Per and polyfluorinated substances in the Nordic Countries

Use, occurence and toxicology

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Content

Summary

The Nordic Chemicals Group (NKG), which is subordinate to the Nordic Council of Ministers, has commissioned the authors, through the Climate and Pollution Agency (KLIF), to undertake a Nordic study based on open information sources and custom market research to describe the use and occurrence of the most common perfluorinated substances (PFC), with less focus on PFOS and PFOA.

The study includes three stages:

1. Identification of relevant per-and polyfluorinated substances and their use in various industrial sectors in the Nordic market.
2. Occurrence in industrial and consumer products and potential emissions to and in the Nordic environment and humans of the substances described in stage 1.
3. A summary of knowledge of the toxic effects on humans and the environment of substances prioritized in stage 2.

Interviews were conducted with more than 50 players in the Nordic market with the aim of obtaining information on use and type of PFC substances. This study, however, gave poor results. In parallel with this survey a net list was therefore produced of PFC substances based on three lists (each separately and together incomplete) from the OECD, REACH pre-registration database, and the Nordic SPIN database. Most production of PFC containing articles is outside the EU and today's legal framework does not provide adequate means to obtain sufficient information about specific PFC substances in imported articles. This net list is therefore not complete so there may be significantly more PFC substances used in the Nordic market.

There are relatively few studies on PFC substances in the environment in the Nordic countries other than PFOA and PFOS which include both biotic (air, land and water) and abiotic (animal and human) data.

Most human data regarding PFCA and PFSA from the years 1992 to 2010 are from Norway and Sweden, with fewer from Denmark and no data from Iceland and Finland. Regarding PFCAs, most studies show the occurrence of PFOA, PFNA and PFHxA. However other PFCA substances (C10–C13) have also been detected in a number of studies. Regarding

PFSA, PFOS and PFHxS are the most studied substances. Human data are missing for PFAL, FTS, PAP/di-PAP and FTMAPs.

In comparison with long-chain PFC substances (≥ C8) the short-chain substances are considered to be less toxic but a number of studies indicate both ecotoxicity and human toxicity. In this area there is a major lack of studies.

In general, since 2002 decreasing levels of PFOA and PFOS are observed in the environment. However, increasing levels of short chained sulfonates have been observed in the environment. In comparison with other countries, the background concentrations of PFOA and PFOS in the environment are lower in the Scandinavian countries especially compared with Central European countries, which is to be expected as populations are smaller and there is less industry in the Nordic countries. However these substances have also been found in the Arctic, far from any sources, which shows that these substances are global contaminants.

One result of this review of the presence of fluorinated substances in the environment is that there are considerable information and knowledge gaps regarding PFCs other than PFOA and PFOS. In addition, there is generally a shortage of human and environmental data about these PFCs. The few data available indicate specific toxic effects on humans and the environment. It takes more and deeper studies to get a clearer picture of these PFC substances before far-reaching conclusions can be drawn about their toxic properties.

Lack of physical-chemical data for PFC substances other than PFOA and PFOS is an obstacle to environmental fate modelling calculations.

The lack of analytical reference substances is currently also a barrier to extended studies of these substances in the environment and humans.

1. Background

Polyfluorinated substances have been used for a long time, but there was no focus on this group until widespread environmental occurrence (e.g., in polar bears) and high reproductive toxicity were found for perfluorooctane sulfonate (PFOS). Because of these properties of the extremely persistent PFOS and by the fact that PFCs do not occur naturally in nature, the substance is restricted under the Stockholm Convention (nominated by Sweden), with only a few allowed remaining uses. Perfluorooctanoic acid (PFOA) was the second substance from this group to attract interest, with hazard and risk assessments being performed, and classification and labelling under discussion in the EU (proposal from Norway). PFOA is a candidate for restriction under Reach. The OECD (Organization for Economic Cooperation and Development) lists a total of 853 different fluorine compounds. Among these some are currently being phased out due to regulations mentioned above.

However, there is a huge number of polyfluorinated substances (including perfluorinated) being used, in many cases leading to substitution of one polyfluorinated substance with others, e.g., perfluorobutansulfonate (PFBS) substituting PFOS. Little is known about the sources of these substances. Many other perfluorinated substances are known to be used, but it is unclear to what extent they are included in monitoring/screening exercises.

Some widely used polyfluorinated substances such as fluorotelomer alcohol-derivatives are precursors to perfluorinated substances. Examples from these groups are polyfluorinated phosphates (diPAPs and PAPs), and fluorotelomer mercaptoalkyl phosphate diesters (FTMAPs), found in food contact materials by Danish scientists (Trier 2011). The polyfluorinated substances are rather persistent but may be degraded to perfluorinated substances, such as PFOA, which in itself is virtually non-degradable and may be problematic as such. In addition, sufficient toxicity data is only available for very few of them.

The overall publicly available knowledge on the use of per- and polyfluorinated substances is very limited, even though we know that there are many such substances on the market. This review aims to increase our knowledge of the uses of these fluorinated substitutes of PFOS/PFOA. This includes emissions and exposures in the Nordic envi-

ronment, and if available, more information on the toxicity and monitoring results of these substances. Of special concern is whether some of the perfluorinated substances already have contaminated the Arctic environment, with PFOS now being recognized as a global POP. Because of the potent surfactant properties of these substances, they are generally used at low concentrations in products and the use of them may therefore not always be clearly known. However, a better knowledge on how these substances are used will increase the possibilities to decrease the environmental emissions directly at the sources.

In conclusion, the aim of this study is to find more information on how per- and polyfluorinated substances are used in the Nordic society and to what extent they may be emitted to the Nordic and Arctic environment. These data will be useful in the process of regulating these substances within REACH or by other international forums like the Stockholm Convention.

2. Introduction

The Nordic Chemical Group (NKG), which is subordinate to the Nordic Council of Ministers, has commissioned the authors, through the Climate and Pollution Agency (KLIF), to undertake a survey that aims to present an overview of the most used PFCs in the Nordic countries besides PFOS/PFOA.

This survey contains three stages namely 1) Identification of relevant per- and polyfluorinated substances and their use in different applications on the Nordic market, 2) Potential emissions and exposure of substances in applications identified in stage 1 and, 3) A summary of knowledge on toxicity of the most important and prioritized substances in this survey.

Table 1. Focus categories of per- and polyfluorinated substances (PFC)

PFCA (Perfluoroalkyl carboxylates)
PFSA (Perfluoroalkyl sulfonates)
PFAL (Perfluoroalkyl aldehydes)
FTOH (Fluorotelomer alcohols)
FTS (Fluorotelomer sulfonates)
PAP/di-PAP (Polyfluoroalkyl phosphates)
PFPE (Perfluoropolyethers)
Other fluorotelomers

The substances in Table 1 were reviewed concerning their use, occurrence, environmental fate and impact along their life cycle in the Nordic countries (Finland, Sweden, Denmark, Iceland and Norway) including use, exposure and unintentional occurrence in industrial manufacturing and applications and other possible public and industrial sources such as long range transport by air.

3. Introduction to fluoro-chemistry

Polyfluoroalkylated substances (PFCs) belong to a large and complex group of organic substances that are extremely versatile and used in a variety of industrial and household applications.

The main characteristics of the polyfluorinated compounds are the replacement of most hydrogen by fluorine in the aliphatic chain structure. Some of these organic fluorine compounds are known as perfluorinated, which means that all hydrogens have been replaced with fluorine. PFCs are synthetically produced compounds which do not occur naturally, and have been manufactured for 50 years (Kissa, 2001).

An understanding of the chemistry of fluorinated surfactants must consider three distinct structural aspects, namely the hydrophobic/oleophobic "tail" that contains a high proportion of fluorine, the hydrophilic group, and the "spacer" organic group linking these two portions of the surfactant together. As with hydrocarbon surfactants, the important fluorinated surfactants include a diverse range of hydrophilic groups:

- Anionic (e.g. sulfonates, sulfates, carboxylates, and phosphates).
- Cationic (e.g. quaternary ammonium).
- Nonionic (e.g. polyethylene glycols, acrylamide oligomers).
- Amphoteric (e.g. betaines and sulfobetaines).

The practical and commercial range of the hydrophobic/oleophobic "tail" of the fluorinated surfactant is limited. Perfluoroalkyl ($F(CF_2)_n$– or RF-), or perfluoropolyether ($(RFO)_n(RFO)_m$-) groups are the hydrophobic/ oleophobic portion of most commercially available fluorinated surfactants. Perfluoroalkyl-containing fluorinated surfactants generally originate from either electrochemical fluorination (ECF) with hydrogen fluoride (HF) or telomerisation of tetrafluoroethylene (TFE). Perfluoropolyether-based fluorinated surfactants typically originate from either oligomerisation of hexafluoropropene oxide (HFPO), photooxidation of TFE or hexafluoropropene (HFP), or oligomerisation of fluorinated oxetanes.

3.1 Production of fluoro-chemicals

There are two main production processes for PFCs; electrochemical fluorination (ECF) and telomerisation. In the electrochemical fluorination process, a technical mixture of hydrocarbons (different carbon chain lengths including branched isomers) with a functional group is subjected to fluorination, leading to a mixture of perfluorinated products with the same homologue and isomer pattern. Telomerisation involves coupling tetrafluoroethene, which leads to straight-chained products with an even number of carbon atoms. Fluorotelomer products often possess two carbon atoms adjacent to the functional group which are not fluorinated that yields linear, even carbon number substances. Telomers are produced and used commercially as mixtures, in which the typical length of the chains is between four and eighteen carbon atoms. Fluoro-compounds can be further reacted and will then occur in other chemical compounds, e.g. acrylate polymers. This means that perfluorinated compounds and fluorinated telomers may occur in a large number of different chemical compounds either added as final treatments, impurities and unreacted monomers of the production process or chemically bound to the polymeric structure (Knepper *et al*., 2011).

3.1.1 Electrochemical fluorination

The ECF of organic compounds using anhydrous HF was the first significant commercial process for manufacturing ECF-based fluorinated surfactants. Typically, a hydrocarbon sulfonyl fluoride (R-SO_2F, for example, $C_4H_9SO_2F$ or $C_8H_{17}SO_2F$) is transformed into the corresponding perfluoroalkyl sulfonyl fluoride (Rf-SO_2F, for example, $C_4F_9SO_2F$ or $C_8F_{17}SO_2F$).

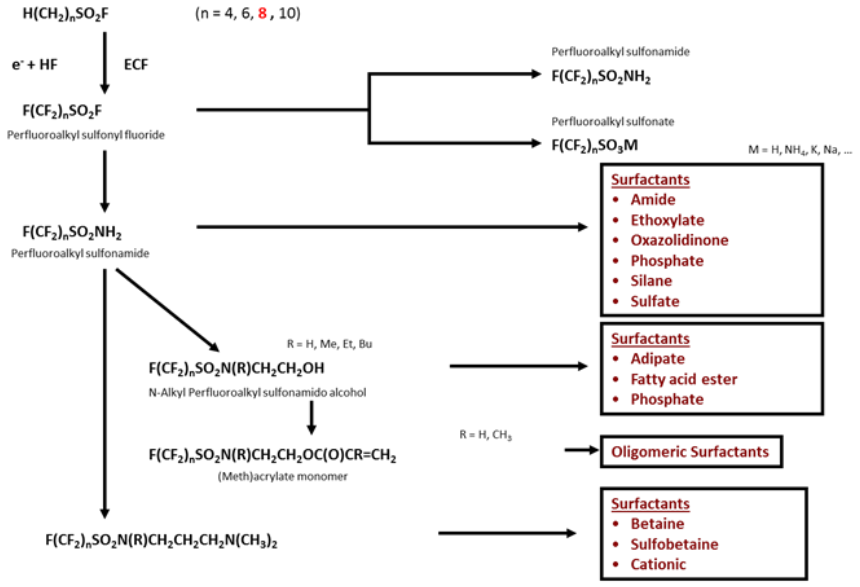
The perfluoroalkylsulfonyl fluoride is the fundamental raw material which is further processed to yield fluorinated surfactants. Commercially relevant perfluoroalkylsulfonyl fluorides are derived from 4, 6, 8, and 10 carbon starting materials yielding perfluorobutanesulfonyl fluoride (PBSF), perfluorohexane sulfonyl fluoride (PHxSF), perfluorooctane sulfonyl fluoride (POSF), and perfluorodecane sulfonyl fluoride (PDSF), respectively.

In the ECF process, fragmentation and rearrangement of the carbon skeleton occurs and significant amounts of cleaved, branched, and cyclic structures are formed resulting in a complex mixture of fluorinated materials of varying perfluoroalkyl carbon chain length and branching as well as trace levels of perfluorocarboxylic acid impurities. The most

basic surfactant derived from the perfluoroalkyl sulfonyl fluoride raw material is the corresponding sulfonate, $RFSO_3$.

