission data on CPs in the Netherlands are available. It is expected that the total emission from application industries will decrease as a result of emission reducing measures. The major emissions are likely to occur through use and disposal of products containing chlorinated paraffins.

Information on the persistence and bioconcentration is scanty. CPs are expected to (bio)degrade in sediment and soil, whereas secondary photolysis in the atmosphere cannot be ruled out. Reported calculated half-life times are approx. 5 days and 1 day, respectively. However, the actual half-lives in the natural environment will be much lower. The biodegradation rate has been shown to decrease with increasing carbon length and chlorine content.

The bioconcentration factor decreases with increasing molecular weight and decreasing lipophilicity; short-chain CPs with a high level of chlorination show the highest accumulation/retention potential. However, the relatively low levels found in higher organisms do not indicate the occurrence of biomagnification.

No figures for the occurrence of CPs in the Dutch environment have been found. Indicative exposure levels were obtained from a survey held in the largest producer of western Europe (UK) and from rough model calculations.

The toxicity of CPs is inversely related to chain length. Therefore in risk evaluation the attention has primarily been focused on the shorter chain grades.
No data on effects on humans are available. Animal studies indicate that short chain CPs may have carcinogenic potential but are non-genotoxic. A tolerable daily intake (TDI) of 0.01 $mg.kg^{-1}.bw.day^{-1}$ at life-time exposure was calculated. The total daily intake in the Netherlands is expected to be (far) below this level.

Ecotoxicological data are limited to acute toxicity data for freshwater organisms. For short-chain CPs provisionally maximum permissible concentrations of 1 $\mu g.l^{-1}$ and 50 $mg.kg^{-1}$ were derived for surface water and sediment/soil, respectively. Environmental concentrations in the Netherlands are expected not to exceed these levels.

It is recommended to make a quantitative inventory of the emissions in the Netherlands and preferably in Europe.

1. INTRODUCTION

This report concerns chlorinated paraffins (CPs), those are characterized by the carbon-chain length range of their n-alkanes and by their chlorine content. A general classification of chlorinated paraffins by carbon-chain length and degree of chlorination is presented in table 1.1. The chain lengths of commercial paraffin products are between 9 and 38 carbon atoms and chlorine contents between 10 and 72% (WHO, 1990).

Table 1.1 Chlorinated paraffins categories (WHO, 1990; Willis, 1991; Beenen, 1991; Annema, 1989)

Carbon chain length	Average % chlorine
C10-C13	50-70
C14-C17	45-60
C20-C30	40-70

This scoping report is part of the preparation for drawing up a integrated criteria document chlorinated paraffins in the framework of the national environmental policy. The objective of this report is to bring the knowledge of the participants in the scoping meeting to the same level, and to put forward points for discussion and decision-making as to the contents of the integrated criteria document.

The report, however is triggered by international activities. In the working group on diffuse sources of the Paris commission (PARCOM) Sweden prepared a document on chlorinated paraffins and proposed to abandon the use of these substances within a 3-10 year period (this was published in the KEMI-report, 1991). An industrial point of view presented by Draka-Polva motivated the Directorate-General for Environmental Protection to request the author to extend the Draka-Polva report (Beenen, 1991). Both reports present different views. This information was used as a basis for the present scoping document, whereas additional information was obtained by the CEFIC sector group "Chlorinated Paraffins".

It should be emphasized that the present report does not aim to be exhaustive: the actual standards and recommendations, the sources and exposure levels in the Netherlands are merely outlined, whereas on the other hand the principal effects and indicative (no) effect levels are described. Subsequently, based on a risk assessment the problems will be pointed out and gaps in knowledge will be identified.

2. ACTUAL STANDARDS AND GUIDELINES

There are no standards and guidelines for chlorinated paraffins in force in The Netherlands. There are, however, standards for aliphatic halogenated hydrocarbons, that are applicable to drinking water and chemical waste.

Table 2.1 gives an overview of the actual standards and guidelines for halogenated hydrocarbons in The Netherlands (VROM, 1991). Chlorinated paraffins are not considered "black-list" substances or "priority substances", but have been placed on the list of "attention substances".

Table 2.1 Actual standards and guidelines for halogenated hydrocarbons in The Netherlands

Environmental compartment/ type of standard	Concentration	Reference
SOIL AND GROUNDWATER		
Soil		Leidraad Bodem-bescherming (1990)
- Reference value	detection limit	
- (A value, multifunctional)		
- B value, suspicion of risk *	5 $mg.kg^{-1}$ d.w.	
- B value, suspicion of risk **	7 $mg.kg^{-1}$ d.w.	
- C value, unacceptable risk *	50 $mg.kg^{-1}$ d.w.	
- C value, unacceptable risk **	70 $mg.kg^{-1}$ d.w.	
Groundwater		Leidraad Bodem-bescherming (1990)
- Reference value (A value) *	0.01 $\mu g.l^{-1}$	
- B value *	10 $\mu g.l^{-1}$	
- B value **	15 $\mu g.l^{-1}$	
- C value *	50 $\mu g.l^{-1}$	
- C value *	70 $\mu g.l^{-1}$	
FOOD AND DRINKING WATER		
Drinking water *	1.0 $\mu g.l^{-1}$	WB (1984)
PRODUCTS		
Chemical waste * (minimal concentration)	5,000 $mg.kg^{-1}$ d.w.	BACA (1991) WCA (1977)

* : per substance
** : total
d.w.: dry weight

3. APPLICATIONS, SOURCES AND EMISSIONS

3.1 PRODUCTION

Chlorinated paraffins are not known to occur as natural products (WHO, 1990).

Chlorinated paraffins have been produced commercially since the 1930s. These mixtures of chlorinated n-alkanes are produced by reacting normal paraffin fractions obtained from petroleum distillation with gaseous chlorine exothermically at 80-120 °C in the liquid phase (WHO, 1990; Beenen, 1991) (50-100 °C according to Willis (1991)). Generally no catalyst is used but some manufacturers may promote the process with ultraviolet light (Beenen, 1991). The chlorination process generates HCl, and the maximum chlorine content which can be achieved is 75-76% (Willis, 1991). Production of resinous chlorinated paraffins (70% chlorine content) requires the use of a solvent such as carbon tetrachloride during chlorination (WHO, 1990), which can still be present in the product at concentrations typically below 0.2%.

No production of chlorinated paraffins takes place in The Netherlands.
Estimated data on the production in Europe and world-wide are shown in table 3.1. No data are available on the production in the former USSR and Japan.

Table 3.1 Production of chlorinated paraffins (estimated)

Country/continent	Production (tonnes.year^{-1})	References
Netherlands	0	Annema, 1989
United Kingdom (1983)	100,000	Willis, 1991
(1990)	50,000	CEFIC, pers. comm.
Western Europe (1985)	95,000	WHO, 1990
USA (1977)	37,000	Kirk-Othmer, 1979
USA (1983)	45,000	NTP, 1986
USA (1987)	45,000	WHO, 1990
USA (1987)	91,000-98,000	SRI, 1991
USA (1990)	43,000	CEFIC, pers. comm.
World (1977)	230,000	Campbell & McConnell, 1980
World (1978)	250,000	NTP, 1986
World (1985)	300,000	WHO, 1990; Beenen, 1991

0 : no production

From table 3.1 it can be concluded that the United Kingdom and the United States of America are the major chlorinated paraffins producers in the world. Although the figures indicate that the production of chlorinated paraffins seems to increases world-wide, this apparent increase may very well be due to the publication of more accurate data in later years.

3.2 APPLICATIONS

The use volume of chlorinated paraffins in The Netherlands is estimated to be between 1,300 and 2,900 tonnes per year (Annema, 1989). The last estimations however, are between 1,300 and 3,400 tonnes per year (Beenen, 1991a; Beenen, pers. comm. 1991; Ceha, pers comm. 1991). The most important applications of chlorinated paraffins are shown in table 3.2.

Table 3.2 Applications of chlorinated paraffins in The Netherlands, and the estimated quantity of chlorinated paraffins used for these applications in 1987/1988 (Annema, 1989)

Application	Quantity (in tonnes.year^{-1})	
Secundary plasticizer for polyvinyl chloride	1,000-2,000	(2,500)[a,b]
Plasticizer in chloro rubber paints	100-500	(≈ 400)[b]
Plasticizer in acrylic resins	100-150	
Extreme pressure lubricant additives in the metal working industry	100-150	(100-200)[c], (100)[b]
Flame retardants	10-40	
Additives in poly-urethane coatings	≈ 50	(≈ 100)[b]
Handle modifier in fat liquors for improving the suppleness of leather	10-100	

a : according to last estimations (Beenen, 1991a)
b : Ceha, pers. comm. 1991
c : Beenen, pers. comm. 1991b

The main application for chlorinated paraffins in the USA is currently as extreme pressure lubricant additives in the metal working industry (45%) and flame retardants (20%) (Beenen, 1991). In contrast, 50% of the chlorinated paraffins consumed in western Europe are used as secondary plasticizers in PVC and other plastics (WHO, 1990). In The Netherlands the percentage in PVC is even higher (70%) (Beenen, 1991a; Ceha, pers. comm. 1991).

The C10-C13 and the C20-C30 groups of compounds are used most in the application of extreme pressure lubricant additives in the metal working industry and flame resistant additives in paints and sealants (world consumption in 1977 (except for Eastern Europe and the former USSR) of 120,000 tonnes per year). The C14-C17 group of compounds are mostly used in secondary plasticizers in PVC (world consumption in 1977 of 110,000 tonnes per year) (Willis, 1991; Annema, 1989). At this

moment it is not clear which chlorinated paraffins are mostly used in The Netherlands.

The flame retardant properties of highly chlorinated paraffins (70%) are important for their use in plastics, fabrics, paints and coatings, this often in combination with antimony trioxide and sometimes to replace brominated additives (WHO, 1990; CEFIC, pers. comm.). Although it has been reported that CPs are used to replace PCBs in condensators and transformers, (Ham et al., 1991; NTP, 1986a; NTP, 1986b), this application is doubted as CPs are unsuitable for such an application because of their limited thermal stability.

The percentage of chlorinated paraffins in products varies considerably. Chloro rubber paints consist for 50% of chlorinated paraffins. The percentage of chlorinated paraffins in extreme pressure lubricants varies between 5-70%, depending on what purpose they are used for (Annema, 1989).

At this moment there is no insight into which groups of chlorinated paraffins are most applied.

3.3 EMISSIONS AND WASTE STREAMS

No emission data on chlorinated paraffins for The Netherlands are available.

Atmospheric emissions have been reported from chlorinated paraffin using processes. In Germany the observed emissions were in the order of 30 $mg.Nm^{-3}$. In England these emissions are estimated to be considerably lower. In the process a loss of about 0.1 $g.kg^{-1}$ of chlorinated paraffins produced was observed (Beenen, 1991). In Sweden the largest single consumer of chlorinated paraffins (1,400 tonnes per year) has estimated that its emission of chlorinated paraffins amounted to 90 kg per year (0.06 $g.kg^{-1}$) (KEMI, 1991).

Table 3.3 summarizes the possible emissions of chlorinated paraffins in the United Kingdom. The Dutch situation may be similar, except for the fact that no production takes place.

It may be stated that the major environmental release of chlorinated paraffins originates from the use and disposal of products which contain them. This occurs through application, through disposal to landfill, or through discarded lubricant oils being washed into watercourses (Willis, 1991).
Based on the application data, it is estimated that maximally 1,000 tonnes of chlorinated paraffins are released into the environment annually in The Netherlands (most of the chlorinated paraffins containing products are likely to be exported; no data are available of the import/export of chlorinated paraffins in The Netherlands), of which 900 tonnes (waste) will be disposed of under controlled circumstances and

maximally 100 tonnes are emitted uncontrolled. These worst case estimations are used in a McKay model (see Chapter 4).

Table 3.3 Process causes for the emission of chlorinated paraffins to the different environmental compartments in the UK (after Willis, 1991) (no data available)

Process	Emission (quantity) water	soil	air	waste
Production losses	0	-	0	-
Application losses	0	0	+	+
Consumption losses	+	0	+*	+
Disposal losses	0	0	+*	+
Transportation and storage	0	0	0	+

- : probably no emission
0 : possible emission (relatively low)
+ : emission
* : mainly emission of lower chlorinated carbons

In contrast to the qualitative emission estimates in the United Kingdom (table 3.3), an American field study conducted in 1986 in a waterway that carried discharges from a producer of chlorinated paraffins showed the presence of chlorinated paraffins contents ranging in orders of magnitude from ppb to ppm. These were found, in most cases, in sediments, suspended particles and biological matter (KEMI, 1991; Beenen, 1991). Chlorinated paraffins have also been found in sewage sludge sediments in the cities Liverpool, Manchester and Zürich (see chapter 4).
As secondary plasticizers, chlorinated paraffins are not chemically bound into plastics, and are able to volatilize to some extent (Willis, 1991). Howard (cited in Willis (1991)) estimated that 10% of the chlorinated paraffins produced could be released in this way. According to Campbell and McConnell (1980) however, no leaching to the environment is possible, because chlorinated paraffins are interlocked with organic matrixes. According to these authors the volatilities of CPs are very low, reporting a vapour pressure of 3×10^{-9} mm Hg above a PVC compound containing CPs. It is also stated by the Dutch chemical industry that there is probably no or little emission by the application of secondary plasticizer for polyvinyl chloride in the environment.

According to Svanberg and Bergman (cited in Beenen (1991)) chlorinated paraffins may leach from paint products. Howard and Zitko (cited in Beenen (1991)) suggest that due to their low water solubility and strong binding capacity, chlorinated paraffins are not likely to leach from landfills and dumps to any great extend.

Chlorinated paraffins are used as extreme pressure lubricant additives in the metal working industry, it is estimated that a maximum of 55% of the amounts of cutting and lubricating oils sold to engineering industry becomes waste (KEMI, 1991).

