THE TIP OF THE ICEBERG

State of knowledge on

persistent organic pollutants

in Europe and the Arctic

GREENPEACE

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CONTENTS

SUMMARY

Persistent organic pollutants (POPs) are a group of chemicals which are very resistant to natural breakdown processes and are therefore extremely stable and long-lived. Most do not occur in nature but are man-made. Once released into the environment, many POPs persist for years, even decades. Many POPs are also highly toxic and build up (bioaccumulate) in the fatty tissues of animals and humans. These 3 properties – persistent, toxic and bioaccumulative, make them, arguably, the most problematic chemicals to which natural systems can be exposed. In recent decades, vast numbers of POPs have been produced, used worldwide and are still in production and use. These chemicals have become widespread pollutants in the environment and even contaminate regions remote from their source, such as the Arctic, deep oceans and mountain areas.

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Arctic Glacier

Certain POPs have been responsible for some catastrophic effects in wildlife, ranging from interference with sexual characteristics to dramatic population losses. Moreover, possible health effects of POPs on wildlife are still becoming apparent today. In humans, levels of POPs in some women of the general population in European countries and Indigenous Arctic Peoples may be sufficient to cause subtle undesirable effects on the immune system and growth of their babies. The developing young of both wildlife and humans are the most vulnerable to toxic effects of POPs. They are exposed to these chemicals because POPs are passed to the foetus in the womb from a mother's body via the placenta and through breast milk to the infant.

Numerous potentially harmful POPs are currently in production and used in everyday products. Only a few POPs have been banned in most countries. However, even these banned POPs continue to contaminate the global environment because of their long-lived nature and because some are still manufactured and used in some countries.

This report draws together scientific findings on possible health effects of POPs on wildlife and humans in Europe and in the remote circumpolar Arctic. Special emphasis is placed on marine mammals since they carry high levels of POPs in their tissues and are particularly susceptible to the toxic effects of these chemicals. The report reveals that we already know that many POPs are widespread environmental contaminants and they may cause detrimental effects in humans and wildlife, particularly on the next generation. Moreover, our current understanding of the effects of POPs may only be the tip of the iceberg - future research could reveal yet more toxic effects attributable to environmental contamination by POPs. The majority of POPs and other chemicals in use have not even been tested to assess their potential hazards. With these factors in mind the report demonstrates that there is an urgent need for the complete phase out of all releases of POPs to the environment.

What are POPs?

POPs encompass many different and varied groups of man-made chemicals. Some POPs have been highlighted by national and international organisations as being chemicals of concern. For instance, the United Nations Environment Program (UNEP) has listed certain POPs, which are organochlorines, as being chemicals of clear concern. Organochlorines are substances containing chemically combined chlorine and carbon. They are a huge group of chemicals that include many POPs. The UNEP list notes 12 organochlorines – known as the dirty dozen. They are the dioxins and furans, (produced as unintentional by-products of combustion and processes involving the manufacture, use and disposal of organochlorines); PCBs, (industrial chemicals that have been banned but are still released to the environment in significant amounts from old sources); HCB (used in the manufacture of

pesticides and produced as an unwanted by-product of various industrial processes involving organochlorines); several organochlorines used as pesticides, - DDT, chlordane, toxaphene, dieldrin, aldrin, endrin, heptachlor and mirex. Use of these organochlorine pesticides is banned or is severely restricted in most countries, but not in all.

The Convention for the Protection of the Marine Environment of the North East Atlantic (OSPAR) has made a commitment for the cessation of emissions, discharges and other losses to the environment of *ALL hazardous substances* in ONE GENERATION (ie. by the year 2020). Although the mechanism for prioritising hazardous substances is still under development, an initial list of 15 chemicals or chemical groups has been selected for priority action. This list includes a variety of POPs and other hazardous substances, eg. heavy metals. Some of these POPs have already been phased out of mainstream production such as pentachlorophenol, but others are still produced and released to the environment on a daily basis. These include dioxins and furans, brominated flame retardants, HCH isomers - such as the organochlorine pesticide lindane, musk xylene (used as a synthetic fragrance), organotin compounds (for example, used as anti-fouling agents for ships), short chained chlorinated paraffins (for example, used in cutting oils and lubricants) and certain phthalates – DBP and DEHP, which are not particularly persistent but are none the less hazardous (main uses as plastic softeners, especially in PVC).

POPs included in the above lists are of immense concern given that they contaminate the environment and have potentially toxic effects. Most research on POPs is limited to a few of these chemicals and information on the remaining ones has only begun to be generated more recently. Similarly, there are vast numbers of other hazardous chemicals for which there is little scientific data. It should also be stressed that the UNEP and OSPAR lists are not complete lists of POPs and hazardous chemicals. There are numerous others which are also environmental contaminants and are of great concern.

Where are they found?

POPs contaminate local areas close to sites where they are released into the environment from industry and agriculture. However, volatile and semi-volatile POPs also contaminate regions remote from their source because they can be transported for thousands of kilometres on air currents. These POPs migrate on air currents from warmer regions of the globe towards colder polar regions. Once they reach colder temperatures they condense, precipitate out, and are deposited again on the Earth's surface. POPs may also be transported for long distances by rivers, ocean currents and as contaminants in wildlife.

Due to global transport of POPs, substantial levels of these chemicals are present in the Arctic, in some cases at levels similar to those in heavily industrialised countries. Thus, while countries from lower latitudes act as a source of POPs, the Arctic acts as a sink for some of these contaminants. Under cold Arctic conditions, POPs are likely to degrade at a slower rate than in temperate regions and will therefore persist in this environment for even longer. Contamination of the Arctic environment will continue for many decades after cessation of POPs source emissions as a consequence of long-distance transport and the persistent nature of these chemicals.

POPs in Food Webs

Many POPs which pollute the environment become incorporated into food webs. They accumulate and persist in the fatty tissues of animals and humans because they are soluble in fats and are not easily broken down in the body. Even low environmental levels of POPs can lead to high levels in the body tissues of animals and humans. For many POPs, the levels in fat increase as one animal eats another, so that the highest levels are found in predator animals at the top of food webs, such as polar bears, seals, toothed whales, birds of prey and humans. Marine mammals accumulate particularly high levels of POPs because of their large quantities of fatty blubber and a reduced capacity to break down some POPs compared to other species.

Wildlife exposure to POPs

In the North East Atlantic, organochlorines such as DDT and PCBs are detectable in all marine mammal species that have been tested. In addition to organochlorines, other POPs which are still produced and used, such as brominated flame retardants and organotin compounds, have also been found in several marine species. This indicates that the oceans are widely contaminated with these POPs, even the deep seas.

In the Arctic, levels of PCBs and DDT in a range of marine mammals were shown to be around 15 times lower than in animals from the North Atlantic. However, this North to South difference in levels is not apparent for some other organochlorines such as HCHs, chlordanes and toxaphene which are present in the Arctic at levels similar to those in North Atlantic marine mammals. In addition, levels of PCBs in polar bear, and Arctic fox are, at least in some regions, exceptionally high. Scientific data on POPs other than organochlorines in Arctic marine mammals are virtually non-existent.

Effects on Wildlife

Europe

A variety of effects on the reproductive and immune system of marine mammals have been associated with levels of POPs in their tissues including:

- Reproductive failure and population collapse in common seals in the Wadden Sea (1950-75).

- A disease-complex (adrenocortical hyperplasia) in ringed and grey seals from the Baltic Sea in the 1970s which involved poor fertility, skull abnormalities and enlargement of the adrenal gland. Some of these effects persist today.

- Mass mortality of seals around European coasts in 1992 caused by a virus. Organochlorines were suspected to exacerbate the problem by weakening the immune system.

- Decline of harbour porpoise populations in the North Sea and Baltic Sea, possibly related to organochlorines.

- Mass mortalities of striped dolphins in the Mediterranean 1990-92 caused by a virus. Organochlorines and organotins suspected to exacerbate the problem by weakening the immune system.

In addition to marine mammals, other wildlife have suffered from the toxic effects of POPs. Effects include dramatic population declines of the European otter possibly linked to PCB exposure, reproductive problems and population declines in seabirds and birds of prey, reproductive effects in marine fish, and regional extinctions of dog whelks due to TBT.

The Arctic

In the Arctic, limited field studies that are available indicate that some effects on wildlife may be associated with exposure to POPs:

- Levels of PCBs in Arctic Beluga are associated with subtle effects on health – induction of cytochrome p-450 enzymes at times of poor feeding when fat reserves have been mobilised.

- Feminised gonads in male Bowhead whales. It is not known, but it is possible that POPs are involved in the high incidence (2 in 155 whales) of this condition.

- Levels of PCBs in polar bears are associated with induction of the cytochrome p-450 enzyme system.

- A high death rate in polar bear cubs possibly due to high levels of organochlorines.

- Abnormally formed genitalia in four polar bears. Possibly due to exposure to high levels of PCBs in the womb.

- A recent study conducted by the Arctic Monitoring Assessment Program (AMAP) compared tissue levels of Arctic wildlife species with levels that are known to cause detrimental effects in

experimental animals. It reported that current levels of organochlorines in polar bear, Arctic fox and harbour porpoise places them at risk from reproductive and immunosuppressive effects on health and neurobehavioral effects in offspring. Beluga, Narwhal and seals are potentially at risk from neurobehavioural effects in offspring.

Exposure to POPs and Effects in Humans

For the general population, the greatest exposure to POPs is from food intake. Since many POPs are fat soluble, the highest levels are present in meat fish and dairy products. Dioxins, PCBs, several organochlorine pesticides and brominated flame retardants are detectable worldwide in human breast milk, adipose tissue and blood, which reflects widespread exposure to these chemicals.

Individuals who are more highly exposed to POPs than the general population include those who consume large amounts of fish or sea mammals from contaminated waters. This includes a group of people who would least be expected to have high exposure to POPs – Indigenous Peoples from the Arctic who consume a traditional diet. Levels of organochlorine pesticides in breast milk from Inuit women residing in Arctic Quebec were 4 times higher than women from Southern Quebec while levels of PCBs were 7 times higher. This is due to a relatively higher exposure to POPs from the consumption of sea mammals in the diet. Furthermore, tissue levels of PCBs and dioxins in Greenland Inuit were 3 times higher than Inuit from Arctic Quebec.

Research indicates that levels of dioxins and some organochlorine pesticides in humans have decreased or stabilised in some European countries in recent years. However, other POPs may now add increasingly to the already existing levels of these POPs in breast milk. For instance, levels of brominated flame retardants have doubled every 5 years over the past 25 years in breast milk from Swedish women. Nitro musks are also detectable in breast milk in European countries.

A substantial proportion of the POPs which have accumulated in a woman's body during her whole lifetime are passed to her child during development in the womb and through breast-feeding. Studies show that current levels of PCBs/dioxins in some women of the general population are associated with subtle, but significant undesirable effects on the nervous system, immune system and postnatal growth of their babies. These effects were evident in healthy infants from the Netherlands whose mothers' had slightly higher levels of PCBs/dioxins in their bodies. Similarly, in the Arctic, babies born to women who had higher levels of PCBs in their breast milk had subtle undesirable effects on their immune system and on postnatal growth. This is obviously of great concern, particularly as effects are occurring at current background levels and effects caused during foetal/infant development are likely to be irreversible.

What Must Be Done

That POPs have become widespread environmental contaminants is abundantly clear from the presence of many POPs in animal and human tissues in both industrialised Europe and in remote areas such as the Arctic and deep oceans. Currently, this problem of global POPs contamination is set to continue because the majority of POPs from man's activities are still being released into the environment. Only a few have been banned and even this is in limited parts of the world. Furthermore, the decrease in environmental levels of a few banned POPs, such as DDT, gives no room for optimism or complacency. Levels of banned POPs are still high enough to be of concern, and moreover, levels of other POPs which are still being widely produced, such as the brominated flame retardants, add to the already heavy environmental burden of POPs. In addition, there may be many other POPs in breast milk which have not yet been studied.

Because the release of POPs into the environment is continuing, there is a potential for further severe impacts on the health of wildlife and humans. Continuing uses of brominated flame-retardants, for instance, could lead to devastating effects on wildlife similar to those caused by PCBs. Problems

may even worsen as the number and quantity of chemicals produced is increasing and most chemicals currently in use have never been tested to assess their potential hazards.

Many POPs are passed from a mother's body to the developing young in wildlife and humans alike. This intergenerational transfer of POPs threatens the health of future populations. Given the persistent nature of POPs, there is only one way forward to safeguard future generations. This is to phase out the production and use of all POPs and other hazardous substances and implement clean production technologies. Action must be taken now to address the existing POPs problems, prevent new problems and start on the road to a Toxics Free Future.

GREENPEACE DEMANDS...

- The production and use of all POPs must be phased out at an international and, ultimately, at a global level.

- This must be achieved through the substitution of POPs (or the processes which generate them) with non-hazardous alternatives.

- Industry and agriculture must pursue clean production technologies and manufacture clean products, recognising that the only way to prevent releases of POPs into the environment is to avoid their production and use.

- As a matter of urgency, action must be taken to stop production, eliminate all discharges, emissions and losses of those chemicals prioritised for action by OSPAR/UNEP, many of which are POPs. This is an essential first step if the target of cessation of emissions, discharges and losses of ALL hazardous substances in OSPAR countries as agreed at ministerial level, is to be achieved within one generation (by 2020).

- Presume that all chemicals are hazardous until demonstrated otherwise, ie. until hazard identification is completed, or in those instances where hazard identification is limited by lack of information, chemicals must be assumed to present hazards of unknown proportions.

- Ultimately, measures to eliminate releases of POPs and other hazardous substances to the environment will need to be taken not just on a regional but on a global basis, because chemical contamination of the environment is a global problem and chemicals do not respect national boundaries.

- OSPAR countries therefore should not only give highest priority to implement the one generation elimination goal of all hazardous substances in Europe, but also aim at this elimination goal at the global level through the UNEP convention for the elimination of POPs.

1

INTRODUCTION

The building blocks of living organisms are organic compounds – that is chemical compounds that contain carbon and hydrogen (and in some cases other elements as well). These compounds are never indestructible and many break down relatively easily. On the other hand, man has learnt to manufacture organic compounds which are extremely difficult to break down. These chemicals are termed persistent organic pollutants (POPs).

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Spitsbergen

A large number of hazardous chemicals have been, and continue to be, manufactured by the chemical industry both intentionally, as products, and unintentionally, as by-products and wastes. These hazardous substances include numerous POPs. Some of these POPs, notably the dioxins and furans, are also generated unintentionally as by-products of combustion processes.

The production and use of POPs has led inevitably to the pollution of the environment with these substances. Because they are not easily degraded by natural processes, many persist in the environment for years. Therefore, even if production of all POPs ceased today, they would continue to pollute the environment for many years to come. Numerous POPs have become very widespread contaminants in the environment because they can be transported for thousands of kilometres on air currents, and in rivers and oceans. As a result of this long-distance transport, some POPs even contaminate remote regions such as the deep oceans, high mountain areas and even the Arctic. Indeed, they may be considered as global pollutants.
In addition to being persistent, many POPs are, by their chemical nature, highly soluble in fats (lipophilic). Consequently they have a tendency to concentrate in the fatty body tissues of living organisms and, over time, can build up (bioaccumulate) to high levels in such tissues. In some cases the levels increase (biomagnify) as one animal consumes another in the food chain so that the highest levels are present in top predator species. Some POPs, such as organotin compounds, accumulate to particularly high levels in the liver and other tissues.

Many POPs are toxic and their long-lives in living tissues may lead to adverse effects on health. Although over time POPs may be metabolised (transformed or broken down) in the body to other compounds (metabolites), some of the metabolites produced are more toxic and persistent than the original chemical. For example, the pesticides heptachlor and chlordane are respectively broken down to heptachlor epoxide and oxychlordane which are more toxic than the original chemicals.

Man-made chemicals occur in the environment and in our bodies not as single entities but as complex mixtures. We are exposed, therefore, not to individual hazardous chemicals, but to many; not to individual POPs, but to diverse mixtures. The significance of such multiple exposure remains poorly understood. Moreover, a substantial proportion of the chemicals which occur in the environment and to which we may be exposed simply cannot be identified. This further complicates the problem.

1.1 The Chemicals of Concern

POPs may be defined in general terms as persistent organic chemicals, including synthetic substances from a range of chemical groups. A prominent and diverse group of POPs are the organohalogens, i.e. organic compounds of fluorine, chlorine, bromine and iodine. Of the halogens, chlorine has been particularly widely used by the chemical industry, in order to manufacture organochlorine chemicals for use as pesticides, industrial chemicals, solvents, cleaning agents and plastics, particularly PVC.
Indeed, all of the 12 POPs so far prioritised for action to reduce or prevent emissions under the

United Nations Environment Programme (UNEP) Draft POPs Convention (see Chapter 8) are organochlorine chemicals (UNEP 1995). These chemical are described in Box 1.1

Environmental and health problems caused by POPs included on the UNEP list have been recognised for some years and, as a consequence, the PCBs and many of the pesticides have been banned or have restricted use in most countries. However, POPs do not respect national boundaries, such that their continued production and use in some countries continues to add to the global burden of these chemicals. In the case of dioxins, still produced unintentionally by many industrial and waste combustion processes throughout the globe, measures have been taken to try to reduce emissions of these chemicals to the air, without properly addressing the processes which lead to their production.

The 12 UNEP POPs are only part of the problem we face. Many more persistent organic chemicals are still in widespread production and use, even in the developed world. While the chemical industry continues to manufacture such chemicals to solve day-to-day problems, they may be creating other, long-term or even irreversible problems and compromising the ability of future generations to meet their own needs. They may also be threatening the fundamental processes which support the diversity of life itself.

Although the greatest attention to date has focused, understandably, on persistent organochlorine chemicals, the general problem of the widespread contamination of the environment with persistent chemicals extends across other chemical groups. A number of more regional intergovernmental organisations, such as those responsible for the protection of regional sea areas (Mediterranean, North Sea, Baltic Sea, North East Atlantic), have recognised that, in order to ensure protection of the environment, action must be taken to reduce and ultimately prevent emissions of all hazardous substances, particularly those which are persistent and bioaccumulative.

BOX 1.1 POPs listed by UNEP

- Dioxins and furans: Polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) are commonly known as dioxins and furans respectively. There are more than 200 individual congeners, although some are more toxic, and some more abundant, than others. 2,3,7,8-TCDD is probably the most toxic form and is now recognised as a human carcinogen. Dioxins are produced as unintentional by-products of many manufacturing and combustion processes, especially processes that use, produce or dispose of chlorine or chlorine derived chemicals. Important sources of dioxins to the environment include waste incineration and many organochlorine production processes, including PVC production.

- Polychlorinated Biphenyls (PCBs): PCBs comprise of a group of 209 different congeners. Around half this number has been identified in the environment. The more highly chlorinated PCB congeners are the most persistent and account for the majority of those polluting the environment. PCBs were produced as industrial chemicals that were mainly used for insulation in electrical equipment. Production of PCBs has almost totally ceased worldwide, although there are reports of it continuing in Russia. At least one third of PCBs that have been produced are estimated to have entered the environment (Swedish EPA 1999). The other two thirds remain in old electrical equipment and in waste dumps from where they continue to leach into the environment. Although this is the major source of PCB pollution in the environment today, some PCBs are also produced as by-products of incineration and certain chemical processes involving chlorine.

- Hexachlorobenzene (HCB): This chemical was previously used as a fungicide for seed grain. It is also produced unintentionally as a by-product during the manufacture of chlorinated solvents, other chlorinated compounds, such as vinyl chloride, and several pesticides. It is a by-product in waste streams of chlor-alkali plants and wood preserving plants, and in fly ash and flue gas effluents from municipal waste incineration. Its major source today remains the manufacture of pesticides (Foster 1995, ATSDR 1997).

- Organochlorine Pesticides: There are eight pesticides in this category listed by UNEP. These are aldrin, dieldrin, endrin, DDT, chlordane, mirex, toxaphene and heptachlor. The majority of these are banned or restricted in many countries, although not all. For example, DDT is still widely used in developing countries particularly for mosquito control (*e.g.* Lopez-Carrillo *et al.* 1996).

The Convention for the Protection of the Marine Environment of the North East Atlantic (the OSPAR Convention) covers 15 States of the North East Atlantic Region and the European Union. In 1998, Ministers of the OSPAR countries agreed to a target for the cessation of discharges, emissions and losses of all hazardous substances to the marine environment by 2020. As a first step, OSPAR has selected a list of 15 hazardous chemicals or chemical groups of particular concern (OSPAR 1998a). This "list of chemicals for priority action" consists mainly of POPs. In addition to dioxins, furans and PCBs, the priority list includes numerous persistent organic chemicals (Box 1.2) which remain in widespread production and use in the OSPAR region.

BOX 1.2 POPs on the OSPAR list of chemicals for priority list

- Hexachlorocyclohexane isomers (HCH): γ-HCH, or lindane, is an organochlorine pesticide and a component of some shampoos for treatment of headlice. Its use as a pesticide in agriculture has declined in recent years, but it nevertheless continues to be used for this purpose in some countries of Europe (Swedish EPA 1999). Use of technical HCH, a mixture of HCH isomers including alpha-HCH, is yet more restricted. Nevertheless, as a result of some continued releases and its persistence in the environment, alpha-HCH remains widespread in the environment, including the Arctic.

- Brominated flame retardants: These chemicals are widely used as fire retardants in electronic equipment *e.g.* electronic boards in computers, radios and television sets, in plastics, textiles, building materials, carpets and in vehicles and aircraft. The production and use of some these chemicals is increasing. Brominated flame retardants include polybrominated diphenyl ethers (PBDEs), and polybrominated biphenyls (PBBs), as well as the more recently developed tetrabromobisphenol-A. It is becoming increasingly clear that PBDEs are widely distributed in the global environment and can accumulate in the tissues of humans and wildlife; similar evidence is growing for other brominated flame retardants

- Organotin Compounds: Organotin compounds are used as active ingredients in anti-fouling agents, fungicides, insecticides and bactericides. One of the chemicals in this group, tributyltin (TBT), has been used as an anti-fouling agent in paints for boats and aquaculture nets since the 1960s, although its use is now restricted to large vessels and a global ban is under discussion. TBT is perhaps best known for its hormone disrupting effects in marine invertebrates, although it is also highly toxic to other organisms. It has been described as perhaps the most toxic chemical ever deliberately introduced into natural waters and has become widespread in the marine environment.

- Short Chain Chlorinated Paraffins: These chemicals have for many years been used to produce a range of products, including use as fire retardants and plasticisers in PVC, rubber and other plastics, varnishes, sealants and adhesives, leather treatment chemicals and as extreme pressure additives in lubricants and metal cutting oils (Campbell and McConnell 1980). It should be noted that it is not just the short chained chlorinated paraffins that are problematic but the whole group of chlorinated paraffins.

- Musk Xylene: This chemical is a nitro-musk which, along with musk ketone, is widely used as a fragrance additive in place of natural musk extracts in cosmetics and detergents. Polycyclic musks are being used increasingly in Europe instead of nitro musks. However, all synthetic musks are persistent in the environment.

- Certain Phthalates – dibutylphthalate (DBP) and diethylhexylphthalate (DEHP): These chemicals are just two of many other phthalates which are used in industry and which have become ubiquitous environmental contaminants. While phthalates are not as persistent as some of the other POPs, they are continuously released to the environment from use in consumer products, particularly as plasticisers in soft PVC (in toys, furniture, flooring, vehicles, clothing, etc.) which accounts for approximately 90% of phthalate use. Phthalates are also used, however, in some other plastics and in non-plastic applications *e.g.* as additives in paints, pesticides, inks, perfumes, cosmetics and insect repellents.