Perfluorooctane sulfonate (PFOS) has historically been made in the largest amounts. Perfluorohexane sulfonate (PFHxS) and perfluorodecane sulfonate (PFDS) are also commercially relevant. Recently, the major historic manufacturer of long-chain perfluoroalkyl sulfonyl chemistry, including PHxSF, POSF, and PDSF, ceased their production and moved to the manufacture of PBSF-based fluorinated surfactants (e.g., $C_4F_9SO_2$-R) which are growing in commercial use (Knepper *et al*., 2011).

Figure 1. Synthesis of ECF-based fluorinated surfactants (Knepper et al., 2011)



Note: n = 8 is PFOS and related substances.

By using the perfluoroalkyl sulfonyl fluoride, for example PBSF, as a basic building block, different products are created through the sulfonyl moiety using conventional hydrocarbon reactions. Perhaps the most versatile intermediates from the ECF process are those containing the perfluoroalkyl sulfonamido functionality, $RFSO_2N(R)$-. For example, $C_4F_9SO_2N(CH3)CH_2CH_2OH$, n-methyl perfluorobutylsulfonamido ethanol (MeFBSE).

These primary alcohols can readily be functionalized into fluorinated ethoxylates, phosphates, sulfates, and (meth)acrylate monomers. Fluorinated (meth)acrylates undergo free-radical polymerizations to give oligomeric fluorinated surfactants. In addition, perfluoroalkyl carboxylic acids (PFCAs) and their derivatives have also been synthesized using the

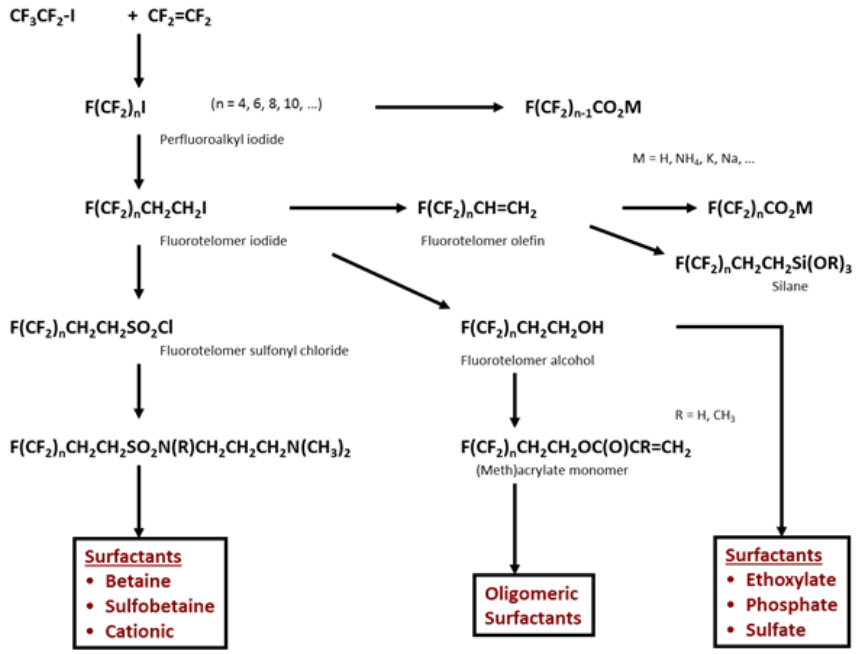
ECF process. Typically, an alkyl carbonyl fluoride (for example $C_7H_{15}COF$) is transformed into the corresponding perfluoroalkylcarbonyl fluoride (for example $C_7F_{15}COF$). The carbonyl fluoride is then reacted to yield esters, amides, or carboxylic acid salts which have all been commercially produced and used as surfactants. The most widely known is the ammonium salt of perfluorooctanoic acid ($C_7F_{15}COOH{\cdot}NH_3$), whose major historical use has been as a processing aid in the manufacture of fluoropolymers.

3.1.2 Telomerisation

The free-radical addition of tetrafluoroethylene (TFE) to pentafluoroethyl iodide yields a mixture of perfluoroalkyl iodides with even-numbered fluorinated carbon chains. This is the process used to commercially manufacture the initial raw material for the "fluorotelomer"-based family of fluorinated substances. Telomerisation may also be used to make terminal "iso-" or methyl branched and/or odd number fluorinated carbon perfluoroalkyl iodides as well.

The process of TFE- telomerisation can be manipulated by controlling the process variables, reactant ratios, catalysts, etc. to obtain the desired mixture of perfluoroalkyl iodides, which can be further purified by distillation. While perfluoroalkyl iodides can be directly hydrolysed to perfluoroalkyl carboxylate salts the addition of ethylene, gives a more versatile synthesis intermediate, fluorotelomer iodides. These primary alkyl iodides can be transformed to alcohols, sulfonyl chlorides, olefins, thiols, (meth) acrylates, and from these into many types of fluorinated surfactants. The fluorotelomer-based fluorinated surfactants range includes nonionics, anionics, cationics, amphoterics, and polymeric amphophiles (Knepper *et al*., 2011).

Figure 2. Synthesis of fluorotelomer-based fluorinated surfactants, (Knepper et al., 2011)



Note: n = 8 is PFOA and related substances

3.1.3 "Per- and Poly- Fluorinated Ethers"

Per- and polyfluorinated ether-based fluorinated surfactants typically have 1, 2, or 3 perfluorinated carbon atoms separated by an ether oxygen, depending on the route to the perfluoropolyether intermediate. The photooxidation of TFE or HFP gives oligomers or polymers with mono- or di-acid end groups. These perfluoropolyethers have random sequences of $–CF_2O–$ and either $–CF_2CF_2O–$ or $–CF(CF_3)\ CF_2O-$ units, from TFE or HFP, respectively (Knepper *et al*., 2011).

In general, the photooxidation of TFE yields mostly difunctional perfluoropolyether acid fluorides, while the photooxidation of HFP yields mostly the monofunctional perfluoropolyether acid fluoride.

The fluoride catalyzed oligomerisation of HFPO, an epoxide, yields a mixture of perfluoropolyether acid fluorides, which can be converted to many types of surfactants, analogous to the fluorinated surfactants from the ECF syntheses. Per- and poly-fluorinated ether surfactants are the newest commercially available substances in this rapidly expanding group of fluorinated surfactants. For example, the phosphate is used as a grease repellent for food contact paper. Per- and polyfluorinated poly-

ether carboxylates are also used as processing aids in the synthesis of fluoropolymers. Per- and polyfluorinated polyether silanes are used as surface treatments (Knepper *et al*., 2011), e.g. for stones or as anti-biofouling agents for ships.

3.1.4 Fluorinated oxetanes

An alternative route to fluorinated surfactants originates from the reaction of polyfluorinated alcohols with oxetanes bearing a $–CH_2Br$ group in their side-chains to create fluorinated oxetane monomers that undergo ring-opening polymerisation to give side-chain polyfluorinated polyethers. Oxetane-based fluorinated surfactants are offered in many forms and functionalities, such as phosphates and ethoxylates (Knepper *et al*., 2011).

4. Methodology and limitations

This chapter gives an overview of how the investigation is carried out as a whole and how the three stages 1) Identification of relevant per- and polyfluorinated substances and their use in different applications on the Nordic market, 2) Potential emissions to and occurence in the Nordic environment of the substances described in stage 1, and 3) A summary of knowledge on toxicity of the most important and prioritized substances in this survey, are linked to each other.

4.1 Methodology

This project is aiming to seek information about uses of less discussed per- and polyfluorocompounds beside PFOA and PFOS. In order to evaluate uses, occurrence and finally toxicity of some prioritised substances the project was structured and performed in three stages, namely:

- *Stage 1 – Identification of relevant per- and polyfluorinated substances and their use in different applications on the Nordic market*
  In stage 1 the following were carried out: a) establishing a database of poly- and perfluorinated substances that may be used on the Nordic market by extraction of a net list which is based on three other lists: A list from OECD, the REACH preregistration database and the Nordic SPIN database and b) a mapping of Nordic market information through a questionnaire to more than 50 market actors in the Nordic market within the following sectors:
  - Aviation hydraulic fluids .
  - Fire fighting foams .
  - Pesticides.
  - Metal plating (hard metal plating and decorative plating).
  - Electronic equipment and components.
  - Chemically driven oil and mining production.
  - Carpets, leather and apparel, textiles and upholstery.
  - Paper and packaging.

    - Coating and coating additives.
    - Construction products.
    - Medical and healthcare products.
- *Stage 2 – Occurence of per- and polyfluorinated substances*
  Identified poly- and perfluorinated substances from stage 1, both from the net list practice and/or answers from Nordic market were meant to be further studied concerning their occurrence in industrial and consumer products, in environment and humans. However, the results from stage 1 did not really give a basis to perform stage 2. Stage 2 was therefore carried out by compiling the occurrence data for per- and polyfluorinated substances that could be found in literature. Findings from this stage resulted in a priority list of the most frequently occurring groups of PFCs in the Nordic environment and in humans which summarises our current knowledge. This priority list – for the stage 3 work – was prepared in consultation with KLIF/NORAP.
- *Stage 3 – Toxic effects of per- and polyfluorinated substances on humans and the environment*
  The priority list from stage 2 was elaborated in ranking order to describe known toxicity data from publicly available literature sources to support future possible regulatory measures from the Nordic authorities.

4.2 Limitations

One major and primary limitation in the intial mapping study is the lack of reliable specific substance data from the market due to the lack of both substance identification and trade secrets. Therefore only publicly available information sources are applied.

There is a major focus of PFCs in the Nordic environment in this survey, consequently literature sources used relate to environmental compartments in the Nordic environment, including in the Arctic.

However, there are limitations in the monitoring data as well, since only PFCs with commercially available analytical reference substances can be analysed and identified in the various studies.

Since there is a strong progress in research in this field especially over the last few years there may be a few very recent publications (also currently unpublished) that have by necessity been left out due to the timing of this survey.

5. Mapping of use of per- and polyfluorinated substances on the Nordic market

The mapping of the use of per- and polyfluorinated substances on the Nordic market was carried out by use of the following instruments:

- Producing a "net list" of PFCs in use on the Nordic market by use of public available lists of PFCs in use.
- Contacting a selection of producers, suppliers and users of PFCs on the European and Nordic market.
- Using information in literature and knowledge from the institutions and persons performing this study.

The first two steps are described in more detail below.

5.1 "Net list" of PFCs in use on the Nordic market

An extraction of a "net list" of PFCs in use in the Nordic countries was performed by use of databases available on the Nordic/European market. There are mainly three lists of PFCs publicly available:

- OECD list from 2007.[1]This list covers substances and polymers that were used on the global market at that time. It is not considered to be up-to-date.
- REACH Pre-registration database.[2] This list covers phase-in [3]substances and polymers intended to be registered under REACH

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[1] Lists of PFOS, PFAS, PFOA, PFCA, Related Compounds and Chemicals that may degrade to PFCA (as revised in 2007). Organisation for Economic Co-operation and Development, 21 August 2007. ENV/JM/MONO(2006)15.

[2] http://echa.europa.eu/information-on-chemicals/pre-registered-substances

[3] Definition according to REACH Article 3. 20)

(i.e. substances manufactured or imported (and/or used) in the EU that are covered by Article 23 concerning transitional provisions).
- SPIN database[4] that covers per- and polyfluorinated substances and polymers contained in dangerous chemical mixtures used in the Nordic countries. The data has its origin in the national product registries.

Initially, PFOS and PFOA and their related substances (C_8-chemistry) have been excluded in the mapping practice of these lists. Other non-PFOS/PFOA substances and additionally polymers have been matched between the lists in order to get a net list of common per- and polyfluorinated substances and polymers, that may be used on the Nordic market. It is important to emphasise that neither of these lists are complete, often due to company trade secrets, but they may provide a selection of categories of per- and polyfluorinated substances and polymers that may be used in the Nordic market.

The next step in the practice of these three lists mentioned above was to extract the common per- and polyfluorinated substances and polymers on each list to receive a "net list" of substances and polymers that are used in EU and the Nordic countries respectively.

A combination of the OECD list and the REACH pre-registration database (and excluding PFOS and PFOA and related substances) resulted in the so-called "European net list" of substances that were on the OECD list and were pre-registered in the REACH system. The "European net list" consisted of 518 substances, i.e. 518 PFCs may be in use on the European market. Of these 79 were polymers or not-precisely defined mixtures which are listed at the end.

A combination of this "European net list" and the Nordic SPIN database resulted in a so-called "Nordic net list" of 118 substances, i.e. 118 PFCs may be in use on the Nordic market. Of these 27 were polymers or not-precisely defined mixtures, which are excluded from the schemes but listed at the end. 91 CAS numbers were therefore included in the sorting as the final "Nordic net list (excluding polymers or not precisely defined mixtures)". We conclude that these PFCs for which there is publicly available information may be used on the Nordic market.

Since neither of these databases contains complete information on the market use of PFCs, the net list is necessarily incomplete and there

[4] http://www.spin2000.net/

may be other PFCs used on the Nordic market in addition to those found in the net list.

A more detailed categorization of the pre-registered 518 non-PFOS/PFOA PFCs in REACH (the "European net list") is found in Appendix B. This includes the polyfluorinated substances that potentially can be used on the Nordic market.