The discarded oils are likely to be disposed of in dumps or landfills, as lubricants are rarely reused (Willis, 1991). The rest is consumed or released into the air and water (KEMI, 1991). The estimates of chlorinated paraffins that reach the environment also include amounts distributed via treated products, metal chips, grindings, leaching, messing around and evaporation (KEMI, 1991; Bremmer, 1988).

Products in which chlorinated paraffins are used as flame retardant agents have long life expectancies (KEMI, 1991). Howard (cited in Willis (1991)) however, states that these chlorinated paraffins are able to volatilize in the environment to some extent (Willis, 1991).

In the leather industry in The Netherlands chlorinated paraffins are used in fat liquors for improving the suppleness of leather. Chlorinated paraffins belong to the group of EOCl compounds. The emission of EOCl (extractable organic chlorine) in the sludge from a sewage plant in Rijen is estimated at 75-100 kg per year. This is 2-3% of the total amount (3,750 kg) which is used locally. It is expected that an important part of the EOCl originates from the use of chlorinated paraffins (Ham et al., 1991). According to the leather industry approximately 95% of the grease fraction will stick to the leather (Ham et al., 1991). This results in a possible water emission of chlorinated paraffins from this source in The Netherlands of approximately 0.5-5 tonnes per year (5% of 10-100 tonnes per year) (after Ham et al., (1991) and Annema (1989)).

Incineration of chlorinated paraffins in properly designed installations results in degradation under liberation of hydrochloric acid. The resulting hydrocarbons with some remaining chlorine decompose further to form carbon dioxide, water, hydrogen chloride and trace amounts of chlorine (Beenen, 1991). Bergman et al. (cited in Beenen (1991) states that the low chlorinated compounds, when heated, tend to deliver aromatic compounds such as: toluene, benzene, biphenyl, naphthalene and even traces of chlorobenzene and dichlorostyrene in the first stage. Higher chlorinated compounds produced a certain number of polychlorinated aromatic compounds such as: polychlorinated benzenes, toluenes, PCBs and a trace of polychlorinated dibenzofuranes (PCDFs) at temperatures between 300 and 700 °C. In the case of combination with flame retardants based upon metal oxides interaction was shown to produce volatile chlorides such as antimonychloride and organic fragments (Beenen, 1991).

3.4 TRENDS

Few data are available on trends on the application and emission of chlorinated paraffins in The Netherlands.

Chlorinated paraffins are not produced in The Netherlands. There are no indications that chlorinated paraffin production will take place in the near future. On the basis of table 3.1 (production of chlorinated paraffins) it seems that world-wide the production of chlorinated paraffins is increasing.

At this moment no increase in application of chlorinated paraffins is expected (Ceha, pers. comm. 1991).
The impact of the dioxine discussions on the production of PVC in The Netherlands is unknown. The application in the PVC industry will probably stay on the same level, depending on the application of PVC (Ceha, pers. comm. 1991). According to Annema (1989) however, application might decrease, because of the slight profit for the chlorinated paraffin producers and distributors.
Since 1980, the use of chlorinated paraffins in rubber paints decreases in The Netherlands, this trend will continue according to the VVVF (Vereniging van Verf- en Drukinkt Fabrikanten) (cited in Annema, 1989).
The use of chlorinated paraffins as extreme pressure lubricant additives in the metal industry is also decreasing in The Netherlands (Annema, 1989).
The application of chlorinated paraffins in handle modifier in fat liquors for improving the suppleness of leather is expected to be static at this moment. The production of these chlorinated paraffin fat liquors in The Netherlands is estimated at 1,000 tonnes per year, but most of it will be exported (Annema, 1989). There are no data, however, on the percentage of chlorinated paraffins in these fat liquors.

Total emission of chlorinated paraffins caused by application industries are expected to decrease, this because of emission reducing measures. However, emissions caused by consumption and disposal losses completely depend on the use of products containing chlorinated paraffins in The Netherlands. In this respect the possible world-wide increase of the production of chlorinated paraffins may give rise to concern.

4. OCCURRENCE AND CONCENTRATIONS

Chlorinated paraffins are non-volatile and highly lipophylic, which restricts their transportation in the environment (Beenen, 1991). Very little data are available regarding the contents of chlorinated paraffins in the environment. Chlorinated paraffins are considered particularly stable (Willis, 1991).

The environmental distribution of chlorinated paraffins, calculated by use of a computer model based on the fugacity model by Mackay is shown in table 4.1.
The input data used for the Mackay model are presented in appendix 1 and are partly based on Willis (1991). The release of chlorinated paraffins in the environment in The Netherlands is estimated roughly at 100 tonnes per year maximally (see Chapter 3.3). Since no estimate on the ratio in which CPs are emitted in the air and the aquatic environment can be made, the calculations are performed on a 50:/50% base.
Recent industry calculations (CEFIC, pers. comm.) resulted in estimated half lives in air ranging from 0.85 to 7.2 days. Taking into account the assumption that air borne CPs are bound on atmospheric particulate matter and therefore are less available for degradation, the average half life was divided by a factor 10. The result was taken as the degradation rate in air. For soil a degradation rate constant of 0.1 day^{-1} was reported (Campbell and McConnell, 1980). Since biodegradation decreases with chain length and chlorination (decreasing bioavailability) and is low under aerobic conditions in soil, this degradation rate was divided by a factor 500. The result was used in the model calculations; catalytically dehydrochlorination was not accounted for.

Table 4.1 Rough estimates of concentrations of chlorinated paraffins in the environment in the "Netherlands" at steady state equilibrium

Medium	Concentrations
Air	8.0 $ng.m^{-3}$
Water	0.8 $ng.l^{-1}$
Suspended matter	2.4 $mg.kg^{-1}$
Biota	2.8 $mg.kg^{-1}$
Sediment	2.0 $mg.kg^{-1}$
Soil	0.05 $mg.kg^{-1}$

The calculated mass balance (based on these rough estimations) of chlorinated paraffins is given in table 4.2.

Table 4.2 Rough estimate of the environmental mass balance of chlorinated paraffins in the "Netherlands" (in tonnes)

Process	Air	Water	Sediment	Soil	Total
Emissions	50	50	-	-	100
Advective import	93	0.1	13	-	107
Advective export	129	0.1	10	-	140
Transformation	2	-	52	9	62
Accumulation	-	-	5	<0.1	5

From these indicative values it is concluded that advective transport and transformation of chlorinated paraffins may play a relatively important role. It should be noted however that these model calculations are made for a specific chlorinated paraffin compound, whereas the input data are based on highly uncertain estimates (see also appendix 1).

4.1 SOIL AND GROUNDWATER

No data on the occurrence of chlorinated paraffins in soil and groundwater in The Netherlands are available. A very rough estimate (table 4.1) indicate concentrations below 0.1 $mg.kg^{-1}$.

No specific data of the degradation of chlorinated paraffins in soil and groundwater are available. There are some general biodegradation data which also may be applicable to soil. Madeley and Birtley (cited in Annema (1989) and Willis (1991)) found that under aerobic conditions micro-organisms acclimatized to chlorinated paraffins could degrade short chain (C10-C13) grades up to 50% Cl rapidly and completely. Slower degradation occurred with C14-C17 grades (up to 45% Cl) and C20-C30 grades (up to 42% Cl). Chlorine contents above 58% inhibited degradation. They also found that the short chain grades were also attacked by non-acclimatized micro-organisms. Zitko and Arsenault (cited in Willis (1991) and Beenen (1991)) found that the rate of biodegradation of chlorinated paraffins under anaerobic conditions was higher than under aerobic conditions. The degradation rate constant of chlorinated paraffins for soil is 0.0041 hr^{-1}, calculated by Campbell and McConnell (1980).

4.2 SURFACE WATER AND SEDIMENT

4.2.1. Surface water

No data are available on chlorinated paraffins levels in surface water in The Netherlands.

Chlorinated paraffins have been identified in marine and fresh water in the United Kingdom, the largest chlorinated paraffin

producer in Europe. Table 4.3 shows chlorinated paraffins levels in various water bodies determined in 1978. Concentrations in nonindustrialized areas are generally below the detection limit of 0.5 $\mu g.l^{-1}$, in industrialized areas they range from <0.5-6.0 $\mu g.l^{-1}$. Relatively high levels were found in seas and sounds.

Table 4.3 Chlorinated paraffins in marine, fresh and non-marine waters (after Campbell & McConnell, 1980)

Sources	Location	Concentration in $\mu g.l^{-1}$ C10-C20	C20-C30
Fresh and non-marine waters			
* Remote from industry	Rivers	<0.5-0.5	<0.5-0.5
	Lakes	<0.5-1.0	<0.5-0.5
	Estuaries	NM	NM
* Close to industry	Rivers	<0.5-6.0	<0.5
	Lakes	NM	NM
	Estuaries	<0.5-1.5	<0.5-0.5
Marine waters	Harbours	0.5	<0.5
	Bays	<0.5	<0.5
	Seas and sounds	<0.5-4.0	<0.5-2.0

<0.5 : below detection limit (<0.5 $\mu g.l^{-1}$)
NM : not measured

From table 4.3 it may be derived that the concentrations of short chain chlorinated paraffins are generally somewhat higher than the longer chain chlorinated paraffins.

More recent data (unpublished) were obtained from CEFIC (pers. comm.) on the Trent/Humber river system in the UK. An improved analytical method permitted identification of short chain (C10-13) and medium chain (C14-17) types. The average concentration of C10-13 CPs was 2.0 (n=4) and 0.56 (n=9) $\mu g.l^{-1}$ in 1985 and 1986, respectively. For C14-17 CPs the average concentration was 7.6 (n=4) and 1.6 (n=16) $\mu g.l^{-1}$, respectively. The concentration of C14-17 was consistently higher compared to the concentration of C10-13 CPs.

The predicted concentration in The Netherlands was estimated far below the detection limit: about 0.001 $\mu g.l^{-1}$, corresponding to about 0.007 $\mu g.l^{-1}$ in surface water containing 30 mg of suspended matter per l (see table 4.1). These estimates are well below the concentrations measured in the UK.

4.2.2. Sediment

No data are available on chlorinated paraffin levels in sediment in The Netherlands.

Chlorinated paraffins have been found in sediments in the

United Kingdom. Table 4.4 shows chlorinated paraffin levels in sediments of various water bodies, the results being in agreement with those found in the surface water. Concentrations in non-industrialized areas ranged from less than 0.05-1.0 $mg.kg^{-1}$ wet weight. In industrialized areas they range from 0.05-15.0 $mg.kg^{-1}$ wet weight.

The Analytical Section of SNV (Scandinavian Environmental Protection Agency) (cited in KEMI (1991) and Beenen (1991)), has found short chain, highly chlorinated paraffins in sediments from an exposed area. The contents were found to be around 1 $mg.kg^{-1}$ dry weight. This is in accordance with the rough estimate in table 4.1: 0.4 $mg.kg^{-1}$ dry weight.

Table 4.4 Chlorinated paraffins in marine, fresh and other non-marine sediments (after Campbell and McConnell, 1980)

Sources	Location	Concentration in $mg.kg^{-1}$ (w/w) C10-C20	C20-C30
Fresh and non-marine sediments			
* Remote from industry	Rivers	<0.05-1.0	<0.05-0.05
	Lakes	NM	NM
	Estuaries	NM	NM
* Close to industry	Rivers	0.05-15.0	0.05-3.0
	Lakes	NM	NM
	Estuaries	<0.05-8.0	<0.05-3.2
Marine sediments	Harbours	0.05	<0.05
	Bays	<0.05	<0.05
	Seas and sounds	<0.05-0.1	<0.05-0.6

<0.05 : lower than detection limit (0.05 $mg.kg^{-1}$ w/w)
NM : not measured

The concentrations of chlorinated paraffins in sewage sludge are shown in table 4.5.

Table 4.5 Concentrations of chlorinated paraffins in sewage sludge sediments (after Beenen, 1991; Annema, 1989; Campbell and McConnell, 1980)

Sources	Concentration in $mg.kg^{-1}$ (w/w) C10-C20	C20-C30
Sewage sludge sediments in Liverpool	4-10	<0.05
Sewage sludge sediments in Manchester	<0.05	<0.05
Sewage sludge sediments in Zürich	<0.05	NM

<0.05 : lower than detection rate (<0.05 $mg.kg^{-1}$ w/w)
NM : not measured

Based on the rough model calculations the CP concentration in sediment in The Netherlands are estimated at about 1 $mg.kg^{-1}$.

Chlorinated paraffins can be catalytically dehydrochlorinated in the presence of iron oxides as well as other inorganic

compounds and the possibility of this process occurring in nature seems feasible (Willis, 1991).
Zitko and Arsenault (cited in Willis (1991)) states that chlorinated paraffins are biodegraded in spiked sediments under aerobic conditions and it is likely that the main route is dehydrochlorination. However, with reference to the higher chlorinated CPs, biodegradation under anaerobic conditions is more important.

4.3 AIR (OUTDOOR AND INDOOR)

No data on the occurrence of chlorinated paraffins in the air (outdoor and indoor) in The Netherlands are available.

Although CPs in their pure form are non-photoreactive, they usually contain impurities in trace amounts that make CPs photosensitive. The degradation time has been shown to be -to some extend- inversely proportional to the carbon chain length. The values of theoretical studies on photochemical degradation of chlorinated paraffins in the atmosphere are shown in table 4.6.

Table 4.6 Photochemical degradation of chlorinated paraffins in the atmosphere (Beenen, 1991; CEFIC, pers. comm.)