BOX 1.3 Toxic Effects of POPs

Exposure to POPs has been associated with a wide range of impacts on health in wildlife and in humans. Effects include carcinogenicity, toxicity to the reproductive, nervous and immune systems and adverse effects on development (e.g. Silberhorn *et al.* 1990, Colborn and Clement 1992, US EPA 1994, Allsopp *et al.* 1997, Longnecker *et al.* 1997).

There are many mechanisms by which POPs can exert biological effects on health; these depend on the specific chemical in question. One mechanism which appears to be common to a wide range of POPs is disruption of the hormone (endocrine) system. Chemicals that have the capacity to disrupt hormones are known as endocrine disruptors.

These chemicals may affect one or more steroid hormones in the body, such as the male sex hormone testosterone, female sex hormones estradiol and progesterone and various hormones produced by the pituitary and adrenal glands. These hormones are common to fish, reptiles, birds and mammals including humans. They are essential for the regulation of many body functions and, in the early life stages, they regulate the development of many tissues and organs. It has been hypothesised that endocrine-disrupting chemicals can interfere with the levels of circulating hormones in the body and in so doing, they may elicit a wide range of adverse effects (Colborn *et al.* 1993, Colborn 1996, EPA 1997).

Other mechanisms by which POPs could cause effects on health include those mediated via enzyme systems. Cytochrome p450 enzymes located in the liver are involved in both the regulation of steroid hormones and in the detoxification of chemicals. This enzyme system is common to all animals with backbones. Some POPs, notably persistent organochlorines, are known to stimulate the production of certain cytochrome p450 enzymes. Various biological effects, including reproductive and immunological effects, may result from induction of these enzymes by POPs (Reijnders 1994, Tanabe *et al.* 1994).

Dioxins and some PCBs can bind to a site in cells known as the Ah receptor. Binding can trigger a whole cascade of different biological effects (see US EPA 1994).

The creation of such a list is only the start of the process – it is clearly not a solution in itself. The diversity of uses of these chemicals, and their releases to the environment, serve to illustrate the scale of the problem to be addressed. We are all exposed to hazardous chemicals, including POPs, during daily life, both because of their pervasive distribution throughout the environment, including the food we eat, and because of their use in consumer products. We know that these chemicals can be harmful. We know something of the threats to human and wildlife health of long-term exposure to complex mixtures of these chemicals (Box 1.3). What we already know is cause enough for grave concern, but this might only be "the tip of the iceberg".

This report collates existing information regarding the long-range transport and distribution of POPs in the environment, with a particular focus on Europe, as an important source of many of the chemicals of concern, and on the Arctic, a remote environment which bears a particularly high burden of contamination as a result. It goes on to discuss the state of knowledge on the biological effects of such contamination in both wildlife and humans. Finally it outlines the action which must be taken to address the problem of chemical contamination at source, highlighting existing commitments to phase out POPs, and hazardous substances more generally, and presenting the fundamental elements of the way forward to a more sustainable, toxics-free future.

2

GLOBAL POLLUTION AND TRANSPORT OF POPs

The Arctic is generally considered to be one of the last pristine regions on Earth. It is populated by few people, has little commercial fishing and industrial activity and there are few local inputs of POPs to the environment (Bard 1999, Fellin *et al.* 1996). However, over the past 20 years or so, considerable levels of POPs, in particular organochlorines, have been detected in the Arctic environment. These chemical pollutants are present throughout the whole of the Arctic environment - in air, water, soil, plants, terrestrial and aquatic wildlife and humans (Iwata 1993). For other POPs less information is available. One study has reported the presence of brominated flame retardants (PBDEs) in seal tissue, but there are no reports on organotins, chlorinated paraffins, phthalates or nitro-musks.

In general, levels of POPs are lower in the Arctic than in other regions of the Earth. An exception is Antarctica, where POPs are present at lower concentrations than in the Arctic (Wania and Mackay 1993). Nevertheless, POPs have reached appreciable levels in the Arctic. For some POPs, such as HCH, levels are even higher in the Arctic than elsewhere (Muir 1997). Concentrations of POPs in the tissues of some wildlife species, such as polar bear and Arctic fox, are also notably high in the Arctic.

Cold temperatures in the Arctic slow down the rate at which POPs are naturally degraded in the environment. Pollutant removal processes are inefficient in the Arctic due in part to low temperature and low solar input (see Bard 1999). Hence POPs are even more persistent in the Arctic environment than in warmer climates. It has been noted that even if all inputs of POPs to the Arctic ceased, existing concentrations of these chemicals would only decline very slowly (Harner 1997).

Studies have shown that the fatty tissues of marine mammals contain many organochlorine POPs. These animals have large amounts of fat as an adaptation to the cold (AMAP 1998 p 196). Many organochlorines are lipophilic (fat-soluble) and the most strongly lipophilic ones, such as DDT, PCBs, chlordanes and toxaphene, strongly partition into the fatty tissues of animals. As a result of the high fat content of Arctic marine mammals, they have high burdens of organochlorine contaminants. In Arctic food webs, large amounts of fat are transferred as one animal consumes another and consequently, levels of organochlorines can increase (biomagnify) through the food chain. In general, the longest-lived animals at the top of food chains, such as beluga, polar bear and humans accumulate the highest levels (Harner 1997).

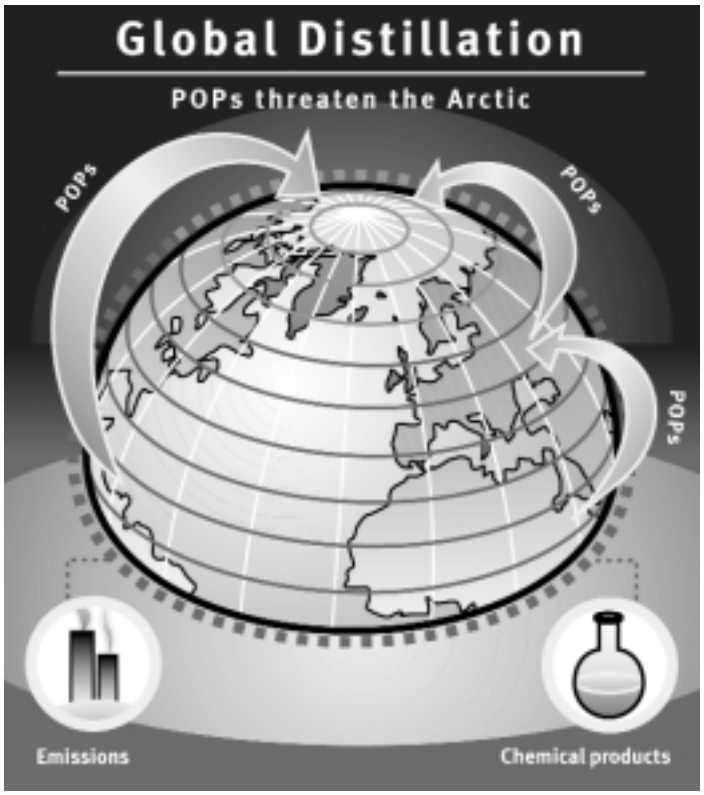
2.1 Transport of POPs on a Global Scale

The extent of organochlorine pollution throughout the Arctic environment implicates the source of these chemicals to originate in other countries. Local sources in the vicinity of the Arctic are far too few in number to cause current contamination levels. It is assumed that long-range transport on air currents is the major route by which POPs have reached the terrestrial and marine Arctic environment (Wania and Mackay 1996). In addition to airborne contaminants, some POPs enter the Arctic environment through transport on ocean currents and in North-flowing rivers from Eurasia and North America (see AMAP 1998 p191).

Due to geographical and meteorological phenomenon, the Arctic is a global sink for volatile and semi-volatile contaminants transported from distal sources, particularly from the manufacture, use and disposal of industrial and agricultural chemicals from lower latitudes (Bard 1999). The Arctic Ocean is considered to be a sink for POPs. Marine mammals receive high concentrations of these chemicals from worldwide contamination (Tanabe *et al.* 1994).

Scientists have proposed a hypothesis that explains how POPs could be transported for long distances on air currents from warmer regions of the globe towards cooler polar regions. The hypothesis is known as global distillation (or global fractionation) because once released to the environment, chemicals appear to become fractionated with latitude according to their volatility as they condense at different temperatures (Wania and Mackay 1993, Wania and Mackay 1996).

POPs are released into the environment, for example, from incinerator stacks to air, as industrial discharges to rivers, as pesticides sprayed onto crops and soil and as losses from a variety of consumer products. Subsequent movement of POPs between air, water, soil or vegetation depends on temperature, and on the physical and chemical properties of POPs. The global distillation hypothesis assumes that warmer temperatures favour evaporation of POPs from the Earth's surface to air, whereas cooler temperatures favour their deposition from air back onto soil, vegetation or water. The overall effect is that POPs volatilise to air in warmer climates and then condense and are deposited again on the Earth's surface in cooler climates.



This cycle of evaporation to air, movement on air currents and deposition to the surface can repeat itself for an individual chemical molecule so that POPs may migrate towards colder polar regions in a series of short hops. This process is known as the "grasshopper" effect (Harner 1997, Wania and Mackay 1996). The compounds migrate, rest, and migrate again in time with seasonal temperature changes at mid-latitudes (Wania and Mackay 1996).

It appears that the more volatile a chemical, the greater tendency it has to remain airborne and the faster and farther it travels on air currents towards remote polar regions. Conversely, chemicals of low volatility are unable to attain high atmospheric levels and are thus deposited close to where they are initially released. Therefore, POPs of higher volatility like α- and γ-HCH migrate faster towards the poles than those of lower volatility like DDT which tend to remain closer to their source (Wania and Mackay 1993, Wania and Mackay 1996). There is evidence that this phenomenon does happen (see Muir *et al.* 1995, Ockenden *et al.* 1998, Wania and Mackay 1996,). A striking example is that of α-HCH levels in seawater. Levels were measured at various intervals between the Western Pacific and the Canadian Arctic. It was found that the level at the equator was very low, but levels gradually increased along more northerly latitudes and reached the highest levels in the Arctic (Wania and Mackay 1996).

Observations suggest that certain POPs such as HCBs and HCHs, preferentially deposit in polar latitudes, while DDT and others primarily deposit at lower latitudes (Wania and Mackay 1996). The more volatile POPs are consequently more abundant in Arctic air than less volatile POPs. An example of the variation of POPs levels and their relative abundances in Arctic air is illustrated by the following measurements that were taken: HCHs (385-577 pg/m^3) > toxaphene (44-36 pg/m3) > PCB (14–20 pg/m^3) > chlordanes (4–6 pg/m3) > DDT (< 1–2 pg/m^3), (see Bard 1999). The same trend of contaminant abundance is present in Arctic seawater.

The transport of POPs to the Arctic in the model proposed by Wania and Mackay (1996) predicts that chemicals migrate northwards in a series of short hops. However, a recent study has suggested that chemicals may be transported long distances northwards by just one single movement rather than several movements. The study found that there was a large year to year variation in concentrations of organochlorines in fish from remote Arctic lakes. The annual variations were similar both in fish from Arctic lakes in the far north of Sweden and in lakes located in the south of the country. These

results were interpreted by the authors as an indication that transport of POPs most likely occurs by an initial one-step process from agricultural and urbanised areas to Arctic regions. The resulting concentrations of chemicals in northern regions is thus controlled by the prevailing winds and the chemical fallout of each specific year (Bignert *et al.* 1998). However, the study also indicates that much is unknown about transport and fate of POPs.

Another way in which persistent organochlorines are deposited in the Arctic is via pollution aerosols that come from Eastern Europe and Asia. Each winter and spring in the Canadian Arctic there is an influx of pollution aerosols that appears as a thick haze. Sulphate is the main constituent, 75% of which is estimated to come from man's activities, in particular from combustion of fossil fuels. These combustion aerosols are rich in organic compounds and particulates and have a high capacity for lipophilic compounds such as certain persistent organochlorines. The aerosols therefore act as a transport medium for POPs. Organochlorines that are in the gas phase of the atmosphere partition onto particles in the aerosol haze and are deposited on the ground as the particles settle (Harner 1997). Studies on atmospheric transport rates have demonstrated that polluted air masses can reach the Arctic fairly rapidly from within 48-72 hours to 7-10 days (see Bard 1999).

Most of the global inventory of POPs will not eventually reach polar regions but will be retained and/or undergo degradation close to their source or en route to polar regions. Nevertheless, levels in polar regions can still be very high. The polar region is a small area when compared to tropical and temperate zones and consequently high concentrations can occur if even a minor share of the global inventory migrates to the polar regions (Wania and Mackay 1996). Maximum levels of some POPs in the Arctic environment may only be realised in years to come (Harner 1997). Furthermore, even if all contamination ceased today, many pollutants such as organochlorines are resistant to degradation, will be transported to the Arctic via global distillation and will persist in the northern environment for decades to come (Bard 1999).

2.2 Sources of POPs to the Arctic Environment

Studies on lake sediments in subarctic Finland show that detectable levels of PCBs and dioxins are present in surface sediments, representative of current day levels, but they decrease to below detection limits in deeper sediments representative of the 1940s or 1920s (Vartiainen *et al.* 1997). Therefore, these chemicals do not appear to be present in the Arctic before the chemical industry began to boom in the 1950s.

Studies show that sources of POPs pollution in the Arctic are most likely to come from mid-latitudes of the Northern Hemisphere such as Europe, Russia and North America (Barrie *et al.* 1989, Muir *et al.* 1997). For example, lindane and chlordane found in the Arctic environment have been correlated with long-range transport episodes from use areas in the mid-latitudes of Europe and North America (Muir *et al.* 1997). In addition, studies have shown that POPs discharged in the tropics are redistributed on a global scale and so may eventually reach Arctic regions (Iwata *et al.* 1994). Monitoring sites in the Arctic have even detected specific discharges of POPs that have taken place in lower latitudes. For example, a sudden use of DDT in former East Germany in the summer of 1983/4 was detected as a pulse by a Swedish monitoring program in the Arctic (Bignert *et al.* 1990).

Within the Arctic environment itself, there are only a few known sources of POPs. These include dioxins and furans from smelters in Norway and PCBs from decommissioned military Distant Early Warning (DEW) sites in Canada (AMAP 1997, pvii). A recent study provided evidence for short-range redistribution of PCBs from the DEW sites up to distances of 20km (Bright *et al.* 1995).

Many of the organochlorine chemicals found to date in the Arctic have been banned or severely restricted in industrialised countries. However, this has not resulted in the substantial or rapid decrease in their environmental levels (Wania and Mackay 1993). For example, initial falls in PCB levels following cessation of manufacture throughout the 1970s and 80s have not been maintained and

levels have generally only stabilised. PCBs are still reaching the environment through evaporation and leaching from landfills, incineration of hazardous waste and remobilisation from existing contaminated areas. Most of the PCBs that have been discharged into the environment are retained in coastal sediments and open ocean waters. It was estimated in 1988 that around 31% of the total tonnage of PCBs produced have entered the environment (Tanabe 1988).The Swedish EPA (1999) give a similar estimate of the proportion already released to the environment. In both cases, however, the uncertainties surrounding such estimates are likely to be high. A further 4% have been accounted for by degradation and incineration, leaving a remainder of 65% still in dumps, landfills and current electrical applications. The OECD predicted in 1987 that in the next 10-15 years many of these electrical capacitors and transformers will come to the end of their useful lives (OECD 1987). Without a rigorous plan to address the problem, future losses could be double the estimated releases to date (Johnston *et al.* 1998a).

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Spitsbergen

Despite controls on some POPs in many developed countries, such as several organochlorine pesticides, these POPs are still being produced and used in developing countries. Although atmospheric and seawater concentrations of organochlorines are higher in the Northern than the Southern Hemisphere, the distribution pattern now suggests a shift or expansion of major sources towards the south (Iwata *et al.* 1993). Many developing countries are importing industrial processes that make use of toxic chemicals. Furthermore, some pesticides are being increasingly used in agriculture and in public health programs to control pests and vector-borne diseases (Forget 1991). This premise is illustrated by the results of an extensive global sampling program. It demonstrated that high residue levels of HCH and DDT were present in the atmosphere and waters of tropical Asia (Iwata *et al.* 1993, Tanabe *et al.* 1994). This study also suggested that chlordane and PCB emissions are increasing in tropical countries. Considering pollution in the Arctic, it has been noted, probably correctly, that organochlorine pollution in remote areas can only be controlled by a complete ban on their use throughout the world (Falandysz *et al.* 1994). Otherwise, transport from areas of use to remote regions appears to be inevitable (Johnston *et al.* 1996a). In conclusion to scientific assessment of the Arctic in 1997, AMAP noted that:

"The long-term reduction of exposure to persistent organic pollutants can only be accomplished through international conventions on bans and restrictions in production and use of these substances".

2.3 Levels of POPs in the Arctic Environment

Levels of organochlorines in Arctic air are comparable to those in more populated and industrialised regions of Europe and North America (Fellin *et al.* 1996). Monitoring at sites in the Canadian, Russian and Norwegian Arctic show that the most frequently detected organochlorines in air are α- and γ-HCH, toxaphene, chlordane-related compounds, PCBs and chlorobenzenes (AMAP 1998 p217). A study in 1995 in the Russian Arctic showed alarmingly that PCB levels in precipitation were about 10 times higher than levels recorded at the Great Lakes of Canada, an area that is known to be heavily contaminated with PCBs (Muir *et al* 1997).

In the Arctic Ocean, levels of POPs are comparable to uncontaminated ocean waters of the mid-latitudes (Muir *et al.* 1992). An exception is HCH for which the highest levels in the world are found in the Arctic Ocean (Muir *et al.* 1997). Present studies

show that the predominant POPs in ocean waters are α- and γ-HCH, HCB and toxaphene (Muir *et al.* 1992). Following these are chlordanes, PCBs and then DDT (Muir *et al.* 1997).

In the Arctic environment, ecologically important sites include melt holes and polynyas (large permanent open areas of water in pack ice). These areas are important winter refuges for marine mammals and spring staging grounds for migratory birds. Research has shown that these areas may be more highly contaminated than other areas in the Arctic, and this is therefore of concern (see Bard 1999).

2.4 Time Trends of POPs in the Arctic Environment

Information on time trends of POPs in the Arctic environment is very limited. A study in northern Sweden has provided the most detailed information on trends since samples have been taken on a yearly basis. This study showed that levels of POPs in the environment and in animals can vary greatly from year to year (Bignert *et al*. 1998). Sampling in other areas of the Arctic has been less frequently performed so in many cases it is not possible to draw conclusions on whether levels of POPs have declined or not. Reviews of the literature have noted that overall, it seems that there were sharp declines in levels of PCBs and DDT in the 1970s and 80s and this corresponds to the declining use of these chemicals in industrialised countries at that time. It appears that for some chemicals, declines are continuing into the 1990s whereas for others such trends are not apparent (see AMAP 1997 p90, AMAP 1998 p279).

Studies show that levels of HCHs in Arctic air have declined in recent years and levels of PCBs may have declined in the subarctic regions. For instance, levels of total HCH in air decreased 9-fold from 1979 to 1993 in the Canadian Arctic and Bering and Chukchi Seas. Levels also decreased 2-fold in the European Arctic, but lindane (γ-HCH) levels here increased between 1984 to 1992 (AMAP 1997). Levels of HCHs in seawater have however not declined in recent years.

Snow is an effective scavenger of atmospheric contaminants and is representative of deposition of contaminants from air. Studies on the Agassiz ice cap in Canada show no change in the levels of PCBs over the past 30 years (Gregor *et al.*1995). However, studies on mosses in Norway indicate there has been an atmospheric decline of PCBs in this subarctic region (see AMAP 1998). Research on Russian polar seas showed decreases for some PCBs but no change in levels of DDT and chlordanes. There was also an increase in the levels of HCHs (Chernyak *et al*. 1995).

Studies on Arctic wildlife show that levels of organochlorines have declined in some animals but not in others. Research on peregrine falcons showed decreases in DDT levels in North America and in DDT and PCBs in Eurasia, but not in northern Canada. However, no reduction in the levels of other organochlorines was evident (see AMAP 1998). A Swedish study, which represents the subarctic region rather than the high Arctic, provides strong evidence for a decline in POPs levels in freshwater fish from lakes and marine fish from the Baltic Sea. Levels of DDT, PCBs, dioxin-like substances, HCH, lindane and HCB all declined between 1967 and 1995. However, the authors commented that despite apparent declines over the past 30 years levels are still unacceptably high. This is especially true for PCBs, for which decreases are slow, indicative of ongoing releases of PCBs to the environment (Bignert *et al*. 1998). Other studies that have monitored marine wildlife have shown declines in HCH, HCB, DDE and PCBs in several seabird species that inhabit the European and North American Arctic (see AMAP 1998).

For marine mammals and polar bears, data on trends are very limited. Seals from eastern Canada and Greenland showed no declines in levels of POPs over the past 10-12 years (see AMAP 1998). A study on ringed seals in the Northwest Territories, Canada, conducted between 1972 and 1991 showed that concentrations of organochlorines are declining in the seal population but only slowly (Addison and Smith 1998). DDT did not fall for the first ten years but declines of 20% for DDT and 50% for DDE were recorded between 1981 and 1991.

PCBs fell in the first ten years but have remained stable since. HCB levels declined by 40% between 1981 and 1991, but little change was seen in HCHs, chlordanes, dieldrin, heptachor epoxide and mirex.

In polar bears, levels of PCBs, chlorobenzenes, HCHs and dieldrin appeared to increase two-fold in animals from Hudson Bay and Baffin Island between 1969 and 1983/4 (Norstrom and Muir 1994). Levels of chlordane were 4 times higher and only DDT appeared to have decreased by about 20%. The methods used to derive these data could not take the age and sex of the bears into account and this may influence the results. Nevertheless, analysis of data which did take these factors into consideration found similar trends. PCBs and HCHs increased by 25% between 1969 and 1983/4, and DDT decreased by 30% (see Norstrom and Muir 1994).

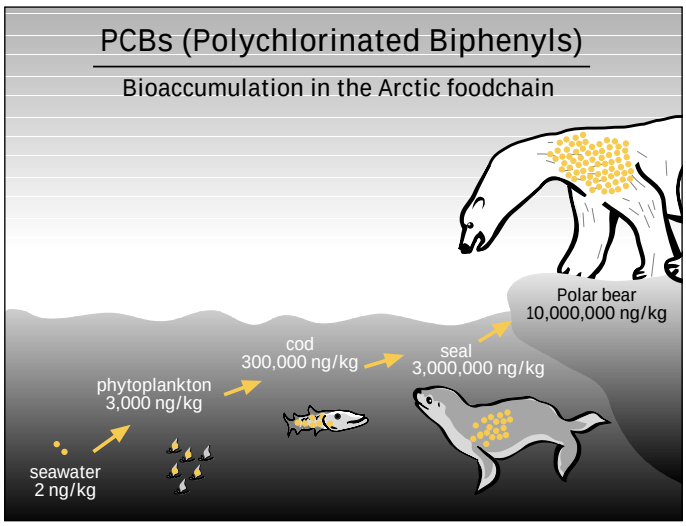
Apparent downward trends of POPs in some regions of the Arctic and in some species are encouraging. It appears that levels have decreased in some animals over the past 20 years but, due to the relatively small number of samples taken and differences in the quantification techniques used, conclusions can only be drawn with caution (Fromberg *et al*. 1999). Furthermore, uncertainties and variability in the data make it difficult to speculate about future trends. As already noted in this report, it is also possible that levels of some POPs may increase in Arctic regions in the future due to their long distance transport (Harner 1997). Moreover, observed levels of POPs in animals and humans living in the Arctic are still close to, or above the thresholds for which biological effects may be predicted (AMAP 1997).