Table 2. The 35 categories of PFCs that were identified in the "net list" exercise

Identified PFC categories	Possible fluoro process
Perfluoroalkane sulfonic acids (PFASs)	ECF
Perfluoroalkane sulfonates (salts)	ECF
Perfluoroalkane sulfinic acid/sulfinates	ECF
Perfluorocycloalkane sulfonic acid and derivatives	ECF
Perfluoroalkane sulfonamides (FASAs)	ECF
Perfluoroalkane sulfonamide, quaternary ammonium salts	ECF
Perfluoroalkanesulfonamide acrylates (MeFASACs)	ECF
Perfluoroalkane sulfonamide methacrylates	ECF
Perfluoroalkane sulfonamide phosphates	ECF
Perfluoroalkane sulfonyl halides	EFC
Other polyfluoroalkyl sulfur compounds	ECF
Perfluoroalkyl carboxylic acids (PFCA)	Telomerisation
Perfluoroalkyl carboxylic salts	Telomerisation
Perfluoroalkyl alcohols/ketones	Telomerisation
Perfluoroalkyl carboxylic acid halides	Telomerisation
Perfluoroalkyl halides	Telomerisation
Perfluoroalkyl alkyl ethers	Telomerisation
Perfluoroalkyl amines	Telomerisation
Perfluoroalkyl amino acids/salts/esters	Telomerisation
Perfluoroalkyl phosphates	Telomerisation
Perfluoroalkyl acrylates	Telomerisation
Perfluoroalkyl methacrylates	Telomerisation
Other perfluoroalkyl carboxylic esters	Telomerisation
Perfluoroalkyl heterocyclic compounds	Telomerisation
Perfluoroalkyl silanes	Telomerisation
Fluorotelomer alcohols	Telomerisation
Fluorotelomer halogenides	Telomerisation
Fluorotelomer sulfonates, sulfonyl chlorides and sulfonamides	Telomerisation
Fluorotelomer acrylates	Telomerisation
Fluorotelomer methacrylates	Telomerisation
Other acrylates	Telomerisation
Fluorotelomer phosphates	Telomerisation
Other fluorotelomers	Telomerisation
Polymers	No information
Undefined mixtures	No information

Additionally structure formulas, synonyms, acronyms, trade names, physical-chemical data and use data have been collected. Only a few of these data, however, are included in the tables that were further developed in project phase 2.

The applied names are as simple as possible and we have chosen to use the most easy to understand. Those are not necessarily the most correct ones, but we have made this choice to make it easier to get an overview and see homologue rows and relationships. That is

also why the "perfluor" prefix and fluorotelomer names have been used where possible.

5.1.1 Discussion about the "correctness" of the "net list"

It must be emphasised that this "Nordic net list" that has been presented in Appendix B only represents some of the "truth". The real picture may very well be very different.

First of all, there is no guarantee that the pre-registered substances are going to be registered in the REACH system. This means that this list may contain substances that may not be used in Europe. On the other hand, new substances were not covered by the transitional provisions and were normally not pre-registered. Therefore the list of the pre-registered substances is probably not complete. Finally the substances used for treatment of articles with per- or polyfluorinated substances outside EU are normally not to be registered within the REACH system. Such per- and polyfluorinated substances are therefore not included in the pre-registration list.

Secondly, the SPIN database is only a database of substances used in chemical products (i.e. substances and mixtures) that are classified as dangerous and used (imported or produced) in the Nordic countries. This means that only chemical products that are classified as dangerous are included – thereby excluding chemicals only containing PFCs that are not classified as dangerous. Moreover, the SPIN database does not contain information about articles treated with e.g. per- or polyfluorinated substances such as impregnated textiles.

Finally, the OECD list is from 2007 and may very well not include all per- and polyfluorinated substances in use today.

5.2 Contacts to producers, suppliers, users and other players on the PFC market

Based on a search and on the knowledge within the project group, a number of producers, suppliers, users and trade organizations in the different Nordic countries were contacted. Global producers and trade organizations were contacted as well. The main contact was carried out by email. But some of the main players on the market were contacted by phone/interviews.

Appendix C contains a list of the about 50 companies and organizations that have been contacted in this project. The questionnaire used for the phone/web interviews are also presented in Appendix C.

5.3 Conclusions

Parallel with the mapping of the Nordic market extracted net lists (Appendix B) based on a list from OECD, the REACH preregistration database and the Nordic SPIN database, identified 518 per and polyfluorinated substances ("European net list") and 118 per and polyfluorinated substances ("Nordic net list") that might be used on the Nordic market (in blue font in Appendix B). Since neither of these databases contain comprehensive information of per- and polyfluorinated substances, there may be several more per- and polyfluorinated substances that may be used on the Nordic market. These per- and polyfluorinated substances were divided into 35 chemical categories. For these 35 per- and polyfluorinated categories their process origin and possible fate into principal degradation products were estimated for a better understanding of the findings concerning occurrence and impact of per- and polyfluorinated substances in the Nordic environment and to humans.

6. Mapping of uses and applications of PFCs on the Nordic market

The mapping carried out in this project has covered the following uses on the markets of the Nordic countries:

- Aviation hydraulic fluids.
- Fire fighting foams.
- Pesticides (insect baits for control of leaf-cutting ants from *Atta spp*. and *Acromyrmex spp.* and insecticides for control of red imported fire ants and termites).
- Metal plating (hard metal plating and decorative plating).
- Electronic equipment and components.
- Chemically driven oil and mining production.
- Carpets, leather and apparel, textiles and upholstery.
- Paper and packaging.
- Coating and coating additives.
- Construction products.
- Medical and healthcare products.

6.1 Aviation hydraulic fluids

Alternative hydraulic fluid additives must undergo extensive testing to qualify for use in the aviation industry to sustain severe conditions during use.

In the manufacturing process for aviation hydraulic fluids, a PFOS-related substance or precursor, such as potassium perfluorooctane sulphonate, was used as an additive to the aviation hydraulic fluids with a

content of about or less than 0.1%.[5] According to the manufacturers, this formulation helps prevent evaporation, fires, and corrosion.

Aviation hydraulic fluids without fluorinated chemicals but based on, for example, phosphate esters are used. These substances can absorb water and the subsequent formation of phosphoric acid can damage metallic parts of the hydraulic system. For this reason, phosphate ester-based hydraulic fluids are routinely examined for acidity as this determines its useful lifetime. Additionally fluorinated chemicals other than PFOS can be used. The potassium salt of perfluoroethylcyclohexyl sulphonate (CAS number. 67584-42-3)[6] is not a PFOS precursor, and it has been used in hydraulic oils instead of PFOS in the past. However, like other C_6 compounds it is likely to be persistent and 3M which formerly produced this chemical has ceased to do so. A search for other alternatives is said to have been going on for 30 years, starting before PFOS was considered a problematic substance. However it is not possible to get any specific chemical composition of alternatives due to trade secrets. Consequently there is no way to describe their potential feasibility and impact to health and environment in a comprehensive way.[7]

6.1.1 Identity and properties

Information gaps

6.1.2 Type of uses, quantities, producers, downstream users and traders

There are several trade names and traders on the market. Some are as follows: Arnica, Tellus, Durad, Fyrquel, Houghto-Safe, Hydraunycoil, Lubritherm Enviro-Safe, Pydraul, Quintolubric, Reofos, Reolube, Valvoline Ultramax, Exxon HyJet, and Skydrol.[8]

The fire-resistant aviation hydraulic fluids principally contain trialkyl phosphates, tri-aryl phosphates, and mixtures of alkyl-aryl-phosphates. However, the products only provide rough descriptions of

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[5] The potassium salt of PFOS was used in such a small quantity that it was not listed on the MSDS at Boeing (Boeing 2001). http://www.boeingsuppliers.com/environmental/TechNotes/TechNotes2001-02.pdf

[6] In the U.S. this chemical is considered a C8 PFOS equivalente and its use in hydraulic fluids is regulated under a Significant New Use Rule: https://www.federalregister.gov/articles/2002/12/09/02-31011/perfluoroalkyl-sulfonates-significant-new-use-rule

[7] UNEP/POPS/POPRC.8/INF/17

[8] http://www.atsdr.cdc.gov/toxprofiles/tp99-c3.pdf

their chemical composition such as “contain phosphate esters”. Consequently there are several information gaps concerning the specific chemical composition of each aviation hydraulic fluid but similarly the traders need to know in detail of these oil characteristics since these characteristics are important to aviation security.

Since very little is published concerning the chemical composition of these aviation hydraulic oils there is currently no possibility to assess their environmental and health impact.

There is currently no, scarce or uncertain data available concerning quantities used on the market.[9]

6.1.3 Efficacy and availability

There is no available information on cost-effectiveness, efficacy, availability, accessibility and socio-economic considerations.

6.2 Fire fighting foams

Fluorinated surfactants are used in fire fighting foams as they are very effective for extinguishing liquid fuel fires at airports, oil refineries etc. Fire fighting foams are divided into:

- Fluoro-protein foams used for hydrocarbon storage tank protection and marine applications.
- Aqueous film-forming foams (AFFF) developed in the 1960s and used for aviation, marine and shallow spill fires.
- Film-forming fluoroprotein foams (FFFP) used for aviation and shallow spill fires.
- Alcohol-resistant aqueous film-forming foams (AR-AFFF), which are multi-purpose foams.
- Alcohol-resistant film-forming fluoroprotein foams (AR-FFFP), which also are multipurpose foams; developed in the 1970s.

[9] As aviation hydraulic fluids are essential to the military in Convention member countries they may be a source of information regarding the alternative substances and their quantities used.

PFOS-containing fire fighting foams has a long shelf life (10–20 years or longer) which is why PFOS-containg fire-fighting foams may still be used around the world in accidental oil fires. However, in recent years fire-fighting foams are not manufactured with PFOS, but with fluorotelomers based on a perfluorohexane (C_6) chain. However, in China PFOS-containing fire fighting foams are still produced.[10]

6.2.1 Types of uses, quantities, producers, downstream users and traders

Information received from the industry during this project confirms that fluorinated surfactants are still used in fire fighting foams. The use of PFOS in fire fighting foams has been discontinued – in new products. However, as PFOS-containing fire fighting foams have a very long shelf life, PFOS-containing fire fighting foams may still be in use globally. EU Regulation from 2008 has, however, ensured that most PFOS stocks have been destroyed.[11]

According to the fire fighting foam industry that has been contacted during this project, the perfluorotelomer used in fire-fighting foams (AFFF, AR-AFFF, FFFP and AR-FFFP) are named C_8-C_{20}-γ-ω-perfluoro telomer thiols with acrylamide (CAS number 70969-47-0) and is used in the most common fluorosurfactants in use in fire-fighting foams since the discontinuation of the PFOS based surfactants. According to the industry most of the manufacturers are committed to continuing use of this chemistry until 2016.[12]

Furthermore, the following summarized information and statements have been received from the fire fighting foam industry about the so-called pure C6 (6:2) fluorotelomers (betaines and aminoxides).

- Production of C6 fluorotelomer in line with the PFOA Stewardship Programme (95% C6 by 2010, 99.9% C6 by 2015) has proved challenging with the end product significantly more expensive than the standard C6/C8 mixture.
- It has proved extremely difficult to achieve acceptable operational efficiency for AFFF fire fighting foams – especially as regards burn-back resistance – using pure C6 fluorotelomer surfactants.

[10] UNEP/POPS/POPRC.6/13/Add.3/Rev.1.
[11] UNEP/POPS/POPRC.6/13/Add.3/Rev.1.
[12] Personal communication with the fire fighting foam industry/producers in summer 2012.

- Approximately 20% more “pure” C6 fluorosurfactant than the older C6/C8 mix is required in order to achieve acceptable performance.
- To date it has proved extremely challenging to formulate an operationally effective fluoroprotein (FP) foam meeting international standards using “pure” C6 fluorotelomer products.
- There are currently very few AFFF manufacturers (one in the Americas, a couple in Europe) whose products are fully C6 compliant and EPA 2015 compliant.
- The majority of manufacturers including a number of major players have taken a conscious decision to stay with the C6/C8 fluorotelomer mixture on grounds of cost and formulation difficulties.
- In particular fluorotelomer surfactants such as CAS number 70969-47-0 (C_8-C_{20}-γ-ω-perfluoro telomer thiols with acrylamide) continue to be used in AFFF formulations with significant potential environmental impact because of the presence of fluorotelomer N:2 chains with N = 8 to N = 20; thus degradation products may include PFOA and its even chain long-chain homologues up to C20 – toxicities are claimed to increase with chain length.
- A major feedstock manufacturer will continue therefore to produce the fluorotelomer betaines 1157N (the C6/C8 homologue mix) as well as 1157D containing the purified C6 fluorotelomer (aminoxide containing pure C6 is also available).
- Of the putative fluorine-free foams on the market relatively few are known to be completely fluorine-free (no organic fluorine present) whereas others are suspected to contain low levels of fluoropolymers.