Carbon chain length	K_{oh} = $cm^3.mol^{-1}.s^{-1}$	$t_{1/2}$ day
C10-C13	9.0-14.9 10^{-12}	1.2 -1.8
C14-C17	14.9-18.9 10^{-12}	0.85-0.8
C15-C30	20.2-31.1 10^{-12}	0.5 -0.8
not specified	2.2-18.8 10^{-12}	0.85-7.2

Most commercially available chlorinated paraffin preparation are involatile and have a low vapour pressure around 20 °C (Willis, 1991; KEMI, 1991). Campbell and McConnell (1980) stated that because of the involatility the CPs cannot be transported and dispersed by atmospheric air movements and assumed that transport by water probably is the only large-scale dispersion mechanism. This contradicts with the McKay model calculations in this report, indicating that advective import and export through air is very important. It also is disagreement with their suggestion (Campbell and McConnell, 1980) that the concentrations of chlorinated paraffins in sediments (up to 1.0 $mg.kg^{-1}$) in remoter areas may be due to atmospheric transport of chlorinated paraffins from the source area.

4.4 FOOD AND DRINKING WATER

4.4.1. Human foodstuff

No data on the occurrence of chlorinated paraffins in the food in The Netherlands are available.

Chlorinated paraffins have been found in human foodstuffs in the United Kingdom. Table 4.7 shows chlorinated paraffins levels in human foodstuffs (Campbell and McConnell, 1980; Willis, 1991; Beenen, 1991).

Table 4.7 Chlorinated paraffins in human foodstuffs

Foodstuff class	C10-C20 chlorinated paraffin average ($mg.kg^{-1}$)	C20-C30 chlorinated paraffin average ($mg.kg^{-1}$)
Dairy products	0.3	-
Vegetable oils + derivatives	0.15	-
Fruit and vegetables	0.025	-
Beverages	<0.05	-
Cheese	-	0.19 (one sample)
Potato crisps	-	0.025 (one sample)
Peach fruit	-	0.025 (one sample)
Mussels	0.1-12 *	<0.1
Sheep	<0.05-0.2	<0.05

<0.05 : below detection limit
<0.1 : below detection limit
Sheep : the sheeps were grazing near a chlorinated paraffin plant
* : range; average 3.25 $mg.kg^{-1}$
- = no data available

4.4.2 Drinking water

No data on the occurrence of chlorinated paraffins in drinking water in The Netherlands are available.
In the United Kingdom chlorinated paraffin levels in five drinking water reservoirs were found to be below the detection limit (0.5 $\mu g.l^{-1}$) (Campbell and McConnell, 1980)

4.4.3 Occurrence in aquatic organisms

Limited data on CP levels in aquatic animals (common seamussel, freshwater and saltwater fish, grey seal) in the United Kingdom (around 1980) show mean levels of C10-20 below 0.1 $mg.kg^{-1}$ and mean levels of C20-30 below 0.05 $mg.kg^{-1}$ (limit of detection). The highest levels of C10-20 were found in marine shellfish (common seamussel): 0.1 - 12 $mg.kg^{-1}$, with a mean value of 3 $mg.kg^{-1}$. The highest level in mussels was found close to the effluent discharge of a CP manufacturing plant. The highest level of C20-30, 0.2 $mg.kg^{-1}$ was found in fish. The levels found in organisms are similar to those found in sediments (Campbell and McConnel, 1980).

Comparison of sediment chlorinated paraffin levels with aquatic organism tissue levels shows little or no bio-accumulation, the tissue levels being close to the levels in the sediment near which the organisms live.

4.4.4. Occurrence in terrestrial organisms

Analyses of seabird eggs in the United Kingdom (around 1980) showed C10-20 levels ranging from < 0.05 (limit of detection) to 2 $mg.kg^{-1}$; C20-30 levels usually were below 0.05 $mg.kg^{-1}$, with a maximum of 0.1 $mg.kg^{-1}$. CP levels in livers of these birds ranged from 0.1-1.2 $mg.kg^{-1}$ fresh weight for C10-20; those for C20-30 usually were lower, with a range of <0.05 to 1.5 $mg.kg^{-1}$ fresh weight. A comparison of these levels with those in potential food sources (aquatic organisms) of these birds indicates little or no potential for biomagnification (Campbell and McConnel, 1980). Analyses conducted in Sweden showed that CP levels in osprey (fish hawk) were lower than those in fish; there was thus no evidence for biomagnification. However, 'surprisingly high' CP levels were found in rabbits and moose in southern Sweden (KEMI, 1991). Further data on the Swedish study were not reported.

5. EFFECTS

5.1. HUMAN TOXICITY

Human data on chlorinated paraffins are limited to the occurrence of these compounds in human tissues and to data on dermal absorption. Hence, almost all data on chemobiokinetics, and all data on metabolism and toxicity reported in this section are based on animal studies.

Since 1980 a number of toxicity studies with selected grades of chlorinated paraffins has been conducted by 'The Working Party of the Chlorinated Paraffin Manufacturers Toxicology Testing Consortium'. In this programme, reviewed by Serrone et al. (1987), the following 4 grades were studied:

1. C_{10-13} *n*-paraffins with 58% chlorine content by weight [notation: C10-13 (58%)];
2. C_{14-17} *n*-paraffins with 52% chlorine content by weight [notation: C14-17 (52%)];
3. C_{20-30} *n*-paraffins with 43% chlorine content by weight [notation: C20-30 (43%)];
4. C_{22-26} *n*-paraffins with 70% chlorine content by weight [notation: C22-26 (70%)].

In this section the results of this programme are summarized, together with additional data.

5.1.1. Chemobiokinetics and metabolism

Oral and parenteral exposure

After oral exposure of mice and rats to a variety of chlorinated paraffins, 'some' to 'considerable' gastrointestinal absorption has been observed. Studies in which animals were exposed to a single oral dose of shorter-chain chlorinated paraffins, C12-C18, resulted in a percentage absorption of 10-75%. The dose level in these studies ranged from 1 to 500 $mg.kg^{-1}$ bw (Biesmann et al., 1983; Yang et al., 1987; Annema, 1989; IARC, 1990).

Both gastrointestinal absorption and chemobiokinetics are dependent on chain length and chlorine content. This is shown in the following studies.

In oral studies in rats, the shorter-chain grades C10-13 (58%) and C14-17 (52%) were 'more readily' absorbed than the longer-chain grades C20-30 (43%) and C22-26 (70%) (Serrone et al., 1987). Quantitative data on gastrointestinal absorption could not be derived from the studies by Serrone et al. (1987).

In studies in female mice, in which grades of [1-^{14}C-] chlorododecanes (C12) with varying chlorine content (17%, 56% and 68%) were administered by gastric intubation or by intravenous injection, the concentration of radiolabel in the tissues, the amount of $^{14}CO_2$ exhaled and the amount excreted in the urine decreased with increasing chlorination. With respect

to distribution and excretion pattern similar results were observed for the two routes of administration, as shown below. After i.v. treatment with C12 (17%), C12 (56%) and C12 (68%), the cumulative amount of $^{14}CO_2$ exhaled 12 hours after treatment was 52%, 32% and 8% of the dose, respectively; after oral administration this amount was 33% and 8% for C12 (56%) and C12 (68%), respectively; C12(17%) was only administered by injection. After i.v. treatment, the cumulative amounts excreted after 12 hours in urine and faeces were 18%, 21% and 5% (urine), and 3%, 4% and 9% (faeces), for C12 (17%), C12 (56%) and C12 (68%), respectively. After oral treatment, the cumulative amounts excreted after 12 hours in urine and faeces were 29% and 4% (urine), and 5% and 21%, for C12 (56%) and C12 (68%), respectively. Only for C12 (68%) faecal excretion was considerably higher after oral treatment than after i.v. treatment (21% *versus* 9%), indicating incomplete absorption at oral exposure. The similarities in excretion pattern of C12 (56%) after oral and i.v. treatment show that this lower chlorinated grade is more readily absorbed than the highly chlorinated C12 (68%) (Darnerud et al., 1982). The study by Darnerud et al. (1982) also showed transplacental passage for all three C12 grades, after treatment of pregnant mice of late gestation. Transplacental passage decreased with increasing degree of chlorination.

(Shortly) after oral or parenteral exposure, the highest CP levels are found in the intestines, liver, kidney, gall bladder and in other tissues with a high metabolic/cell turnover activity or a high excretory activity. In addition, high CP levels are found in adipose tissue and, with respect to the shorter-chain grades, in the ovaries (Darnerud et al., 1982; Biesmann et al., 1983; Serrone et al., 1987, Yang et al., 1987). It is noted that the CP 'levels' reported in these studies are based on radioactivity measurements (whole body autoradiography) after exposure to labelled compounds and, hence, include parent compounds and possible metabolites and tissue-bound residues. In the study by Biesmann et al. (1983) in which female mice were exposed to a single dose of 3.3 $mg.kg^{-1}$ bw of a polychlorohexadecane [C16 (69%)], whole-body autoradiography measurements conducted 30 days after treatment still showed high activities in corpora lutea, adipose tissue and liver. Similar results were observed in the study by Darnerud et al. (1982): also high activities in liver and adipose tissue 30-60 days after administration of chlorododecanes (C12) to female mice. In this latter study, a considerable longer-term retention of radioactivity was also seen in the central nervous system, the gonads and the adrenal cortex after injection of C12 (17%) or C12 (56%), but not after injection of C12 (68%). It is noted that the use of whole-body autoradiographs without data on metabolism may lead to erroneous conclusions with respect to long-term retention of labelled xenobiotics, because metabolic intermediates of the parent compounds may have been incorporated in endogenous compounds with slow turn-over rates. The exhalation of $^{14}CO_2$ can be used as an indication for metabolisme and for the possibility of incorporation of short-chain intermediates in endogenous compounds (Darnerud and Brandt, 1985).

In a feed study in which male rats were exposed for 8 weeks to labelled C14-17 (52%) in the diet at dose levels of 0.4 or 40 $mg.kg^{-1}$ (equivalent to 0.02-0.04 and 2-4 $mg.kg^{-1}$ $bw.day^{-1}$), equilibrium levels in liver and abdominal fat were reached within 1 and 6 weeks, respectively, regardless of the dose level used. At both dose levels, equilibrium levels in abdominal fat (in which the highest levels were found) were very similar, but never exceeded, the dietary dose levels. According to the authors, this is an important difference with a number of other organochlorine compounds such as DDT, Aldrin, Dieldrin, and the polychlorinated biphenyl Aroclor 1254, which have been shown to accumulate to levels in adipose tissue which are several times higher than the dietary concentration fed to animals. Equilibrium levels in the liver were about 5-times lower. After cessation of exposure, the CP level in abdominal fat decreased about 50% in 8 weeks. The lowest dose level choosen in this study is similar to that estimated for man (Birthley et al., 1980).
Data on metabolism of chlorinated paraffins are limited. One metabolic pathway is (cytochrome-P450 mediated) dechlorination prior to ß-oxidation, ultimately yielding carbon dioxide. Conjugation, e.g. with *N*-acetylcysteine or gluthathion has also been observed (IARC, 1990). The study by Darnerud et al. (1982) shows that dechlorination occurs at least at the terminal position.
Chlorinated paraffins are excreted both in the faeces and in the urine. Faecal excretion is the major route of elimination for highly chlorinated compounds which are less absorbed than lower chlorinated compounds and hardly degraded to carbon dioxide. For example, in experiments in which animals were exposed to a single dose of C16 (69%) or C18 (50-53%), 60% and 22% of the dose, respectively, was excreted in the faeces in 24 hours, increasing to 66% and 76% in 96 hours. The amount excreted shortly after treatment mainly consists of unabsorbed parent compound. Both after oral and intravenous exposure the amount of these grades excreted in the urine was below 5% of the dose (Biesmann et al., 1983; Yang et al., 1987).

Dermal exposure
Based on in an *in vitro* study with human skin and an *in vivo* study with rats, Beenen (1991a) concluded that chlorinated paraffins are 'very poorly' absorbed through skin. In the latter study (Yang et al., 1987), less than 1% and less than 0.1% of the dermally applied dose of C18 (50-53%) and C28 (47%) were recovered in excreta, exhaled air and tissues in 96 hours. The dose level was 66 mg/cm^2 shaved dorsal skin (equivalent to 2,000 $mg.kg^{-1}$ bw).

Inhalatory exposure
No data on inhalatory retention of CPs are available.

Occurence in human tissues
Post-mortem findings showed C10-20 levels up to 1.5 $mg.kg^{-1}$ in

liver, up to 0.5 mg.kg^{-1} in kidney and up to 0.6 mg.kg^{-1} in adipose tissue. The majority of samples (from 24 subjects) showed C10-20 levels in these tissues up to 0.2 mg.kg^{-1}. The C20-30 levels in these tissues usually were below 0.05 mg.kg^{-1} (detection limit), with a maximum of 3.5 mg.kg^{-1} in fat of one subject (Campbell and McConnel, 1980). All levels are on fresh weight basis.

5.1.2. Toxicity, carcinogenicity and teratogenicity

Short-term exposure (acute and subacute toxicity)

Acute toxicity
Acute toxicity tests with a variety of chlorinated paraffins with different chain length and degree of chlorination have resulted in oral LD50-values exceeding 4 g.kg^{-1} bw (mice, rats). In most tests the LD50-values exceeeded 10 g.kg^{-1} bw (Birthley et al., 1980; NTP, 1986a,b). These data show that the acute toxicity at oral exposure is very low.

Subacute toxicity
A number of compounds were tested in 2-w oral toxicity studies in rats and/or mice (NTP, 1986a,b; Serrone et al., 1987). These range-finding studies are not discussed in detail herein, because the same compounds were tested in semi-chronic toxicity studies (see 'long-term exposure'). The results of the subacute toxicity studies are in agreement with those of the semi-chronic toxicity studies. For example, the 2-w studies reported by Serrone et al. (1987) showed decreasing toxicity with increasing chain length, based on no-observed-(adverse)-effect-levels, NO(A)ELs, of 30, 500, ≥3,000 and ≥15,000 mg.kg^{-1} bw.day^{-1} for C10-13 (58%), C14-17 (52%), C20-30 (43%) and C22-26 (70%), respectively. The studies with the 2 first-mentioned grades identified the liver as (primary) target organ.