3

LEVELS OF POPs IN MARINE MAMMALS OF EUROPE AND THE ARCTIC

POPs are known to accumulate in the fatty tissues of fish, amphibians, reptiles, birds, and mammals. In some cases, these chemicals build up through food chains, reaching the highest levels in predatory species such as marine mammals.

Levels of contaminants in marine mammals have been reported for decades throughout the world's ecosystems. The most frequently measured POPs in marine mammals are organochlorines such as DDT and PCBs (eg. Norstrom and Muir 1994). Other POPs, such as organotins and polybrominated flame retardants have been less frequently monitored in European marine mammals. In the Arctic, data on polybrominated flame retardants are limited to just one study and no data is available on organotins.



3.1 POPs in the Food Web of European Marine Mammals

Many POPs are lipophilic (fat soluble) and they tend to build up in the fatty tissues of animals, a process known as bioaccumulation. As one animal consumes another in the food chain the levels may become even higher, a process known as biomagnification. Consequently, predatory animals at the top of food chains tend to accumulate the highest levels of POPs.

Levels of organochlorines in marine mammals generally reflect the level in the food chain (the trophic level) in which they feed. Baleen whales (mysticeti) feed off microscopic plants (plankton) and animals (zooplankton) that constitute the lower trophic levels of the food chain. Consequently, baleen whales generally have lower contaminant levels than toothed whales (odontoceti) which feed off fish or larger prey from the middle trophic levels. The highest levels of contaminants occur in dolphins and porpoises. This is because they feed at a high trophic level, have a small body size and a high metabolic rate, and these factors lead to a high accumulation of contaminants (Aguilar and Borrell 1995).

3.2 POPs in the Food Web of Arctic Marine Mammals

In Arctic air and seawater, HCHs are higher than most other POPs. By comparison PCBs and DDT are low. However, the latter compounds are more lipophilic and bioaccumulate to higher levels than HCH in marine animals. They are also of greater toxicological significance (Norstrom and Muir 1994). Extremely low levels of contaminants in seawater can be biomagnified to high levels in marine food chains. For instance, animals at high trophic levels in the food chain such as seals, beluga, seabirds and polar bears have concentrations of toxaphene and HCH in their tissues around 10 million times higher than levels in the surrounding water. For PCBs, the amplification is of the order of 1000 million times (Muir *et al.* 1992).

Arctic marine food chains are generally simple. For example, phytoplankton-zooplankton-fish-seal-polar bear, or phytoplankton-zooplankton-whale (Harner 1997). POPs have been found to be present in microscopic plants (phytoplankton) and microscopic animals (zooplankton), which constitute the lower trophic levels of Arctic food chains (Hargrave *et al.* 1992). Baleen whales such as the Bowhead whale feed on small invertebrates from the lower trophic level. These whales tend to have lower exposure to organochlorines and therefore lower tissue levels than species such as seals, narwhal and beluga whales which feed on fish, (the middle trophic level). Similarly, species such as polar bears, which consume seals that are dependent on fish, tend to have the highest exposures to organochlorine contaminants (O'Hara *et al.* 1998).

3.3 Levels of POPs in European Marine Mammals

3.3.1 Organochlorines

Results of a study which determined PCB and DDT levels in a range of species from the northeastern North Atlantic (Faroe Islands and Iceland) are given in table 3.1. These figures illustrate examples of levels of organochlorines which occur in marine mammals from the northern North Atlantic and also show how levels vary according to an animal's position in the food chain. For instance, the lowest levels were found in Fin whale (Balaenoptera physalus) and Sei whale (Balaenoptera borealis) which are baleen whales. The fact that these whales are massive and have a very low metabolic rate also contributes towards their relatively low levels. Sperm whales (Physeter macrocephalus) are also large, but they are toothed whales that feed mainly on squid. These whales are situated at an intermediate stage of the food web and therefore have higher levels of organochlorines than baleen whales. Higher levels still are found in long-finned pilot whales (Globicephala melas). These whales have a similar diet to sperm whales, but they are much smaller and consequently carry higher loads of organochlorines. Finally, levels of organochlorines would be expected to be the highest of all in harbour porpoises (Phocoena phocoena) and in white-sided dolphins (Lagenorhynchus acutus). In this study however, levels were similar to, or lower than, those detected in long-finned pilot whales. It is possible that this occurred because pilot whales have vast home ranges and may have fed in more polluted waters than the dolphins and porpoises which are perhaps restricted to less polluted coastal waters of the Faroes (Borrell 1993).

Data from other studies shows that organochlorine levels are generally the highest in harbour porpoises and certain species of dolphin due to their size, and because the coastal waters they inhabit are often the most highly polluted. For example, a study on several species collected from around the British Isles in the North East Atlantic confirmed that levels were highest in small coastal species including harbour porpoises and bottlenose dolphins. Levels were somewhat lower in deeper sea species such as pilot whales, striped and white-beaked dolphins, Cuvier's beaked whale (Ziphius carirostris) and Northern bottlenose whale (Hyperoodon ampullatus), (Simmonds *et al.* 1998). Another study also showed that levels of PCBs, HCHs, dieldrin and HCB were higher in harbour porpoises from the Dutch coast than in other whales and dolphins from the open North Sea and the western Atlantic (Duinker *et al.* 1989).

Studies on organochlorine levels in baleen whales consistently show that these whales have lower contaminant loads than toothed whales. For instance the mean level of ΣDDT in blubber of 22 male Fin whales from the Spanish coast was 1.98 ppm (Aguilar and Borrell 1988). In the endangered North Atlantic right whale (Eubalaena glacialis), the mean concentration of ΣDDT in six males from the western North Atlantic was 0.21 ppm, although the sampling technique used in this study may have underestimated levels (Woodly *et al.* 1991). Higher organochlorine levels were found in toothed whales. For instance, the mean level of ΣDDT in blubber of eight male Sperm whales from the Spanish coast was 5.10 ppm (Aguilar 1983). In a female Killer whale (Orcinus orca) stranded on the UK coast the mean ΣDDT level was 6.15 ppm (Law *et al.* 1997).

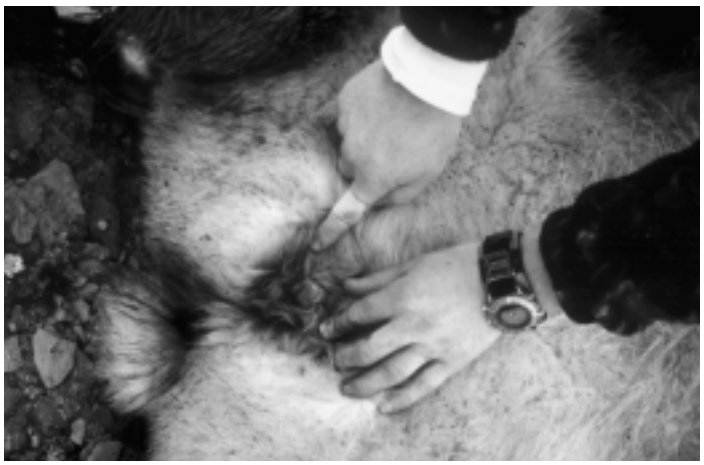
3.3.2 Effects of Age and Sex on Levels

Studies show that organochlorine levels generally increase with age in marine mammals until sexual maturity is reached. Thereafter, concentrations increase with age and body size in male animals. In females however, levels decrease. The decline occurs as a result of transfer of organochlorines from females to their young during pregnancy and lactation (Cockcroft *et al.* 1989, Aguilar and Borrell 1988).

3.3.3 Brominated POPs

Recent studies have started to detect other POPs in marine mammals. For instance, chemicals that are

used as flame retardants in various products, including PBBs and PBDEs, have been detected in a whitebeaked dolphin and harbour seals that were feeding in the North Sea and the Wadden Sea (de Boer *et al.* 1998). Levels of PBDEs were relatively high in the dolphin (>7 ppm) and seals (>1 ppm). This reflects the ongoing production of these chemicals and indicates that PBDEs may create an envi-



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Polar bear with abnormally formed genitalia

ronmental problem similar to that caused by PCBs. Both PBBs and PBDEs are listed as compounds that can affect the regulation of thyroid and steroid hormones.

PBBs and PBDEs were also detected in Sperm whales and a Minke whale which were stranded along the Dutch coast (de Boer *et al.* 1998). The presence of these compounds in Sperm whales indicates that they have reached deep ocean waters. PBDEs have been found in herring, harbour seals, grey seals and ringed seals along the Swedish coastline of the Baltic Sea, and in salmon from the Umea river close to Bothnian Bay (Haglund *et al.* 1997, Sellstrom *et al.* 1993). They were also detected in rabbit, moose, reindeer and Arctic charr in various regions of Sweden. Concentrations found in guillemot eggs (0.13-1.5 ppm TeBDE) and in Osprey (1.8 ppm TeBDE) were particularly high (Sellstrom *et al.* 1993). The studies suggested that levels increase (biomagnify) through the food chain since lower levels were found in fish than in seals (Haglund *et al.* 1997, Sellstrom *et al.* 1997).

A study on pilot whale collected from the Faroe Islands in 1994 and 1996, shows comparably high levels of PBDEs in these animals. Concentrations were 3160 ng/g lipid (3.16 ppm) in young males, 3.038 ppm in young females, 1.61 ppm in adult males and 0.843 ppm and 1.048 ppm in adult females. The higher concentrations in young animals can be explained by a considerable lactational transfer of these compounds from the females to the offspring (Lindstrom *et al.* 1999).

3.3.4 Organotins

Tributyltin (TBT) compounds have been used extensively as the active component of antifouling paints for ships and marine structures since the 1960s. Their use has been restricted in many countries on small craft because of their adverse effects on some shell-fish. TBT is widespread in dog whelks (Buccinum undatum) in the North Sea and is present in populations of dogwhelks along the Norwegian coast (Folsvik *et al.* 1999, Ten Hallers-Tjabbes *et al.* 1994). It is known to have endocrine-disrupting properties and evidence suggests it is likely to be responsible for imposex, the development of male sexual characteristics in female dogwhelks. TBT has been implicated in the local extinction of dog whelks in the Dutch Wadden Sea (Cadee *et al.* 1995). Nevertheless, TBT use on larger vessels continues worldwide to date.

Research on the presence of butyltin compounds in Grey seals and Harbour porpoises from coastal waters of England and Wales has found low levels in these animals (Law *et al.* 1998). In addition to TBT, the study detected other butyltin compounds that are used industrially, namely dibutyltin (DBT) and monobutyltin (MBT). These compounds also result from the breakdown of TBT in the environment. The total level of TBT, DBT and MBT detected in the liver of seals ranged from 12 to 22 µg/kg (ppb) and from 22-640 ppb in harbour porpoises. This confirms the presence of organotins in UK coastal waters. An extension of this work has investigated butyltin levels in marine mammals that feed in deep sea waters (Law *et al.* 1999). Levels of butyltins were determined in a range of whales and dolphin species that were found stranded around

Table 3.1 Concentration of organochlorine compounds in the blubber of different cetacean species (males) from the Northern North Atlantic. Pollutant concentrations are calculated on mg/kg (ppm) based on the lipid content of the sample

Species	Area	Number of animals	Years	p,p'-DDT	PCBs
Fin whale	Iceland	48	1982,85,86	0.85 (+- 0.46)	1.26 (+-0.61)
Sei whale	Iceland	14	1982,85	0.40 (+- 0.29)	0.46 (+- 0.26)
Sperm whale	Iceland	10	1982	7.80 (+-1.51)	10.51 (+-2.07)
Long-finned pilot whale	Faroe Islands	52	1987	31.39 (+-19.23)	48.81 (+-23.13)
White-sided dolphin	Faroe Islands	8	1987	22.46 (+-10.33)	42.68 (+-18.02)
Harbour porpoise	Faroe Islands	3	1987,88	5.57 (+-0.73)	13.39 (+- 2.38)

Source: Borrell (1993)

the coasts of England and Wales between 1992 and 1998. The total butyltin levels in liver ranged from 19 to 312 ppb. These data indicate the widespread distribution of butyltin residues in deep offshore waters and the oceanic food chains of both plankton-eating baleen whales (mysticetes) and fish –eating toothed whales (odontocetes). The impact of this contamination is difficult to assess. Laboratory studies have shown they are toxic to the immune system and it is possible that they may act additively with other contaminants such as organochlorines (Law *et al.* 1999).

A recent study detected butyltin compounds in tissue samples from eight marine mammal species in open oceans and Asian waters (Iwata *et al.* 1997). It suggested that contamination by butyltins was spread on a global scale. The study found that butyltins appeared to bioaccumulate in marine mammals. Levels of butyltins in marine mammals implied that these animals have less capacity to break down these chemicals in the body compared to other animals and may therefore accumulate higher levels.

3.4 Levels of POPs in Arctic Wildlife

There are four major classes of organochlorines which are commonly found in Arctic marine mammals. These are DDT, PCBs, chlordanes and toxaphene. Research shows that out of all Arctic marine mammals, beluga and narwhal most often have the highest levels of these POPs in their tissues. In addition, polar bears have exceptionally high levels. In the North American Arctic, polar bears were found to have slightly higher levels of PCBs and chlordanes than whale and dolphin species, and in Spitzbergen their PCB levels were approximately five times greater (Norstrom and Muir 1994).

An insight into tissue levels and biomagnification in marine food chains is given in a recent study on wildlife of Greenland and the Faroes (Fromberg *et al.* 1999). The study measured the sum of individual PCB congeners (chemicals). The highest levels were reported for porpoise blubber (700 - 4500 µg/kg wet weight (ppb)), which occupies the top level of a food chain. Seal blubber contained less (170-750 µg/kg ww (ppb)) and fish liver, representing a lower level in the food chain, had a lower level of these chemicals (40-75 µg/kg ww (ppb)).

The most frequently studied organochlorine POPs in Arctic marine mammals are the PCBs and DDT-related compounds. HCHs, HCB, chlordanes, toxaphene, dioxins and furans, dieldrin and endrin are less frequently measured (see AMAP 1998, p265). In addition, there are many other organohalogens that are known or suspected to be toxic to wildlife for which there is little or no data for Arctic wildlife. Such chemicals include about 22 chemicals that may exert similar effects to dioxin, including the brominated flame retardants, poly-brominated diphenyl ethers (PBDEs) (Giesy *et al.* 1994). A Swedish study found PBDEs were present in ringed seal (Pusa hispida) in Svalbard (Sellstrom *et al.* 1993), albeit at fairly low concentrations (47

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ppb l.w. TeBDE) relative to PCBs. PBDEs are used as flame retardants in plastics and their presence in the Arctic confirms they have become widespread in the environment, even in remote regions.

In addition, many organohalogens which can be found in environmental samples simply cannot be reliably identified. For instance a recent study on Arctic Beluga found a large proportion of chlorinated chemicals in tissue that were not identifiable as known pollutants, but were suspected to be conjugates of known POPs such as DDT (Kiceniuk *et al.* 1997). In areas of the Baltic Sea only 10-15% of the organochlorines extracted from fish can be attributed to known polluting chemicals (Wesen *et al.* 1990).

3.4.1 Effect of Location in the Arctic on Tissue Levels

In some cases, tissue levels of organochlorines in marine mammals vary geographically within the Arctic. For example, ringed seals in the Russian Arctic have noticeably higher levels of DDT and PCBs than seals in the Canadian Arctic, suggesting the presence of significant local sources for these chemicals in Russia or nearby areas. Interestingly, levels of HCHs and chlordane-related compounds in seals were similar at both locations and elsewhere in the Arctic probably reflecting their uniform distribution through atmospheric transport to the Arctic (Nakata *et al.* 1999).

Of all the organochlorines monitored in the Arctic it has been noted that the dioxins and furans (PCDD/Fs) are the least uniform in distribution, with the highest levels reported in marine mammals from the Canadian Arctic Archipelago (Norstrom and Muir 1994). A possible explanation is that dioxins and furans are deposited in this area from the Arctic haze (Norstrom *et al.* 1990). Research on levels of dioxins and furans in seals showed that the main isomer in Canadian Arctic samples (2,3,7,8-TCDD) was different to European Arctic samples (1,2,3,7,8-PeCDD and 2,3,7,8-TCDF), (Oehme *et al.*1990). This indicated a difference in source compositions of dioxins and furans for these areas.

3.4.2 Levels of POPs in Marine Mammals in the Arctic versus Temperate Zones

There is evidence that marine mammals from Arctic waters have somewhat lower levels of organochlorines than animals from northern temperate latitudes, as may be expected from their location remote from sources. For instance a study that monitored levels of PCBs, HCHs, DDT and DDE in a range of marine mammals concluded that animals from the western North Atlantic were contaminated with about 15 times higher levels of organochlorines than animals from the Arctic Ocean (Mossner and Ballschmiter 1997). Similarly, levels of PCBs in common seals were two times higher in Iceland and 35 times higher in the North Sea and Baltic Sea

compared to ringed seals from Spitzbergen in the Arctic (Norstrom and Muir 1994). North to south differences are also seen in beluga. Isolated populations of beluga in the St. Lawrence estuary, North America, which is highly polluted with industrial chemicals, have levels of PCBs in blubber about 25 times higher than beluga in the Canadian Arctic (Muir *et al.* 1992c).

A recent study found marine mammals near to the Faroe Islands in the North Atlantic had higher levels of organochlorines than animals around Greenland (Fromberg *et al.* 1999). For example, long-finned pilot whale blubber from the Faroes had levels of PCBs and ΣDDT (PCBs: 17000-39000 μg/kg ww (ppb), and ΣDDT: 10000-24000 μg/kg ww) that were 3 to 10 times higher than previously measured levels in white whale blubber from Greenland (PCBs: 3700 μg/kg – 5400 μg/kg ww, and ΣDDT: 2700 – 4100 μg/kg ww).

Most evidence shows that north to south differences are apparent for levels of PCBs and DDT in marine mammals. However, levels of some other organochlorines seem to show less geographical variation. For example, there is less variation in levels of HCHs, toxaphene and chlordanes in ringed seal and beluga between the Arctic and temperate northern latitudes. This indicates a more uniform global distribution of these chemicals as predicted by long-range transport models such as that proposed by Wania and Mackay (1996), (see section 2.1).

4

EXPOSURE AND BIOLOGICAL EFFECTS OF POPs IN MARINE MAMMALS

The accumulation of POPs in marine mammals, in particular organochlorines has given rise to concern about the long-term worldwide impact of these chemicals on these animals (see Reijnders 1996). It is difficult to assess the impact of contaminants on wildlife populations. Nevertheless, in some instances, immunological and reproductive disorders have been linked to organochlorine levels in tissues of marine mammals. These effects have resulted in dramatic declines in some populations of marine mammals from northern temperate latitudes.

It is of concern that POPs are transferred from females to their young because the early stages of life are the most vulnerable to some of these chemicals (e.g. Vom Saal *et al.* 1992, Colborn *et al.* 1993). In egg-laying birds and fish, fat reserves are mobilised into egg production, with the subsequent passage of POPs to the egg (de Wit *et al.* 1997). In mammals, POPs are passed via the placenta and via milk to the young. Exposure to POPs during the developmental stages of life may lead to irreversible effects on health (Colborn *et al.*1993).

A number of different effects in marine mammals from northern temperate latitudes are suspected to be caused by POPs. These include weakening the immune system such that marine mammals become more susceptible to moribilliviruses. These viruses have resulted in mass mortalities of seals and dolphins. There is evidence that sperm whales and fin whales may be susceptible to moribilliviruses, although no link has been made between POPs in these whales and disease. In the Arctic, levels of POPs in marine mammals are generally lower than levels found in marine mammals from northern temperate latitudes. Nevertheless, studies indicate that some marine mammals may still be at risk from adverse biological effects on neurobehaviour and reproduction. In addition, risks of biological effects in Arctic mammals may worsen during periods of fasting or poor feeding when fat reserves decrease and the concentration of POPs in remaining fat simultaneously increases (de Wit *et al.* 1997).

4.1 Susceptibility of marine mammals to POPs

Marine mammals such as polar bears, seals and whales accumulate relatively high body burdens of POPs due to their position in the food chain, their high quantities of fatty blubber tissue and the passage of POPs from females to their young. In particular, whales and dolphins are especially susceptible to bioaccumulation of organochlorines due to three factors:

- Firstly, they have a thick layer of fat or blubber in which POPs can readily accumulate. For example more than 90% of the whole body burden of POPs in striped dolphins is found in the blubber. Once organochlorines are retained in such fat-rich tissue they are not easily eliminated.

- The second factor relates to the transfer of POPs from females to their offspring. Considerable quantities of organochlorines have been shown to be passed to the young in milk. This is due to the very high lipid content of the milk in these animals. For instance, over 60% of the PCBs in the body of a female striped dolphin were found to be transferred to her newborn by lactation (Tanabe *et al.* 1994).

- Finally, whales and dolphins are susceptible to bioaccumulation of organochlorines due to their relatively limited metabolic ability to break down the contaminants. It has been demonstrated that these animals have a lower capacity for degrading PCBs through the relevant cytochrome p450 enzyme systems than many other mammals, and that co-planar PCBs present a particular problem (Tanabe *et al.* 1988). Scientists believe that this third factor may be the most important in determining why whales and dolphins retain a wide variety of organochlorines at very high concentrations (Tanabe *et al.* 1994).

The transfer of POPs from females to their young is of great concern given the increased vulnerability during developmental stages of life. Several studies on seals have shown that organochlorine chemicals are transferred from the female to the foetus in the womb (reviewed by Hutchinson and Simmonds 1994). Research on beluga whales

showed that the foetus had about 10% higher concentrations for all organochlorines compared to the mother (Wade *et al.* 1997). In ringed seals from the Arctic, it has been estimated that a female will lose about 38% of her total body burden of DDT, 25% of PCBs and 30% of chlordane-like compounds during lactation (Nakata *et al.* 1998). For marine mammals, the transfer of organochlorines to their young suggests continuous exposure to the chemicals over many generations, even if inputs to the marine environment can be reduced in the near future (Tanabe *et al*. 1994).

Methods of assessing the impacts of POPs on marine mammals, with particular reference to the Arctic, are discussed in box 4.1.

BOX 4.1 Assessment of Biological Effects of POPs in Marine Mammals

Little is known about biological effects of organochlorine chemicals in Arctic marine mammals. It is rarely feasible to use large Arctic animals for conventional laboratory studies to identify toxic effects of chemicals. A few studies have been conducted on birds, seals and dolphins but they are costly, logistically difficult and, moreover, they are ethically questionable. In the absence of experimental data on species of interest, other approaches have been used to provide insights into the biological implications of contaminants in animals. The most commonly available data describe levels of chemical contaminants in body tissues. In cases in which a relationship between contaminant levels and a biological response has been established (usually from other species of animals in the laboratory), information on levels of contaminants in the species of interest has been used to predict effects in the field (Lockhart 1995).