Within the petroleum industry PFSA (perfluoroalkyl sulfonates) and FTS (fluorotelomer sulfonates) are used (according to the petroleum industry). However, no information about quantity or the specific fluorinated compounds used have been received.[13]

6.2.2 Efficacy

Fluorinated surfactants are used within fire fighting because of very good fire fighting properties and because they can be stored for many years under harsh conditions. Furthermore, the fluorinated surfactants

[13] Personal communication with the petroleum industry in summer 2012.

are not too expensive and they are available.[14] Generally, the fluorinated C_6-chemistry used is considered to be effective, however, not as effective as the C_8-chemistry, and higher concentrations or amounts may therefore be needed.

6.2.3 Availability

The described fluorinated C_6-technology are commercially available worldwide and therefore also on the Nordic market.

6.3 Pesticides

Pesticides exist as formulations containing active ingredients (the pesticide) and additives (adjuvants) that can help in the application of the pesticide or to enhance the efficiency of the pesticide.

PFCs are used both as active pesticides and as adjuvants in the pesticide formulation.

6.3.1 Identity and properties

N-Ethyl perfluorooctane sulfonamide (known as sulfluramid or sulfuramid), a PFOS related substance, has been used as an active ingredient in ant baits to control leaf-cutting ants, as well as for control of red imported fire ants, and termites. PFOS and other fluorinated substances have also been used as inert ingredients in pesticides.

There are a number of chemical alternatives to *N*-Ethyl perfluorooctane sulfonamide (known as sulfluramid or sulfuramid), with a multitude of uses: Chlorpyrifos, Cypermethrin, mixture of Chlorpyrifos and Cypermethrin, Fipronil, Imidacloprid, Abamectin, Deltamethrin, Fenitrothion, mixture of Fenitrothion and Deltamethrin but none of these are fluorochemicals.

In addition there are a number of other pesticides which contain one or several fluorine atoms, typically as $-CF_3$ groups.

PFCs adjutants are marketed and patents exist on them, but so far no studies have been conducted on their identity, levels of use or exposure to the environment.

[14] Personal information received during this project from a user of fluorinated AFFF's.

6.3.2 Types of uses, quantities, producers, downstream users and traders

PFC adjuvants can have various functions such as being dispersion agents for the pesticide, as a means to better spread the pesticide on leafs/the insect or to increase the uptake through the leafs/insects. PFC adjuvants are typically used in smaller amounts (0.1%) than other adjuvant surfactants because they are more effective surfactants. So far there is no overview of producers of these compounds, and it is not known if or to which extent the PFC adjuvants are used in the Nordic countries.

6.4 Metal plating (hard metal plating and decorative plating)

Fluorinated surfactants are able to lower the surface tension in chrome acid baths used for chrome plating by forming a thin foamy layer on the surface of the chrome bath. This mist suppressant layer dramatically reduces the formation of chromium-(VI) aerosols (Cr^{6+}), which are well-known as carcinogenic, sensitizing and dangerous for the environment (Poulsen *et al.*, 2011). The challenges to this application are to have a surfactant that are stable in the presence of hot chromic acid and can resist decomposition during the electrolysis as well. Under these demanding conditions perfluorinated surfactants such as PFOS is stable and maintains its activity under a long period.

Previously, PFOS was used for both decorative chrome plating and hard chrome plating processes but new technology applying chromium-(III) instead of chromium-(VI) has made PFOS use in decorative chrome plating outdated and unnecessary. For hard chrome plating, however, the process with chromium-(III) does not function. Instead larger closed tanks, or increased ventilation combined with an extraction of chromium-(VI), are suggested as alternative solutions for the applications where a use of chromium-(III) is not possible yet (Poulsen *et al.*, 2011).

6.4.1 Identity and properties

The most common fluorinated surfactant used for hard metal chromium plating has been tetraethyl ammonium heptadecafluorooctane sulfonate (CAS number 56773-42-3; Fluortensid-248), a PFOS-related substance are used in Europe and the Nordic countries[15] within the metal plating industry. However, in recent years some substitution of PFOS seems to have taken place worldwide with polyfluorinated surfactants instead such as (Poulsen *et al.*, 2011):

- Potassium 1,1,2,2-tetrafluoro-2-(perfluorohexyloxy)ethane sulfonate (CAS number not known) – commercial name F-53 Chromic Fog Inhibitor (Hangzhou Dayangchem Co. Ltd., China).
- Potassium 2-(6-chloro-1,1,2,2,3,3,4,4,5,5,6,6-dodecafluorohexyloxy)-1,1,2,2-tetrafluoroethane sulfonate (CAS number not known)). Commercial name F-53B Chromic Fog Inhibitor (Hangzhou Dayangchem Co. Ltd., China).
- *1H,1H, 2H,2H*-Perfluorooctane sulfonic acid/6:2 Fluorotelomer sulfonic acid (CAS number 27619-97-2). Commercial names: Fumetrol® 21 (Atotech Skandinavian AB, Sweden) or MiniMist Liquid (MacDermid, USA).

6.4.2 Types of uses, quantitites, producers, downstream users and traders

The chromic acid bath that is used for hard chrome plating is extremely reactive and oxidizing, and PFOS is used because it is very resistant to that harsh environment and has an extremely low surface tension. It is very difficult to find another chemical with such useful properties. However, there are PFOS-free fluorinated alternatives on the market based on e.g. fluorotelomers and also fluorine free alternatives as described above, which do not seem to have large market shares today [Poulsen *et al.*, 2011]. In a substitution project for the Danish EPA carried out in 2010 [Poulsen *et al.*, 2011] it was proven that PFOS-free fluorinated alternatives could be used for hard chrome plating instead of PFOS.

Producers and suppliers of mist suppressants for the metal industry have been mapped in (Poulsen *et al.*, 2011).

[15] Information received in this project from the contacted suppliers of mist suppressants for the metal plating industry in Europe (Nordic countries).

- Atotech Skandinavien AB (Sweden).
- EngTech Scandinavia A/S (Denmark).
- Surtec Scandinavia ApS (Denmark).
- Galvano Kemi (Denmark).
- Enthone (Cookson Electronics) (Sweden).
- Kiesow Dr. Brinkmann GmbH (Germany).
- GalvaNord (Elplatek) (Denmark).
- Dr. Günter Dobberschütz (Germany).
- CL Technology GmbH (Germany).
- Schlötter Galvanotechnik (Germany).
- Chembright (China).
- MacDermid Scandinavian (Sweden) Plating Resources, Inc. (USA).

A selection of these companies that in 2009/2010 replied that they delivered to the Nordic market has been contacted to get newer information for this Nordic project. However, replies have not been received from all the companies that participated in the 2009/2010 survey.

In the above mentioned Danish EPA project [Poulsen *et al.*, 2011] it was estimated that the global use of PFOS (calculated as 100% pure PFOS) was between 32 and 40 tons for the entire metal plating industry based (but with emphasis on non-decorative hard chrome plating) on different information from 2004–2010. The use of pure PFOS in the Nordic countries was estimated to be at least 90 kg (calculated amounts from contacted suppliers).

Information received by contact to the suppliers of mist suppressants to the Nordic countries in this project shows an actual confirmed use of 3 kg of pure PFSA (perfluoroalkyl sulfonates) – i.e. tetraethyl ammonium heptadecafluorooctane sulfonate (CAS number 56773-42-3) being sold to the Nordic countries in 2011 as wetting agent for chromium baths (this is only based on information from limited number of suppliers for the Nordic market). Further contact to one hard chromium plater in Denmark confirms that the use of PFOS-based (PFSA) has not changed since the survey carried out in 2009/2010 [Poulsen *et al.*, 2011]. The use of PFSA in Denmark can therefore still be estimated to be around 10 kg annually. Based on the limited replies from suppliers of mist suppressants to the Nordic countries in this project it is estimated that the total use of PFSA in the Nordic countries is 90 kg or less as estimated in the 2009/2010 survey. Further concerning brands see Appendix D.

6.4.3 *Efficacy*

The performance of the non-PFOS fume suppressant is considered as not equal to that of the PFOS based fume suppressants. To achieve the same reduction in surface tension, more products may be necessary and it may have to be replenished more frequently. The project funded by the Danish EPA about substitution of PFOS in non-decorative hard chrome plating (Poulsen *et al.*, 2011) showed that non-PFOS fume suppressant can be used. However, more fume suppressants may be necessary thus enhancing the costs.

6.4.4 *Availability*

Alternatives to PFOS-based mist suppressants are available and to some extent in use in the Nordic countries. The primary alternative identified in the Nordic countries is:

> CAS number 27619-97-2: 1*H*,1*H*, 2*H*,2*H* perfluorooctane sulfonic acid – commercial name Fumetrol® 21 (Atotech Skandinavian AB, Sweden)

Other commercial alternatives are available as well, but there is not information about the exact identification of the fluorinated surfactant used. Similarly some non-fluorinated alternatives have been introduced as well, but no information of the chemical identification is available (these alternatives are not discussed any further here) (Poulsen *et al.*, 2011).

6.5 Electronic equipment and components

Electrical and electronic equipment often requires several parts and processes. PFOS and related chemicals are used in the manufacturing of printers, scanners and similar products. The PFOS-related substances are process chemicals, and the final products are mostly PFOS-free. PFOS have many different uses in the electronic industry and is involved in a large part of the production processes needed for electric and electronic parts that include both open and closed loop processes. Open processes are applied for solder, adhesives and paints. Closed loop processes mostly include etching, dispersions, desmear, surface treatments, photolithography and photomicrolithography.

PFOS can be used as a surfactant in etching processes in the manufacture of compound in semiconductors and ceramic filters. PFOS are then added as part of an etching agent, and rinsed out during the subsequent washing treatment. Desmear process smoothes the surface of a through-

hole in printed circuit boards. PFOS can be used as a surfactant in desmear agent, i.e. etching agent. PFOS is added in a desmear agent, and rinsed out during washing treatment.

According to information from OECD survey (2006) less than 1 tonne of *N*-ethyl-*N*-[3-(trimethoxysilyl)propyl] perfluorooctane sulfonamide (CAS number 61660-12-6), a PFOS related substance, had been used as an additive in toner and printing inks. Low volumes of PFOS-related substances were also used in sealants and adhesive products.[16]

6.5.1 Identity, properties, types of uses, producers, downstream users and traders

Information gaps.

6.6 Chemically driven oil and mining production

It is reported that PFOS is used in some parts of the world as surfactants in oil well stimulation to recover oil trapped in small pores between rock particles. Oil well stimulation is in general a variety of operations performed on a well to improve the wells productivity. The main two types of operations are acidization matrix and hydraulic fracturing.

Alternatives to PFOS are PFBS, fluorotelomer-based fluorosurfactants, perfluoroalkyl-substituted amines, acids, amino acids, and thioether acids. In most parts of the world where oil exploration and production is taking place, oil service companies engaged in provision of well stimulation services predominantly use a formulation of alcohols, alkyl phenols, ethers, aromatic hydrocarbons, inorganic salts, methylated alcohols, alipathic fluorocarbons for oil well stimulation. Oil well stimulation services also involve corrosion control, water blocks/blockage control, iron control, clay control, paraffin wax and asphaltene removal and prevention of fluid loss and diverting.

6.6.1 Identity, properties, types of uses, producers, downstream users and traders

Information gaps.

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[16] UNEP/POPS/POPRC.8/INF/17.

6.7 Carpets, leather and apparel, textiles and upholstery

Fluorinated finishes are a technology known to deliver durable and effective oil and water repellence and stain and oil release properties. Historically, fluorinated polymers based on perfluorooctane sulfonyl (PFOS) electrochemical fluorination chemistry have been used. PFOS was not directly used to treat textiles but used to be present at up to 2 wt% in products. In addition, fluorotelomer-based polymers have also been used.

A restriction of use of PFOS in textiles was introduced within EU legislation in 2008. As in other areas there is no longer a use of C_8-chemistry, but has been replaced by C_6-chemistry.[17]

Fluorotelomer alcohols, when used for waterproof and dirt-repellent finishes, are supposed to ensure that PFC degradation products such as PFOS are formed. FTOHs were found in eight of the 14 samples. The highest concentration of fluorotelomer alcohols was 464 µg/m^2. Test results showed that some manufacturers are already using C6 telomer alcohols (i.e. 352 µg/m^2 of 6:2 FTOH). Long-chain C10 telomers were also used in the products (10:2 around 200 µg/m^2). Next to the fluorotelomer alcohols, fluorotelomer acrylates (FTAs), also known as polyfluorinated acrylates, were also detected in some samples (8:2 and 6:2). These acrylates are intermediates in the production of fluorinated polymers. Like the C8 telomers, they can be converted into PFOA through oxidation. No perfluorooctane sulfonate (PFOS) was found in the investigation (Schultze *et al.*, 2006).

6.7.1 Identity and properties

Major manufacturers in conjunction with global regulators have agreed to discontinue the manufacture of "long-chain" fluorinated products and move to "short-chain" fluorinated products. Novel short-chain fluorinated products, both short-chain fluorotelomer-based and perfluorobutane sulfonyl-based, have been applied for manufacture, sale and use in carpets, textiles, leather, upholstery, apparel, and paper applications.[18]

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[17] Personal information from the Finnish Textiles and Clothing Industry.
[18] UNEP/POPS/POPRC.8/INF/17.