Developmental toxicity

Teratogenicity
The embryo/foetotoxicity and teratogenicity of C10-13 (58%), C14-17 (52%), C20-30 (43%) and C22-C26 (70%) was investigated in oral studies in which the test compound was administered by gavage to rats on gestation days 6 to 19 and to rabbits on gestation days 6 to 27 (Serrone et al., 1987; table 5.1). Exposure of rats to 2,000 mg.kg^{-1} bw.day^{-1} of C10-13 (58%) resulted in severe maternal toxicity (8 out of 25 treated dams died), in embryo-/ foetotoxicity (increased number of post-implantation losses, increased early and late resorptions, decrease in viable fetusses per dam) and in irreversible structural changes (digital malformations, viz. adactyly and/or shortened digits). The mid-dose level of 500 mg.kg^{-1} bw.day^{-1} of C10-13 (58%) resulted in 'signs of maternal toxicity' (no further details reported), but not in embryo-/

foetotoxic or teratogenic effects; the low-dose level of 100 $mg.kg^{-1}$ $bw.day^{-1}$ was without effect. Teratogenic effects were not observed in the studies with the other grades tested (see table 5.1).
Based on the results of these studies (which were not reported in detail by Serrone et al., 1987), the grade C10-13 (58%) is considered to be embryo-/foetotoxic and teratogenic, although the latter effect was accompanied by severe maternal toxicity. The grades C20-30 (43%) and (possibly) C22-26 (70%) are considered to be embryo-/foetotoxic, but not teratogenic. The results of these studies are in agreement with those of other toxicity studies, showing decreasing toxicity with increasing chain length (regardless of degree of chlorination).
It is noted that the exposure of the animals was somewhat longer than laid down in the "segment II study" protocol (rat: gestation days 6 to 15; rabbit: gestation days 6 to 18).

Table 5.1. Oral teratology studies with chlorinated paraffins

Grade	Species	LED mg/kg bw/day	NO(A)EL mg/kg bw/day	Effects and remarks	Ref.
C10-13 (58%)	rat	500	100	Maternal toxicity at 500 and 2,000; embryo-/foetotoxicity and digital malformations at 2,000.	Serrone et al., 1987
C10-13 (58%)	rabbit	30	10	Embryo-/foetotoxicity at 30 and 100.	Serrone et al., 1987
C14-17 (52%)	rat	5,000	2,000	Maternal toxicity at 5,000.	Serrone et al., 1987
C14-17 (52%)	rabbit	-	≥100	Mean maternal body weight losses were seen during treatment at the high dose level in a range-finding study.	Serrone et al., 1987
C20-30 (43%)	rat	5,000	2,000	Maternal toxicity at 5,000.	Serrone et al., 1987
C20-30 (43%)	rabbit	2,000	500	Embryo-/foetotoxicity at 2,000 and 5,000	Serrone et al., 1987
C22-26 (70%)	rat	-	≥5,000		Serrone et al., 1987
C22-26 (70%)	rabbit	-	≥1,000	In preliminary rabbit studies an increase in post-implantation loss was observed at ≥1,000, but this effect was not observed in the main teratology study.	Serrone et al., 1987

LED: Lowest-effect-dose; NO(A)EL: No-observed-(adverse)-effect-level

Fertility and reproduction

The effect of C14-17 (52%) on fertility and reproductive parameters was investigated in a feed study in which groups of male and female rats were exposed to dose levels of 0, 100, 1,000 or 6,250 $mg.kg^{-1}$ feed (equivalent to 0, 5, 50 and 312 $mg.kg^{-1}$ $bw.day^{-1}$, using a standard conversion factor of 20). Animals of both sexes were exposed for 28 days before mating and during the period of mating. Exposure of female rats was continued through gestation up to postnatal day 21. Pups were exposed to the same diet from weaning until the age of 10 weeks. Exposure did not result in effects on fertility or reproductive performance of the parent generation. No adverse effects on pups were observed untill lactation day 7, but none of the pups of high-dosed parents survived to weaning. Survival and body weights of mid-dosed pups were decreased by lactation day 21, as well as body weights of low-dosed pups. The reduced weight continued after weaning in the female pups, whereas in male pups the difference with the control group became less pronounced as the study progressed. From these and preliminary results (not published) it appears that the effect of postnatal exposure via milk is considerably greater than prenatal exposure *in utero* (Serrone et al., 1987). It is noted that detailed information (quantitative data; statistics) on this reproduction study is not reported by Serrone et al. (1987).

Long-term exposure (semi-chronic and chronic toxicity)

Semi-chronic toxicity

A number of compounds was tested in 90-d oral toxicity studies (Birtley et al., 1980; NTP, 1986a,b; Serrone et al., 1987). These feed and gavage studies are summarized in table 5.2.
Most 90-d oral studies with shorter-chain grades (C12, C10-13, C14-17) resulted in NO(A)ELs of (about) 10 $mg.kg^{-1}$ $bw.day^{-1}$. In one study with C14-17 (52%), relative liver weight was increased in male rats (but not in female rats) at 10 $mg.kg^{-1}$ $bw.day^{-1}$, but this effect was not supported by microscopic findings. Most studies with longer-chain grades (C20-30, C22-26, C23) resulted in NO(A)ELs of 900 $mg.kg^{-1}$ $bw.day^{-1}$ and higher. However, two studies showed histopathological changes in the liver of female rats at dose levels of 100 and 165 $mg.kg^{-1}$ $bw.day^{-1}$, respectively. These changes, especially the occurence of granulomatous inflammation that was observed in both studies, are considered to be compound-related adverse effects.
The results of these oral studies show that the toxicity of shorter-chain grades is, in general, higher than that of longer-chain grades (regardless of degree of chlorination). These studies identified the liver and kidney as primary target organs of chlorinated paraffins. Effects included increases in liver and kidney weights (both absolute and relative to body weight) and histopathological changes in these organs. In addition, the thyroid was identified as a target organ for shorter-chain grades.

Table 5.2. Oral semi-chronic (90-d) toxicity studies with chlorinated paraffins

Grade	Species	Oral route	LED mg/kg bw/day	NO(A)EL mg/kg bw/day	Target organs	Reference
C12 (60%)	mouse (m,f)	G	180*	90*	liver	NTP, 1986b
C12 (60%)	rat (m,f)	G	225*	-	liver,kidney [1]	NTP, 1986b
C10-13 (58%)	rat (m,f)	D	100	10	liver,kidney,thyroid	Serrone et al., 1987
C10-13 (58%)	rat (m,f)	G	100	10	liver,kidney,thyroid	Serrone et al., 1987
C14-17 (52%)	rat (m)	D	10	-	liver,kidney,thyroid,adrenal [2]	Serrone et al., 1987
	rat (f)	D	100	10	liver, kidney	
C14-17 (52%)	rat (m,f)	D	25 **	12 **	liver,kidney	Birtley et al., 1980
C14-17 (52%)	dog (m,f)	D	30	10	liver	Birtley et al., 1980
C20-30 (43%)	rat (f)	G	100	-	liver [3]	Serrone et al., 1987
	rat (m)	G	3,750	900	kidney [4]	
C22-26 (70%)	rat (m,f)	D	3,750	900	liver	Serrone et al., 1987
C23 (43%)	mouse (m,f)	G	-	≥5,320*		NTP, 1986a
C23 (43%)	rat (f)	G	165*	-	liver [5]	NTP, 1986a
	rat (m)	G	-	≥2,660*		

D: Administered in the diet
G: Administered by gavage
LED: Lowest-effect-dose
NO(A)EL: No-observed-(adverse)-effect-level

*: Reported dose level multiplied by 0.71 (5/7) because the animals were treated 5 days/week.
**: Dietary dose level divided by a standard conversion factor of 20.

1) A dose-related increase in relative liver weights was observed in both male and female rats (statistically significant at all dose levels).
2) An increase in relative liver weight at 10 $mg.kg^{-1}$ $bw.day^{-1}$ only occured in male rats (not in female rats) and was not supported by microscopic findings. So, Serrone et al. (1987) considered 10 $mg.kg^{-1}$ $bw.day^{-1}$ to be the NO(A)EL for C14-17 (52%) in both male and female rats.
3) Dose-related increase in absolute liver weights in female rats, together with multifocal granulomatous hepatitis, inflammation and necrosis at all dose levels.
4) Because of the (marginal) effect observed at the dose level of 3,750 $mg.kg^{-1}$ $bw.day^{-1}$ (mild necrosis in kidneys only), Serrone et al. (1987) considered 3,750 $mg.kg^{-1}$ $bw.day^{-1}$ to be the NO(A)EL for C20-30 (43%) in male rats.
5) A dose-related increase in the incidence of granulomatous inflammation in the liver was observed in all dosed groups of female rats (increasing from 1/10 at 165 $mg.kg^{-1}$ $bw.day^{-1}$ to 9/10 at 2,660 $mg.kg^{-1}$ $bw.day^{-1}$; controls: 0/10).

Chronic toxicity (carcinogenic and noncarcinogenic effects)
The toxicity and carcinogenicity of C12 (60%) and C23 (43%) have been investigated in 2-yr oral studies within the U.S. National Toxicology program (NTP 1986a,b; see also Bucher et al., 1987). In these studies, B6C3F1 mice and F344/N rats were treated by gavage 5 days per week; the vehicle used was corn oil.

In the studies with C12 (60%) dose levels were 0, 125 and 250 $mg.kg^{-1}$ bw for male and female mice, and 0, 312 and 625 $mg.kg^{-1}$ bw for male and female rats. According to the NTP-classification, there was *'clear evidence of carcinogenicity* of C12 (60%) for B6C3F1 mice as shown by increased incidences of hepatocellular adenomas and of adenomas or carcinomas (combined) in dosed male and female mice and increased incidences of adenomas and of adenomas or carcinomas (combined) of thyroid gland follicular cells in dosed female mice'. Further, there was *'clear evidence of carcinogenicity* of C12 (60%) for F344/N rats based on increased incidences of hepatocellular neoplasms (primarily neoplastic nodules) in male and female rats, of adenomas or adenocarcinomas (combined) of the kidney tubular cells in male rats, and of follicular cell adenomas or carcinomas (combined) of the thyroid gland in female rats. Mononuclear cell leukemia in dosed male rats may have been related to administration of C12 (60%)'. The main results with respect to the carcinogenic effects observed in these studies are summarized in table 5.3. With respect to noncarcinogenic lesions, very high incidences of liver hypertrophy were most evident, in male and female rats. Especially in dosed male rats, survival was much lower than that of vehicle controls, perhaps due to kidney injury. Hence it was noted that the maximum tolerated dose may have been exceeded in rats, especially in male animals (NTP, 1986b). Because of both the carcinogenic effects (in mice and rats) and the noncarcinogenic effects (in rats only) observed at the dose levels tested, NO(A)ELs for C12 (60%) in mice or rats could not be established.

In the studies with C23 (43%) dose levels were 0, 2,500 and 5,000 $mg.kg^{-1}$ bw for male and female mice, 0, 1,875 and 3,750 $mg.kg^{-1}$ bw for male rats, and 0, 100, 300 and 900 $mg.kg^{-1}$ bw for female rats. According to the NTP-classification, there was *'clear evidence of carcinogenicity* of C23 (43%) for male B6C3F1 mice as shown by an increase in the incidence of malignant lymphomas'. There was *'equivocal evidence of carcinogenicity* of C23 (43%) for female B6C3F1 mice as shown by a marginal increase in the incidence of hepatocellular neoplasms' (It is noted that this increase was not statistically significant). Further, there was *'no evidence of carcinogenicity* of C23 (43%) for male F344/N rats' and there was *'equivocal evidence of carcinogenicity* of C23 (43%) for female F344/N rats as shown by an increased incidence of adrenal gland medullary pheochromocytomas'.
The main results with respect to the carcinogenic effects observed in these studies are summarized in table 5.4.

Table 5.3. Incidences of treatment-related primary tumours in B6C3F1 mice and F344/N rats orally exposed to C12 (60%)

Mice ♂♂	**0 mg/kg bw [1]**	**125 mg/kg bw [1]**	**250 mg/kg bw [1]**
Lung Alveolar/bronchiolar carcinoma	0/50 (0%)	3/50 (6%)	6/50 (12%) *
Liver Hepatocellular adenoma Hepat. adenoma or carcinoma	11/50 (22%) 20/50 (40%)	20/50 (40%) * 34/50 (68%) *	29/50 (58%) * 38/50 (76%) *
Mice ♀♀	**0 mg/kg bw [1]**	**125 mg/kg bw [1]**	**250 mg/kg bw [1]**
Haematopoietic system Lymphoma (malignant)	0/50 (0%)	3/50 (6%)	6/50 (12%) *
Liver Hepatocellular adenoma Hepat. adenoma or carcinoma	0/50 (0%) 3/50 (6%)	18/50 (36%) * 22/50 (44%) *	22/50 (44%) * 28/50 (56%) *
Thyroid Follicular cell adenoma or carcinoma	8/50 (16%)	12/49 (24%)	15/49 (31%)
Rats ♂♂	**0 mg/kg bw [2]**	**312 mg/kg bw [2]**	**625 mg/kg bw [2]**
Haematopoietic system Mononuclear cell leukemia	7/50 (14%)	12/50 (24%)	14/50 (28%)
Liver Neoplastic nodules Neoplastic nodules or hepatocellular carcinoma	0/50 (0%) 0/50 (0%)	10/50 (20%) * 13/50 (26%) *	16/48 (33%) * 16/48 (33%) *
Kidney Tubular cell adenoma Tub. cell adenoma or carcinoma	0/50 (0%) 0/50 (0%)	7/50 (14%) * 9/50 (18%) *	3/49 (6%) 3/49 (6%)
Rats ♀♀	**0 mg/kg bw [2]**	**312 mg/kg bw [2]**	**625 mg/kg bw [2]**
Liver Neoplastic nodules Neoplastic nodules or hepatocellular carcinomas	0/50 (0%) 0/50 (0%)	4/50 (8%) 5/50 (10%) *	7/50 (14%) * 7/50 (14%) *
Thyroid gland Follicular cell adenoma or carcinoma	0/50 (0%)	6/50 (12%) *	6/50 (12%) *

* Statistically significant increase ($p < 0.05$; Fisher Exact Test)

[1] Male and female mice were treated by gavage with 0, 125 or 250 mg/kg bw on 5 days/weeks for 104 weeks; these dose levels are equivalent to 0, 90 and 180 mg/kg bw/day.
Numbers of surviving mice: 34, 30 and 30 for males, and 35, 31 and 25 for females, respectively.