A scientific program in the Arctic, known as the Arctic Monitoring and Assessment Program (AMAP), has used this approach to identify which wildlife species are most at risk from biological effects of POPs. There are however drawbacks with this approach. For instance, laboratory animals are usually exposed to a single chemical over a relatively short time period, whereas wild animals are exposed to mixtures of chemicals over a long time period. In addition, different species vary in their sensitivity to a chemical and this can make it difficult to know which tested species best represents those in the Arctic (see AMAP 1998, p290). It must be noted, therefore, that while such exercises can yield useful insights into possible cause/effect relationships, they are by no means empirical and their results must be interpreted with caution.

The AMAP report made comparisons of PCB levels in Arctic wildlife with known effect levels in other species. These included neurobehavioural effects in the offspring of monkeys and humans whose mothers had a known level of PCBs, reproductive effects in otters, survival of new-borns in mink, and immunosuppression in seals. A further assessment of possible biological effects in Arctic wildlife was made using the guidelines for protecting fish eating wildlife. Levels in a fish diet that caused reproductive effects in mink were compared to estimated levels in the diet of Arctic animals. Results of these assessments are discussed below.

A second alternative approach used in the AMAP study to identify possible biological effects involved the use of an indicator of biological response to a chemical, known as a biomarker. A widely used biomarker for organochlorine POPs is the cytochrome p450 enzyme system. It is common to mammals, birds and fish. The enzyme system plays a prominent role in breaking down toxins and regulating steroid hormones (Reijnders 1994). Steroid hormones are metabolised by enzymes from this system and this aids in the control of circulating levels of hormones in the body. Toxic persistent chemicals induce cytochrome p-450 enzymes and some are metabolised by these enzymes. Some toxic effects of organochlorines appear to be mediated through induction of this enzyme system (Tanabe *et al*. 1994). Thus, examining animals for induction of cytochrome p-450 in association with levels of contaminants in the body, has been used to provide further information on the correlation between contaminant levels and biological effects.

4.2 SEALS

4.2.1 European Populations of Seals

Biological Effects

Reproductive and immune disorders have occurred in wild populations of seals for which PCBs, DDT and their respective metabolites have been implicated. It seems that disruption of endocrine systems is common to both of these effects (Reijnders 1994), and could be the underlying mechanism of observed toxicity.

Effects on Reproduction

A population collapse of common seals (*Phoca vitulina*) in the western most part of the Wadden Sea, The Netherlands, occurred as a result of reproductive failure. This effect may be attributable to organochlorine pollution. Between 1950 and 1975 the population dropped from more than 3000 to less than 500 animals due to a decline in the numbers of seal pups produced. An experimental study was set up to investigate the problem. Seals were either fed a diet of fish from the western Wadden Sea or less contaminated fish from the northeast Atlantic Sea. The study found that seals fed on the more polluted fish from the Wadden Sea had a

BOX 4.2 Exposure OF European Seals

Levels of organochlorine chemicals in seals in the Northern Hemisphere are generally higher than in the Southern Hemisphere. For example, levels of DDE in seal milk from the Northern Hemisphere were 14 to 118 times larger than milk from the Southern Hemisphere (Bacon *et al.* 1992). Within the Northern Hemisphere, levels are particularly high in seals from the Baltic Sea reflecting proximity to industrial and agricultural sources. For instance, DDT and PCB levels here are 15-20 times higher than in Scotland, Norway and Canada (see table 4.1), (Jenssen 1996). Regarding the dioxins and furans, levels were found to be among the lowest concentrations recorded in seals for grey seals from Sable Island, Nova Scotia in the northwest Atlantic (total PCDDs mean 6.95 pg/g lipid, total PCDFs mean 2.18 pg/g lipid). These levels are similar to, or slightly lower than, levels found in ringed seal (Phoca hispida) blubber from the Canadian Arctic, from Spitzbergen, and in harp seal (Phoca groenlandica) from the Greenland Sea (Addison *et al.* 1999).

Table 4.1 Concentrations (µg/g l.w. (ppm)) of PCBs and DDT compounds in blubber of newborn (i.e. <21 days) and juvenile grey seals (Halichoerus grypus).

Location	Year	Age	ΣPCB	ΣDDT
NE Scotland	1988	1.5 - 4.5 years	4.5 (7)	1.8 (7)
W Norway	1992	< 21 days	2.0 (12)	1.2 (12)
Canada (Sable Island)	1988	< 21 days	2.4 (4)	1.4 (4)
	1989	1 year	4.1 (9)	2.5 (9)
Baltic	1984 - 1988	1 year	77.0 (5)	30.0 (5)
	1981 - 1988	1 year	82.0 (5)	48.0 (5)

Values are presented as means. (n) indicates number of individuals.

Source: Jenssen (1996)

significantly higher rate of reproduction failure. The failure resulted from problems at the implantation stage of pregnancy, but the mechanism of action was unknown. This study suggested that the population collapse of common seals in the Wadden Sea may have been caused by pollutants in the fish they ate (Reijnders 1986).

Baltic Sea seal populations have suffered with reproductive and immunological disorders in recent decades. In the 1970s there was a sharp decline in ringed seals (*Phoca hispida*) and grey seals (*Halichoerus grypus*). Only 27% of females were pregnant compared to a usual figure of 80-90% (Helle *et al.* 1976b). The seals were found to have pathological changes in their uteri that caused the poor fertility (Helle *et al.* 1976a). Several

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Seal

other symptoms were observed including changes in skull shape (Zakharov and Yablokov 1980), skull bone lesions (osteoporosis) and enlargement of the adrenal gland (adrenocortical hyperplasia). Studies revealed that all the problems were part of a disease-complex known as hyperadrenocorticism. Data suggested that organochlorines, in particular PCBs, can interfere with the hormonal system to cause these effects (see Reijnders and Brasseur 1992).

Today, grey seals of the northern Stockholm archipelago have recovered in numbers, but recovery is slower in grey and common seals of the southern Baltic. In the Gulf of Finland, uterine occlusions are still occurring, even in young seals, and adrenal damage continues to be reported. Another feature of the disease complex, namely intestinal ulcers, has become even more frequent in grey seals. These ulcers are probably caused by parasites, but they heal abnormally slowly. In many cases the ulcers have become so deep they pierce the intestinal wall which subsequently causes death (see Swedish EPA 1998).

Effects on the Immune System

In 1988, an estimated 18,000 harbour seals and several hundred grey seals died around the coasts of Denmark, Sweden, The Netherlands, Norway, United Kingdom and Ireland. The primary cause of this mass mortality, or epizootic, was a moribillivirus called phocine distemper virus, similar to that which causes distemper in dogs (see Kennedy 1996). There is some evidence that the concentration of organochlorine contaminants in the seal tissue may have influenced the ability of the animals to mount an effective immune response to this disease (Simmonds and Mayer 1997). For instance, levels of organochlorines in the blubber of the animals that died were higher than in those animals that survived (Hall *et al.* 1992). In addition, seals living in less contaminated parts of Britain had lower mortality rates than those in more polluted waters (Simmonds *et al.* 1993). Other studies of organochlorine contaminants found in seals from UK waters (Law *et al.* 1989, Mitchell and Kenndy 1992), have also lead to the conclusion that organochlorines may have exacerbated the effects of the virus that caused the epizootic, and the secondary infections which were often the final cause of death in affected animals (Heide-Jorgensen *et al.* 1992, Munro *et al.* 1992). Furthermore, research shows that exacerbation of lung infections, such as the moribillivirus in seals and in other marine mammals, may be partly caused by damage to certain cells in the lungs as a result of the build up of PCB degradation products (PCB methyl sulphones), (Troisi *et al.* 1997).

Recent experimental studies on seals have shown that pollution may have played a key role in mass mortalities such as the 1988 seal epizootic (Ross *et*

al. 1995, de Swart *et al.* 1994, 1996). The studies also show that marine mammals that inhabit more polluted coastal environments of Europe may have an increased susceptibility to infections. The studies involved feeding herring from the highly polluted Baltic Sea or from the relatively unpolluted Atlantic Ocean to two groups of harbour seals (*Phoca vitulina*) for two and a half years. Results showed that certain cells of the immune system were reduced in animals fed the more contaminated fish. This indicated impaired immunological functions from long-term exposure to pollutants in fish. In particular, the part of the immune system that deals with defence against viral infection was affected.

The experimental studies on harbour seals also showed that levels of retinol (vitamin A) and thyroid hormones in blood, were reduced in seals that were fed the more highly PCB-contaminated fish from the Baltic Sea as opposed to fish from the North Atlantic (Brouwer *et al.* 1989). Vitamin A deficiency can lead to alterations in immunity, and thyroid hormones are important in development and growth, including neurological development (Jenssen 1996). Effects of reduced levels of thyroid hormones and retinol in seals could lead to an increased susceptibility to microbial infections, reproductive disorders and other pathological alterations. It has been suggested that reduced retinol and thyroid hormones were involved in reproductive disorders and viral infections of seals and other marine mammals in the Baltic Sea, North Sea and Wadden Sea (Brouwer *et al.* 1989).

Recent studies on wild populations of Norwegian grey seal pups indicate that lower levels of retinol and thyroid hormones are correlated with higher concentrations of PCBs in seal blood. Evidence from animal studies and the experimental studies with harbour seals suggest that PCBs in the Norwegian seal pups may be causing these effects (Jenssen 1996).

A recent study on wild grey seal pups at the Isle of May in the Firth of Forth, Scotland, investigated whether exposure to PCBs in the mothers' milk was causing immunosuppression (Hall *et al.* 1997). The study found that pups receiving relatively higher doses of PCB contaminants during early life were no more likely to succumb to infection than those receiving lower doses. It is possible that grey seals are less sensitive to immunosuppression by organochlorines than harbour seals. However, it is also possible that the study was conducted over too short a time period to detect effects on the immune system (Hall *et al.* 1997).

4.2.2 Arctic Seals

Arctic seals are known to have lower PCB levels than those associated with adverse effects in seals from the Baltic Sea or experimental seals. Levels in Arctic seals are 50 times lower than levels linked with the disease complex in Baltic seals and about 10 times lower than levels associated with reduced reproductive success (Norstrom and Muir 1994). Nevertheless, it is not possible to rule out such effects as little is known about relative sensitivities between species and populations. Moreover, more subtle effects of organochlorines may occur. For instance, it has been shown that PCB and dieldrin levels in blubber of ringed seals in the Canadian Arctic were correlated with the cytochrome P-450 enzyme system and other enzymes used as biomarkers (Lockhart and Ferguson 1994).

According to the AMAP assessment of biological effects, levels of PCBs in Arctic harp, ringed, harbour and grey seals were above levels for subtle neurobehavioural effects in the offspring of monkeys, but were below levels for reproductive effects and reduced young survival in otters and mink, and immunosuppressive effects in seals. Levels of PCBs in diet also exceeded guidelines for protecting fish eating wildlife based on reproductive effects in mink. According to this assessment, seals in the Arctic may therefore be at risk from neurobehavioural and reproductive effects on health as a result of current tissue levels of PCBs (see AMAP 1998).

4.3 WHALES, DOLPHINS AND PORPOISES OF EUROPE

4.3.1 Sperm Whales (Physeter macrocephalus)

Biological Effects

In recent years there has been an unprecedented number of live strandings of sperm whales in Europe from the North Sea. However, the North Sea is probably not within the normal range of sperm whales. Their migration route from colder water feeding grounds is southwards around the western side of the British Isles and not the eastern side (Simmonds and Mayer 1997). During the past eight years, 85 stranded animals have been reported on North Sea coasts (Sonntag and Lutkebohle 1998).

BOX 4.2 Exposure OF Sperm Whales

The sperm whale is distributed worldwide and is commonly found in northeastern Atlantic waters (Aguilar 1983). Sperm whale bulls are among the largest of all whales, old bulls growing to be about 20 meters long (Sonntag and Lutkebohle 1998). Several organochlorine compounds have been detected in sperm whale tissues, including PCBs, DDT, HCHs, HCB, dieldrin, and chlordanes (Law et al. 1996). It has been noted that levels of organochlorines in sperm whales are intermediate in position in relation to other whales and dolphins (Aguilar 1983). For example, mean levels of PCBs and DDT in muscle of sperm whales (4.68 and 26.8 ppm) were found to be higher than in fin whales (Balaenoptera physalus) (0.74 and 12.65 ppm), but lower than in the common dolphin (Grampus griseus) (61.33 and 47.20 ppm) from the same region. Similarly, another study on sperm whales that were stranded around coasts of the North Sea found that levels of PCBs, HCH, HCB and dieldrin were relatively low in comparison with small cetaceans such as the harbour porpoise. However, levels of DDT were about the same in both species (Law *et al.* 1996). Organotin compounds have been detected in livers from sperm whales that were found stranded on the coasts of the Netherlands and Denmark in the North Sea in 1997 (Ariese *et al.*1998). It is unlikely that the whales fed in the North Sea and the presence of organotins in the whales is most likely due to food intake from deeper off-shore parts of the North Atlantic.

The frequency of the strandings is far greater than previously recorded by the historical record. For instance, 12 dead sperm whales were reported on the Eire and UK west coasts in 1990, whereas historical figures for the United Kingdom show only 12 whales were reported stranded in the entire period from 1913 to 1966 (Simmonds and Mayer 1997).

With present knowledge, it is not possible to understand the exact cause of the strandings and why the animals have been driven off route into the North Sea. However, studies show that the stranded whales have various lesions. The lesions in the mouth and skin may have a viral origin. The possible types of virus involved include poxviruses and moribilliviruses, which are like the viruses implicated in mass mortalities of seals and dolphins. External ear lesions were also found and if these extended to the inner ear, it is possible that hearing and balance may be affected which could affect their ability to echo-locate and communicate (Simmonds and Mayer 1997). With regard to viral infection, Simmonds (see Simmonds and Mayer 1997) has noted that "there is no reason to hope that bigger whales are exempt from the factors causing the recent mass mortalities in other marine mammals".

It has also been hypothesised that the whales may have been driven off course because of the large number of drilling rigs north and northeast of the Shetland Islands which create a great deal of sound and may confuse the animals (Sonntag and Lutkebohle 1998). Once in the North Sea, the relatively shallow waters and sandbanks towards the south could be extremely problematic for them since they are deep sea creatures.

4.3.2 Fin Whales (Baleanoptera physalus)

An immature Fin whale was found stranded along the Belgian coast in 1997. It was emaciated and had severe parasitic lesions. On studying the animal, researchers found evidence of lesions that were associated with a moribillivirus infection, the type of virus that has caused mass mortalities in seals

and dolphins (Jauniaux *et al.* 1998). This was the first such viral infection ever found in a baleen whale. It was concluded that moribillivirus infections are potential threats to baleen whales, given their frequent association with severe epizootics in marine mammals. In this study, no information was available on levels of contaminants in the whale.

4.3.3 Harbour Porpoise (Phocoena phocena)

Biological Effects

A review on harbour porpoises has indicated that levels of organochlorines, and in particular PCBs, are high enough to cause concern about the possible effects of these chemicals on the maintenance of the population (Aguilar and Borrell 1995). Indeed, a marked decline has taken place in the Baltic population of harbour porpoises (see Granby and Kinze 1991), and there are several reports suggesting a decline in the North Sea region (see Kleivane *et al.* 1995). Since these animals are vulnerable to toxicity from long-term exposure to persistent organochlorines it cannot be excluded that present levels of these contaminants have contributed to the reported decline of harbour porpoises (Kleivane *et al.* 1995).

Currently, there is little data on effects of organochlorines in harbour porpoises. One study associated enlarged adrenal glands (adrenocortical hyperplasia) with long-term exposure to toxic substances such as organochlorines in two harbour porpoises found stranded on the Belgian coast (Joiris *et al.* 1992). However, another study on 28 animals from the British coast found no association between organochlorines and this condition (Kuiken *et al.* 1993).

A study on harbour porpoises from British waters which investigated possible immune system effects of organochlorines, reported no evidence that organochlorines were associated with an increased risk of dying from infectious or parasitic diseases in these animals (Kuiken *et al.* 1994). However, a more recent study on a larger number of animals reported the converse. In this preliminary study, Jepson *et al.* (1998) suggested that exposure to PCBs had a negative effect on the health status of harbour porpoises by pre-disposing them to death due to infectious diseases. Animals that died from infectious diseases had significantly higher mean PCB levels (29.4 mg/kg (ppm), n=33) than animals that died due to physical trauma (11.3 mg/kg (ppm), n=29).

BOX 4.3 Exposure of Harbour Porpoises

The harbour porpoise is widely distributed in coastal waters of the Northern Hemisphere (see Granby and Kinze 1991). In the western North Atlantic however, it is considered as a threatened species by the Committee on the Status of Endangered Wildlife in Canada (see Wesgate and Tolley 1999). Harbour porpoises feed at high trophic levels and have a small body size. These factors contribute to the animals having high body burdens of organochlorine contaminants. As previously mentioned, concentrations of organochlorines in the blubber of harbour porpoise are many times higher than in plankton feeders such as the large baleen whales, substantially higher than in toothed whales such as sperm and pilot whales, and in the upper range of levels in other dolphins and porpoises.

Comparisons of data on organochlorine levels in harbour porpoises between different geographical localities are hindered by the fact that techniques used to analyse samples may be different. In addition, levels vary with age and sex of animals and such biological data is often not given in published studies. A review of studies shows that DDT levels in harbour porpoises are similar in the eastern and western North Atlantic, although PCB levels are somewhat higher in the eastern North Atlantic (see Aguilar and Borrell 1995). Research in Scandinavian waters has detected HCHs, DDTs, PCBs, HCB, chlordanes, dieldrin and endrin in harbour porpoises (Kleivane *et al.* 1995). Levels of PCBs and DDTs were the highest ranging from 3.7-65.3 and 3.2-45.0 µg/g (ppm) lipid weight respectively. Mean concentrations of dieldrin, endrin and trans-nonachlor were 1-3 ppm and of HCHs, heptachlor epoxide and HCB were <1 ppm. Data based on a limited number of samples suggests that levels in Scandinavian and Welsh waters are similar. However, levels of lindane were higher in southern Scandinavian waters possibly reflecting recent use of this pesticide (Kleivane *et al.* 1995). A study in the Danish North Sea and Baltic waters commented that levels of DDT recorded in 1977 from the Dutch coast were 3 times higher than from the Danish North Sea. The study also noted that levels in the late 1980s appeared somewhat lower than the early 1980s (Granby and Kinze 1991).

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Dolphins

4.3.4 Dolphins

Significant declines in the sightings of dolphins from the Cornish coast have been recorded in recent years (see Simmonds *et al.* 1998).

Mass mortalities of dolphins have occurred since the late 1980s. Huge numbers of striped dolphins died between 1990 and 1992 around the whole of the Mediterranean, possibly due to the virus dolphin morbillivirus. This virus is the same sort of virus, a morbillivirus, that caused mass mortalities of seals. The dead dolphins were found to have higher body levels of PCBs than healthy animals (Aguilar and Borrell 1994). Similarly, in the late 80s and early 90s, many bottlenose dolphins died along the eastern coast of the United States. They were also contaminated with high levels of PCBs and other organochlorines.

In both mass mortalities, it is possible that these contaminants could have contributed to suppression of the immune systems, rendering the dolphins more susceptible to infections that subsequently led to their death. This is further supported by a study on bottlenose dolphins resident in Florida. Blood samples taken from the animals found that suppression of the immune system increased with increasing levels of pollutants in the dolphins especially DDT, DDE and PCBs. More research is necessary to confirm these findings (Lahvis *et al.* 1995), but the indications are strong. Particularly high levels of organotins, TBT and DBT, were also detected in dolphins found stranded along the US Atlantic and Gulf coasts during 1989-1994, (Kannan *et al.* 1997). These chemicals are also potential immunosuppressing agents and may have contributed to dolphin mass mortality in these regions.

4.4 WHALES AND PORPOISES OF THE ARCTIC

4.4.1 Beluga (Delphinapterus leucas)

Several organochlorines including PCBs, DDT, toxaphene, HCBs, HCHs, chlordanes and mirex have been detected in beluga blubber from four different locations in the Canadian Arctic (Hudson Bay, Cumberland Sound, High Arctic (Jones Sound) and Beaufort Sea). Levels of contaminants at these sites, including the more remote High Arctic, were similar. This may indicate the diffuse atmospheric transport of these contaminants (Muir *et al.* 1990). Another study has detected PCBs, DDTs and HCHs in brain, liver and muscle in beluga whales from the Beaufort Sea (Metcalf *et al.* 1999). Organochlorine profiles in this study also indicated the source of contaminants was long-range transport to the Arctic from lower latitudes. Data on time trends showed that levels of chlordane and toxaphene were higher in the 1980s than the late 1960s whereas levels of DDT were slightly lower. These results also concur with data on polar bear tissue levels (Muir *et al.* 1990). Blubber samples taken from beluga of Alaska's north coast showed that toxaphene, PCBs, DDTs and chlordane were present with toxaphene reaching the highest levels (Wade *et al.* 1997). Once again, results suggested that the source of these organochlorines is global distillation from lower latitudes.

Most research on biological effects of POPs on belugas has been conducted on an isolated population of about 500 animals in the St. Lawrence River estuary, Quebec, Canada, a region not within the Arctic. These belugas have been chronically exposed for more than 50 years to a complex mixture of industrial pollutants from nearby sources, including many persistent organochlorines, butyltin compounds, polycyclic aromatic hydrocar-

bons and heavy metals (Martineau *et al.* 1994, Yang *et al.* 1998). Levels of organochlorines in these belugas are extremely high as may be expected from their proximity to industrial sources. PCBs, DDTs and mirex were 25, 32 and 100 times higher than in Arctic belugas (Muir *et al.* 1992c).

Post-mortem examination of dead St. Lawrence belugas has confirmed frequent infections with mildly pathogenic bacteria and gastric ulcers, which is indicative of a suppressed immune system. Occurrence of frequent infections and ulcers is consistent with toxicity caused by organohalogens, particularly organochlorines, and strongly suggests that immunotoxicity could be related to high concentrations of contaminants in their tissues (De Guise *et al.* 1995, Martineau *et al.* 1994). It is also suspected that the whales have a reduced reproductive rate since the number of pregnant females is dramatically low. In addition, one adult true hermaphrodite beluga has been found that had two ovaries, two testes and genital tracts of both sexes. It is possible that this developmental effect is related to pollutants that have estrogenic activity (De Guise *et al.* 1994). Another study on a lone neonate beluga that was found showed that organochlorines are transferred via the placenta to the developing animal in the womb. This conclusion was reached because the neonate had not been fed with milk and yet it had tissue levels of PCBs and other organochlorines which were lower than, or in the lower range of, levels published for adult female beluga from this region (Gauthier *et al.* 1998).

Research on beluga in the Canadian Arctic was undertaken after a group of animals entered a freshwater lake system, became trapped there with the onset of winter, and were subsequently taken by hunters. A correlation was found between the levels of several PCB congeners in the animals and cytochrome p-450 enzyme activities. It is thought that the beluga lost weight because they were trapped, and consequently organochlorine contaminants such as PCBs in their fat stores were released and induced the cytochrome p-450 enzyme system. It was noted that this may be the first statistical inference that current body burdens of contaminants in Arctic species are associated with subtle effects on health (Lockhart 1995). However, a later study on normal Arctic beluga found no correlation between PCBs and cytochrome p-450 activity (Addison *et al.* 1997). Nevertheless, a review by AMAP (1998, p301), notes that the study on the trapped beluga demonstrates that current body burdens of contaminants in Arctic animals can be associated with subtle effects under certain conditions. Existing body burdens of organochlorines may be sequestered effectively when blubber reserves are high, but may become a significant problem during a poor feeding season (AMAP 1998).