6.7.2 Types of uses, quantitites, producers, downstream users and traders

There is currently no publicly available data concerning quantities used on the market. For a selection of trade names, traders and manufacturers, see Appendix D. A Danish survey funded by the Danish EPA estimated that the use of fluorinated substances in impregnated products and impregnation agents (i.e. covering impregnating agents for footwear, carpets, textiles, leather, furniture etc. and impregnated products such as footwear, carpets, clothing, furniture, etc. and other products such as paints, printing inks, ski waxes, floor polish etc.) was between 14 and more than 38 tons of pure fluorinated substances in Denmark. When assumed that the same products and use patterns are applicable to the other Nordic countries, the total amount used within the Nordic countries may be between about 50 tons or more than 100 tons in the Nordic countries.

This former Danish survey as well as contact to the textile industry in the Nordic countries in this survey illustrates that treatment of textiles with fluorinated compounds is not performed in the Nordic countries of any kind of textiles, except maybe in the carpet industry. For brands see Appendix D.

6.8 Paper and packaging

Fluorinated surfactants have been evaluated for paper uses since the early 1960s. Perfluorooctyl sulfonamido ethanol-based phosphates were the first substances used to provide grease repellence to food contact papers. Fluorotelomer thiol-based phosphates and polymers followed. Currently polyfluoroalkyl phosphonic acids (PAPs/diPAPs) are used in food-contact paper products and as levelling and wetting agents. Since paper fibers and phosphate-based fluorinated surfactants are both anionic, cationic bridge molecules need to be used in order to ensure the electrostatic adsorption of the surfactant onto the paper fiber. These surfactants are added to paper through the wet end press where cellulosic fibers are mixed with paper additives before entering the paper forming table of a paper machine. This treatment provides excellent coverage of the fiber with the surfactant and results in good folding resistance. An alternative treatment method involves application of a grease repellent at the size press and film press stage which consists of impregnating the formed paper sheet with a surface treatment. Fluorinated phosphate surfactants are not preferred for this mode of paper treatment. In this latter case, fluorinated polymers are used instead of

surfactants. In terms of oil and water repellency, it is well recognized in the paper industry that phosphate-based fluorinated surfactants provide good oil repellency but have limited water repellency. Acrylate polymers with fluorinated side chains derived from sulfonamido alcohols and fluorotelomer alcohols are the most widely used polymers because they deliver oil, grease, and water repellence. Most recently, perfluoropolyether-based phosphates and polymers have become widely used treatments for food contact paper and paper packaging.[19]

At least one manufacturer has developed a non-chemical alternative for this use. The Norwegian paper producer Nordic Paper is using mechanical processes to produce, without using any persistent chemical, extra-dense paper that inhibits leakage of grease through the paper.

6.8.1 Types of uses, quantities, producers, downstream users and traders

See Appendix D

6.9 Coating and coating additives

Fluorinated surfactants provide exceptional wetting, leveling and flow control for water-based, solvent-based and high-solids organic polymer coating systems when added in amounts of just 100–500 ppm.

Coating and coating additives include the following uses:

- Cleaning products and polishes.
- Impregnating products.
- Ski waxes.
- Paint and lacquers.
- Dental floss.

Fluorinated surfactants impart various properties to paints and coatings including anti-crater and improved surface appearance, better flow and levelling, reduced foaming, oil repellency, and dirt pickup resistance. They have also been widely used in inks.

[19] UNEP/POPS/POPRC.8/INF/17.

The inclusion of fluorinated surfactants in ink jet compositions has led to better processing through modern printers and excellent image quality on porous or non-porous media. Fluorinated surfactants improved surface wetting during the screen printing of carbon black inks onto Polymer Electrolyte Membrane (PEM) fuel cell electrodes. In addition, fluorinated surfactants improved the cold-water swelling and internal bond strength of wood particleboard bonded with urea–formaldehyde (UF) adhesive resins due to reduced interfacial tension of the resins and improved substrate wetting.[20]

6.9.1 Type of uses, quantities, producers, downstream users and traders

The uses of fluorochemicals are quite varied. Specifically, floor polish, where anionic fluorosurfactants are used and at the 100–200 ppm level based on weight of polish.

Performance of most manufacturers of floor polish considers the addition of fluorosurfactants necessary to wet, flow and level properly on a floor.

Paint and lacquers

According to the Confederation of Danish Industry – Paints & Lacquers section – there is no use of per- or polyfluorinated substances in the Danish paint industry.[21] Similarly no use of per- or polyfluorinated substances have been reported in the Finnish Printing Ink industry.

Ski waxes

The Norwegian National Institute of Occupational Health has in 2009 investigated the exposure of professional users of ski waxes in Norway. This investigation shows that the professional users of ski waxes are exposed to fluorinated chemicals – also airborne. This investigation does not mention the concentration of the fluorinated substances used in ski waxes nor the total amounts used. It is, however, mentioned that ski waxes may contain either a mixture of several perfluoro-n-alkanes (C_{12}-C_{24}) or perfluoro-n-alkanes (C_7 or C_8) (Daae *et al.*, 2009).

[20] UNEP/POPS/POPRC.8/INF/17.

[21] Personal communication in the summer of 2012with the Confederation of Danish Industry.

6.10 Others

6.10.1 Construction products

According to information received from the industry, the same fluor-chemistry that is used in fire-fighting foams (Thiols C8-C20 -gamma-omega-perfluoro tellers with acrylomide (CAS 70969-47-0)) is also used in a variety of building and construction products relating to light weight concrete, concrete sandwich panels, and light weight concrete blocks – at least in Australia. It is not known whether this use is widespread and in use in the Nordic countries as well. Construction products as the above mentioned are often recycled and crushed and placed in a landfill site. Non-fluorinated alternatives for use in light weight concrete and related concrete products do exist.[22]

6.10.2 Medical and healthcare products

According to information received from the Nordic chemical industry within this survey, the following fluorinated compounds have been used and sold in Finland, Denmark and Sweden within processing medical or other healthcare products.

- Tetraethyl ammonium heptadecafluorooctane sulphonate (CAS number 56773-42-3), a PFOS-related substance.
- Tetraethylammonium perfluorobutane sulphonate (CAS number 25628-08-4).

The exact use is not known. Searches on the internet shows that the chemical product can be used for metal chromium plating as well as wetting and flow control agent for coating photographic paper and film. The use was about 150 kg of pure fluorinated substances in the three above mentioned Nordic countries in 2011.

[22] Personal communication with the fire fighting foam industry/producers in summer 2012.

6.11 Other important market information for the Nordic market

3M comments that shorter chain fluorochemical could potentially be used in all application fields as described in the tender document for this project (e.g. metal plating, oil production, carpets, leather, apparel textiles and upholstery, coatings). 3M is at this moment no longer active in the field of paper & packaging applications, fire-fighting foams and pesticides. For more details about other alternatives, including shorter chain fluorotelomers (4:2 and 6:2 FTOH) 3M refers to the manufacturers of this chemistry.

According to information received from the Finnish Plastic Industry, none of their more than 100 member companies are producing fluorinated substances nor importing any. No fluorinated substances are listed in the local buyer's guide of the plastics and rubber industry.

6.12 Conclusions

There are considerable information gaps of most per- and polyfluorinated chemicals concerning the exact chemicals composition in commercial products, their quantities produced and uses on the Nordic market. Based on interviews with more than 50 stakeholders with industrial relevance to the Nordic market, this survey has identified two major reasons for these information gaps. Findings show considerable knowledge gaps and/or trade secrets among manufacturers and importers on the Nordic market, whether they trade with articles or chemical products. However it is hard to distinguish to what extent lack of knowledge is predominant compared to trade secrets but both phenomena exist on the Nordic market.

The survey of the use of per- and polyfluorinated substances on the Nordic market has however, revealed the use of a few specific compounds (listed in the table below):

Table 3. Specific PFCs and their uses in relation to the Nordic market survey and net list practice

CAS number	Specific compound used	Use area	Comment
70969-47-0	Acrylamide, Thiols, C_{8-20}, Gamma, Omega, Perfluoro, Telomers	Fire fighting foams Construction products	Not on the "Nordic net list"
161278-39-3	C_6-fluorotelomers such as perfluorohexane ethyl sulfonyl betaine, often used in combination with hydrocarbons such as FORAFAC™ products (DuPont)	Fire fighting foams	Not on the "Nordic net list"
No information	Dodecafluoro-2-methylpentan-3-one (3M)	Fire fighting dispenser	No information
56773-42-3	Tetraethyl ammonium heptadecafluorooctane sulfonate (Fluortensid-248), a PFOS -related substance (perfluoroalkyl sulfonate)	Fire fighting foams	Not on the "Nordic net list"
27619-97-2	1H,1H, 2H,2H-Perfluorooctane sulfonic acid/6:2 Fluorotelomer sulfonic acid (Fumetrol® 21 or Mini-Mist Liquid)	Metal plating	Fluorotelomer sulfonates – on the "Nordic net list"
61660-12-6	*N*-ethyl-*N*-[3-(trimethoxysilyl)propyl] perfluorooctane sulfonamide	Electronic equipment and components	Not on the "Nordic net list"
No information	Fluorotelomer alcohols – e.g. 6:2 and 10:2 FTOH	Textiles	Fluorotelomer alcohols – on the "Nordic net list"
No information	Polyfluorinated acrylates (FTA 8:2 and 6:2), methacrylates and fluoroacrylate polymers	Textiles and food contact paper	Perfluoroacrylates – on the "Nordic net list"
No information	Polyflouroalkyl phosphonic acids (PAPs/diPAPs)	Food contact paper	Fluorotelomer phosphates – on the "net list"
No information	Perfluoro-n-alkanes (C_{12}-C_{24}) or perfluoro-n-alkanes (C_7 or C_8)	Ski waxes	No information

It is seen from the table above that the identified chemical compounds with a specific CAS number in general are not available on the "Nordic net list" which implies that these compounds have neither been identified through the REACH pre-registration list and through the SPIN database. This does, however, not mean that they are not used. The chemical groups of compounds for which the survey has identified a use category are in most cases on the "Nordic net list". To conclude: some overlap can be found between the chemical groups of per- and polyfluorinated found in the "Nordic net list" and in this survey of the use in the Nordic countries. Some of the chemical groups that have been identified through chemical analysis for use in specific products are also available on the "Nordic net list" (Appendix B). However, the results of Stage 1 were not

a good starting point for Stage 2 as the entire area is characterized by a lack of information.

6.13 Future work

There is a need to improve access to specific PFC substance information from industrial actors on the market. The current legal tools according to CLP and REACH, such as safety data sheets, provisions regarding registration etc. are not sufficient to provide that information, in particular not for PFCs in articles where almost all production occurs outside the EU.

7. Occurrence of per- and polyfluorinated substances

There are to date considerable data gaps concerning potential emissions into the environment and exposure to humans of the suggested PFCs studied in this survey.

The fate of currently used per- and polyfluorinated substances is in many cases not known. Detection of various final breakdown products in the environment is an indication of ongoing reactions/mechanisms/ activities in areas where non-persistent per- and polyfluorinated substances are used.

In the following chapters the environmental occurrence and the health and environmental effects of the different per- and polyfluorinated substances and their degradation products are described to the extent possible based on the currently available information.

7.1 Emissions to and occurrence of PFCs into the environment

PFCs in the Nordic countries have been reported in a number of publications and reports. The current literature covers both biotic and abiotic samples like air, indoor dust, water, wastewater, sludge, sediment and soil.

In the following paragraphs, the existing literature on PFCs is presented, together with the emission estimates for PFOA and PFOS and additional reports of emissions and surface water concentrations of PFOA and PFOS for the whole European territory.

7.1.1 PFCAs (Perfluoro carboxylates)

Abiotic and biotic samples

PFCAs in Nordic countries have been reported in a number of papers and reports. Starting with seawater, PFCAs have been analysed in Greenland, Iceland, Faroe Islands and in Tromsø (Norway). Among PFCAs PFOA has been the most abundant, at concentrations that reached 40 pg/L. PFHxA, PFHpA and PFNA were typically detected at levels of a

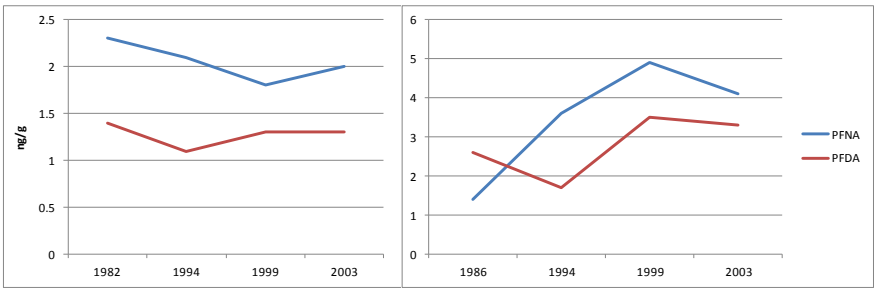
few pg/L (Butt *et al*., 2010). In a study in Greenland, PFCAs were detected in snow with PFOA being again the dominant compound with concentrations up to 520 pg/L (Theobald *et al*., 2007). In Denmark, PFCAs have been analysed and reported for a number of wastewater treatment plants (WWTPs), in the order of a few ng/L. In a few cases, it was reported that concentrations in the effluent wastewaters were slightly higher than the respective in influents (for example, for PFDA). It is interesting to note that there are big variations in the concentrations of PFCAs between WWTPs, but also within the same WWTPs. In particular, in two samples analysed from one WWTP, the concentration of PFOA was below 2 ng/L in the first sample and 23.5 ng/L in the second, while the concentration at the effluents was 10.1 and 16.4 ng/L, respectively. In another WWTP samples taken on the same day, contained 4.5 and 6.4 ng/L of PFOS in the influent WWTP and 8.7 and 21.0 ng/L in the effluent (Bossi *et al*., 2008). The same study reported also sludge concentrations of PFCAs. Only PFOA, PFNA and PFDA were detected, with the latter showing the highest value of 32 ng/g (dry weight, dw). In sewage sludge from Norway, PFUnA, PFTA and PFTrA were detected (Report 2367/2008). Other reports also show findings of PFOA (Report TA3005/2012 and TA 2636/2010). In Iceland and Faroe Islands, PFCAs were regularly below the limit of quantification, and when quantified, their concentrations were normally at <1 ng/g (wet weight, ww).