[2] Male and female rats were treated by gavage with 0, 312 or 625 mg/kg bw on 5 days/week for 103 weeks; these dose levels are equivalent to 0, 220 and 440 mg/kg bw/day.
Numbers of surviving rats: 27, 6, and 3 for males, and 34, 23 and 29 for females, respectively.

In rats the primary noncarcinogenic lesion observed was a diffuse lymphohistiocystic inflammation in the liver of both males and females, with very high incidences (higher than 80% *versus* 0-8% in controls) at all dose levels. In addition, the severity of this lesion increased with increasing dose level. No significant noncarcinogenic lesions were considered to be compound related in mice (NTP, 1986a). The NO(A)EL of C23 (43%) in mice was 2,500 $mg.kg^{-1}$ bw (equivalent to 1,785 $mg.kg^{-1}$ $bw.day^{-1}$). Because of the lymphohistiocystic inflammation in the liver of rats (which is considered to be a compound-related adverse effect) at all dose levels tested, a NO(A)EL of C23 (43%) in rats could not be established.

In the evaluation of these studies by IARC (1990) it was concluded that there is *'sufficient evidence'* for the carcinogenicity of C12 (60%) in experimental animals and that there is *'limited evidence'* for the carcinogenicity of C23 (43%) in experimental animals. The former compound, C12 (60%), was classified by IARC as *'possibly carcinogenic to humans'* (Group 2B).

With respect to the above-mentioned NTP studies, the CEFIC Sector Group 'Chlorinated Paraffins' provided the following comments (which are reported herein without discussion):

* *Short-chain (C10-13) and medium-chain (C14-17) CPs were peroxisomal proliferators in rats and (especially) in mice, while long-chain (C20-30) CPs did not induce this effect (see also 'additional toxicity data' in this section). These data correlated well with NTP liver cancer data.*
* *CPs were not peroxisome proliferators in guinea pigs. Thus the normal species differences in peroxisome proliferation were observed. As is the case with other peroxisome proliferators CPs probably pose no hepatocarcinogenic hazard to man - because of the well documented species difference in response to peroxisome proliferators.*
* *CP-related male kidney tumours appear to be due to a sustained nephropathy and regenerative hyperplasia. These were observed only in the male rat - not in female rats. The renal tumours induced by CPs are unlikely to be relevant to man, because they are male rat specific and then only occur after sustained nephropathy and hyperplasia.*
* *CPs were found to decrease plasma thyroxine levels and increase plasma TSH (thyroid stimulating hormone), this in turn resulted in thyroid hypertrophy and hyperplasia (demonstrated by increased DNA synthesis). The initial decrease in thyroxine appeared to be due to increased thyroxine-UDPG glucuronyl transferase activity. Thus the thyroid tumours induced by CPs in the NTP studies appear to be due to altered thyroid homeostasis and consequential hyperplasia leading to neoplasia. Alterations in thyroid status may be of potential relevance to man (but only if thyroxine-UDPG glucuronyl transferase in induced leading to decreased plasma T_4 levels).*

Table 5.4. Incidences of treatment-related primary tumours in B6C3F1 mice and F344/N rats orally exposed to C23 (43%)

Mice ♂♂	**0 mg/kg bw [1]**	**2,500 mg/kg bw [1]**	**5,000 mg/kg bw [1]**	
Haematopoietic system Lymphoma (malignant)	6/50 (12%)	12/50 (24%)	16/50 (32%) *	
Mice ♀♀	**0 mg/kg bw [1]**	**2,500 mg/kg bw [1]**	**5,000 mg/kg bw [1]**	
Liver Hepatocellular carcinoma Hep. adenoma or carcinoma	1/50 (2%) 4/50 (8%)	1/49 (2%) 3/49 (6%)	6/50 (12%) 10/50 (20%)	
Rats ♀♀	**0 mg/kg bw [2]**	**100 mg/kg bw [2]**	**300 mg/kg bw [2]**	**900 mg/kg bw [2]**
Adrenal Pheochromocytoma	1/50 (2%)	3/50 (6%)	6/50 (12%)	7/50 (14%) *

* Statistically significant increase ($p < 0.05$; Fisher Exact Test)

[1] Male and female mice were treated by gavage with 0, 2,500 or 5,000 mg/kg bw on 5 days/week for 103 weeks; these dose levels are equivalent to 0, 1,785 and 3,570 mg/kg bw/day.
Numbers of surviving mice: 28, 36 and 28 for males, and 21, 22 and 20 for females, respectively.

[2] Female rats were treated by gavage with 0, 100, 300 or 900 mg/kg bw on 5 days/week for 103 weeks; these dose levels are equivalent to 0, 70, 210 and 630 mg/kg bw/day.
Numbers of surviving female rats: 34, 30, 33 and 31, respectively.
(In male rats, the incidences of tumours were not treatment-related)

Additional toxicity data

Liver effects

Amongst the effects observed in the liver due to exposure to chlorinated paraffins are proliferation of hepatocytes, proliferation of smooth endoplasmatic reticulum and peroxisomes, and induction of cytochrome-P450. In general, shorter-chain compounds, especially the more chlorinated compounds, were (much) more effective in this respect than longer-chain compounds. Some examples (with data on dose levels) are described below.

In the two 90-day oral toxicity studies reported by Birtley et al. (1980), C14-17 (52%) caused a dose-related proliferation of the smooth endoplasmatic reticulum in the hepatic cells, at dose levels ≥ 25 $mg.kg^{-1}$ $bw.day^{-1}$ in rats and ≥ 30 $mg.kg^{-1}$ $bw.day^{-1}$ in dogs. Proliferation of peroxisomes was reported in livers of mice and rats exposed by gavage to a dose level of 1,000 $mg.kg^{-1}$ $bw.day^{-1}$ of C12 (60%), C10-12 (56%), or C14-17 (40%). A C23 (40%) grade did not induce peroxisomal effects (IARC, 1990). A cell proliferation assay in rats resulted in a positive response, i.e. an increased percentage of S phase hepatocytes, after a single oral dose (by gavage) of 2,000 $mg.kg^{-1}$ bw (Ashby et al., 1990).

Daily intraperitoneal injections of 1,000 mg.kg^{-1} bw. of C10-13 (49%, 59% or 71%), C14-17 (50%) or C18-26 (49%) in rats for 4 days resulted in one or more of the following liver effects: increases in absolute and/or relative weights, increased cytochroom-P450 content and increased P450-mediated metabolism. A similar treatment regime (intraperitoneal injections on days 1, 4 and 6) resulted in one or more of the following morfological liver effects: proliferation of smoooth endoplasmatic reticulum and peroxisomes, and increased number and size of lysosomes, mitochondria and lipid droplets (Nilsen et al., 1981).

Central nervous system effects
In a study with immature (10-d old) mice, treatment with a single oral dose of a polychlorohexadecane [C16 (69%)] resulted in an effect on the presynaptically sodium-dependent choline uptake system in the cerebral cortex (viz. a significant 65% decrease in V_{max} measured in a crude synaptosomal fraction isolated from the brain). The dose of 1 mg.kg^{-1} bw (the lowest CP dose level used in the toxicity studies described in this section) was administered by intubation; animals were killed 7 days after treatment (Eriksson and Nordberg, 1986).

5.1.3. Genotoxicity

In vitro tests
The chlorinated paraffins C12 (60%) and C23 (43%) did not induce gene mutations in *Salmonella typhimurium* strains TA97, TA98, TA100 and TA1535. Tests were performed both in the absence and presence of metabolic activation (NTP, 1986a,b). Negative responses in this test system were also reported in other studies, viz. for C10-13 (50%), C14-17 (52%) and C20-30 (42%) in strains TA98, TA100, TA1535 and TA1538, both in the absence and presence of metabolic activation (Birtley et al., 1980) and for C10-23 (70%) (IARC, 1990).
A cell transformation test with baby hamster kidney cells (screening test for carcinogenesis) resulted in negative responses for C10-13 (50%), C14-17 (52%) and C20-30 (42%) (Birtley et al., 1980).

In vivo tests
A number of compounds was tested for genotoxicity in *in vivo* tests, especially for chromosomal aberrations in a bone marrow cytogenetic test in rats (Serrone et al., 1987; Ashby et al., 1990). All tests, summarized in table 5.5, resulted in a negative response.

Table 5.5. *In vivo* genotoxicity tests with chlorinated paraffins

Grade	Animal species and test system	Endpoint	Exposure	Result	Ref.
C12 (60%)	rat (m) UDS assay	DNA repair in hepatocytes	single oral dose (gavage), 2,000 mg/kg bw	Negative	Ashby et al., 1990
C10-13 (58%)	rat (m), dominant lethal test	chrom. aberr. or gene mutation in germ cells	oral (gavage), 0-250-750-2,000 mg/kg bw/day, for 5 consecutive days	Negative	Serrone et al., 1987
C10-13 (58%)	rat (m) bone marrow cytogenetic test	chrom. aberr.	oral (gavage), 0-250-750-2,500 mg/kg bw/day, for 5 consecutive days	Negative	Serrone et al., 1987
C14-17 (52%)	rat (m) bone marrow cytogenetic test	chrom. aberr.	oral (gavage), 0-500-1,500-5,000 mg/kg bw/day, for 5 consecutive days	Negative	Serrone et al., 1987
C20-30 (43%)	rat (m) bone marrow cytogenetic test	chrom. aberr.	oral (gavage), 0-500-1,500-5,000 mg/kg bw/day, for 5 consecutive days	Negative	Serrone et al., 1987
C22-26 (70%)	rat (m) bone marrow cytogenetic test	chrom. aberr.	oral (gavage), 0-500-1,500-5,000 mg/kg bw/day, for 5 consecutive days	Negative	Serrone et al., 1987

5.2. ECOTOXICITY: AQUATIC ORGANISMS

5.2.1. Accumulation and food chain transfer

In most bioaccumulation studies reported, organisms were exposed for a relatively short period which was insufficient to reach steady-state (equilibrium) levels. Therefore, very little steady-state bioconcentration factors[1] are available.

Exposure in water (bioconcentration)
Bioaccumulation studies with detailed information on experimental conditions such as exposure time and exposure concentration (Bengtsson et al., 1979; Svanberg and Lindén, 1979; Renberg et al., 1986) are summarized in table 5.6.

Table 5.6. Accumulation of chlorinated paraffins in aquatic organisms exposed in water

Grade	Test Species	Exposure conc. (µg/l)	Exp. time	Test type	BCF *	Ref.
C12 (69%)	M. edulis (mussel)	0.003	21 d	F	95,000	Renberg et al.'86
C12 (69%)	M. edulis	0.13	28 d	F	140,000 **ss** [1]	Renberg et al.'86
C16 (34%)	M. edulis	0.13	28 d	F	7,000 **ss** [1]	Renberg et al.'86
C10-C13 (49%)	A. alburnus (fish)	125	14 d	R [2]	800	Bengtsson et al.'79
C10-C13 (59%)	A. alburnus	125	14 d	R [2]	750	Bengtsson et al.'79
C10-C13 (71%)	A. alburnus	125	14 d	R [2]	150	Bengtsson et al.'79
C11.5 (70%)	A. alburnus	100	14 d	R	100-150	Svanberg & Lindén '79
C14-17 (50%)	A. alburnus	125	14 d	R [2]	30	Bengtsson et al.'79
C18-26 (49%)	A. alburnus	125	14 d	R [2]	15	Bengtsson et al.'79

* Whole-bode BCF, calculated on a fresh weight basis.
[1] Steady-state was reached within 14 days. [2] Renewal every second or third day (at weekends).

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[1] Bioconcentration factor (BCF): $C_{organism}$ / C_{water}

The term 'bioconcentration' is used in general to express the ability of organisms to concentrate a substance from the water in which they are exposed.
In case of an exposure medium other than water, the term 'bioaccumulation' is often used.

In the studies by Renberg et al. (1986), steady-state BCFs in the common seamussel *Mytilus edulis* were 140,000 for C12 (69%) and 7,000 for C16 (34%), at an exposure concentration of 0.13 $\mu g.l^{-1}$. Equilibrium levels were reached in 2 weeks, indicating a fairly rapid uptake. Elimination of the highly chlorinated grade C12 (69%) was slow: about one-third of the amount accumulated in a 28-d exposure period was recovered in the wet tissues (based on radioactivity measurements after a 28-d depuration period). Data on the elimination of C16 (34%) were not reported.
In the studies by Bengtsson et al. (1979), BCF-values in the fresh/brackish water fish *Alburnus alburnus* (bleak) ranged from 800 for C10-13 (49%) to 15 for C18-26 (49%), at an exposure concentration of 125 $\mu g.l^{-1}$. It was not reported whether or not equilibrium levels were reached. After 7 days of depuration, at least 50% of the amount accumulated during the 14-d exposure period was recovered in the fish (70% of C10-C13 (49%), 85% of C10-13 (59%) and 50% of C10-13 (71%); the accumulated amounts of the two other compounds were too low to estimate the amount eliminated. After this depuration period, the retained amounts of C10-13 (49%) and C10-13 (59%) were higher than that of Clophen A 50 (a mixture of more than 50 PCBs with an average chlorine content of 50% on weight basis), although the amount of PCBs accumulated was about 2.5 times greater than that of these two grades of chlorinated paraffins. It is noted that the levels in fish were based on total organic chlorine content measured by neutron activation analysis (NAA), so, possible metabolites and tissue-bound residues were included in the measurements. However, in subsequent work (not published) analyses made by mass spectrometry agreed well with NAA.
These two studies clearly show that the amount of CP accumulated in aquatic animals is decreasing with increasing chain length and, to a lesser extend, with increasing degree of chlorination (both factors determining molecular weight), and that elimination occurs more slowly than uptake.
A third study resulted in a BCF in bleak *A. Alburnus* of about 100-150 for C11.5 (70%), at an exposure concentration of 100 $\mu g.l^{-1}$. It was not reported whether or not equilibrium was reached. In agreement with the afore-mentioned studies, elimination of this highly chlorinated grade was slow: after 5 weeks of depuration in a continuous flow of uncontaminated brackish water there did not seem to be any significant reduction in the amount of CP accumulated during the 14-d exposure period. The residue levels in fish were calculated from the amount of chlorine bound to persistent fat-soluble compounds and determined by NAA (Bengtsson et al., 1979 (report); cited in Svanberg and Lindén, 1979).