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Beluga

The AMAP report made a comparison of PCB levels in Arctic beluga with levels which were associated with detrimental effects in other species. Results showed that PCB levels in beluga were higher than levels that caused neurobehavioural effects in the offspring of monkeys and humans, but were lower than levels that caused reproduction and immunosuppressive problems in other species. Levels of PCBs, but not DDT or chlordanes, in diet of Arctic beluga exceeded guidelines for protecting fish eating wildlife based on reproductive effects in mink (AMAP 1998). According to this assessment, beluga are thought to be at risk from neurobehavioural and reproductive effects of PCBs.

4.4.2 Narwhal (Monodon monoceros)

Several organochlorines have been detected in narwhal that were collected during 1982-83 in the Canadian Arctic. They include toxaphene, DDT, chlordane, dieldrin, HCHs, HCBs, PCBs and mirex (Muir *et al*. 1992b).

No biological effect studies have been carried out on narwhal. Levels of PCBs in narwhal exceeded levels that caused neurobehavioural effects in the offspring of monkeys, but did not exceed levels in other species which caused reproductive and immunosuppressive effects. PCB levels in the diet exceeded guidelines for protecting fish eating wildlife based on reproductive effects in mink (AMAP 1998). Reproductive and neurobehavioural effects in Narwhal may be predicted even at current body burdens of PCBs.

4.4.3 Bowhead Whale (Balaena mysticetus)

The population of bowhead whales in the western Arctic numbers only around 8000 animals. In Alaska the Bowhead whale is a major subsistence food source and is culturally important. Unlike toothed whale species (odonoceti) which feed on prey such as fish and squid, baleen whales such as the bowhead feed on plankton and/or zooplankton. Studies on organochlorines in Arctic bowheads have shown that levels of these contaminants are much lower than in Arctic seals and toothed whales (O'Hara *et al.* 1998, see O'Shea and Brownell 1994). For example, compared to levels of ΣDDT found in male and female beluga in the Arctic (mean 2.81 ppm ww in blubber of male and female whales), (see Norstrom and Muir 1994), levels in Arctic bowheads from Alaska were over 50 times lower (<0.05 ppm ww), (see O'Hara *et al.* 1998). Therefore, even though bowheads are known to have large stores of fats, they do not accumulate organochlorines at a significant rate compared to toothed whales. The difference is most likely due to the fact that they feed at a lower trophic level of the food chain. O'Hara *et al.* (1998) conclude that no adverse effects in Arctic bowhead whales related to organochlorines are expected based on current levels in these animals and current understanding of organochlorine toxicity.

One study has reported an unusual assembly of reproductive structures in two male bowheads (Tarpley *et al.* 1995). Externally these animals appeared to be female but research revealed that the gonads were underdeveloped testes. The gonads had thus become feminised, a disorder known as male pseudohermaphroditism. Two out of 155 whales thus far examined is a very high incidence of this condition. In humans for example, the condition, caused by a genetic defect, occurs naturally as one in 62,400 males (Tarply *et al.* 1995). It is possible that natural incidence of this disorder is much greater in bowheads. However, it is also possible that such disorders result from chemical exposure. As previously mentioned in this report, endocrine disrupting chemicals, including many organochlorines, have been implicated in affecting sexual or gonadal development. In these cases of pseudohermaphroditism in whales, no cause and effect has yet been implicated, and research remains understandably limited (O'Hara *et al.* 1998). Nevertheless, the high incidence of this condition in Bowhead whales is of great concern given that their population is endangered.

4.4.4 Harbour Porpoise (Phocoena phocoena)

Levels of organochlorines in harbour porpoises are relatively high compared to other whales and dolphins possibly due in part to their inshore habit. Studies show that levels of organochlorines are lower in western Greenland than levels characteristic of the eastern North Atlantic (Aguilar and Borrell 1995). It is of concern that research demonstrates that levels may be increasing in animals from West Greenland (Granby and Kinze 1991).

The AMAP study reported levels of PCBs in harbour porpoises that exceeded levels which caused neurobehavioural, reproductive and immunosuppressive effects and reduced young survival in a range of species. Levels of PCBs in diet also exceeded guidelines for protecting fish eating wildlife based on reproductive effects in mink. It was concluded

that current levels of organochlorines in the harbour porpoise may place this species under significant threat from a range of biological effects (see AMAP 1998).

4.5 POLAR BEARS (URSUS MARITIMUS)

Polar bears are distributed throughout the Arctic. This animal is the principal mammalian predator at the top of the Arctic marine food chain, eating mostly ringed seal (Phoca hispida) and some bearded seals (Erignathus barbatus), (Stirling and Archibald 1977, Stirling *et al.* 1977). Levels of organochlorines in polar bear fat have been well documented. As a consequence of their position in the food chain, levels are relatively high (Norstrom *et al.* 1998, see Norstrom and Muir 1994, Norheim *et al.* 1992). Several organochlorine contaminants have been detected in their fat including PCBs, chlordane-related chemicals, HCB, DDT, HCHs, dieldrin (Norstrom *et al.* 1988), dioxins and furans (Norstrom *et al.* 1990), and toxaphene (Zhu and Norstrom 1993). PCBs and chlordanes were found to account for more than 80% of the total organochlorines in polar bear fat (Norstrom *et al.* 1990). In addition, PCBs, dioxins and furans were reported to be detectable in polar bear milk (Oehme *et al.* 1995). The pattern (profile) of dioxin congeners was different in polar bear milk from the Canadian Arctic compared to bear milk from Svalbard and reflected the different source areas of these chemicals.

Other contaminants found in polar bear tissue are the breakdown products of PCBs and DDE, known as methylsulphone (MeSO2) metabolites. The toxicological significance of these compounds in polar bears is presently unknown, but methylsulphone-DDE is toxic in other species. For instance, it has been shown to cause toxicity to the adrenal gland in laboratory mice. It was also found in the adrenal gland of Baltic grey seals and is suspected to be involved in a disease complex in these animals (adrenocortical hyperplasia) which was characterised by reproductive failure and jaw bone erosion (see Letcher *et al.* 1995, Norstrom and Muir 1994).

As part of the AMAP study a comparison was made of the levels of organochlorines in polar bear fat from different Arctic regions with levels in other species known to be linked with adverse biological effects. This showed that in most cases, levels in polar bears exceeded effect levels in other species (AMAP 1998, p303). Therefore, it was noted that some polar bear populations may be at risk for the immunosuppressive effects of PCBs and dioxins (PCDD/Fs), as well as the reproductive and developmental effects of PCBs. Furthermore, analysis of intakes of organochlorines from a polar bear diet also revealed that PCB intakes exceeded levels associated with reproductive effects, and DDT and PCB levels exceeded Canadian and US EPA guidelines for protecting aquatic wildlife.

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Polar bear

Polar bears have lengthy fasts on a seasonal basis during the ice-free period when the bears are forced onto land. During this time they utilise some of their fat reserves and may lose up to 90% of their adipose tissue (fat) stores. This may lead to a substantial mobilisation of organochlorine contaminants (Oehme *et al.* 1995, Letcher *et al.* 1995). For example, concentrations of PCBs, chlordanes and HCBs have been shown to increase in female polar bear fat during the fasting period (Polischuk *et al.* 1995). It is probable that the increased levels of organochlorines in polar bears during the fasting period will increase risks for adverse health effects on the animals.

There is some evidence that current levels of organochlorines in polar bears may cause subtle effects on health. A strong correlation was found between activity of the cytochrome p-450 enzyme system and concentration of PCBs. This suggests that these contaminants may be responsible for inducing the enzyme system. In addition, levels of thyroid hormones in polar bear blood were related to levels of organochlorines (reviewed by AMAP 1998, p303). Thus, these chemicals may alter the levels of thyroid hormones in polar bears.

Observations on polar bears on Svalbard have been carried out to monitor reproductive success. The pregnancy rate was similar to other polar bear populations and did not appear to be affected by PCB levels in the bears (see AMAP 1998, p303). However, in Hudson Bay, there is a slow decline in reproductive performance of female polar bears and the possibility of organochlorine-induced effects on reproduction cannot be ruled out (Derocher 1991).

A further study on Svalbard reported a very high mortality of young polar bears (Wiig 1995). Research on the levels of organochlorines in polar bears documented higher levels in the cubs than in the mothers. It also showed that, while mothers were fasting, levels of PCBs, chlordanes and HCBs in their milk increased with time. The transfer of organochlorine contaminants in milk from mother to young occurs at a crucial point in the growth and development of polar bear cubs (Polischuk *et al.* 1995). This high intake of contaminants could adversely affect the early development in polar bear cubs and lead to a higher death rate of cubs (see AMAP 1998, p303). It should be noted that there are two settlements within Svalbard which have landfill sites containing industrial waste. These sites may leak contaminants, including PCBs and HCB, and add to contamination of the local area (see AMAP 1998).

During studies on polar bears at Svalbard, four genetically female bears have been found which had aberrantly formed genitalia (Wiig *et al.* 1998). Two of the bears had both a vaginal opening and a penis. Because of their condition, all four bears were classified as female pseudohermaphrodites. One explanation for the condition could be excessive secretion of androgen (a male sex hormone) during pregnancy, originating from either an adrenal or an ovarian tumour. However, work on other species of bears suggested that a more likely explanation for high occurrence of female pseudohermaphroditism may be due to ingestion of compounds by the mother during pregnancy that have androgenic activity (see Wiig *et al.* 1998). Polar bears have high body burdens of organochlorine chemicals, such as PCBs, which are endocrine disruptors. It is therefore possible that a high PCB level in the mother could expose the developing foetus to endocrine disrupting effects and disturb organ differentiation.

5

OTHER WILDLIFE

5.1 ARCTIC WILDLIFE

Studies in the Arctic have detected POPs in the tissues of terrestrial species such as caribou, in freshwater fish and aquatic mammals including mink and otter, and in marine fish and marine mammals (Muir *et al.* 1992, see AMAP 1998 259-279). POPs are present at every trophic level of food chains from microscopic marine plants and animals to top predators like toothed whales and polar bears (Hargrave 1992).

Organochlorine POPs have been implicated in causing detrimental effects in birds of prey in Arctic regions. For example, present DDE levels in Canadian tundra peregrines, Fennoscandian merlin and white-tailed sea eagle are still causing significant egg shell thinning (de Wit *et al.* 1997).

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Arctic fox

5.1.1 Arctic Fox (Alopex lagopus)

Levels of PCBs measured in Arctic foxes in Svalbard were found to be essentially unchanged in the period from 1973-1974 to 1983-1984. The levels are very high in these animals (Wang-Andersen *et al.* 1993). Comparison with studies on effects of PCBs shows that levels in Arctic foxes are higher than levels associated with neurobehavioural effects, poor reproductive success, reduced litter size and immunosuppressive effects. Arctic foxes eat ringed seal; the AMAP assessment indicated that levels of DDT and PCBs in their diet exceeded guidelines for the protection of aquatic wildlife. It was concluded that Arctic foxes from Svalbard are therefore at risk from the immunosuppressive effects of PCBs and probably also dioxin-like substances, as well as reproductive and developmental effects of PCBs (see AMAP 1998, p304).

5.2 EUROPEAN WILDLIFE

5.2.1 Terrestrial Mammals

In Germany, PCBs have been detected in tissues of foxes (Brunn *et al.* 1988, Georgii *et al.* 1994). Levels of certain PCBs, the more highly chlorinated PCBs, decreased in recent years in these animals, but levels of low chlorinated PCBs increased. This is not desirable given the toxic potential of these chemicals (Georgii *et al.* 1994). Hares in Germany were found to be contaminated with PCBs, DDT and heptachlor and around 50% were contaminated with lindane (Brunn *et al.* 1985). DDT, HCH and HCB were detectable in wild boar, roe deer, stag and elk from eastern Poland (Rodziewiecz and Hajduk 1995 – abstract only). Chamois that inhabit the Catalan Pyrenees, Spain, were found to be contaminated with HCB (mean 31.03 ppb), PCB and DDE (mean 2.06 ppb). These animals are herbivores and organochlorine levels were consequently relatively low. However, given their privileged wild habitat and their position in the food chain, the levels found were probably higher than expected (Guitart *et al.* 1990).

5.2.2 European Otters, Mink and Polecats

Otters

In the past, the European otter (Lutra lutra) was common in most inland waters and along the coasts of Europe. During the 1950s the otter population in Europe suddenly began to decline throughout most of its range (see Sjoasen *et al.* 1997). Reports from the 1990s indicate that otter populations and range are seriously reduced over much of northwest and central Europe. Populations are thriving mainly on the western seaboards and on the eastern periphery of Europe (see Mason 1995).

There are several possible reasons for the decline in otter populations throughout Europe, but there is strong evidence that the main culprit is PCB pollution. In environments where PCB contamination is relatively low, concentrations can be relatively high in aquatic top predators like the otter. Levels may reach 10 million times those in water (Mason 1995).

It has been found that PCB levels in declining and endangered populations of otters are high, whereas in thriving populations PCB levels are low (Mason 1995). For example, in Latvia, otters are thriving and have high population densities. Levels of PCBs in these otters (mean 2.3 ppm lipid weight), and in the fish they consume, are lower than reported concentrations in other areas of Europe (Sjoasen *et al.* 1995).

Laboratory studies in mink show that reproduction is impaired when muscle concentrations of PCBs exceed 50 ppm lipid weight. Research indicates that the otter is at least as sensitive to PCB toxicity as mink (Leonards *et al.* 1998). It is therefore of great concern that average concentrations of PCBs greater than 50 ppm have been found in otters from South Sweden (population endangered), The Netherlands (virtually extinct), East Anglia, UK (wild population probably extinct) and Czechoslovakia (declining in many parts). Furthermore, in otter tissues, those PCBs that are known to exert toxic effects, make up a substantial proportion of the total PCBs.

Two otters that were examined from East Anglia which had the highest PCB levels, had bleeding feet, deformed toes and claws, uterine tumours and skin lesions. These symptoms are similar to those in Baltic seals which suffered from a disease complex (adrenocortical hyperplasia) that was probably due to PCBs and resulted in reproductive failure and immune system effects (see section 4.2.1). The otters from East Anglia also displayed disorientation behaviour prior to death (walking round in circles). Disorientation behaviour was also seen in Irish otters, some entering riverside shops. These symptoms are consistent with organochlorine poisoning; all these otters had elevated levels of PCBs (see Mason 1995, Mason 1995b).

Otters are thriving and average PCB concentrations are below 50 ppm in Norway, Ireland, Scotland and Denmark (see Mason 1995). In a recent study, Mason (1998) suggested a decline in PCB levels in otters from England and Wales, UK, although results may be confounded by a few individual otters in the 1980s that had particularly high levels. The study reported a decline in PCBs of about 8% per year, similar to the 7% per year decrease noted for Danish otters that are also expanding their habitat range. Nevertheless, even in areas where contamination is generally low, some individual otters are highly contaminated. This occurs particularly where thriving populations exist in uncontaminated upland/ western areas, for example, in Ireland and Scotland, and some individuals are forced through population pressure to inhabit marginal, contaminated areas. In Denmark, PCB levels at the centre of otter populations were found to be generally low, but isolated populations on the periphery had rather high levels. A study showed that while the otter range has consolidated in the population centre, some of the peripheral populations have disappeared (see Mason 1995).

Mink

The European mink (*Mustela lutreola*) is an endangered species which lives in aquatic environments. Studies presented at a European conference on mink held in 1993 showed a more extensive decline in populations than previously suspected (see Lopez-Martin *et al.* 1994). A study in Northern Spain found that levels of PCBs in mink muscle tissue (118 ppm) were higher than levels that caused reproductive failure in laboratory mink (50 ppm). It was concluded that although it cannot be proven, it would appear extremely likely that mink are being affected by organochlorine pollutants (Lopez-Martin 1994).

Polecats

Unlike otters and mink that feed exclusively on aquatic food, such as fish and frogs, the polecat feeds on both aquatic and terrestrial prey, for example, rodents and birds. The polecat has become restricted in distribution over the past four decades in continental Europe. Habitat destruction has been reported as the main reason for its

decline, although effects of pollutants have not been studied and could be important. Decline may, for example, be related to effects of PCBs as seen in the European otter. Studies in The Netherlands (de Voogt *et al.* 1993), Switzerland (Mason and Weber 1990), and Germany (Behnisch *et al.* 1997), have detected PCBs in the tissue of polecats. The most recent of these studies indicates that levels of PCBs in polecats have declined over the past few years. Some of the animals in the Swiss study had levels of PCBs that were greater than 50 ppm, the level that caused reproductive effects in mink. However, Behnisch *et al.* (1997) estimated that dietary intake of PCBs for polecats in Germany were unlikely (as a single factor) to cause a further decline in polecat populations.

5.2.3 Birds

Several organochlorines have caused reproductive problems and population declines in birds. One of the most notorious effects is egg-shell thinning caused by DDE. This results in crushed eggs and breeding failure of many sensitive birds of prey and fish-eating birds (Fry 1995). The pesticides dieldrin and aldrin were also responsible for population declines in many birds of prey during the 1960s and 70s. In the UK, population declines of the Scottish eagle, peregrine falcon and sparrow hawk were caused by these pesticides, although DDE also contributed to their low reproductive success through egg-shell thinning (see Gilbertson 1989). As the use of some organochlorines has fallen over the years, populations of depleted species have gradually recovered and egg-shell thickness and breeding success have improved. For example, the natural recovery of kestrels in the UK occurred by the late 1970s and sparrowhawks by the mid-1980s. These recoveries occurred after dieldrin and aldrin residues in the bird's livers fell below a mean level of about 1.0 ppm. Monitoring levels of organochlorines over nearly thirty years has showed significant declines in dieldrin and aldrin, but not in PCB levels in kestrels and sparrowhawks (Newton *et al.* 1993).

As in the UK, DDE and other pesticides are blamed for dramatic decreases in the populations of several birds of prey in Sweden during the 1950s and 60s (Swedish EPA 1999). However, recovery of populations of peregrine falcons, white-tailed eagle and eagle owl have been slow. These birds would probably not have survived the 1970s and 1980s without human assistance in the form of captive breeding and supplementary feeding. To date, the population of peregrine falcons has not yet regained its original size.

Fish-eating seabirds may also have succumbed to detrimental effects of organochlorines. PCBs were implicated in the death of 15,000 guillemots in the Irish Sea in 1969 and contributed to the death of gannets in 1972 in this region (see Gilbertson 1989). Since these times, studies have shown that levels of organochlorines have generally declined in seabirds of the Northern Hemisphere. For example, downward trends in the levels of DDE, HCB, PCBs, β-HCH, lindane and oxychlordane were recorded between 1983 and 1993 in several seabird species from regions in Svalbard in the Arctic, north Norway and northwest Russia (Barrett *et al.* 1996).

Some effects of organochlorines in seabirds have been observed recently despite the general downward trend in many organochlorines. For example, cormorants that breed in heavily contaminated areas of the rivers Rhine and Meuse have severely reduced breeding success compared to other Dutch colonies. This is due to egg-shell thinning and increased embryo mortality which have been associated respectively with levels of DDE and PCBs found in eggs (Dirksen *et al.* 1995). A study on cormorants in eastern England found that levels of organochlorines had declined in eggs, but higher concentrations of DDE, PCBs and dieldrin in eggs were associated with thinner egg-shells (Mason *et al.* 1997).

A recent study was conducted on over 300 common guillemots that were either found dead on Belgian beaches over six successive winters, or were found alive but failed to recover at rehabilitation centres. The most common cause of death was due to oiling. However, 75% of the birds were in a state of cachexia, (reduced fat reserves and emaciated pectoral muscles). Levels of PCBs and mercury in the birds were clearly correlated with the incidence of

this condition, which may ultimately result in lethal hemorrhagic gastro-enteropathy. The study suggested that PCBs and mercury cannot be considered as the unique and direct cause of death in the birds, but might be an additional source of physiological stress, leading to debilitation and death (Debacker *et al.* 1997).

Water birds that eat fish, such as the heron, previously suffered from DDE impacts of egg-shell thinning and egg breakage. Studies in Italy on the black crowned night heron, *Nycticorax nycticorax,* (Fasola *et al.* 1998-abstract only) and in the UK on the grey heron, *Ardea cinerea* L., (Newton *et al.* 1993) show that levels of DDE in herons or their eggs have significantly declined. A study on grey herons in France noted that levels of DDE in eggs were lower than levels associated with reproductive effects reported in the wild or in laboratory studies (de Cruz *et al.*1997).

5.2.4 Marine Fish

Research has shown that chemical pollution of the sea can adversely affect individual fish, and hence, potentially, whole fish populations. The chemicals to which fish are exposed comprise a complex mixture of contaminants. These contaminants include persistent organochlorines along with numerous other chemical groups. Organochlorines are detectable in fish from the Atlantic (Hellou *et al.* 1997), the Baltic (Strandberg *et al.* 1998), the North Sea (CEFAS 1998a) and in less polluted deep sea waters in the far north such as off the west coast of Greenland (Berg *et al.* 1997). Higher levels of organochlorine contaminants have predictably been found closer to industrialised areas. For instance, higher levels of PCBs and HCB were found in fish in the Gulf of Gdansk, a potentially highly polluted area, compared to fish from other regions of the Baltic Sea (Strandberg *et al.* 1998). Nevertheless, a study on levels of organochlorines in fish oils, sold as dietary supplements, showed that organochlorine pollution of the marine environment to be a very widespread and generic problem (Jacobs *et al.* 1997). Other POPs, including brominated flame retardants, were found in fish that were taken from the North Sea (de Boer and Dao 1993). Organotin compounds were present in fish from the North Sea and other European waters (Belfroid *et al.* 1999).

Monitoring possible impacts of chemical pollution on fish have revealed effects such as skin damage and liver malfunction (e.g. CEFAS 1998a). Other effects include a variety of impacts on reproduction. Long-term exposure to pollutants such as pesticides is also suspected of predisposing fish to diseases through suppression of the immune system (Dunier and Siwicki 1993).

Reproduction in fish may be affected by the direct uptake of chemicals into the ovaries. Even cod from relatively remote areas are known to have organochlorine contaminants in their ovaries (Hellou *et al.* 1993) as well as flatfish from more polluted areas (Loizeau and Abarnou 1994). Many organochlorines can act directly on reproductive tissues. In the North and Baltic Seas, links have been made between the levels of these contaminants and reproductive success. Contamination of the ovaries of fish by organochlorines has been correlated with a lower hatching success in a variety of fish species in the Baltic Sea (von Westernhagen *et al.* 1987a, von Westernhagen 1987b).

Effects on development in fish have been recorded in Atlantic mackerel of the US Atlantic coast. Mortality, malformation and abnormal genetic effects (chromosome division) of fish embryos were found to correlate with levels of pollution. These findings were further substantiated by the fact that abnormal chromosome division was also found in the bodies of various species of adult fish in more polluted areas. Such effects in adults may compromise the quality of their eggs and increase embryo mortality (Longwell *et al.* 1992). A high frequency of sterile burbot with retarded gonad development have also been reported in the Gulf of Bothnia (Pulliainen *et al.* 1992). It has been suggested that these effects are related to pollution in the Baltic Sea.