PFCAs were close to the detection limits in marine sediments in Iceland and Faroe Islands (max concentration of 0.09 ng/g (dry weight) was for PFHxA, Butt *et al*., 2010), and similarly not detected in background sediments in Norway (Report 2367/2008). However in sediments close to a company that manufactures fire fighting foams the concentrations of PFCAs were particularly high, reaching 326.7 ng/g for PFuNA. Similarly, PFHxA and PFOA exhibited high concentrations as well, namely 112.9 and 101 ng/g, respectively. The important impact of local sources such as the fire fighting foam used in airports has been proven to contaminate adjacent soils, groundwater and other environmental compartments. In particular, this can be seen in the comparison between background soils close to the major Oslo airports and soils from the airport areas. For background soils, in Rygge and Gardemoen, PFCAs were not detected, whereas soils from the airports exhibited higher concentrations, particularly those from Gardemoen. In the latter, concentration of PFUnDA was 43.6 ng/g, while for PFOA and PFHpA were around 4 ng/g (Klif Report TA-2444/2008).

In air samples, PFCAs are typically not detected and only PFOA was detected in concentrations that ranged between 0.15–1.51 pg/m^{-3} in the Norwegian Arctic (Butt *et al*., 2010).

The occurrence of PFCAs in biotic samples is quite extensively reported. Concentrations in algae, fresh and marine water fish samples, seabirds, pinnipeds, whales, marine mammals, in eggs, plasma and liver have been reported. A lot of these species have been collected from the Arctic. In algae and fish in most cases, PFCAs are below the limit of detection, or when detected, the concentrations are very low. In ice amphipod samples from the Barents sea only PFOA was detected (3.15 ng/g ww) (Haukas *et al*., 2007) and in marine and fresh water fish, PFCAs when detected, was always below 2 ng/g (Haukas *et al*., 2007, Kallenborn *et al*., 2004, Bossi *et al*., 2005). A number of studies have been performed on sea birds. Bossi *et al*. (2005) did not detect PFCAs in black guillemot and in northern fulmar samples that were collected before 2000. However, Kallenborn *et al*., 2004 found 0.4 ng/g PFHpA and 1.0–1.3 ng/g of PFNA in the same sea bird species studied in 2004. Very high concentrations for PFDA and PFDoDA were detected in guillemots by Løfstrand *et al*. (2008), while PFOA and PFNA were not detected. Concentrations of these PFCs in herring gulls ranging from not detected to <1 ng/g were reported by Verreault *et al*., 2007, however, no temporal trends, meaning no trends over time, could be seen between samples collected in 1993 or 2003. Similarly, no clear temporal trend could be observed for PFNA and PFDA in ringed seals over the last 30 years (Figure 3, Bossi *et al*., 2005).

Figure 3. Temporal trends of PFCAs in ringed seal samples from two areas in Greenland (Bossi et al., 2005)



In more recent samples (from 2000 and 2005), Galatius *et al*., 2011 reported concentrations for harbor porpoises for PFDA and PFUnDA and noticed a small increase between 2000 and 2005. PFDA was the only PFCA that was detected in blue mussels (nd-6.38 ng/g, Klif Report TA-

2367/2008), whereas PFNA was the dominant PFCA in liver and blood from marine mammals (Smithwick *et al*., 2005). In a study by Rotander *et al*., 2012, temporal variations in concentrations of different PFCs were examined in livers of pilot whale, ringed seal, minke whale, harbor porpoise, hooded seal, Atlantic white-sided dolphin and in muscle tissue of fin whales. The sampling spanned over 20 years (1984–2009) and covered a large geographical area of the North Atlantic and West Greenland. In general, the levels of the long-chained PFCAs (C9–C12) increased over the studied period (Rotander, 2012).

7.1.2 PFSA (Perfluoroalkyl sulfonates)

Occurence in the environment

PFSAs have been studied in most of the afore-mentioned studies that reported PFCAs. PFOS is the most important and best studied compound from this class. It is also the one that in almost all cases exhibited the highest concentration levels. In abiotic environmental samples, PFOS was the only PFSA detected in marine sediments in Faroe islands in concentrations just higher than the quantification limit (ND-0.11 ng/g ww, Butt *et al*., 2010), while PFDS was reported from sediments in Norway in two studies, between which though, there was an important difference in the maximum concentrations (0.93 and 0.094 ng/g, by Fjeld *et al*., 2005 and Bakke *et al*., 2007, respectively). In sea water and other aquatic samples, PFOS has ranged between ND and 1.18 pg/L, in the Faroe Islands (Butt *et al*., 2010). The substance was not detected in Iceland, but was found at levels as high as 90 pg/L in Tromsø, Norway. In the same sample, PFHxS was also detected at the concentration of 16.4 pg/L, being many times higher than in other sea water samples, where it was barely detected. These differences underline the importance of the urban discharges. As a matter of fact, in effluent wastewaters in Denmark, PFOS was detected at concentrations up to 1,115 ng/L and PFHxS at concentrations up to 19.8 ng/L. Sewage sludge samples have also been analysed for PFSAs and again the prevalence of PFOS and PFHxS was seen. In a Danish WWTP, PFOS concentrations varied between 4.8 and 74.1 ng/g (dw, Bossi *et al*., 2008), while in Norway, the range was between 1.2 and 5.16 ng/g (Report TA 2367/2008). PFDS was also reported in sludge from Norway, at a range of 0.35 and 6.84 ng/g (dw).

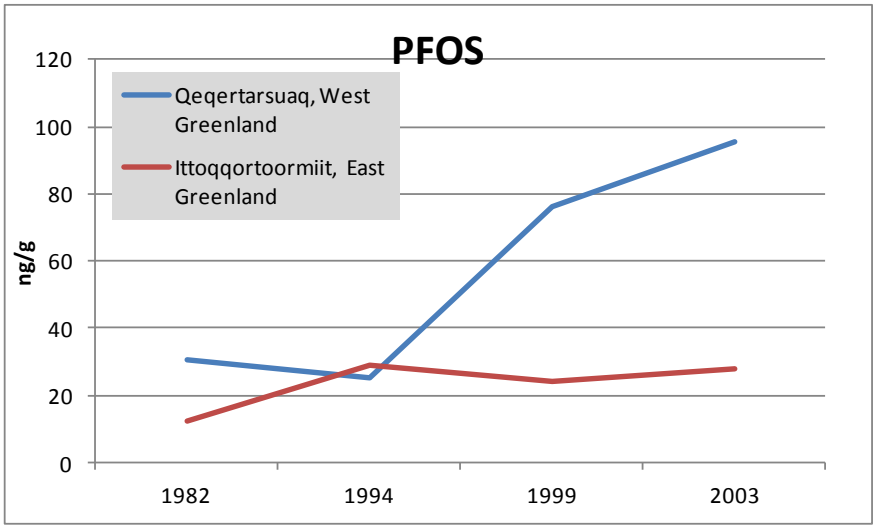
Similarly to what was reported for PFCAs, the importance of local emission sources was assessed by analyzing soils adjacent to airports and remotely from the airports, yet in the same region. The differences in concentrations were 5–10 times for PFOS (40.2 and 226.9 ng/g in

soils in Rygge, and from 109.9 to 959 ng/g dw in soils close to Gardermoen) and even higher for the other PFSAs that were not detected in background soils (Report 2444/2008). Close to the training station of the company manufacturing fire-fighting foams, the concentrations in stream water was 68,886 ng/L for PFOS, 37,312 ng/L for PFHxS, which is many orders of magnitude higher than any influent or effluent wastewater sample.

PFOS was the only PFSA detected in the Norwegian Arctic ambient air, however in very low concentrations, close to the limit of quantification (0.02 – 0.97 pg/m^3, Butt *et al*., 2010). In other air samples from Norway, concentrations of PFDS ranged between nd and 5.13 pg/m^3, PFOS ranged between 0.03–3.32 pg/m^3 and PFHxS between nd and 0.71 pg/m^3.

In biotic samples, again PFOS and PFHxS are the most commonly reported chemicals. PFDS and PFBS are only occasionally detected. Regarding fish samples, PFHxS was only detected in Arctic cod (0.04 ng/g ww, Haukas *et al*., 2007), whereas PFOS was detected in Arctic cod, in dab and shorthorp sculpin, in concentrations that ranged between nd and 28 ng/g, ww (highest concentration in long-rough dab, Kallenborn *et al*., 2004). PFDS was also detected in fish samples, with its highest concentration (11.6 ng/g ww) observed in long-rough dab as well (Kallenborn *et al*., 2004). In sea birds, PFOS exhibited its highest concentrations of 134 ng/g (ww) in glaucous gull samples from the Norwegian Arctic (Verreault *et al*., 2005), 3–4 times higher than in Herring gull from Hornøya or from Røst.Whereas for PFHxS, the levels were similar in all kinds of sea birds, without any visible trend. PFBS was not detected and PFDS was detected in Herring Gull, but always in very low concentrations (average between 0.04 and 0.21 ng/g, ww). The concentrations of PFOS in ringed seal and whales are broadly in the same range as in sea birds, however, there seems to be important spatial differences. As can be seen in Figure 3, in 1999 and 2003, the differences in PFOS concentrations in ringed seals between West and East Greenland are notable, both in terms of concentration levels, but also as temporal trends.

Figure 4. Temporal and spatial differences in the concentration of PFOS in ringed seals in West and East Greenland (Bossi et al., 2005)



Finally, it should be noted that the highest concentrations for PFOS and PFHxS were reported in polar bears' blood from Svalbard (Smithwick *et al.*, 2005). In particular, PFHxS exhibited an average value of 2,940 ng/g (ww) and PFOS had an average of 1,290 ng/g (ww). PFOS was particularly high also in the liver samples (1,170–1,285 ng/g), whereas PFHxS was relatively low (36 ng/g ww).

7.1.3 PFAL (Perfluoro aldehydes)

The literature research has shown that there are no published studies dealing with the environmental concentrations of perfluoro aldehydes in the Nordic countries.

7.1.4 FTOH (fluorotelomer alcohols)

Fluorotelomer alcohols are fluorotelomers that have one alcohol group. They are characterised by high volatility and in the environment act as precursors to PFCAs. FTOHs were broadly produced and it was estimated that between 2000 and 2002, more than 4,000 tonnes were produced annually. Since 2002 their production decreased sharply, after 3M ceased their production and use in their products.

Occurrence in the environment

As FTOHs are volatile compounds, their environmental occurrence is predominantly in the gas phase in the atmosphere. Only very few studies exist that have studied FTOHs in the Nordic countries, and in particular, just for Norway. Concentrations are summarized in Table 4.

Table 4. Occurrence of fluorotelomer alcohols in the Nordic countries

Sample	4:2 FTOH	6:2 FTOH	8:2 FTOH	10:2 FTOH	12:2 FTOH	Unit	Country
Alnabru (Air)[1]	nd	nd-1,720	nd-9,470	nd-3,460	nd	pg/m3	Norway
Birkenes (Air)1	nd	Nd	nd	nd	nd	pg/m3	Norway
Kjeller (Air) [4]	nd	11.7	34.4	17.2		Pg/m3	Norway
Indoor air[2]	4.8	1,492	6,438	4,088		pg/m3	Norway
Indoor air, homes[3]	21	42	10,005	3,405		pg/m3	Norway
Indoor air, Office[3]	165	266	3,151	1,970		pg/m3	Norway
Indoor air, homes[4]	114	2,990	3,424	3,559		pg/m3	Norway
Indoor air, Office[5]	nd	212	637	1,279		pg/m3	Norway
WWTP, influent[1]	nd	Nd	nd	nd	nd	ng/L	Norway
WWTP, effluent[1]	nd	Nd	nd	nd	nd	ng/L	Norway
WWTP, sludge[1]	nd	nd-0.01	nd-0.01	nd-0.02	nd-0.03	ng/g dw	Norway
Sediments (various sites in Norway)[1]	nd	Nd	nd-0.09	nd-0.1	nd-0.6	ng/g dw	Norway
Mussels	nd	Nd	nd	nd-0.09	nd	ng/g dw	Norway

[1]Report 2367/2008; [2]Haug *et al*., 2011; [3]Huber *et al*., 2011; [4]Barber *et al*., 2007; [5]Jahnke *et al*., 2007.