Less detailed data on bioconcentration indicate BCF-values for C11 (58%), C12 (58%0 and C16 ranging from 3,600 up to 41,000 in rainbow trout and seamussel at exposure concentrations in the range of several $\mu g.l^{-1}$. Elimination half-lives for these compounds in these organisms were reported to be 10-20 days (Annema, 1989 and Svanberg, 1983; secundary literature).
A comparison of CP levels in aquatic organisms and in water

(Campbell and McConnel, 1980; see also 4.4.3) provides rough estimates of BCF-values: 100-1,000 for freshwater and saltwater fish, and 1,000-10,000 for saltwater mussels. It is noted, however, than CP levels in organisms and water were below the limit of detection in a number of samples, which makes it impossible to calculate reliable BCF-values on the basis of these data.

Dietary exposure

In mussels *(M. edulis)* fed suspended yeast cells containing C20-C30 (42%) at a dose level of 524 mg.kg^{-1} dry weight, whole-body CP level slowly increased to about 10 mg.kg^{-1} dry weight after 7 weeks of exposure (equilibrium not reached). After termination of exposure, the CP levels rapidly decreased: in the 8-w depuration period, about 90% of the whole-body CP level was eliminated (Madeley and Birtley, 1980).

In rainbow trout (*S. gairdneri*) fed diets containing C20-30 (42%) at dose levels of 47 or 385 mg.kg^{-1} dry weight for five weeks, the calculated whole-body CP levels were about 10 and 100 mg.kg^{-1} dry weight, respectively. In the low-dose group, whole-body CP level approached equilibrium; it was not reported whether or not equilibrium was achieved in the high-dose group as well. After termination of exposure, the CP level in flesh (muscles) and whole body decreased with about 35% and 70%, respectively within 7 weeks (Madeley and Birtley, 1980).

In the two above-mentioned studies, CP tissue levels in mussels and fish were determined by direct ^{14}C-scintillating counting. Comparisons of CP levels in tissues measured by this technique *versus* thin-layer chromatography indicate that fish metabolized the bulk of the accumulated amount of CP (up to 80%), while mussels did not. The data indicate splitting of the paraffin chain in fish before removal of all chlorine atoms from the molecule (Madeley and Birtley, 1980). The structure of the metabolites were not identified. Metabolism of chlorinated paraffins was also observed in a study in which carp (*Cyprinus carpio*) were injected with C16 (34%): after 4 days a significant amount of the dose administered was excreted as carbon dioxide (Svanberg, 1983).

In a second study in which rainbow trouts *S. gairdneri* were fed a diet containing chlorinated paraffins [C12 (59%)], the flesh of the fish contained 18 mg CP.kg^{-1} dry weight, after 12 weeks of exposure to a dietary dose level of 10 mg.kg^{-1} (Svanberg and Lindén, 1979).

The uptake and elimination of chlorinated paraffins was also investigated in long-term studies in which bleaks (*A. alburnus)* were exposed to C10-C13 (49%), C10-C13 (71%) or C18-C26 (49%) in their diet for 3 months, followed by a depuration period of 10 months. Tests with 3 different dose levels of C10-C13 (49%), viz. 590, 2,500 and 5,800 mg.kg^{-1} feed showed 'uptake efficiencies' (percentage of administered dose recovered in the fish, based on recovery of total organic chlorine content) of 45%, 10% and 5%, respectively. In the depuration period, at least 90% of the amount accumulated was eliminated in 7 days in the low- and medium-dose group.

Elination in the high-dose group was slower: about 30% in 7 days and at least 90% in 35 days. The amount of C10-13 (71%) and, especially, C18-26 (49%) accumulated in fish was lower than that of similar dosed C10-C13 (49%), with uptake efficiencies of 6% and 2%, respectively. The C10-13 (71%) level in fish did not or hardly decrease during the depuration period, while that of C18-C26 (49%) decreased about 50% within 5 weeks, but remained on the same level afterwards (Bengtsson and Bauman-Ofstad, 1982). Renberg et al. (1986) found that later analyses of tissue extracts from the bioconcentration experiments by Bengtsson and Baumann-Ofstad (1982), using the mass spectrometric method of Gjös and Gustavsen (1982), proved the presence of native Witaclor 171P [C10-13 (71%)] in the fish still 625 days after transfer to uncontaminated conditions.
Bioaccumulation studies with the marine crustacean *Nitocra spinipes*, exposed to two polychlorododecanes [C12 (56%) and C12 (69%)] and two polychlorohexadecanes [C16 (32%) and C16 (68%)] support the findings from fish experiments. Short-chain CP with a high level of chlorination show the highest 'accumulation/retention potential'. (Bengtsson and Åhlman, 1982, manuscript, cited in Svanberg, 1983).

Biomagnification
In a UK study (Campbell and McConnel, 1980; see also 4.4.3), CP levels measured in fish-eating predators (pike, grey seal) were very similar or lower than those in fish. In a Swedish study in the Baltic, CP levels in grey seal were also found to be lower than those in fish (KEMI, 1991). So, these data do not indicate the occurence of biomagnification[2].

5.2.2. Toxicity

Short-term exposure (acute toxicity)

In the Microtox test 15-min EC50-values of 1-1.5 $mg.l^{-1}$ were found for C10-13 (49%) (Svanberg, 1983, from Tarkpea, 1982).
For the marine crustacean *N. spinipes* 96-hr LC50-values were 0.06 and 0.1 $mg.l^{-1}$ for two formulations of C10-13 (49%), <0.3 and <5 $mg.l^{-1}$ for two formulations of C10-13 (70%), and 9 $mg.l^{-1}$ for C14-17 (45%). For C14-17 (52%) and C22-26 (42%, 49%) the 96-hr LC50-values exceeded 1,000 $mg.l^{-1}$ (KEMI, 1981; Svanberg, 1983; from Tarkpea et al., 1981). In tests with the freshwater crustacean *D. magna* 96-hr LC50-values for short-chain compounds were of the same low magnitude as reported for *N. spinipes* (no further data reported). On the other hand, the crustacean *L. leander* was not affected at 1,000 $mg.l^{-1}$ of C10-13 (49%) (Svanberg, 1983).
Tests with freshwater fish (*S. gairdneri* and L. *macrochirus*)

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[2] 'Biomagnification' is the occurence of a substance at successively higher levels with increasing trophic levels in food chains.

resulted in 96-hr LC50-values exceeding 300 mg.l^{-1} for C10-13 (58%), C20 (34%), C23 (40%), C24 (48%) and C20-30 (Madeley and Birtley, 1980; NTP, 1986). Static tests with the freshwater/ brackish water fish *A. alburnus* in natural brackish water with a salinity of 7 o/oo resulted in LC50-values exceeding 5,000 mg.l^{-1} for C10-13 (49%, 56%, 63%, 71%), C11.5 (70%), C14-17 (50%, 52%), and C22-26 (42%), based on nominal test concentrations (Lindén et al., 1979).
It should be noted that the acute toxicity tests usually have been performed at concentrations that exceed the solubility of the CPs in water with several orders of magnitude. For most grades the solubility will be about 0.01 mg.l^{-1} and for shorter chain grades up to order of 0.1 mg.l^{-1}. However most commercially produced CPs can be emulsified in water (Howard et al., 1975; cited in Beenen, 1991a).

Long-term exposure

Data on effects of chlorinated paraffins at long-term exposure are very limited. Accumulation tests in which fish *A. alburnus* were exposed to nominal concentrations of 125 μg.l^{-1} did not result in mortality. However, behavioural changes (slugish movements, absence of shoaling behaviour, and abnormal vertical postures) were observed at exposure to different grades of C10-13. The effects disappeared within a couple of days after the fish were transferred to clean water. Exposure to C14-17 (50%) or C18-26 (49%) did not result in these effects (Bengtsson et al., 1979; see also Table 5.7). Similar neurotoxic effects were observed in a pilot study in which bleaks were exposed for 4 weeks to 0.1 and 1 mg.l^{-1} of C11.5 (70%) (Svanberg et al., 1978) and in a flow-through study in which rainbow trouts (*S. gairdneri*) were exposed for 3 weeks to 0.04 (0.05 ?) mg.l^{-1} of 'Chlorowax' (Howard et al., 1975; cited in Svanberg and Lindén, 1979 and in Svanberg, 1983). In the latter study, the effects gradually became worse and resulted in immobilization within 15-20 days.
According to Beenen (1991), 'chronic toxicity of the short-chain grades to a range of species was also (together with acute toxicity at concentrations below the water solubility) demonstrated within the range of 10-100 ppb' (μg.l^{-1}). Further details were not reported.

Table 5.7. Aquatic organisms - long-term toxicity of chlorinated paraffins

Grade	Test Species	Test type	Test water	Exp. time	Criterion & result (µg/l)	Ref.
C10-C13 (49%)	A. alburnus	R	seawater (n) 7 o/oo	14 d	NOLC ≥ 125 EC_{bcf} 125	Bengtsson et al.'79 [1,2]
C10-C13 (59%)	A. alburnus	R	seawater (n) 7 o/oo	14 d	NOLC ≥ 125 EC_{bcf} 125	Bengtsson et al.'79 [1,2]
C10-C13 (71%)	A. alburnus	R	seawater (n) 7 o/oo	14 d	NOLC ≥ 125 EC_{bcf} 125	Bengtsson et al.'79 [1,2]
C14-17 (50%)	A. alburnus	R	seawater (n) 7 o/oo	14 d	NOLC ≥ 125	Bengtsson et al.'79 [1]
C18-26 (49%)	A. alburnus	R	seawater (n) 7 o/oo	14 d	NOLC ≥ 125	Bengtsson et al.'79 [1]

[1] Test solutions were renewed every second or third day (at weekends); in this bioaccumulation study, only one test concentration was used.

[2] Behavioural changes were sluggish movements, absence of shoaling behaviour and abnormal vertical postures.

Additional toxicity data (dietary exposure)

Lombardo et al (1975) noted reduced growth in fingerling trout that were exposed for 12 weeks to 10 $mg.kg^{-1}$ of C12 (60%) in the diet (NTP, 1986). Rainbow trout that were exposed for 5 weeks to 47 or 385 $mg.kg^{-1}$ of C20-30 (42%) in the diet remained in good condition, based on measurement of the 'condition factor' (Madeley and Birtley, 1980). It was not reported which condition parameters were measured in the latter study.

In female flounders *P.flesus* (saltwater fish), exposure to C12 (49%) or C12 (70%) at a total dose level of 1,000 $mg.kg^{-1}$ bw resulted in some effects on haematological and biochemical parameters. Sampling was performed 2 and 4 weeks after the first of two administrations in feed capsules (Haux et al., 1982).

Fish *A. alburnus* exposed for 3 months to dietary dose levels up to 5,800 $mg.kg^{-1}$ of C10-13 (49%), or to dose levels of 3,180 $mg.kg^{-1}$ of C10-C13 (71%) or 3,400 $mg.kg^{-1}$ of C18-C26 (49%) showed no significant increase in mortality. Exposure to ≥ 2,500 $mg.kg^{-1}$ of C10-13 (49%) or to 3,180 $mg.kg^{-1}$ of C10-13 (71%) resulted in behavioural changes (sluggish movements and swimming closer to the bottom than usual). The behavioural changes generally were acompanied by folded dorsal fins and minor balance problems. These effects gradually disappeared within a couple of weeks of the subsequent depuration period (Bengtsson and Ofstad, 1982).

5.3. ECOTOXICITY: TERRESTRIAL ORGANISMS

5.3.1. Accumulation and food chain transfer

Japanese quails (*C. coturnix japonica*) orally exposed to a single dose of 330 $\mu g.kg^{-1}$ bw of C16 (69%) excreted a cumulative amount of 17% and 58% in urine and faeces (combined) after 8 and 96 hours, respectively, based on radioactivity measurements. During the first 8 hours, about 2% was exhaled as CO_2. After intravenous exposure to the same dose level, less than 2% was excreted in faeces and urine (combined) and less than 1% was exhaled, 8 hours after treatment (elimination after this period was not determined) (Biesmann et al., 1983).
In mallard ducks (*A. platyrynchus*) exposed to a single oral dose of 10.000 $mg.kg^{-1}$ bw of C14-17 (52%), the highest CP levels were found in fat (67 $mg.kg^{-1}$ fresh weight), gut (15 $mg.kg^{-1}$ fresh weight) and heart (7 $mg.kg^{-1}$ fresh weight). The levels in liver and muscle were much lower: 2.4 and 2.2 $mg.kg^{-1}$ fresh weight, respectively. In ring-necked pheasants (*P. colchicus*) exposed for 5 days to a diet containing 1,000 $mg.kg^{-1}$ of C14-17 (52%), the highest CP levels were found in heart (3.1 $mg.kg^{-1}$ fresh weight) and gut (1.4 $mg.kg^{-1}$ fresh weight). The levels in liver and muscle were lower, 0.6 and 0.3 $mg.kg^{-1}$ fresh weight, respectively. Because of the immaturity of the pheasants, fat was not available for analysis. Tissue CP analyses were conducted by thin-layer chromatography, after appropiate extraction procedures (Madeley and Birtley, 1980).