Many chemicals, including some POPs, are capable of disrupting endocrine systems. They may do this by affecting levels of circulating hormones or induc-

ing liver enzymes that are involved in the regulation of hormones. A recent extensive review has concluded that effects of chemicals upon fish include lesions, haemorrhage or malformations of the gonads. Malformations of these organs together with the pituitary and liver can inhibit secretion, production and metabolism of hormones by the endocrine system (Kime 1995). Fish exposed to contaminated sediments in the northwest Atlantic were found to have reduced levels of circulating hormones and female fish were less likely to spawn (Johnson 1993). Winter flounder exposed to contaminated sediments in the northwest Atlantic were less able to manufacture egg yolk, vital to the survival of fish larvae (Pereira *et al.* 1992). This in turn could lead to reduced viability of eggs and larvae (Spies and Rice 1988, Collier *et al.* 1993). These effects may occur for long periods of time. Consistent low egg viability and larval size has been observed over three consecutive years in one population of winter flounder. The contaminants present varied in nature but included PCBs (Nelson *et al.*1991).

A recent study undertaken on flounder from the North Sea, in offshore and coastal waters of the UK, revealed endocrine-disrupting effects in this species (CEFAS 1998b). Some of the male fish that were tested had detectable levels of yolk protein in their blood. Yolk protein is normally only produced by female fish and its presence in male fish denotes effects of estrogenic chemicals. Some of the male fish were also classified as imposex because they had eggs in their testes. Those fish that were the most highly contaminated with PCBs, organochlorines and other contaminants were the worst effected, though it is not certain which chemicals were causing effects. The authors of this study made a qualitative association between the inputs of industrial effluent and the extent of effects on the fish. It is not known whether the effects on the flounder will have long-term implications for their survival.

5.2.5 River Fish

Since many POPs are widely dispersed in the global environment, they can be detected in river fish and in fish from freshwater lakes. Even in remote regions, such as lakes in northern Finland in the subarctic, appreciable amounts of PCBs (0.1-0.5 ppm) were detected in the fish species Arctic charr (*Salvelinus alpinus*), (Korhonen *et al.* 1997).
Levels of organochlorines in fish are often higher in industrialised areas reflecting local sources of these chemicals. The river Elbe in Germany is one of the most highly polluted rivers in Europe as a consequence of industrial discharges. Fish are still contaminated with organochlorines in this river despite the recent closure of some polluting industries (Marth *et al.* 1997). In Latvia, organochlorine pollution in fish from lakes and rivers was highest in the industrial region of Riga. PCB levels in the fish suggested recent inputs of fresh PCBs to inland waters of this region (Valters *et al.* 1999). In Italy, levels of organochlorines in fish in the river Po indicated there have been recent inputs of DDT into the river, possibly from its use for flower cultivation (Galassi *et al.* 1996). Intensive use of pesticides in agricultural regions may cause pollution in rivers due to run off from fields. For instance, in the agricultural region near the Cinca river in northeastern Spain, lindane was detected in 90% of fish which were tested (Raldua *et al.* 1997).

Other POPs can also be found in aquatic life of rivers and lakes. For example, brominated flame retardants have been found in fish taken from rivers in the Netherlands and Sweden (de Boer and Dao 1993). Nitro musks have been detected in fish and shell-fish in Canada and in Western Europe (Gatermann *et al.* 1999). In Canada, levels of musk ketone exceeded the German tolerance limit for fish (10 ppb ww) in 15 out of 29 samples which were analysed. In German rivers, levels of polycylic musks were higher than musk xylene and musk ketone. This is most likely because these compounds are being increasingly used in Europe to substitute for nitro musks in fragrance formulations.

As in marine fish, pollution can cause adverse effects on reproduction and health in freshwater

fish. The early life stages, embryos and larvae, are the most sensitive stages. Laboratory studies have shown that exposure to organochlorines can result in embryo mortality and developmental abnormalities (e.g. Gorge and Nagel 1990). Another effect of organochlorines in fish, shown by laboratory studies, is suppression of the immune system (e.g. Dunier and Siwicki 1993).

In UK rivers, it is possible that endocrine disrupting properties of phthalates and alkylphenols (chemicals on the OSPAR priority list) have been partly responsible for causing reproductive problems in fish in rivers and estuaries. Studies found that when male fish were placed in cages near to sewage discharges, they became feminised. They produced high levels of egg yolk protein, normally only produced in females, and there was an increased incidence of reduced testis size. This indicated that something in the sewage effluent had estrogen-like activity (Purdom *et al.* 1994, Matthiessen 1996).

Subsequent studies showed that the estrogenic activity of domestic sewage could be due to natural estrogens excreted by women and, to a lesser extent, to synthetic estrogen from the contraceptive pill. However, this could not account for some of the effects being detected in fish, especially from industrial sewage (Desbrow *et al.* 1996). Both phthalates and alkylphenols have estrogenic properties and they are present in industrial sewage. Experiments have shown that two alkylphenols, nonyl phenol and octylphenol, induce yolk protein synthesis in male fish and cause reduction in testes size (Jobling *et al.* 1996). It has been concluded that, at least for the river Aire in the UK where a wool scouring factory discharges these chemicals into the river, alkylphenols could be solely responsible for causing feminisation of male fish (Matthiessen 1996). Initial studies on the Tyne estuary have found feminisation in male fish which is associated with nonylphenol levels, although it is not yet known whether other chemicals may be implicated (Lye and Frid 1997).

6

TISSUE LEVELS AND HEALTH EFFECTS OF POPs IN HUMANS IN EUROPE

The ubiquity of POPs in the global environment is reflected by their presence in human tissues. Research from numerous countries throughout the world has demonstrated that measurable quantities of persistent organochlorines are present in human adipose tissue, blood and breast milk (Jensen and Slorach 1991, Allsopp *et al.* 1998). Some other POPs such as brominated flame retardants and nitro musks are also detectable. Measuring levels of POPs in tissues is one of the most accurate and precise ways of assessing human exposure to these environmental pollutants. This information is also useful for studying the relationship between human exposure to POPs and health effects (Schecter 1998).

In all countries, exposure to POPs in the general population comes largely from contaminated foodstuffs. The highest levels are present in meat, fish and dairy products since many POPs are soluble in fats. Other dietary exposure may arise from pesticide residues sprayed on crops and which contaminate drinking water. Food packaging may also result in phthalates and bisphenol-A leaching into foods. Children's toys, such as teethers, that are made of soft PVC plastic may contain phthalates and so potentially expose them to these chemicals (Stringer *et al.* 1997). For a more complete account of these exposure routes in lower latitudes see Allsopp *et al.* (1997).

Members of the population who are particularly highly exposed to POPs include individuals who consume large quantities of fatty fish from polluted waters such as the Baltic Sea. Other more highly exposed members of the population include nursing infants and those who are occupationally exposed to persistent chemicals. Intake of POPs for nursing infants is high. POPs which have accumulated in a woman's body during her lifetime decrease in the mother's body as they pass to her breast milk, and hence to her nursing infant. As breast milk is very rich in fats it contains higher concentrations of organochlorines than most other diets (Quinsey *et al.* 1996). Organochlorines in breast milk are found in parts per million or parts per trillion levels.

Studies from the Great Lakes, the Netherlands, and from PCB rice-poisoning incidents in Japan and Taiwan, indicate that exposure to PCBs and dioxins is linked to subtle effects on development. These include reduced intellectual capacity, growth retardation and immune system effects. Some of these effects have occurred at current background levels found in the general population of Europe (e.g. Weisglas-Kuperus 1998, Brouwer *et al.* 1998). Other studies from several countries have suggested reproductive effects such as reduced sperm counts, increases in male reproductive abnormalities, shortened duration of lactation and shortened menstrual cycle length may result from chemical exposure (Mendola *et al.* 1997). For a more detailed account of such health effects, see Allsopp *et al.* (1997).

6.1 Highly Exposed Populations and Health Effects

Studies show that consumption of contaminated fish and seafood can lead to high exposure to POPs in individuals from the general population of Europe. Populations who are particularly highly exposed to POPs through their diet include the Faroe Islanders who consume whale meat, and fishing communities of the Baltic Sea, who consume high amounts of fatty fish.

6.1.1 Faroe Islanders

The North Atlantic Faroese pilot whale drive fishery takes between 500 and 3000 whales annually. The average consumption of whale meat by Faroese Islanders results in an intake of PCBs of 2.06 µg/kg body weight/day. This is of concern; it is more than double the US Food and Drugs Administration TDI for PCBs, (1µg/kg/day). Pilot whale blubber also contains DDE, lindane and dieldrin, and an average consumption also results in exceedance of the ADI for dieldrin (Simmonds *et al.* 1994, Simmonds and Johnston 1994).

Exposure of the Faroe Islanders to PCBs and other organochlorines through the consumption of pilot whale blubber appears to be comparable to exposure in the Great Lakes populations consuming contaminated sports fish. Studies on fish-eaters from Lake Michigan found that higher fish consumption in women correlated with a lower birth

weight in their children (Fein *et al.* 1984). It was also associated with small, but significant, deficits in cognitive function in their children (Jacobson and Jacobson 1993 and 1996). For the Faroe Islanders, the long-term health implications of consuming whale meat cannot be predicted with certainty. However, given that this does not represent an essential subsistence food, ingestion of pollutants in whale meat may be best avoided (Simmonds and Johnston 1994).

6.1.2 Swedish Fishermen

Fatty fish from the Baltic Sea contain high levels of PCBs and other organochlorine contaminants such as toxaphene (e.g. de Boer and Wester 1993). Swedish fishermen from the South-east part of the Swedish coast eat more than twice as much fish as individuals from the general population. Families of fishermen from the east coast of Sweden, known to have consumed PCB- and dioxin-polluted fish from the Baltic Sea, gave birth to children with lower birthweights than their counterparts from the west coast of Finland whose diet was less contaminated (see Brouwer *et al.* 1995). Other studies have also found an association between low birth weight and exposure to PCBs. Maternal consumption of PCB contaminated fish from Lake Michigan was linked to lower birth weight of their babies (Fein *et al.* 1984). Exposure to high levels of PCBs and dioxins at a rice-poisoning incident in Japan was also associated with lower birth weights (Wong and Hwang 1981). Similar associations can also be identified in otherwise healthy populations in the Netherlands (Patandin *et al.* 1998). The authors commented that although it is not clear what the impact of a lower birth weight has for later development, all factors that negatively influence growth of the foetus and babies should be regarded as harmful.

In Finland, with the exception of Baltic fish, PCB concentrations in food are relatively low (see Vartianinen *et al.* 1998). However, a study on pregnant women living in the Aland and Turku archipelago in Finland, concluded that consumption of fatty fish from the Baltic Sea was not associated with higher blood and umbilical cord levels of PCBs (Hagmar *et al.* 1998). Moreover, no association was apparent in these communities between PCB and PCDD/F intake and birth weight (Vartiainen *et al.* 1998). The reasons underlying the departure of these observations from those of other studies are not clear. They demonstrate, nevertheless, that understanding the nature and extent of impacts which may result from chemical exposure remain limited.

6.1.3 Europe

Some individuals in European countries may have a comparably higher exposure to certain POPs than other individuals because of their consumption of contaminated fish. For example, a recent review on organotins in seafood noted that for most countries of the world, including those with a high seafood consumption, no data on organotin levels in seafood are available (Belfroid *et al.* 1999). However, information is available for 22 countries. Of these, samples of seafood from more than one third of the countries exceeded the proposed "safe" level (tolerable average residue level) for TBT in food based on average seafood consumption in each country. This included Italy and France. Calculations were based on consumption for an adult weighing 60 kg. Those adults consuming more than the estimated average intake in a country will be even more highly exposed. In addition, since children weigh less than 60 kg the risk to health posed by consumption of TBT in seafood could be greater for them than for adults.

6.2 Tissue Levels of POPs

Levels of dioxins and organochlorine pesticides in human milk from European countries that have been published in the scientific literature are presented in tables 6.1a/b and 6.2 respectively. There are however some discrepancies in the sampling methods and analytical techniques that have been used in different studies. This means that the figures are not directly comparable. However, the figures can at least give an approximate indication of the variation in levels of organochlorine contaminants between different countries (Allsopp *et al.* 1998).

6.2.1 PCDD/Fs and PCBs

The World Health Organisation has conducted an extensive sampling program on levels of dioxins and PCBs in many countries (WHO 1996). Levels of dioxin-like chemicals (PCDD/Fs) are generally higher in industrialised countries due to the release of these compounds from many industrial processes, notably those involving production, use or disposal of organochlorines. The highest levels of PCDD/Fs (10-20 ppt) were apparent in Belgium, Canada, Finland, Spain and The Netherlands. The lowest levels (4-10 ppt) were reported Albania, Hungary and less industrialised regions in Croatia, Norway and the Russian Federation (see table 6.1a and 6.1b).

Comparison of PCDD/F levels in breast milk collected in 1992/3 have been made with samples collected in 1987/8 (WHO 1996). In some European countries levels declined substantially during this period. In others such as Denmark and Finland there was no decline. Overall, reductions in levels may reflect measures taken to reduce dioxin emissions.

The World Health Organisation (WHO) has set a TDI for PCDD/Fs of 1-4 pg/kg/day (van Leeuwen and Younes 1998). US EPA has set a much lower value for ADI (0.01 pg/kg/day), (US EPA 1994). Both ADI and TDI are exceeded for a breast-fed infant in most developed countries. For example, if an infant was breast-fed for one year on milk containing 20 ppt TEQ its average daily dose would be 60 pg TEQ/kg/day (US EPA 1994). This exceeds the US EPA recommended ADI 6000-fold.

Differing laboratory methods have made comparisons of PCB levels difficult between different countries. Nevertheless, it appears that PCB levels in breast milk in Sweden (Noren 1993), Germany (Furst *et al.* 1994) and The Netherlands (Liem and Theelen 1997) have remained fairly stable in recent years.

6.2.2 Organochlorine Pesticides

Levels of organochlorine pesticides in breast milk are given in table 6.2. DDT and its metabolite DDE are widespread contaminants in human milk. The highest levels are found in countries where DDT is still in use. Levels of DDE are also relatively high, between 1 and 2.5 ppm, in the Faroe Islands (2.01 ppm), France (2.18 ppm), former East Germany (1.13 ppm), Italy (2.2ppm), Kazakstan (1.96 ppm), Russia (1.26 ppm) and Slovak Republic (1.66 ppm). The lowest levels (<1ppm) are evident in other western and northern European countries. Levels have declined in recent years in countries where DDT has been banned (see Allsopp *et al.* 1998).

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Sperm whale

Levels of HCB in human milk are less than 0.2 ppm in most countries and in some, have shown a declining trend in recent years. The highest levels of HCB have been recorded for the Czech Republic (0.639 ppm) and Slovakia (0.829 ppm). It is suggested that this is a result of HCB use in agriculture and its formation during the production of chlorinated solvents in these countries (Kocan *et al.* 1995). The ADI for HCB is exceeded in the Czech Republic, Slovakia and in Russia. In addition, a recent study suggested that HCB exerts dioxin-like toxicity. Current levels in breast milk would add substantially overall dioxin-like toxicity (van Birgelen 1998).

β-HCH and γ-HCH have been detected in breast milk from several countries with β-HCH being the most persistent. Relatively high levels were evident in Russia and Kazakstan. Levels of β-HCH were reported to decline in Sweden between the mid-1970s and 1990 (Noren 1993, Vaz *et al.* 1993).

Table 6.1a Mean Levels of PCDD/Fs in Breast Milk from Various Countries, 1987/88 and 1992/3

Country	Area	No. Samples in pool (n) in 1987/88	1992/3	ppt TEQ [PCDD/F] 1987/88	ppt TEQ [PCDD/F] 1992/3
Albania	Tirana		10		4.8
	Librazhd		10		3.8
Austria	Vienna (urban)	54	13	17.1	10.7
	Tulln (rural)	51	21	18.6	10.9
	Brixlegg (industrial)		13		14.0
Belgium	Brabant Wallou		8	33.7	20.8
	Liege		20	40.2	27.1
	Brussels		6	38.8	26.6
Croatia	Krk	14	10	12.0	8.4
	Zagreb	41	13	11.8	13.5
Czech Republic	Kladno		11		12.1
	Uherske Hradiste		11		18.4
Denmark	7 different cities	42	48	17.8	15.2
Finland	Helsinki	38	10	18.0	21.5
	Kuopio	31	24	15.5	12.0
Germany	Berlin	40	10	32.0	16.5
Hungary	Budapest	100	20	9.1	8.5
	Scentes	50	10	11.3	7.8
Lithuania	Palanga (coastal)		12		16.6
	Vilnius city (urban)		12		13.3
	Anykshchiai (rural)		12		14.4
Netherlands	mean of 17 individual samples	10	17	34.2	22.4
Norway	Tromso (coastal)	11	10	18.9	10.1
	Hamar (rural)	10	10	15.0	9.3
	Skien/Porsgrunn (industrial)	10	10	19.4	12.5
Russian Federation	Arkhankeisk		1		15.2
	Karhopol		1		5.9
Slovak Republic	Michalovce		10		15.1
Spain	Bizkaia		19		19.4
	Gipuzkoa		10		25.5
Ukraine	Kiev nr.1		5		11.0
	Kiev nr.2		5		13.3
United Kingdom	Birmingham		20	37.0	17.9
	Glasgow		23	29.1	15.2

Source: WHO (1996)

Table 6.1b Mean Levels of PCDD/Fs in Breast Milk from Various Countries

Country/ Year Samples Taken	Number of Samples	ppt TEQ [PCDD/Fs]	Reference
Estonia Tarto & Tallinin 1991	12	17.5 (Nordic)	Mussalo-Rauhamaa and Lindstrom 1995
France	15	20.1	Gonzalez *et al.* 1993
Sweden Sundsvall, Umea, Goteborg, Borlange 1987	40	22.4 (Nordic)	Mussalo-Rauhamaa and Lindstrom 1995

Table 6.2 Mean Concentration of Organochlorine Pesticides in Breast Milk on a Lipid Basis (ppm)

Country	p,p'DDT	p,p'DDE	Dieldrin	HCB	HE	β-HCH	γ-HCH
Belarus	0.672[1]						
Czech Republic	0.998[1]			0.639		0.071	
Faroe Islands	0.064	0.981 2.01		<0.1		<0.04	
France	0.044	2.183	0.19	0.147	0.097	0.287	0.037
Germany							
a Eastern Germany	0.134	1.130	0.042	0.167	0.008	0.083	0.0098
b Western Germany	0.061	0.589	0.009	0.218	0.014	0.075	0.016
c Lower Saxony	0.38[2]		0.14[2]	0.223[2]		0.045	0.016
Italy	0.15	2.2		0.18			
Kazakstan	0.3	1.96	nd	0.091	nd	2.210	0.23
The Netherlands		0.705	0.013	0.083			
Norway	0.338[1]			0.041			
Russia							
Kola Peninsula	0.178	1.269		0.129			
5 different regions	0.387	1.408	<0.002-3.0	0.245	0.0118	1.589	0.0094
Slovak Republic	0.126	1.667		0.829			
Spain	0.012	0.6041	0.0039	0.0008	0.0311	0.235	0.0105
Sweden	0.03	0.35		0.037		0.02	
UK	<0.02	0.40	0.03	0.02		0.08	<0.02

Source: Allsopp *et al.* (1998) [1] ΣDDT [2] median values

6.2.3 Other POPs

Polybrominated flame retardant chemicals, polybrominated diphenyl ethers (PBDEs) have also been detected recently in German breast milk and Swedish blood samples (Wehler *et al.* 1997). A recent study in Sweden is of great concern since it showed that PBDEs levels in breast milk have continually increased since 1972 (Noren and Meironyte 1998). The increase is exponential. Levels in the milk have doubled every 5 years.

Polybrominated dibenzo-*p*-dioxins and furans (PBDD/Fs) in breast milk of Swedish women were found to be either not detectable or present at low concentrations (less than 20 ppt), (Wiberg *et al.* 1992). These compounds are of particular concern because they also add to the body burden of dioxin-like compounds.

Nitro musks are synthetic compounds that are used as fragrances in cosmetic and household products. These chemicals have been shown to bioaccumulate in aquatic food webs. Human exposure results from consumption of contaminated fish and probably from skin absorption from products containing the chemicals. As a consequence of their persistence and lipophilicity and the worldwide production of nitro musks they can be assumed to be ubiquitous in human fat and milk (Rimkus *et al.* 1994).

Studies in Germany have detected nitro musks, namely musk xylene and musk ketone in the majority of breast milk samples that have been tested (Failing *et al.* 1999, Liebl and Ehrenstorfer 1993, Rimkus *et al.* 1994). Human milk samples taken in 1995 had mean concentrations of musk xylene and musk ketone of 41 µg/kg (ppb) and 10 ug/kg (ppb) milk fat respectively (Failing *et al.* 1999). An acceptable daily intake of musk xylene for adults, based on a lifetime exposure to the chemical, has been proposed as 3 ng musk xylene/kg body weight per day by Maekawa and coworkers (in Failing *et al.* 1999). The daily exposure for an infant from breast milk containing 41 ppb musk xylene would be 742 ng per day. This is a 50-fold exceedance of the proposed ADI. However, the application of an infants exposure in this context is again questionable because the proposed ADI is based on a lifetime intake and breast feeding normally continues for less than 1% of the average life-span of a human (Failing *et al.* 1999). Nevertheless, nitro musks are persistent, bioaccumulative chemicals that are known to have toxic effects in laboratory animals and their presence in human milk is therefore not desirable. It has been suggested that legal measures are urgently required to reduce and avoid such chemicals in human fat and milk. Musk xylene has not been used in Japanese products under voluntary restriction since 1982 because of its high bioaccumulation potential in aquatic organisms (Rimkus *et al.* 1994).

7

TISSUE LEVELS AND HEALTH EFFECTS OF POPs IN ARCTIC INDIGENOUS PEOPLES

Indigenous Peoples of the Arctic are highly exposed to POPs as a consequence of the consumption of high amounts of sea mammals in their traditional diet. Studies on Inuit babies have revealed a higher consumption of PCBs in breast milk is associated with possible effects on the immune system and on growth (Dewailly *et al.* 1993b, Patandin *et al.* 1998). These effects are consistent with studies from other countries.

7.1 Exposure to POPs

The last decade has witnessed increasing concern for Indigenous Peoples residing in the Arctic because of potential health effects from contaminants in their diet. The traditional diet of these peoples may expose them to pollutants, such as organochlorines and brominated flame retardants, that have accumulated in the animals they consume.

Approximately 95,000 people live in the Canadian Arctic and 51% of these people are of aboriginal descent (e.g. Inuit/Dene/Metis), (Oostdam *et al.* 1997). Dietary surveys have been carried out to investigate exposure to food contaminants for Inuit people (Eskimos) who live on Baffin Island in the eastern Canadian Arctic (Kuhnlein *et al.* 1995, Chan *et al.* 1997) and Broughton Island in the Northwest Territories (Kinloch *et al.* 1992). Studies have also been conducted with Sathu Dene/Metis Indigenous people of the western Northwest Territories (Kuhnlein *et al.* 1995, Berti *et al.* 1998). Results of these studies show that estimated "safe" intake levels, known as Acceptable or Tolerable Daily Intakes (ADI, TDI) that have been set by the Canadian government for some organochlorines in diet, were frequently exceeded by the Inuit diet. For example, a survey on Baffin Island showed that over 50% of women exceeded the ADI for toxaphene and chlordane, and the ADI for PCBs was also frequently exceeded (Kuhnlein *et al.* 1995). In contrast, the percentage of exceedance among the Dene/Metis communities in the western NWT was very low (Kuhnlein *et al.* 1995, Bertis *et al.* 1998). These differences are due to the balance of foodstuffs consumed in the diet. The traditional Inuit diet consists of a high proportion of food derived from whales and seals. In contrast, the diet of the Dene/Metis is based on terrestrial species such as caribou and fish which are lower in the food chain and therefore generally contain lower levels of organochlorines (Kuhnlein *et al.* 1995).