It can be seen that fluorotelomer alcohols are in very low concentrations in effluents, sludge or sediments, however in air, and particularly in the indoor environment, they can occur at very high levels. 8:2 FTOH is the most abundant FTOH with its levels that reach 10,000 pg/m^3, followed by 10:2 FTOH. In one case 10:2 FTOH exhibited higher concentrations than 8:2 FTOH. The study of Barber *et al*. (2007) compared FTOH concentrations between Kjeller and various sites in the United Kingdom and it was shown that the concentrations in Norway were always lower than in the UK, and in particular ΣFTOHs in Kjeller (63.3 pg/m^{-3}) were 7–8 times lower than in Manchester.

7.1.5 FTS (fluorotelomer sulfonates)

Occurrence in the environment

Table 5 presents results from the occurrence of fluorotelomer sulfonates in the Nordic environment. 6:2 FTS is the most studied member and has been detected in soil, sediment, groundwater, indoor dust and in some few cases also in biota. Its' concentrations are relatively low and the high concentrations presented in Table 5 for soil, sediment and groundwater, are from an area that was located close to contaminated sites. In the

various biota analysed, 6:2 FTS was detected only in ice amphipod, sea snails and in ivory gull eggs.

In the indoor air, 6:2 FTS has been detected in two cases in dusts, in concentrations that ranged between nd and 38.7 ng/g. 8:2 FTS was detected only in indoor air dust in concentrations ranging from nd to 50.2 ng/g.

Table 5. Occurrence of FTSs in the Nordic environment

	6:2 FTS	8:2 FTS	Unit	Country
Groundwater[1]	3,220		ng/L	Norway
Soil[1]	1,020,688		ng/kg	Norway
Sediment[1]	1,090		ng/g dw	Norway
Contaminated stream water[1]	9,388		ng/L	Norway
Indoor dust, homes[2]	nd-38.7	nd-50.2	pg/g	Norway
Indoor air, homes[7]	nd		pg/m^3	Norway
Background soil[1]	nd		ng/g dw	Norway
Sea snail[1]	2.4-129		ng/g dw	Norway
Indoor Air dust[3]	9	15	ng/g	Norway
Ivory gull egg[4]	0.25		ng/g	Russia
Ivory gull egg[4]	0.28		ng/g	Russia
Ivory gull egg[4]	0.37		ng/g	Russia
Ice amphipod[5]	0.48		ng/g ww	Barents Sea
Polar cod[5]	nd		ng/g ww	Barents Sea
Black guillemot[5]	nd		ng/g ww	Barents Sea
Glaucus gull[5]	nd		ng/g ww	Barents Sea
Influent[6]	Nd	nd	ng/L	Norway
Effluent[6]	Nd	nd	ng/L	Norway
Sludge[6]	Nd	nd	ng/g ww	Norway
Sediment[6]	nd-2.37	nd	ng/g ww	Norway
Air[6]	nd-0.11	nd	pg/m^3	Norway
Sediment[6]	Nd	nd	ng/g ww	Norway
Mussel[6]	Nd	nd	ng/g ww	Norway
Cod liver[6]	Nd	nd	ng/g ww	Norway

[1]Report 2444/2008; [2]Huber *et al*., 2011; [3]Haug *et al*., 2011; [4]Miljeteig *et al*., 2009; [5]Haukas *et al*., 2007; [6]Report 2367/2008; [7]Barber *et al*., 2007.

7.1.6 Other fluorinated telomers

In 2002, polymer production consumed 80% of the fluorotelomers produced worldwide, according to the Telomer Research Programme (2002). A study of carpets treated with various polymeric and surfactant fluorocoatings found residual FTOHs ranging from 0.04–3.8% of the dry mass of the commercial fluorochemicals. 6:2, 8:2, 10:2 and 12:2 FTOH and NMeFOSE were measured, and the contribution of the FTOHs varied greatly between the products; however 8:2 FTOH was the most predominant compound in the study (Dinglasan-Panlilio and Mabury, 2006). It is common that the FTOHs are linked via esterbonds to a polymer backbone, as in the case of fluoroacrylates. FTOHs can be released to its surroundings if the ester bond is hydrolysed, upon exposure to water, heat and a catalyst, such as acids.

The literature research has shown only few data about compounds like FTCA, FTUCAS and mainly in air and water samples.

7.1.7 PAP/di-PAP (polyfluoroalkyl phosphate esters)

The literature research has shown that there are no published studies dealing with the environmental concentrations of polyfluoroalkyl phosphate esters in the Nordic countries. No studies have so far investigated point pollution sources of PAPs in Nordic countries, and at present it is not known if PAPs are produced in the Nordic countries. PAPs might however be applied as coating agents on paper and board, and can be released to the environment from e.g. paper manufacturing plants or paper conversion industries.

Canadian studies have thus shown that PAPs are present in waste water treatment plant sludge and in paper fibres (D'eon 2009). PAPs have also been found in Danish and Swedish paper and board food packaging (Trier 2011), where 57% of the samples taken in 2009 from Danish, Swedish and Canadian retailers (Trier 2011, thesis) contained di-PAPs in the material (Trier 2011, thesis). It is therefore likely, that also wastewater sludge in Nordic countries contain PAPs, which upon hydrolysis (e.g. catalysed by heat and acidic conditions) release fluorotelomer alcohols. Sludge which are used as fertilizers on fields, and containing PAPs, are thus likely to release FTOHs and PFCAs. Other possible exposure routes of PAPs to the environment are via household waste sites and during storage of recycled paper.

7.1.8 Other fluorinated compounds of interest

The data on other fluorinated compounds of interest in the Nordic countries are scarse, almost non existent. This lack of data supports one of the conclusions of this report that screening projects should be further encouraged.

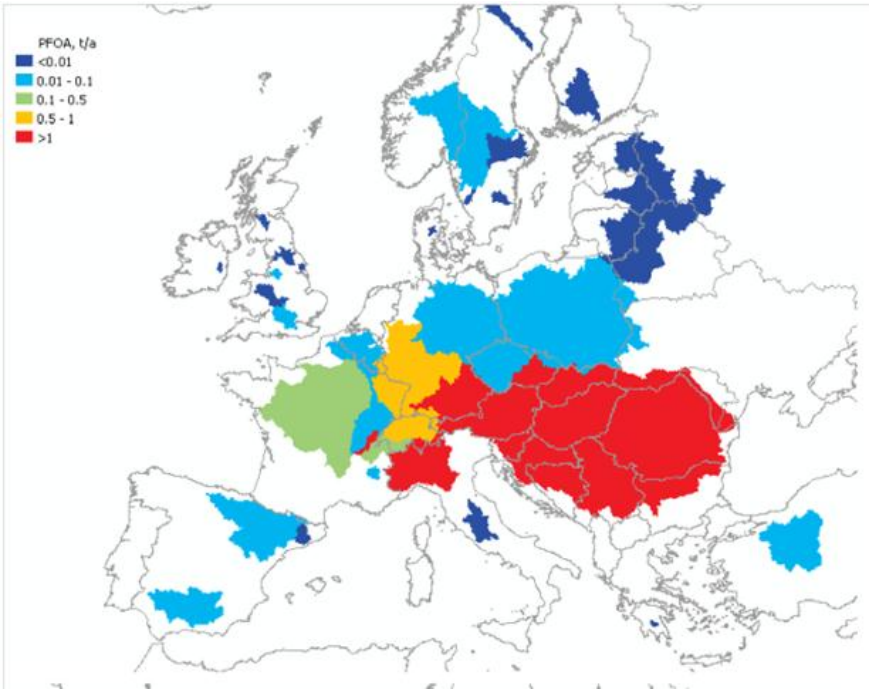
7.1.9 Emission calculations for fluorinated compounds in the Nordic countries

The compilation of emission inventories for the various chemicals that occur in the environment is a challenging task that requires detailed information.,Even when much information is available, the final estimates are highly uncertain values. An emission inventory needs to contain accurate information on the emissions directly from materi-

als/products, or from industrial discharges during manufacturing etc. but also needs to provide information regarding spatial and temporal distribution (Breivik *et al*., 2007). In other cases, the use of environmental monitoring data is used together with climatologic conditions and other relevant input parameters in order to estimate emissions (Pistocchi and Loos, 2009). As seen from the results presented in the relevant chapters, there are only few data on the occurrence of fluorinated chemicals in the Nordic countries, something that makes the estimate of emissions based on the Pistocchi and Loos method quite difficult (Pistocchi and Loos, 2009).

In the literature, one can find information about the emissions of PFOS and PFOA. Based on the report TA-2354/2007 prepared for the Norwegian Pollution Control Authority (2007), the PFOA estimated emissions (primary and secondary and due to long range transport) for Norway were 130–380 Kg per year. Important primary sources were considered to be carpets (12Kg/y), coated and impregnated paper (1.3 Kg/y) and textiles (0.5 Kg/y). Pistocchi and Loos (2009) also estimated PFOA concentrations and emissions for almost the whole European territory, thus Nordic countries were included.

Figure 5. PFOA emission estimates for Europe (Figure taken from Pistocchi and Loos, 2009)



It can be seen that for the sites for which it was possible to estimate the emissions and concentrations, the Nordic countries were amongst the lowest found in all of Europe. This is also supported by the literature review of Eschauzier *et al*. (2008) who stated that "The low population density and fewer industrial activities in Scandinavian countries compared to central Europe could explain the lower concentrations found in the North of Europe."

Although it is difficult for someone to extrapolate and estimate exact annual emissions for entire countries, the data shows that the emission both in Norway and Sweden is much less than 100 Kg/y (Table 6).

The latter value is calculated by using the average European emission factor value Pistocchi and Loos (2009) estimated for PFOA (and also PFOS). The estimated emissions are given in Table 6; it should be noted that these emissions represent a relatively negative scenario and are overestimated. The reason for this overestimation is the fact that the average European emission factors are used, although, as it can be seen from figures 5 and 6, the emissions in Sweden and Norway are far lower than the average European emissions. For this purpose, it is safe to say that the emissions are below 100 Kg/y in each country, showing a declining trend for the Nordic emissions.

Table 6. Average emission factors and annual emission estimates for Norway (assuming population of 5 milions) and Sweden (assuming population of 9.5 milions)

		Emission Factors (µg/d*capita)	Annual emissions (Kg/y)
Norway	PFOA	27	49.28
	PFOS	19	34.68
Sweden	PFOA	27	93.62
	PFOS	19	65.88

Finally, the study of Pistocchi and Loos (2009) presented estimates for the surface water concentrations all over Europe, both for PFOA and PFOS (Figures 7 and 8).

It is interesting to note that the levels of PFOS and PFOA concentrations that were estimated (predicted) by Pistocchi and Loos (2009) are in very good agreement with the measured concentrations (few tens to few hundred pg/L).