5.3.2. Toxicity

Short-term exposure (acute toxicity)
In oral acute toxicity studies with birds, single doses up to 10,280 $mg.kg^{-1}$ bw (mallard duck, *A. platyrynchos*) and 24,600 $mg.kg^{-1}$ bw (ring-necked pheasant, *P. colchicus*) of C14-17 (52%) did not result in mortality, abnormal clinical signs, or effects on body weight gain. Administration of higher doses was not possible in these gavage studies.
In subacute dietary toxicity studies with 2-w old birds of these species, groups were exposed to dietary dose levels of 1,000 or 24,000 $mg.kg^{-1}$ of C14-17 (52%) for 5 days. In these studies, treatment did not affect body weight gain or health condition. At autopsy no abnormalities were noted in either species (Madeley and Birtley, 1980).

Developmental toxicity
The effect of C10-C13 (58%) on reproductive parameters of birds was investigated in a feed study in which groups of young adult mallard ducks (males and females) were exposed to dose levels of 0, 28, 166 and 1,000 $mg.kg^{-1}$ feed. Parameters studied were egg production, egg weight, eggshell thickness, eggshell cracks, egg fertility, hatchability and 14-d post-

hatch survival. No effects were seen in adults or hatchlings at any of the dose levels. A slight effect on egg fertility (not statistically significant over the entire study period) and eggshell thickness was observed at the high-dose level; the two lower-dose levels were without effect resulting in a NO(A)EL of 166 $mg.kg^{-1}$ feed (Serrone et al., 1987).

Additional data toxicity

Injection of C10-13 (49%), C10-13 (70%) or C22-26 (42%) into the yolk of chick eggs at dose levels up to 200 $mg.kg^{-1}$ egg did not affect the hatchability (Brunström, 1983; abstract). A sublethal dose (300 $mg.kg^{-1}$ egg, according to Annema, 1989) of these grades injected after 4 days of incubation resulted in liver effects such as increased organ weight and effects on microsomal enzyme activities. In these respects, C10-13 (70%) was the most potent and C22-26 the least potent grade studied (Brunström, 1985, abstract).

5.4. TOXICOLOGICAL LIMIT VALUES

5.4.1. Humans

All data in this section are based on animal studies; human data are not available, with exception of one study on *in vitro* percutaneous absorption.
Gastrointestinal absorption and metabolism (ultimately yielding carbon dioxide) of chlorinated paraffins are inversely related to chain length and degree of chlorination. The highest levels are found in tissues with a high metabolic or excretory activity, in adipose tissue and, with respect to shorter-chain grades, in the ovaries. Metabolism to carbon dioxide that is exhaled is a major route of elimination for low-chlorinated compounds. Faecal excretion increases with increasing chain length and chlorination, due to decreasing absorption and decreasing metabolism to carbon dioxide.

Oral exposure (tolerable daily intake)

The grade C10-13 (58%) is considered to be embryo-/foetotoxic and teratogenic, although the latter effect (observed in rats) was accompanied by severe maternal toxicity. Of the other three grades tested in teratogenicity studies [C14-17 (52%), C20-30 (43%) and C22-26 (70%)], C20-30 (43%) and (possibly) C22-26 (70%) are considered to be embryo-/foetoxic, but not teratogenic. Embryo-/foetotoxic effects of C10-13 (58%) and C20-30 (43%) were also observed at dose levels that did not result in signs of maternal toxicity (at short-term exposure during gestation). There is insufficient evidence for carcinogenicity of a long-chain grade [C23 (43%)] in experimental animals, but there is sufficient evidence for the carcinogenicity of a short-chain grade [C12 (60%)] in experimental animals. Both *in vitro* and *in vivo* genotoxicity tests showed no evidence for genotoxicity of chlorinated

paraffins, C12 (60%) and C23 (43%) included. Therefore, C12 (60%) is considered to be a non-genotoxic carcinogen, which implies that there is a threshold level for carcinogenicity. The data on the different toxicity parameters studied consistently show that the toxicity of chlorinated paraffins is primarily dependent on chain length, showing decreasing toxicity with increasing chain length (which is in agreement with a lower accumulation potential of long-chain compounds). Therefore, and because chronic toxicity and carcinogenicity studies have been conducted only with C12 (60%) and C23 (43%), only two categories of chlorinated paraffins have been distinguished below to derive toxicological limit values, namely shorter-chain grades (i.c. C≤17) and longer-chain grades (i.c. C≥18).

Shorter-chain grades (C≤17)
The chronic toxicity and carcinogenicity studies with C12 (60%) resulted in a LED (lowest-effect-dose) of 90 $mg.kg^{-1}$ $bw.day^{-1}$; lower dose levels were not tested. Semi-chronic toxicity studies and teratology studies with several shorter-chain grades indicate a NO(A)EL (no-observed-(adverse)-effect-level) of 10 $mg.kg^{-1}$ $bw.day^{-1}$. A fertility and reproduction study, in which the progeny was exposed both pre- and post-natally, resulted in a LED of 5 $mg.kg^{-1}$ $bw.day^{-1}$ (reduced body weight gain of the pups at a dietary dose level of 100 $mg.kg^{-1}$ of C14-17 (52%), the lowest dose tested). Extrapolation of the NO(A)EL of 10 $mg.kg^{-1}$ $bw.day^{-1}$ from the semi-chronic toxicity studies to a tolerable daily intake for humans at life-time exposure results in a TDI of 0.01 $mg.kg^{-1}$ $bw.day^{-1}$ (safety factor: 1,000 [10 x 100], i.e. a factor 10 for extrapolation from 'semi-chronic' to 'chronic' and a factor 100 for extrapolation from animals to man). The TDI of 0.01 $mg.kg^{-1}$ $bw.day^{-1}$ is 500-times lower than the LED from the reproduction study.

Longer-chain grades (C≥18)
Toxicity studies with C20-30 (43%) grades consistently resulted in an adverse effect in the liver in rats, namely granulomatous or lymphohistiocystic inflammation, at all dose levels tested; a NO(A)EL could not be derived for this effect. The LOEL (lowest-observed-effect-level) with respect to this effect is 70 $mg.kg^{-1}$ $bw.day^{-1}$, the lowest dose tested in the chronic toxicity and carcinogenicity study in rats. Extrapolation of this LED of 70 $mg.kg^{-1}$ $bw.day^{-1}$ to a tolerable daily intake for humans at life-time exposure results in a TDI of 0.07 $mg.kg^{-1}$ $bw.day^{-1}$ (safety factor: 1,000 [10 x 100], i.e. a factor 10 for extrapolation from LOEL to NO(A)EL and a factor 100 for extrapolation from animals to man).

<u>Exposure by inhalation</u>
Data on exposure by inhalation are lacking. Therefore, acceptable airborne concentrations can not be established. Because of the low vapour pressure of these compounds (KEMI, 1991) this route of exposure is not expected to threaten human health (outside occupational settings).

5.4.2. Ecosystems

Aquatic organisms (surface water)
It is concluded that the toxicity of chlorinated paraffins is inversely related to chain length (which is in agreement with the higher accumulation potential of short-chain compounds compared to long-chain compounds), and that some species of crustaceans are much more sensitive to short-chain chlorinated paraffins than fish.
Because of the limited number of toxicity data (a few LC50-values and NOLC-values), only a 'preliminary effect assessment' is possible. This kind of effect assessment is based on applying assessment factors according to the EPA (1984) method modified by the OECD (OECD, 1991). The results obtained are considered as tentative values for 'maximum permissible levels', MPLs (Slooff, 1992).
Because of the limitations of the data and to be consistent with the toxicological limit values derived for humans, only distinction is made between the shorter-chain grades (i.c. C≤17) and longer-chain grades (i.c., C≥18). No distinction between freshwater and seawater has been made.

Shorter-chain grades (C≤17)
The data available clearly indicate that crustaceans are much more susceptible to some shorter-chain CPs than fish, especially to C10-13 grades. Assuming that algae will be less susceptible to CPs as well (based on the chemical structure of CPs), an assessment factor of 100 is applied to the lowest LC50 value for crustaceans, 60 $\mu g.l^{-1}$. Rounded off, this implies a tentative maximum permissible level of 1 $\mu g.l^{-1}$ for the shorter-chain grades. With respect to C14-17 grades it is noted that this MPL will overestimate the actual risk, because LC50-values for these grades are orders of magnitudes higher than the water solubility.

Longer-chain grades (C≥18)
For longer-chain grades it is not possible to derive a MPL because LC50-values are orders of magnitudes higher than water solubility.

Taking into account the high Kow-values, consideration is given to the potential of CPs to cause food-chain effects, resulting in unacceptable accumulation and effects in higher species.
Short-chain CPs show a high bioconcentration potential as may be derived from the log Kow in combination with the molecular weight. Therefore this grade may have an impact on predators at the top scale of the ecosystem (Romijn et al., 1991). Bioconcentration, however, is also dependent on the metabolic capacity of the organisms. In contrast to mussels fish are able to metabolize CPs (except highly-chlorinated grades containing about 70% chlorine), resulting in lower BCFs than those for mussels (5.2.1). Fish-eating birds and mammals therefore are exposed to lower CP levels than may be predicted from Kow estimates. The risk of biomagnification is considered

low; field observations showed that concentrations of CPs in fish predators are lower than those in fish. However, the possible effects on birds and mammals that are specialized in eating mussels need further consideration.
In subacute toxicity studies with birds, dietary dose levels of 1,000 $mg.kg^{-1}$ and 24,000 $mg.kg^{-1}$ of C14-C17 (52%) did not result in effects. In a fertility and reproduction study with birds, a dietary dose level of 166 $mg.kg^{-1}$ of C10-13 (58%) did not result in effects. For mammals, the lowest-effect-dose (LED) observed in studies with laboratory animals is 5 $mg.kg^{-1}$ $bw.day^{-1}$ for shorter-chain grades and 70 $mg.kg^{-1}$ $bw.day^{-1}$ for longer-chain grades (5.4.1). These levels are equivalent to a dietary dose level of 100 and 1,400 $mg.kg^{-1}$, respectively. Applying (in accordance with Slooff, 1992) an assessment factor of 10 to the dietary dose level of 100 $mg.kg^{-1}$, a level of 10 $mg.kg^{-1}$ feed is considered a maximum, all the more the factor of 10 is applied normally to a NO(A)EL. For shorter-chain CPs, experimental steady-state bioconcentration factors up to 10,000 to 100,000 have been reported. Based on these data, shorter-chain CP concentrations of 0.1-1.0 $\mu g.l^{-1}$ seawater may result in unacceptable effects in mussel-eating mammals. Birds appear to be less sensitive, but semi-chronic or chronic toxicity studies are not available for this kind of animals.
Data on the toxicity of CPs to sediment dwelling species are not available. Therefore a tentative maximum permissible concentration is derived by the equilibrium-partition method, deriving the partition coefficient (Kp) according to the formula Kp = 0.05 x Kow, as described in Slooff et al. (1992). Based on a Kow of 6 and a maximum permissible short chain CP-concentration in water of 1 $\mu g.l^{-1}$, the tentative maximum permissible concentration in sediment for shorter-chain grades is estimated at 50 $mg.kg^{-1}$. The permissible level for longer-chain grades, which are less toxic, will be much higher.

Terrestrial organisms (soil)
Data on microbe-mediated processes and on terrestrial invertebrates and plants are lacking. Therefore, MPLs for chlorinated paraffins in soil can not be derived. Provisionally they may be set equal to the levels derived for the sediment.

6. EVALUATION

Information on the effects of chlorinated paraffins is scarce whereas data on exposure in The Netherlands are lacking. This lack of information seriously hampers a sound risk evaluation. It is evident, however, that the toxicity of chlorinated paraffines is inversely related to chain length. Therefore, as far as the availability of the data allows, distinction is made between the shorter-chain grades (C≤17) and longer-grades (i.c., C≥18).

6.1. RISKS TO HUMANS

The data indicate that oral exposure is the major and critical route of exposure. Data on effects on humans are not available. Hence the risk evaluation is based on tolerable daily intakes (TDI) which have been derived from animal studies.

For shorter-chain grades (C≤17) a TDI of 0.01 $mg.kg^{-1}.bw.day^{-1}$ has been established, corresponding to a daily intake of 0.6 mg for a person weighing 60 kg. No data on the occurrence of CPs in food are available for The Netherlands. If the levels in food are comparable to those measured a decade ago in Europe's largest producer, the UK, the tolerable daily intake is not likely to be exceeded (see chapter 4). Consumption of seamussels (which contained 0.1-12 $mg.kg^{-1}$ C10-20 CPs, the average being 3 $mg.kg^{-1}$) may result in a relatively high intake of short-chain CPs, but taking into account the food pattern of the Dutch population it seems unlikely that the TDI will be exceeded more than occasionally. Therefore, the risk for humans is considered to be small.

Although the data on longer-chain CP levels in food are even more limited than those for shorter-chain CPs, they indicate that the levels are likely below those of the shorter-chain CPs. Taking into account the lower toxicity of the longer-chain CPs (TDI 0.07 $mg.kg^{-1}$ $bw.day^{-1}$, corresponding to a daily intake of 4.2 mg for a person weighing 60 kg), they do not present a risk to the general population in The Netherlands.