A dietary study on Indigenous people inhabiting Taymir Autonomous Region in the Russian Arctic showed that intake of total PCBs was about 3-4 times less compared to the Inuit population of Broughton Island in the Canadian Arctic. Intake of DDT and chlordanes were also higher for the Inuit. Again, the difference is attributable to variations in traditional diet. The Indigenous people of the Russian Arctic eat predominantly reindeer meat and fish whereas the Inuit consume mainly meat and fat of sea mammals (Klopov 1997).

Traditional Inuit foods are therefore primarily contaminated with chemical residues from industrial and other activities often at a considerable distance from the Arctic. The potential for health effects from consuming such high levels of contaminants are largely unknown. However, available imported foods are nutritionally inferior and the traditional diet also has important social and cultural benefits (Kinloch *et al.* 1992, Kuhnlein *et al.* 1996). Therefore, it has been suggested that it is necessary to act at national and international levels to control the entry of pollutants into the environment in order to make a sustained contribution to the health of Inuit, rather than to discourage the use of certain traditional foods (Kinloch *et al.* 1992). This is, in any case, a more responsible approach. These issues are discussed further in section 8.

7.2 Tissue Levels

A study has been conducted in Arctic countries by the AMAP Monitoring Program to assess geographical variations of organochlorines in maternal blood from women inhabiting these regions (Gilman *et al.* 1997). The levels of organochlorines mirrored the relative amounts of traditional food consumed, especially where marine mammals make up a large amount of the diet. Hence mothers in Greenland had the highest levels of HCB, mirex and 3 chlordane metabolites, followed by Inuit women in

Table 7.1 Maternal Plasma Concentrations of Pesticides: Circumpolar Study 1994-1996 (geometric means µg/kg lipid (ppt))

Country/ Pesticides	Canada [1] (n = 67)	Greenland [2] (n = 117)	Sweden [3] (n = 40)	Norway [4] (n = 60)	Iceland (n = 40)	Russia [5] (n = 51)
β–HCH	9.3	18.5	9.2	8.1	32.1	222.5
α-chlordane	1.0	1.1	1.0	1.3	1.3	1.6
γ-chlordane	1.1	1.3	1.0	1.3	1.3	1.4
Cis-nonachlor	6.6	20.9	1.2	1.8	2.7	5.3
p,p'-DDE	133	407	84.0	79.4	113.2	411.9
p,p'-DDT	7.9	15.0	2.4	3.0	4.0	48.3
HCB	55.1	97.6	15.6	23.1	41.2	62.8
Mirex	4.5	9.1	1.1	1.4	1.9	1.4
Oxychlordane	27.8	60.8	1.9	3.7	6.6	3.3
Transnonachlor	30.5	110	3.8	6.8	12.2	11.5

[1] Inuit women from west/central NWT, [2] women from Disko Bay region, [3] women from Kiruna, [4] women from Hammerfest and Kirkenes, [5] women from Nikel.

Source: Gilman *et al.* 1997

Table 7.2 Umbilical cord blood concentrations of PCBs and DDE (geometric means, µg/kg lipid (ppt)): Circumpolar study 1994-1996

	Canada (Inuit) [a] (n = 319)	Canada, general population (n = 502)	Greenland (n = 102)
PCBs (as Aroclor 1260)	780	211	1388
PCBs (∑ 14 congeners)	309	115	504
p,p'-DDE	384	173	424

a Nunavik (Northern Quebec, Canada)

Source: see AMAP (1998 p820)

Table 7.3 Organochlorine concentrations in human milk from Inuit (Arctic Quebec) and Southern Quebec Women

Organochlorine compounds	Inuit women (n = 107) from Arctic Quebec Mean (ppt)	Southern Quebec women (n = 50) Mean (ppt)
DDE	1212 ± 170	336 ± 18
HCB	136 ± 19	28 ± 3
Dieldrin	37 ± 5	11 ± 1
Mirex	16 ± 4	1.6 ± 0.3
Heptachlor epoxide	13 ± 2	8 ± 1
Trans-chlordane	3.7 ± 0.4	<6
Endrin	<8	<6

Mean ppt represents arithmetic mean and 95% confidence interval.

Source: Dewailly *et al.* (1993)

Canada, (see table 7.1). Levels were lower in maternal blood from Sweden, Iceland and Norway because these peoples primarily consume marine fish species and terrestrial mammals such as reindeer and very few marine mammals. Blood samples taken from Russian mothers had notably high levels of DDE and HCH which suggests that there are significant local uses of these chemicals in northwestern Russia, or that there are significant amounts in their food supply (Gilman *et al.* 1997).

Umbilical cord blood levels of PCBs and DDE were also analysed in the AMAP study, (see table 7.2), (see AMAP 1998). The highest levels were found in samples from Greenland, followed by Inuit from Canada. Concentrations were markedly lower in samples from the general population in Southern Canada.

In keeping with AMAP studies, other research has found high levels of organochlorine contaminants in tissues from Inuit people of Greenland and Canada in comparison to other countries. Research on organochlorine levels in human tissues at autopsy in Nuuk and Ilulissat, found higher levels of PCB, DDT and HCB in adipose (fat) tissue than observed in other countries (Mulvad *et al.* 1995). Another study showed that levels of PCBs in tissues at autopsy were higher than levels of dioxins and furans (PCDD/Fs), (Ryan *et al.* 1996). This is typical of people who consume large quantities of marine wildlife since it contains significantly more PCBs than PCDD/Fs. The study concluded that exposure of Greenland Inuits to PCBs and PCDD/Fs as measured in fat and liver tissue appears to be about 3 times greater than Inuit from northern Quebec and markedly higher than urban dwellers from populated areas. It noted that the Greenland Inuit are one of the most exposed non-occupational groups to these chemicals. It is of interest that tetra-, penta-, and hexadioxins and furans were not detectable in well-preserved frozen remains of Eskimo women from Alaska, dated between 100 and 400 years old. This shows that these chemicals were not previously present in humans from the Arctic and implicates their current source as being from the combustion of synthetic chemicals in countries from lower latitudes (Schecter *et al.* 1988).

A study on adult Inuit from Nunavik (Arctic Quebec) found that body burdens of PCBs and dioxin-like compounds largely exceeded those of Southern Quebec adults. For example, levels of PCBs in Inuit blood (2.0 ppm lipids) were 15-fold greater than for Southern Quebec (0.13 ppm lipids). Levels of dioxin-like compounds were 3.4-fold greater for Inuit (89 ppt TEQ lipids) than from Southern Quebec (26.1 ppt TEQ lipids), (Ayotte *et al.* 1997).

Another study on Inuit women from the same region indicated that levels of PCBs in breast milk were 7 times higher than in breast milk from Southern

Quebec women. Similarly, concentrations of DDE, HCB and dieldrin were on average 4 times higher in Inuit women (see table 8.3), (Dewailly *et al.* 1993).

A study in the Russian Arctic reported that higher concentrations of organochlorines in breast milk of indigenous women compared to non-aboriginal women. It was estimated that the average intake for native infants during the first month of breast-feeding was 1.87 times as high as for non-indigenous ones (Klopov 1997).

7.3 Health Effects

Acceptable Daily Intakes (ADIs) are set by regulatory authorities in order to protect public health. The ADI is defined as the daily intake of a chemical that, during a lifetime, appears to be without appreciable risk on the basis of all facts known at that time (see Stevens *et al.* 1993). Although ADIs should be interpreted with caution, relying as they do on numerous extrapolations and assumptions, they nevertheless provide a useful indication of the scale of the problem facing Arctic communities.

For an Inuit newborn, the average daily intakes of most organochlorines from breast milk exceeds current ADIs. However, the application of ADIs to the short period of breast-feeding is unclear because ADIs are formulated on the basis of a whole lifetime of exposure in an adult. Nevertheless, it is of great concern when ADIs are exceeded by breast-feeding even if relevance to health is unknown. Given that infants may be more susceptible to the toxic effects of chemicals than adults, because for example they are undergoing rapid tissue growth and development, then it is most likely that current ADIs are higher than appropriate for infants (Quinsey *et al.* 1996).

It is questionable whether ADIs are protective of health in general. The process of risk assessment used in their derivation involves many uncertainties. Furthermore, many POPs have been listed as endocrine disruptors, and current endpoints used in toxicity tests may not be sensitive enough to detect adverse effects of such chemicals. Research shows that some of these chemicals may have greater effects at lower doses than higher doses (vom Saal *et al.* 1995). As noted by Professor vom Saal (University of Missouri), (ENDS 1997), " *there are no safe doses of endocrine disruptors, just as there are no safe doses of carcinogens".*

Despite the potential disadvantages of breast-feeding because of contaminants in the milk, breast-feeding is known to convey many advantages such as important nutritional and immunological benefits (Sonawane 1995). Breast-feeding is therefore recommended by health experts even though there is concern over chemical contaminants in human milk (WHO 1996, MAFF 1997). Again, control of contaminants at source must be the key focus.

One study has attempted to provide a health risk assessment specifically for Inuit newborns by using a mathematical model to estimate an infant's exposure to organochlorines from breast milk. The study compared the estimated infant's organochlorine

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Breastfeeding infant

exposure with exposure to these chemicals in laboratory animals that is known to cause adverse effects such as cancer. The study estimated that exposure to organochlorines for an Inuit newborn was less than exposures in animals that caused various adverse health effects (Ayotte *et al.* 1994).

A study on Inuit babies has been carried out to investigate whether exposure to organochlorines via breast milk is associated with effects on the immune system (Dewailly *et al.* 1993b). The study showed that a higher level of PCBs in milk was

associated with changes in certain cells of the immune system. Similarly, a study on healthy women and their babies from the general population of the Netherlands found that exposure to higher levels of PCBs in the womb and via breast milk was related to changes in cells of the immune system. The level of PCBs in Inuit milk was twice that of the Dutch women. The implication of changes in immune system cells is unknown (Weisglas-Kuperus *et al.* 1995, Weisglas-Kuperus 1998).

The study on Inuit babies also showed that infants who were exposed to higher levels of PCBs and dioxins from breast milk had more episodes of ear infections (acute otitis media) than those exposed to lower levels. It is possible that an increased disposition to this infection is due in part to lack of maturity of the child's developing immune system (Dewailly *et al.* 1993b). Indeed, a study in the Netherlands on healthy mothers and their babies from the general population also found that there was an increased incidence of ear infections in children at the age of 3 and a half years who were exposed to higher PCB levels. It is possible that an increased incidence of ear infections in Inuit babies and in Netherlands children may not be due to an effect on the immune system. Rather it could be caused by a direct effect of PCBs on the cells lining the Eustachian tube of the ear. PCBs may preferentially accumulate in these cells and ultimately result in increased infections (Patandin 1999).

A study was undertaken to assess the growth of Inuit newborns (Dewailly *et al.* 1993c). The study reported that higher exposure to organochlorines through breast milk was associated with slightly reduced height in males, but not in females. A reduced height due to PCB exposure is consistent with previous studies in both human and animals. For instance, children born to mothers who were highly exposed to PCBs in a rice-poisoning incident in Taiwan were shorter than unexposed children (see Dewailly *et al.* 1993c). In addition, a study on healthy women and their babies from the general population of the Netherlands found that exposure to higher levels of PCBs in the womb predicted a lower growth rate in babies up to 3 months of age (Patandin *et al.* 1998).

8

THE SOLUTION: ADDRESSING THE PROBLEM AT SOURCE

It is abundantly clear from the preceding review that the manufacture, use and disposal of synthetic chemicals has resulted, and continues to result, in widespread environmental contamination. Although the highest levels of contamination may be recorded immediately adjacent to sources of pollution, some contaminants have reached areas of the globe, including the open ocean and the Arctic, which are remote from industrial, agricultural and urban development. Of particularly concern in this regard are those man-made chemicals which are environmentally persistent, i.e. POPs, many of which are also highly toxic and have the ability to accumulate to high levels in biological tissues. Carriage in the atmosphere, on ocean currents and in migratory wildlife continues to deliver these undesirable chemicals and their consequences to otherwise remote and pristine areas.

8.1 Global Exposure

We are all exposed to these chemicals. In some cases this results from occupational exposure or the use of products containing POPs in the home. In all cases, however, exposure also occurs as a result of the widespread distribution of such contaminants in the environment, particularly in our food and, as a consequence of the properties of these chemicals, the long-term persistence of POPs in our body tissues. Much of this exposure goes unnoticed, and is currently unavoidable given that contamination is so widespread and, moreover, that we continue to manufacture, use and release to the environment a range of persistent organic chemicals.

Although exposure is ubiquitous, some of the highest exposures of humans to POPs occur, remarkably, in populations observing traditional lifestyles in remote regions of the Arctic. This is by virtue of the high bioaccumulation potential for POPs in Arctic marine foodwebs combined with the high proportion of marine mammal tissue consumed in traditional diets. The scale and severity of the problem of widespread POPs contamination of the Arctic is crystallised by DeMarche *et al.* (1998) in summarising their extensive review, prepared under the AMAP programme:-

"All POPs considered under the AMAP monitoring programme have been found in air, snow, water, sediments and/or biota in the Arctic. In some cases, a number of Arctic species have POP levels high enough to cause effects".

Clearly, further research and monitoring of levels, trends and impacts will be essential if we are to improve our understanding of the long-term fate and effects of chemicals in the environment. Nevertheless, as stressed by DeMarche *et al.* (1998), there is an urgent need, and a universal responsibility, also *"to continue to promote measures to reduce levels of POPs in the environment*".

8.2 Lessons Unlearned?

History should have taught us a number of fundamental lessons:

- We have generally been very poor at predicting threats of chemicals to wildlife and human health, and even worse at taking decisive measures in order to avoid or reverse them. We have often seen the benefits of chemicals long before we have realised the disbenefits. Moreover, even in some cases where those disbenefits have been well documented (for example, with dioxins or brominated flame retardants), there remains a tremendous reluctance to address the problem effectively.

- Once released to the environment, chemicals become practically impossible to manage. Avoidance of exposure becomes an impossibility. We can only then attempt to reduce exposure to POPs through, for example, changes in diet or lifestyle. For most human populations, enforced or advisory changes in diet would be grossly unpopular and difficult to implement. For Arctic communities consuming traditional foodstuffs, such initiatives would challenge an entire way of life. For wildlife, of course, options to reduce exposure simply do not exist.

- Our historic and continuing mismanagement of chemicals, and failure to address the problems at source has, with little doubt, already led to consequences for environmental health which will remain as our legacy for generations to come.

In spite of this, it seems inevitable that we will continue to make the same mistakes. Current market shifts towards the substitution of one group of hazardous brominated flame retardants (the polybrominated diphenyl ethers, PBDEs) with another (tetrabromo-bisphenol-A) tends to suggest that we have yet to learn from these lessons.

The POPs which we have long recognised to be a problem (e.g. PCBs, DDT) will not be addressed simply by such recognition. Nor will a ban on the production and use of these chemicals in some regions resolve the totality of the problem. For example, PCBs and other banned chemicals will continue to enter the environment from stockpiles and waste dumps, as well as through continued unregulated use and generation as byproducts, as long as we do not address these sources on a global basis. We must also recognise that many POPs, particularly the dioxins and furans but also PCBs and others, are still generated as unintentional byproducts of some of the manufacturing and disposal practices which we continue to pursue

Moreover, with the POPs that we continue to develop and produce, we must recognise that the history of widespread environmental contamination is already repeating itself, and must act now to avoid further worsening the emerging problem. These substances do not only enter the environment in discharges from manufacturing plants. Indeed, in many cases, diffuse losses from products in which hazardous chemicals are used as ingredients or additives (e.g. brominated flame retardants, phthalates, synthetic musks), both during use and/or after disposal, may account for by far the greatest overall releases of such chemicals, leading directly to widespread contamination and exposure.

8.3 The need for global solutions

For all hazardous substances, what is required is effective action to address the problems at source. In its report on the state of the Arctic environment prepared in 1997, AMAP recognised this need in its concluding statement:

BOX 8.1 UNEP POPs Convention and UNECE LRTAP POPs Protocol

Draft UNEP POPs Convention

In 1995, the Governing Council of UNEP agreed upon a list of chemicals for which urgent measures were required (UNEP 1995). Of these chemicals (tabulated below), all are organohalogens and the majority are pesticides. Although still in the drafting stages, the intention of the Convention is that the pesticides and industrial chemicals on the list should be phased out and replaced with alternatives on a global scale and that emissions of those generated as unwanted by-products (e.g. dioxins and furans) be reduced or eliminated at source. Finalising the text of the Convention will require sustained negotiations. Nevertheless, the declared intention to ban the production and use of some compounds globally is a significant step in the regulation of persistent organic pollutants. As it stands, however, the Convention will permit some continued production and use of many of these chemicals for an unspecified transition period, during which alternatives will be identified or developed (UNEP 1997).

The 12 UNEP POPs

PCBs	DDT
Dioxins	Chlordane
Furans	Hexachlorobenzene
Aldrin	Mirex
Dieldrin	Toxaphene
Endrin	Heptachlor

UNECE POPs Protocol

A Protocol to address POPs on an international basis has already been adopted under the Convention on Long-Range Transboundary Air Pollution (LRTAP), a Convention of the United Nations Economic Council for Europe (UNECE 1998) which also covers North America. The Protocol, adopted under LRTAP in June 1998, expresses the ultimate aim to control, reduce or eliminate discharges of POPs, listing 16 substances or groups of substances for which measures are required. The focus is on compounds which are not only toxic, persistent and bioaccumulative but which also have a high capacity for transport across national boundaries on atmospheric currents.

Understandably there is a high degree of overlap with the list of chemicals under consideration by UNEP. In some cases, such as for the pesticides chlordane, dieldrin, mirex and toxaphene, the Convention demands an immediate ban on production and use. For other compounds listed, the requirements are for severe restrictions on uses and/or emissions.

"The long-term reduction of exposure to persistent organic pollutants can only be accomplished through international conventions on bans and restrictions in production and use of these substances". AMAP (1997)

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Environment Ministers from EU countries at OSPAR conference

To some degree, this statement reflects the already widely recognised need for, and ongoing development of, regional and, in particular, global initiatives to address certain POPs at source. For example, the ongoing development of regional and global mechanisms to address POPs, under the auspices of the United Nations Environment Programme (UNEP) and Economic Council for Europe (UNECE), mark significant steps forward in terms of the recognition of the scale of the problem and the nature of the solutions required (see Box 8.1).

Neither the UNEP POPs Convention nor the UNECE POPs Protocol are likely to enter into force for many years to come. For example, only one country (Canada) has so far ratified the UNECE Protocol; many more ratifications will be required prior to entry into force. The finalisation and implementation of these Conventions will involve complex and difficult negotiations, even for those chemicals already listed.

Moreover, the development of these Conventions is not a solution in itself. While they represent enormous steps forward in terms of recognising the need for international and global efforts, these processes will address only a subset of the most problematic chemicals and, even then, may not lead to the total phase out of these chemicals. Even if effective measures can be introduced for all of the 12 UNEP POPs and/or 16 UNECE POPs, this will in no way solve the totality of the problem of environmental contamination with persistent organic chemicals. Ultimately it will be essential to extend the provisions of these Treaties to cover other groups of substances which possess similar properties, a large proportion of which are still in widespread production and use, and to ensure that the measures introduced lead as rapidly as possible to the elimination of these chemicals as environmental contaminants. It is also necessary to substitute those processes which unintentionally generate chemicals such as dioxins, furans and, indeed, PCBs, even if this implies a major shift in the chemical industry.

8.4 Progress Under Regional Programmes

At the same time, however, there is a recognition at a number of more regional levels (e.g. North Sea, North East Atlantic, Mediterranean) that measures to address widespread chemical contamination can, and must, go further than may currently be possible within global initiatives, both in terms of the range of substances which must be addressed and the extent and urgency of controls which are required. Perhaps the best example of such a regional process is the development of the OSPAR Convention and its related strategies and measures, although similar provisions have been adopted within other regional seas programmes, for example, the North Sea Conference process and associated Ministerial Declarations (MINDEC 1995).

8.4.1 The OSPAR Convention (1992) and the "One Generation Goal"

The Convention for the Protection of the Marine Environment of the North East Atlantic (the OSPAR Convention, formed from the amalgamation of the former Oslo and Paris Conventions) entered into force in March 1998 and covers the 15 States of the North East Atlantic Region and the European Union.

The Convention (OSPAR 1992) requires that all Contracting Parties:-

"...take all possible steps to prevent and eliminate pollution and shall take the necessary measures to protect the maritime area against the adverse effects of human activities..."

At the OSPAR meeting held in Sintra in June 1998, Ministers from each of the Contracting States and a representative of the European Commission agreed to a common statement, the Sintra Statement (OSPAR 1998a), which sets out clear commitments to address hazardous and radioactive substances, eutrophication, the offshore oil and gas industry and the overall protection of biological diversity. With respect to hazardous substances, the Ministers agreed:

"to prevent pollution of the maritime area by continuously reducing discharges, emissions and losses of hazardous substances (that is, substances which are toxic, persistent and liable to bioaccumulate or which give rise to an equivalent level of concern), with the ultimate aim of achieving concentrations in the environment near background values for naturally occurring substances and close to zero for man-made synthetic substances".

Note that these measures are intended to apply to a much wider range of hazardous substances than those currently under consideration within the developing POPs processes outlined above. Furthermore, the Ministers agreed to make:

"every endeavour to move towards the target of cessation of discharges, emissions and losses of hazardous substances by the year 2020".

This latter commitment, commonly referred to as the "one generation goal", is perhaps the most important element of the OSPAR approach. It represents a substantial step forward in relation to the way in which chemical substances are regulated and a fundamental departure from the more conventional (though misguided) wisdom that chemicals can be "managed" at "safe" levels in the environment. Inevitably, meeting the target of cessation of discharges, emissions and losses will necessitate the phase out of hazardous substances (or the processes which generate them) and their substitution with non-hazardous alternatives. Further discussion of this is presented in Box 8.2.

8.4.2 Hazardous Substances for Priority Action

Although the details of the implementation of the OSPAR Convention and the Hazardous Substances Strategy remain under development, the commitment to a progressive and precautionary approach to the prevention of environmental contamination is clear. As a first step towards implementation, OSPAR also agreed in 1998 on a "List of Chemicals for Priority Action" (the Priority List), a list of 15 chemicals or chemical groups of particular concern (Table 8.1). The list includes dioxins, furans and PCBs which are widely recognised to present problems and for which further measures to prevent releases to the environment and address stockpiles are urgently required. Also included are a number of chemicals or chemical groups which are still manufactured and marketed as chemical products in the OSPAR region and throughout the world, including chemicals such as brominated flame retardants, phthalates and synthetic musks which are widely used in consumer products. The urgency of measures to address these diverse chemicals can be illustrated by consideration of one such group, the brominated flame retardants (Box 8.3).

In addition to the general commitment to address all hazardous chemicals by 2020, Ministers at Sintra agreed to a specific timetable to address these priority substances, i.e.:

"the drawing up of programmes and measures by 2003 for the control of discharges, emissions and losses of substances on [the Priority] list, and their substitution with less hazardous or non-hazardous substances where feasible;" (OSPAR 1998a)

Furthermore, mechanisms are currently under development in order to identify other hazardous chemicals and thereby update the Priority List.

BOX 8.2 Implementing OSPAR: The Precautionary Principle and Principle of Substitution

The target of cessation of discharges, emissions and losses has developed from a recognition that, in order to achieve the specific aim of the protection of the marine environment, it will be necessary to avoid the potential for substances that are intrinsically hazardous to reach the marine environment. By adopting an approach which necessitates the prevention of chemical releases at source, even where concerns are related only to the potential for such substances to cause adverse impacts in the marine environment, the Convention recognises the central importance of the Precautionary Principle in ensuring environmental protection.

"Effective action is to be taken when there are reasonable grounds for concern that hazardous substances introduced into the marine environment, or which reach or could reach the marine environment, may bring about hazards to human health, harm living and marine ecosystems, damage amenities or interfere with other legitimate uses of the sea, even when there is no conclusive evidence of a causal relationship between the inputs and effects". (OSPAR 1998b)

In other words, it is enough for a hazardous chemical to have the potential to reach the marine environment, by whatever route, for action to be required. It is not necessary to demonstrate that marine or other organisms will be exposed to a certain concentration of that chemical, nor to demonstrate that inputs of that chemical are causing adverse effects. Underlying this precautionary approach is the recognition that the fate and effects of chemicals in the marine environment are extremely difficult to predict, a truism which is clear from the preceding review. Moreover, once they have entered the marine environment these substances cannot be recalled. Those which are persistent and, therefore, capable of wide dispersion, may contaminate water, sediments and wildlife for decades and have the potential to exert effects which are, in practical terms, irreversible. It is, therefore, the hazardous properties of substances and their potential to reach the marine environment which are considered undesirable.

The implementation of the OSPAR commitments is elaborated further in the Strategy with Regard to Hazardous Substances (OSPAR 1998b). Central to this strategy is the principle of substitution, that is the necessity for:

"...the substitution of hazardous chemicals with less hazardous substances or preferably non-hazardous substances where such alternatives are available...".

Furthermore, except through the implementation of the principle of substitution,:

"emissions, discharges and losses of new hazardous substances shall be avoided".

In other words, the development and manufacture of new chemicals which are intrinsically hazardous and for which the potential exists to reach the marine environment by any route will no longer be permitted, except where such chemicals are produced as interim substitutes for substances which are more hazardous. This should effectively "close the door" on the production of yet more hazardous chemicals such that attention can be focussed on addressing the problem with which we are already faced.

There is a recognition that the mechanism must be capable of addressing substances which, while not meeting all the criteria of toxicity, persistence and bioaccumulation, give rise to equivalent concern with respect to the potential for effects in the marine environment, with particular regard to endocrine disrupting chemicals.

8.5 From Objectives to Action: the Challenge of OSPAR Implementation

Of course, the objectives and goals set out above will only be achieved if the programmes and measures developed to address hazardous substances are both effective and consistent with the OSPAR Strategy. Cessation of the discharges, emissions

and losses of the chemicals on the existing Priority List must be the first, and most urgent, challenge for OSPAR, and a necessary first step towards meeting the "one generation goal". In addition to prioritising further substances for action, OSPAR also recognised the need to ensure that no new hazardous substances are introduced and added to the existing problem (see Box 8.2).

It is recognised that the implementation of such measures will require substantial changes in both national and international legislation with respect to chemicals and environmental protection. Sweden is the first (and, so far, only) OSPAR country to adopt a bill under its national legislation which will implement the "one generation goal". The other OSPAR states must follow Sweden's lead.

The European Union, in the form of the Parliament and the Commission, will inevitably play a substantial role in the practical implementation of the OSPAR Convention. The prospect of this raises some fundamental problems, not least of which is the fact that the current system of chemicals regulation within Europe has developed from an entirely different perspective, and with fundamentally different goals, than the OSPAR Convention.

8.5.1 Regulation of Chemicals in the European Union

The manufacture, marketing and use of chemicals within the European Union is currently regulated under a complex system of Directives and Regulations which have, as their fundamental basis, the objective of maintaining the free circulation of chemical products for economic purposes. Furthermore, this legislation is based on the assessment and management of the "risks" presented by chemicals. In simple terms, risk is determined as a product of the hazardous properties of a chemical and an estimation of the degree to which different environmental compartments, including humans, are exposed to that chemical. In other words, even if a chemical is known to possess highly hazardous properties, it is possible that the overall assessment will conclude that the risks are acceptable as exposure will only occur at levels below those at which adverse effects might be predicted, or that a causal relationship between dose and effects has yet to be firmly established. In such cases, despite the hazardous nature of the chemical, it is likely that no further regulation would be required. This represents a fundamental departure from the more precautionary, hazard-based approach required by OSPAR. The limitations of such an approach are summarised in Box 8.4.

Table 8.1 the OSPAR List of Chemicals for Priority Action, hazardous chemicals which have been prioritised for urgent action to address discharges, emissions and losses by 2003.

The OSPAR List of Substances for Priority Action
Polychlorinated dibenzodioxins (PCDDs)
Polychlorinated dibenzofurans (PCDFs)
Polychlorinated biphenyls (PCBs)
Polyaromatic hydrocarbons (PAHs)
Pentachlorophenol
Short chain chlorinated paraffins
Hexachlorocyclohexane isomers
Mercury and organic mercury compounds
Cadmium
Lead and organic lead compounds
Organic tin compounds
Nonylphenol ethoxylates and related substances
Musk xylene
Brominated flame retardants
Certain phthalates –dibutylphthalate and diethylhexylphthalate

8.5.2 Failure of existing EU Chemical Regulations

In addition to the fundamental limitations to risk-based regulation outlined in Box 8.4, the existing system of chemical regulation in Europe is widely seen to have failed to deliver timely and effective measures for other institutional reasons. This is particularly true with respect to the so-called "existing chemicals", i.e. more than 100,000 chemicals which were listed as being marketed or used in Europe prior to 1982. Since the adoption of Regulation 793/93 on the evaluation and control of the risks of existing substances, only 38 of the 110 substances prioritised as particularly hazardous have been, or are undergoing discussion. Risk

assessment reports have been completed for only 19. Of these, Commission Recommendations concerning the results of the risk evaluation are available for only 4 (EC 1998). Moreover, even for those 4, not a single risk reduction measure has yet been formally agreed. Even if such measures are ever proposed, other institutional barriers are likely to delay implementation still further. The limitations of the current regulatory system have been elegantly summarised by the European Environment Agency (EEA 1998).

Indeed, the failings of the existing mechanism have, to some extent, also been recognised by the European Union itself, prompting the ongoing review of key instruments of EU chemicals legislation (EC 1998). This should give the opportunity for the Commission to ensure that the main elements of the OSPAR Convention (to which the EU is signatory), particularly the "one generation goal", are incorporated into the revised approach such that the Ministerial commitments can be met. On the contrary, however, it appears that the revisions currently proposed by the Commission will amount to little more than a "streamlining" of the existing approach, such that the underlying and fundamental weaknesses will not be addressed (EC 1998).

8.6 The Way Forward

In order that the OSPAR objectives can be achieved, a more fundamental overhaul of EU chemicals policy and legislation will be essential. It is essential also to ensure that the target of cessation of discharges, emissions and losses of all hazardous substances by 2020 is incorporated into the EU Water Framework Directive (currently in draft form, EC 1999) which aims to protect all surface and

BOX 8.3 The Brominated Flame Retardants

The need for such measures is well illustrated using the example of the brominated flame retardants. The toxicity, persistence and ability to bioaccumulate of this group of substances has been recognised for some time (Kamrin and Fischer 1991, Jansson *et al.* 1993, Kholkute *et al.* 1994), although understanding of the diversity and severity of effects continues to emerge (Hornung *et al.* 1996, Kang *et al.* 1996, Eriksson *et al.* 1998). Some are suspected endocrine disruptors (Olsson *et al.* 1998, Meerts *et al.* 1998). It is also clear that several brominated flame retardants, particularly the PBDEs, have become widespread contaminants of the urban and natural environment, being detectable even in marine mammals from remote areas (de Boer *et al.* 1998, Lindstrom *et al.* 1999) as well as more generally in marine and freshwater fish (Sellstrom *et al.* 1998, Allchin *et al.* 1999) and in human blood and breast milk (Wehler *et al.* 1997). Release to the environment occurs not only during the manufacture of these chemicals but also during the routine use and final disposal of a wide range of household and other consumer products (e.g. computer components, other electrical goods) in which they are included as additives. This sort of widespread, or diffuse, loss of hazardous chemicals can only be addressed effectively at source.

In recognition of the fundamental problems posed by these chemicals, and the availability of alternatives for most applications, The Swedish National Chemicals Inspectorate (KEMI 1999) has recently proposed a national ban on the manufacture or use of these chemicals. Moreover, the World Health Organisation has proposed their substitution with less hazardous alternatives wherever these are available (WHO 1998).

In spite of these concerns, manufacture of certain brominated flame retardants continues to grow on a global basis. Groups such as the polybrominated biphenyls (PBBs) and polybromiated diphenyl ethers (PBDEs) are increasingly being replaced by tetrabromo-bisphenol-A, despite evidence that the latter may exhibit hazards as severe as those of those it replaces. This trend must be halted.

Undoubtedly there are already manufacturers of consumer goods which have succeeded in substituting these chemicals without compromising the fire safety of their products. The difficulty for the consumer is in identifying which are the progressive manufacturers because of the lack of information on the presence of hazardous constituents in products. This situation must be reversed and a ban on the manufacture, marketing and use of brominated flame retardants imposed throughout the OSPAR region. Ultimately, of course, their substitution will need to be tackled on a global basis. It is important to note that, despite the growing concerns over the effects and widespread distribution of these chemicals, they are not currently included within either the UNEP or UNECE POPs processes.

groundwater resources of the European Community. Only then will the Precautionary Principle (and the principle of preventative action), which is already enshrined as a guiding principle within the Treaty establishing the European Community (the Treaty of Amsterdam, EC 1993), begin to be implemented. An alternative approach to the regulation of the manufacture, marketing and use of chemicals in Europe, based on the precautionary principle and aimed at implementing fully the OSPAR commitments, is outlined in more detail by Santillo *et al.* (1999). More specific measures which may be required in order to address the particular problem of endocrine disrupting chemicals are proposed by Santillo *et al.* (1998b).

Although OSPAR is aimed at the protection of the maritime area, meeting the objectives, in particular the "one generation goal", will in turn ensure an overall reduction in the chemical burden on wildlife and society. In the case of persistent chemicals and the contamination of remote environments, the benefits may not be felt for some time to come. Nevertheless, effective action taken now will begin to reverse the problem such that future generations may be liberated from our toxic inheritance and may avoid new toxic problems in the future.

8.6.1 A New Approach to the Regulation of Hazardous Chemicals

We must take urgent action NOW if we are to prevent the existing problem of global contamination with persistent organic pollutants (POPs) and other hazardous chemicals from getting still worse and to start on the road to a Toxics Free Future. To do so, we must:

- recognise that we cannot solve the problems caused by chemicals once they are released to the environment, and that the only effective way to address the problem is to take action to prevent release of these chemicals at source.

- take steps to ensure, for ALL hazardous chemicals, that discharges from pipelines, emissions to atmosphere and losses from manufacturing processes, products and disposal operations may be continuously reduced and eliminated before 2020. In order to do so, it will be essential that the "one generation goal" adopted by OSPAR is incorporated and implemented within national and international legislation;

- as a first step, take urgent action on the Priority List of chemicals identified by OSPAR, of which the majority are POPs. In order to achieve the cessation of their discharges, emissions and losses it will be necessary to phase out the production (or unintentional generation) of these chemicals through their substitution (or the processes which generate them) with less hazardous or preferably non-hazardous alternatives (see Box 8.5);

BOX 8.4 Limitations to the Assessment and Management of Risks

Risk-based approaches extend from the view that environmental risks of chemicals can be quantified and managed at levels which are both sustainable and "acceptable", either in absolute terms or relative to the benefits which are derived from the production and/or use of that chemical. Such approaches assume that it is possible to know enough about hazard and exposure to assign a reliable, quantitative risk to a particular activity.

In practice, risk assessments tend to employ necessarily simplistic and subjective representations of environmental processes and frequently lack the breadth and quality of data necessary to facilitate forecasting of impacts. For example, there remain enormous gaps in our understanding of the identity, let alone the individual or combined toxicity, of chemicals released to the marine environment. This is a substantial problem, as organisms are invariably exposed to complex mixtures of chemicals (Johnston *et al.* 1996b). It is also extremely difficult, if not impossible, to determine the actual doses involved in environmental exposures and the most appropriate and sensitive ways of detecting impacts. Uncertainties and indeterminacies arising from ecosystem complexity are rarely made explicit, although they may fundamentally compromise the accuracy of the assessments and, consequently, the degree of protection provided by any risk management measures. In addition, acceptability of risk is largely a subjective issue and one which is likely to depend in part on the degree of inequity between those bearing the risks and those deriving the benefits. It also depends on the past experience and interests of those involved in the assessment and decision making processes. These issues are discussed in more detail by Johnston *et al.* (1996c, 1998b) and by Santillo *et al* (1998a).

BOX 8.5 Action Required for Specific Substances on the OSPAR Priority List

For those hazardous chemicals which currently remain in widespread production and use, the target must be their rapid substitution with less hazardous or, preferably, non-hazardous alternatives. Action is particularly urgent in the cases of those chemicals used in dispersive applications, such as the brominated flame retardants, phthalates and synthetic musks used as additives in consumer products and organotins used in antifouling paints on ships. In some cases (e.g. brominated flame retardants), less hazardous alternative are already available for most, if not all, applications. In other cases (e.g. phthalates used as plasticisers in soft PVC products) alternative additives may simply be unnecessary as the problem may be addressed through more fundamental material, or even product, substitution.

For dioxins and furans, it will be necessary to develop a "cradle-to-grave" dioxin elimination strategy in order to eliminate those anthropogenic sources which are already identified (including incineration of hazardous and municipal wastes, manufacture and disposal of PVC, secondary smelting of metals contaminated with chlorinated chemicals) and to identify and address further sources;

For chemicals such as PCBs and PCP, for which manufacture in the OSPAR region at least has long been banned, the prevention of further releases to the environment will only be achieved through measures to identify and address "hotspots" of existing contamination (including obsolete equipment or materials containing these substances, either still in use or in waste-dumps). Treatment of such contaminated materials must be conducted in a manner which ensures complete destruction of the chemicals without the generation of hazardous waste streams;

- take into account the role served by chemicals in society when evaluating their necessity, such that the potential for non-chemical solutions can be properly evaluated;

- ensure that chemicals for which the hazards are not known are no longer produced and that no new hazardous substances are added to the existing problem;

- ensure that, in the interim, products which contain hazardous chemicals as additives are clearly and comprehensibly labelled, such that consumers have the choice to avoid such products should they wish. The importance of this was also recognised by the Ministers in Sintra (OSPAR 1998a).

- further the development of clean production technologies which are more efficient in terms of material and energy usage, produce cleaner products with less overall, and less hazardous, wastes and, ultimately, which operate in "closed loop" configurations to serve the needs of society in a more equitable and sustainable manner;

- implement fully the Precautionary Principle, such that in future we may be better able to avoid problems before they occur. The continuation and further development of scientific research has a fundamental role to play in identification of potential problems and solutions, but we must be ready to take effective precautionary action to prevent environmental contamination and degradation even in the face of considerable and often irreducible uncertainties.

This new approach will entail acceptance of much greater responsibility for the chemicals we, as society, produce and use. The bulk of this responsibility must, ultimately, be borne by those who profit from the manufacture and/or marketing of chemicals or the processes which generate them. In general terms, we must aim at continuous reduction of the overall chemical burden borne by society and the wider environment. The development of regional and global Conventions to address persistent organic pollutants (POPs) and, in particular, the commitment by North East Atlantic states under the OSPAR Convention to cease discharges, emissions and losses of all hazardous chemicals within one generation, has demonstrated an ability to change our way of thinking. Nevertheless, if we are to solve the problems we face, we must also be ready to change the way we act. To do otherwise would be to ignore our transnational and transgenerational responsibilities to ensure the sustainability of the bases for productivity and the diversity of life itself, not only in industrialised regions, but also in remote regions such as the Arctic and deep oceans.

9

CONCLUSIONS

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Polar bears

Over 100,000 chemicals are currently in use worldwide and the number and amount of chemicals used is still increasing. Many of these are hazardous chemicals. Of particular concern are the persistent organic pollutants or POPs. Despite the extensive global production and use of POPs, our current understanding of the toxic effects of POPs on wildlife and humans is limited to only a handful of chemicals. The POPs that we have extensive information on, like PCBs and DDT, are known to cause serious negative impacts on wildlife. There is also evidence to suggest that human health in different parts of the world is influenced by exposure to these well-known POPs. About other POPs, like brominated flameretardants or TBT, we have less information when it comes to the effects on wildlife and humans. However, we do know that these chemicals have similar properties when it comes to their persistence in the environment, their potential to accumulate in organisms and their inherent toxicity. On top of that, there are even more poorly described chemicals of which nobody knows the possible effects. In essence therefore, our present state of knowledge on exposure to POPs and their health effects in wildlife and humans may only be the tip of the iceberg. This is an alarming conclusion, but one that must, nevertheless, be drawn. Future research may reveal many more toxic effects caused by POPs contamination of the environment. However, because these chemicals are resistant to break down and can accumulate in the environment and organisms, the potential for negative health effects is impossible to avoid once the chemicals are released.

POPs are not only a problem in countries where they are produced and released into the environment. Some are also present in remote regions far removed from their source due to long-distance transport on air currents and ocean currents. Therefore, POPs are a global problem. Research in Europe and the Arctic shows that POPs contaminate the physical environment, wildlife and humans. Some of the most extensively studied POPs (e.g. PCBs, DDT, dioxins) are suspected to be responsible for many different effects on wildlife including effects on reproduction and the immune system and as contributing factors in the dramatic declines in populations of some marine mammals. In humans, studies suggest that current background levels are sufficient to cause subtle undesirable effects on the foetus in the womb and on breast-fed infants.

The release of POPs into the environment is ongoing. Only a few POPs, of which the devastating effects have been known for a long time, have been banned in some countries, and even then production continues in other countries. Furthermore, the persistent nature of POPs means that even if production ceased today, they will remain as contaminants in the environment for years to come. Even more striking is the fact that the production of chemicals like brominated flame retardants and other POPs continues, in some cases increasing globally, although it is clear that they possess similar hazardous properties to the PCBs and chlorinated pesticides which have been the focus of most attention to date.

The only solution to the problem is to phase out the production of all POPs, and ultimately all hazardous chemicals, globally. The substitution of one problematic chemical with another is no improvement. The implementation of clean production technologies which do not depend on hazardous chemicals is the way forward.

To achieve this will require acceptance of much greater responsibility for the chemicals we, as society, produce and use. The bulk of this responsibility must, ultimately, be borne by those who profit from the manufacture and/or marketing of chemi-

cals or the processes which generate them. In addition, ongoing and further development of regional and global Conventions, and the transposition of these commitments into national law, will be necessary to implement and enforce legislation to phase out POPs and hazardous chemicals.

The way forward is illustrated by the following Greenpeace demands on the issue.

GREENPEACE DEMANDS...

- The production and use of all POPs must be phased out at an international and, ultimately, at a global level.

- This must be achieved through the substitution of POPs (or the processes which generate them) with non-hazardous alternatives.

- Industry and agriculture must pursue clean production technologies and manufacture clean products, recognising that the only way to prevent releases of POPs into the environment is to avoid their production and use.

- As a matter of urgency, action must be taken to stop production, eliminate all discharges, emissions and losses of those chemicals prioritised for action by OSPAR/UNEP, many of which are POPs. This is an essential first step if the target of cessation of emissions, discharges and losses of ALL hazardous substances in OSPAR countries as agreed at ministerial level, is to be achieved within one generation (by 2020).

- Presume that all chemicals are hazardous until demonstrated otherwise, ie. until hazard identification is completed, or in those instances where hazard identification is limited by lack of information, chemicals must be assumed to present hazards of unknown proportions.

- Ultimately, measures to eliminate releases of POPs and other hazardous substances to the environment will need to be taken not just on a regional but on a global basis, because chemical contamination of the environment is a global problem and chemicals do not respect national boundaries.

- OSPAR countries therefore should not only give highest priority to implement the one generation elimination goal of all hazardous substances in Europe, but also aim at this elimination goal at the global level through the UNEP convention for the elimination of POPs.

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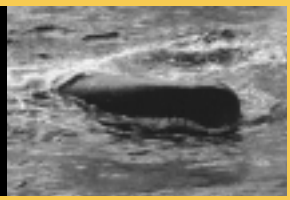

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List of abbreviations

α-HCH	alpha-hexachlorocyclohexane, HCH isomer, component of technical HCH pesticide formulations
β-HCH	beta-hexachlorocyclohexane , HCH isomer, component of technical HCH pesticide formulations
ΣDDT	Total quantity or concentration of DDT and it's breakdown products DDD and DDE
γ-HCH	gamma-hexachlorocyclohexane, HCH-isomer used alone as a pesticide (lindane) and a component of technical HCH pesticide formulations
µg	Microgramme (10-6 grammes)
ADI	Acceptable Daily Intake
AMAP	Arctic Monitoring and Assessment Programme
DBP	Dibutyl phthalate
DBT	Dibutyltin, breakdown product of tributyltin
DDD	Dichlorodiphenyldichloroethane, break down product of DDT
DDE	Dichlorodiphenyldichloroethylene, breakdown product of DDT
DDT	Dichlorodiphenyltrichloroethane, pesticide
DEHP	Diethylhexyl phthalate, sometimes called dioctyl phthalate (DOP)
DEW	Distant Early Warning
dw	Dry weight
EPA	Environment Protection Agency
EU	European Union
HCB	Hexachlorobenzene
HCH	Hexachlorocyclohexane (see above for individual isomers)
HE	Heptachlor epoxide
kg	Kilogramme (103 grammes)
l.w.	Lipid weight
LRTAP	Long-Range Transboundary Air Pollution
MBT	Monobutyltin, breakdown product of tributyltin
mg	Milligramme (10-3 grammes)
nd	Not detected
ng	Nanogramme (10-9 grammes)
NWT	North West Territories
OECD	Organisation for Economic Co-operation and Development
OSPAR	Intergovernmental organisation which oversees the implementation of the 1992 OSPAR Convention (Convention for the Protection of the Marine Environment of the North East Atlantic)
PAHs	Polyaromatic (polynuclear) hydro carbons
PBBs	Polybrominated biphenyls
PBDEs	Polybrominated diphenyl ethers (also called polybrominated diphenyl oxides)
PCBs	Polychlorinated biphenyls
PCDDs	Polychlorinated dibenzo-p-dioxins
PCDFs	Polychlorinated dibenzofurans
PeCDD	Pentachloro dibenzo-p-dioxin
PCP	Pentachlorophenol
pg	Picogramme (10-12 grammes)
POPs	Persistent Organic Pollutants
ppb	Parts per billion
ppm	Parts per million
ppt	Parts per trillion
PVC	Polyvinyl chloride
TBT	Tributyltin
TCDD	Tetrachlorodibenzodioxin
TCDF	Tetrachlorodibenzofuran
TDI	Tolerable Daily Intake
TeBDE	Tetrabromodiphenylether
TEQ	Toxicity equivalent
UK	United Kingdom
UNECE	United Nations Economic Council for Europe
UNEP	United Nations Environmental Programme
US	United States of America
WHO	World Health Organisation
ww	Wet weight