6.2. RISKS TO ECOSYSTEMS

For the aquatic environment a tentative maximum permissible concentration of 1 $\mu g.l^{-1}$ has been derived for the shorter-chain grades (C≤17), based on LC50-values for crustaceans which appear to be much more sensitive than fish.
No data on the occurrence of chlorinated paraffins in Dutch surface waters are available. Rough model calculations indicate levels of 0.001 $\mu g.l^{-1}$ in 'filtered' water, corresponding to about 0.01 $\mu g.l^{-1}$ surface water containing 30 $mg.l^{-1}$ suspended matter. Higher levels have been reported in

the UK, the average levels for C10-17 being 9.6 and 2.2 $\mu g.l^{-1}$ in 1985 and 1986, respectively. Based on this quantitatively and qualitatively poor information, one may assume that nation-wide the CP levels in The Netherlands will not exceed the proposed tentatative maximum permissible concentration of 1 $\mu g.l^{-1}$. Based on the diffuse character of the sources one may even argue that it is not very likely that there are locations at which this level is exceeded. Also the indication that shorter-chain CP levels of 0.1-1.0 $\mu g.l^{-1}$ may result in unacceptable effects in mammals (and possibly birds) that are specialized in eating seamussels, does not give rise to much concern. This is strengthened by the observation that the level of shorter-chain CPs in seamussels that may affect mussel-eating animals, i.e. 10 $mg.kg^{-1}$, has been found only in seamussels living close to an effluent discharge of a CP manufacturing plant. Hence, the risk of the shorter-chain CPs is very likely acceptable.
Taking into account the lower toxicity and lower environmental levels of longer-chain CPs, the risk associated with these compounds is considered negligible.

No data on effects on soil organisms are available, nor there are data on the occurrence of CPs in soil. Therefore the risk cannot be evaluated. It is noted, however, that the provisionally MPL for shorter-chain CPs derived from the aquatic MPL is estimated at 50 $mg.kg^{-1}$, whereas the model calculations indicate a 'total' CP concentration in soil (table 4.1) of 0.03 $mg.kg^{-1}$. In spite of the lack of data, the risk of chlorinated paraffins at the present assumed level of emissions is considered small, if present at all.

6.3 DEGRADATION AND BIOCONCENTRATION

The most important reason for proposing to abandon chlorinated paraffins (KEMI, 1991) is based on the intrinsic properties of the chlorinated paraffins, indicating that these substances are highly persistent to chemical or biological degradation in the environment and that they are highly bioaccumulative, giving rise to effects due to biomagnification.

It is noted that CPs usually enter the environment as a mixture of various molecular species. Thus, the chemical properties of CPs are the average of the chemical properties of the constituents. Differences have been observed in both persistence and bioconcentration potential. The following observations were made:

- Biodegradation is expected to occur in sediment and soil, the biodegradation rate being higher under anaerobic conditions than under aerobic conditions. The degradation rate has been shown to decrease with increasing carbon chain length and chlorine content. For soil a half-life time of about 5 days has been calculated, but due to a low bioavailability the actual half-life is presumably much higher.

- CPs do not ondergo hydrolysis or oxidation or otherwise react under natural conditions, which is in accordance with some applications which require chemical resistance. However, CPs can be catalytically dehydrochlorinated in the presence of iron oxides as well as other inorganic compounds and the possibility of this process occurring in nature seems quite feasible.
- Secondary and sensitized photolysis cannot be ruled out. In the atmosphere a half-life time of 0.85-7.2 days has been reported. However, actual the photodegradation rate will be lower because air borne CPs are presumably bound on atmospheric particulate matter.
- Bioconcentration potential decreases with increasing molecular weight (determined by carbon chain length and chlorination) and with increasing Kow (chlorine content). The bioavailability of substances with a molecular weight > 600 is generally considered to be small. Further the bioconcentration also depends on the ability to metabolize chlorinated paraffins; this ability has been shown in fish (except for highly chorinated CPs) but not in mussels. Accordingly the highest bioconcentration factors were observed for mussels. There are no indications that biomagnification, the occurence of a substance at successively higher levels with increasing trophic levels in food chains, occurs.

7. RECOMMENDATIONS

Based on scanty information on exposure and effect levels it is concluded that chlorinated paraffins do not seem to present a significant risk to humans or ecosystems in The Netherlands. However, since chlorinated paraffins are rather toxic (especially those with a short carbon chain and a high chlorine content) and may be persistent under natural conditions, these compounds may need further attention.
In order to indicate the potential exposure it is recommended to make an inventory of the use and associated release of (short-chain) chlorinated paraffins in The Netherlands, and preferably in Europe.

8. REFERENCES

Annema, J.A.A. (1989)
Milieu-aspecten van Gechloreerde Paraffines
Stichting Natuur en Milieu, Utrecht

Ashby, J. et al. (1990)
Cell replication and unbscheduled DNA synthesis (UDS) activity of low molecular chlorinated paraffins *in vivo* Mutagenesis 5, 515-518

Beenen, J.H. (1991a)
Chlorinated paraffins
Review of the literature and discussion of the KEMI-report 1/91.
Draka-Polva B.V., Enkhuizen

Beenen, J.H. (1991b)
Personal communication. Draka-Polva B.V., Enkhuizen

Bengtsson, B-E. and E. Baumannn-Ofstad (1982)
Long-term studies on uptake and elimination of some chlorinated paraffins in the bleak, *Alburnus alburnus*
Ambio 11 (1), 38-40

Bengtsson, B-E. et al. (1979)
Structure related uptake of chlorinated paraffins in bleaks (*Alburnus alburnus* L)
Ambio 8 (2-3), 121-122

Biesmann, A. et al. (1983)
Chlorinated paraffins: disposition of a highly chlorinated poly-chlorohexadecane in mice and quail
Arch. Toxicol. 53, 79-86

Birtley, R.D.N. et al. (1980)
The toxicological effects of chlorinated paraffines in mammals
Toxicol. Appl. Pharmacol. 54, 514-525

Bremmer, H.J. (1988)
Halogeenverbindingen in metaalbewerkingsvloeistoffen
Report no 738608004. RIVM, Bilthoven, June 1988

Brunström, B. (1983)
Toxicity in chick embryos of three commercial mixtures of chlorinated paraffins and of toxaphene injected into eggs (Abstract in MEDLINE)
Arch. Toxicol. 54, 353-357

Brunström, B. (1985)
Effects of chlorinated paraffins on liver weight, cyto-chrome P-450 concentration and microsomal enzyme activities in chick embryos (Abstract in MEDLINE)
Arch. Toxicol. 57, 69-71

Bucher, J.R. et al. (1987)
Comparative toxicity and carcinogenicity of two chlorinated paraffins in F344/N rats and B6C3F1 mice
Fundam. Appl. Toxicol. 9, 454-468

Campbell, I. and G. McConnell (1980)
Chlorinated paraffins and the environment.1 Environmental occurence
Environ. Sci. Technol. 14, 1209-1214

CEFIC (19920
Information was obtained by the CEFIC sector group "Chlorinated Paraffins", an association regrouping the major European producers of chlorinated paraffins (i.e. Hoechst, ICI, Caffaro, CECA and Quimica del Cinca) through the Sector Group Manager, R. van Sloten

Ceha, M.J. (1991)
Personal communication. ICI Nederland, Rotterdam

Darnerud, P.O. and I. Brandt (1985)
Pitfalls in the interpretation of whole-body autoradiograms: long-time retention in brain and adrenal cortex caused by metabolic incorporation of ^{14}C from various labelled xenobiotics
Acta Pharmacol. et Toxicol. 56, 55-62

Darnerud, P.O. et al. (1982)
Metabolic fate of chlorinated paraffins: degree of chlorination of [1-^{14}C]-chlorododecanes in relation to degradation and excretion in mice
Arch. Toxicol. 50, 217-226

Eriksson, P. and A. Nordberg (1986)
The effects of DDT, DDOH-palmitic acid, and a chlorinated paraffin on muscarinic receptors and the sodium-dependent chloline uptake in the central nervous system of immature mice
Toxicol. Appl. Pharmacol. 85, 121-127

EPA (1984)
Estimating 'Concern Levels' for Concentrations of chemical substances in the Environment
Environmental Effects Branch, Health and Environmental Review Division, United States Environmental Protection Agency

Ham, A.T.J., P.C.M. Frintrop and P. Regoort (1991)
Bijdrage van de lederindustrie aan het gehalte aan extraheerbaar organochloor verbindingen in zuiveringsslib. Een rapport van het Hoogheemraadschap West-Brabant en het RIZA
RIZA-report no 91.011, March 1991

Haux, C. et al. (1982)
Sublethal physiological effects of chlorinated paraffins on the flounder, *Platichthys flesus* L.
Ecotox. Environ. Saf. 6, 49-59

IARC (1990)
IARC Monographs on the Evaluation of the Carcinogenic Risks to Humans, Volume 48: Some Flame Retardants and Textile Chemicals, and Exposures in the Textile Manufacturing Industry. International Agency for Research on Cancer, World Health Organization, Lyon

KEMI (1991)
Risk Reduction of Chemicals, 167-187 (Chapter 9: Chlorinated paraffins. KEMI report no. 1/91 (Editor: L. Freij), The Swedish National Chemicals Inspectorate (KEMI) and The Swedish Environmental Protection Agency (SNV)

Kirk-Othmer (1979)
Encyclopedia of Chemical Technology. Third edition, volume 5, pp. 786-790

Lindén, E. et al. (1979)
The acute toxicity of 78 chemicals and pesticide formulations against two brackish water organisms, the bleak (*Albernus alburnus*) and the harpacticoid *Nitocra spinipes*
Chemosphere 8, 843-851

Madeley, J.R. and R.D.N. Birtley (1980)
Chlorinated paraffins and the environment. 2. Aquatic and avian toxicology
Environ. Sci. Technol. 14, 1215-1221

Nilsen, O.G. et al. (1981)
Effects of chlorinated paraffins on rat liver microsomal activities and morphology - Importance of the chain length and the degree of chlorination of the carbon chain
Arch. Toxicol. 49, 1-13

OECD (1991)
Draft report on the OECD Workshop on the extrapolation of laboratory aquatic toxicity data to the real environment held in Washington (USA)

NTP (1986a)
Toxicology and Carcinogenesis Studies of Chlorinated Paraffins (C23, 43% Chlorine) in F344/N Rats and B6C3F1 Mice (Gavage Studies)
National Toxicology Program Technical Report Series No. 305, NIH-Publication 86-2561, U.S. Department of Health and Human Services, Public Health Service, National Institutes of Health

NTP (1986b)
Toxicology and Carcinogenesis Studies of Chlorinated Paraffins (C12, 60% Chlorine) in F344/N Rats and B6C3F1 Mice (Gavage Studies)
National Toxicology Program Technical Report Series No. 308, NIH-Publication 86-2564, U.S. Department of Health and Human Services, Public Health Service, National Institutes of Health

Renberg, L. et al. (1986)
The use of the bivalve *Mytilus edulis* as a test organism for bioconcentration studies, II. The bioconcentration of two ^{14}C-labeled chlorinated paraffins
Ecotox. Environ. Saf. 11, 361-372.

Romijn, C.A.F.M. et al. (1991)
Presentation of a general algorithm for effect-assessment on secondary poisoning. I. Aquatic food chains
RIVM report no 67912002

Schmid, P.P. and M.D. Müller (1985)
Trace level detection of chlorinated paraffins in biological and environmental samples, using gas chromatography/mass spectrometry with negative-ion chemical ionization
J. Association off Anal Chem, 68 (3)

Serrone, D.M. et al. (1987)
Toxicology of chlorinated paraffins
Fd Chem. Toxic. 25, 553-562Svanberg, O. et al. (1978)
Chlorinated paraffins - A case of accumulation and toxicity to fish
Ambio 7 (2), 64-65

Slooff, W. (1992)
RIVM Guidance Document. Ecotoxicological effect assessment: Deriving Maximum Tolerable Concentrations (MTC) from single-species toxicity data
RIVM report no. 719102018, Bilthoven, The Netherlands

SRI International (1987)
Directory of Chemical Producers United States of America California, USA

SRI International (1991)
Directory of Chemical Producers Western Europe. Volume 2, California, USA

Svanberg, O. and E. Lindén (1979)
Chlorinated paraffins - an environmental hazard?
Ambio 8 (5), 206-209

Svanberg, O. (1983) (Ed.)
Chlorinated Paraffins - A review of Environmental Behaviour and Effects
Report SNV PM 1614, National Swedish Environmental Protection Board, Solna. ISBN 91-7590-114-5, Berlings, Arlöv

VROM (1990)
Stoffen en Normen. Overzicht en achtergronden van belangrijke stoffen en normen in het milieubeleid.
Directoraat-Generaal Milieubeheer. Handboek milieuvergunningen, augustus 1991.

WHO (World Health Organization) (1990)
IARC monographs on the evaluation of carcinogenic risks to humans. Some flame retardants and textile chemicals. and exposures in the textile manufacturing industry
Volume 48, International Agency for Research on Cancer, Lyon, pp. 55-72

Willis, B. (1991)
Chlorinated paraffins: Environmental exposure
Note N112/91. Building Research Establishment, Garston, Watford

Yang, J.J. et al. (1987)
Percutaneous and oral absorption of chlorinated paraffins in the rat
Toxicol. Ind. Health 3, 405-412

APPENDIX 1

The values of the parameters used in the Mackay model of chlorinated paraffin compound are given in table appendix 1

Table appendix 1.1 Physico-chemical parameters used in the Mackay model of chlorinated paraffin compound

Water solubility[a]	10×10^{-6} g/l
Molecular weight	399
Octanol-water partiton coefficient[b]	Log P_{ow} = 7.89
Vapour pressure[a]	10^{-6} mm Hg
Soil degradation rate[a]	0.0002 day^{-1}
Air degradation rate[a]	0.02 day^{-1}

a: Campbell and McConnel (1980)
b: Pilotte (cited in Willis (1991))

Specific characteristics of the environmental compartments of The Netherlands were used. Since no information was available on the contribution from abroad, these were estimated by calculating the environmental CP concentrations in The Netherlands at equilibrium and these concentrations (in surface water and air) were assumed to be present in the environment that is transported into The Netherlands.
Calculations were made according to SIMPLESAL version 5.3 (890322).