Figure 6. PFOS emission estimates for Europe (Figure taken from Pistocchi and Loos, 2009)

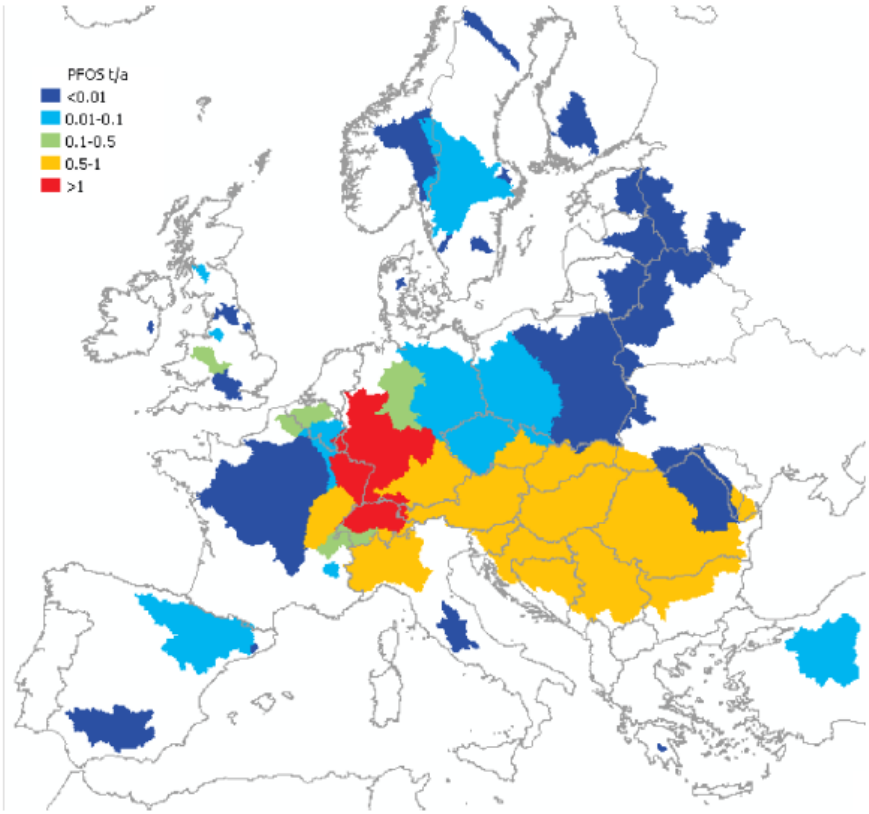


Figure 7. Surface water PFOA concentrations for Europe (Figure taken from Pistocchi and Loos, 2009)

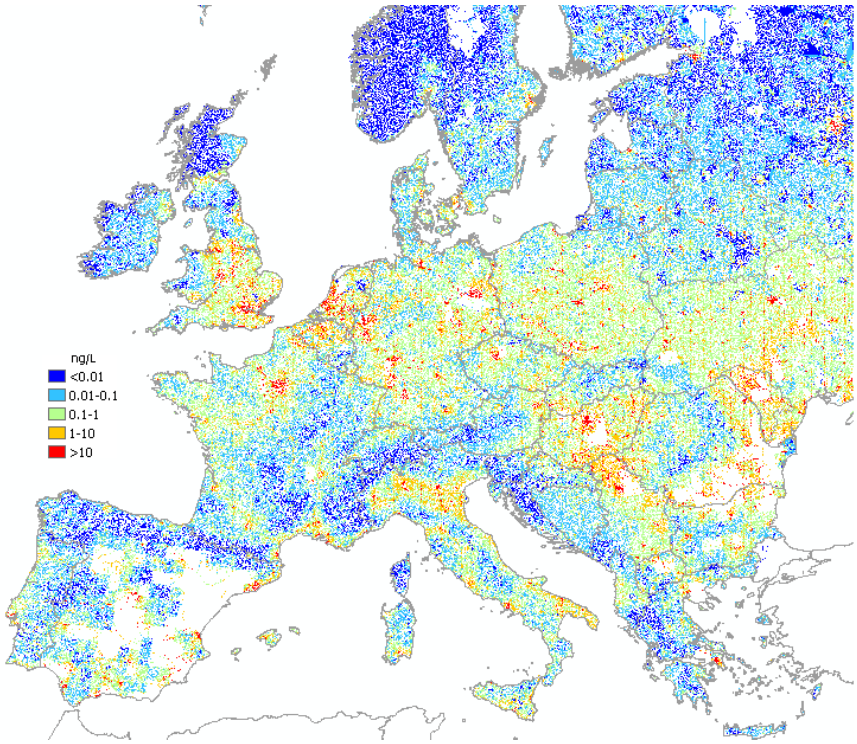
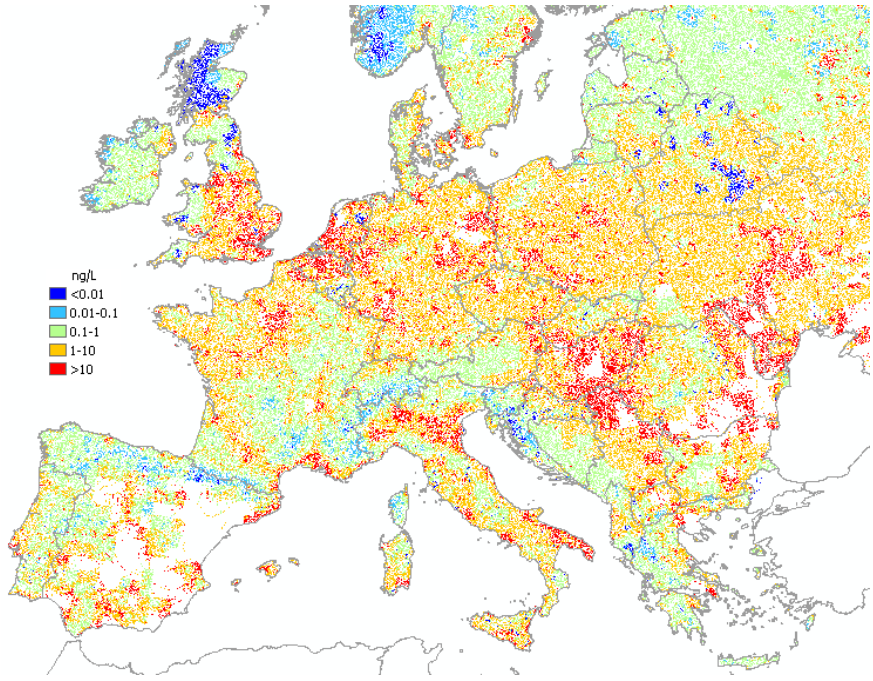


Figure 8. Surface water PFOS concentrations for Europe (Figure taken from Pistocchi and Loos, 2009)



7.1.10 Estimation of long range atmospheric transportation potential and fugacity modelling

It is well known that the potential of individual organic contaminants to be transported over large distances (LRAT, long range atmospheric transport) will differ widely, reflecting differences in physico-chemical properties and reactivity in the atmosphere (Wania, 2006; Wania and Mackay, 1996). The L_A (characteristic travel distance or long range atmospheric transport potential – LRATP) of any chemical at any point in time will be limited by atmospheric reaction and (net) atmospheric deposition. L_A is an easy measure of the chemicals' mobility in the environment and is defined as the distance over which the initial air concentration of a chemical is reduced to 1/e (~37%) (Bennett *et al*., 1998; Beyer *et al*., 2000; Breivik *et al*., 2006). The LA can be estimated by the formula:

$$L_A = u \,.\, M_A \,/\, [N_{RA} + N_{AS}]$$

(Eq. 1), where u is the wind speed, and N_{RA} and N_{AS} are the rates or reaction and air to surface deposition, respectively.The calculation of L_As for

any substance allows us to estimate the concentrations at any distance from the source, by using the equation:

$$C(x) = C_0 \cdot e^{-x/L_A}$$

(Eq. 2) where, C(x) is the concentration of the chemical in a distance "x" from the emission point, C_0 is the initial concentration of the chemical at the point of the emission (distance is 0 km) and L_A is the characteristic travel distance of the chemical.

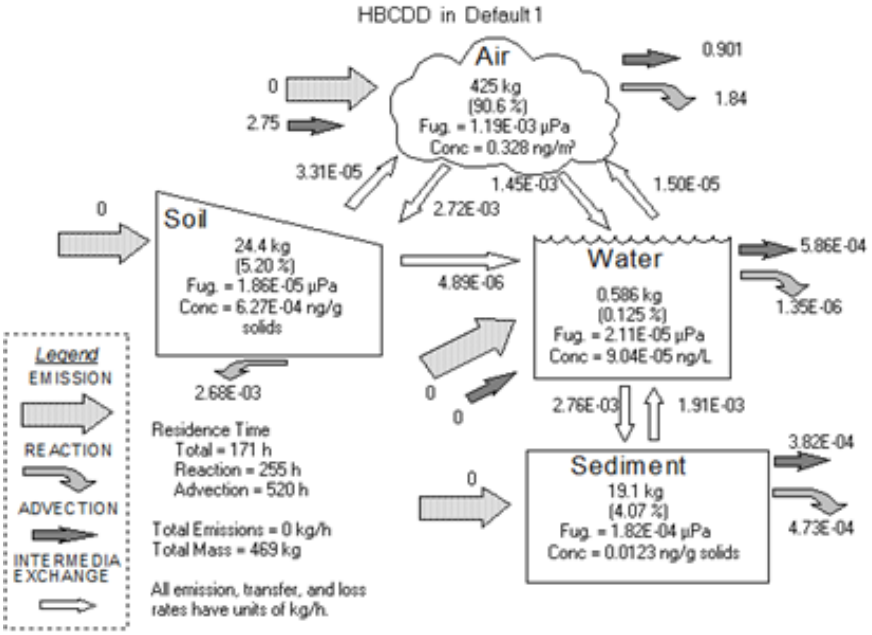
This approach could be used to estimate LRATP for fluorinated compounds that are volatile and can be found in the gas phase. It would improve our understanding about the secondary emissions and about the contributions from non-Nordic countries. However, in order to do so, we need a rich set of monitoring data and sound and internally consistent physico-chemical properties (reaction rates, deposition rates) and in addition, values that are representative for the climatologic conditions in the Nordic countries. Both are important data gaps that should be better positioned/addressed within the future research priorities for the Nordic countries. Estimation of LRATP for all fluorinated chemicals is also beyond the scope of this report.

Another interesting tool for the estimation of the distribution of our target compounds between the various environmental compartments and thus, again, for the estimation of fluxes of fluorinated compounds that partition between air, water, soil, sediment etc, is the fugacity model (e.g. the LEVEL III model). This model requires as input the important environmental concentrations (air, water, soil etc.), emission rates and physico-chemical properties, in particular, the half-lives of the compounds of interest in the various environmental compartments. The LEVEL III model can then estimate directly loads, fluxes and concentrations, and can provide in a graphical form all the information about the "circle" of the pollutants between the various environmental compartments (Figure 9).

Application of the LEVEL III fugacity model is not possible for most fluorinated compounds of interest because of the lack of consistent and reliable data for the aforementioned parameters. Before such modelling approaches can be applied for the less studied fluorinated substances (e.g the long carbon chain ones, >C8), we would need to have internally consistent physico-chemical parameters representative for the climatologic conditions of the Nordic countries. In addition, we would require emission factors and environmental concentrations in order to estimate

accurately their environmental fate. At this point, for the less studied fluorinated compounds, this appears to be premature.

Figure 9. Graphical output of the LEVEL III model (the case of HBCDD)



7.1.11 Conclusion on emissions and occurrence

PFCs in the environment in Nordic countries have been reported in publications and reports covering both biotic and abiotic samples, like air, indoor dust, water, wastewater, sludge, sediment and soil. Regarding PFCAs, most studies report finding PFOA, PFHxA and PFNA. Similarly, for PFSAs, PFOS and PFHxS are the most studied compounds. Compared to other countries, the concentrations in the Nordic countries are lower, especially when compared with central European countries with high GDPs. However these substances have also been found in the Arctic, far from any sources, which shows that these substances are global contaminants.

The outcome of this review of the environmental occurrence of fluorinated substances is that there is urgent need for new data, on more PFCs in order for decision makers to have a complete picture about the PFC levels in all environmental compartments and an in-depth knowledge of spatial and temporal distribution, and clear temporal trends. The detailed environmental fate study of PFCs is hindered in many cases by the lack of reliable (or in some cases total lack of) physical-chemical properties for many fluorinated compounds.

Existing data on the emissions and surface water concentrations of PFOA and PFOS show that the Nordic countries are among the least contaminated regions in Europe from PFCs, as is to be expected due to lower population density and less industry.

7.2 Sources of exposure of PFCs to humans

In general there are two important sources of exposures of PFCs to humans namely via food and drink intake and through exposure to house dust.

Food intake is assumed to be a main source of exposure of the general population to PFCs. However most of the data are given on PFOS and PFOA whereas only limited data are available on other PFCs in food. In a recent study by Haug *et al*., 2010a, 12 different PFCs were detected in 21 samples of different food and beverages on the Norwegian marked (data in Appendix F). Calculation of intake was done by use of consumption data given by Norkost 1997 (Haug *et al*., 2010a). The study found that in general the highest dietary intake of PFCs in Norway was from PFOA (31 ng/day) followed by PFOS (18 ng/day), PFDA (13 ng/day) and PFNA (9.5 ng/day). For all the given substances 85% of the measured data was >LOD. (A value of ½ x LOD was used for data below LOD). The estimated total intake of the 12 PFCs for the Norwegian adult population was 103 ng/day. Consumption of *fish*, *meat*, *seafood products* and *cereals* represented 75–92% of the total estimated intake of the PFCs.

In the UK the highest levels of 11 PFCs (PFHxA, PFHpA, PFOA, PFNA, PFDeA, PFUnA, PFDoA, PFBS, PFHxS, PFOS and PFOSA) were in fish and offal food (Clarke *et al*., 2010). Other kinds of food, including shellfish, meat, milk, butter, cheese, cereals and vegetables were found to be almost free of PFCs in the UK foodstuffs.

The intake from cereals is higher in the Nordic study (Haug *et al*., 2010a) than in a Spanish study (Ericson *et al*., 2008). This may be due to different consumption patterns in Norway and Spain or the Nowegian data on cereals may be overestimated due to analytical uncertainties, according to Haug *et al*. 2010a.

The estimated human intake of PFCs decreases with increasing age and the intake was found to be higher in males than in females according to Haug *et al*., 2010a.

7.2.1 PFCA (Perfluoro carboxylates)

Food and drinking water

The median human intake of PFOA in several regions studied world wide is estimated to 2.9 ng/kg bw/day (Fromme *et al*., 2007) and 2.5 ng/kg bw/day (range 0.3–140 ng/kg bw/day) (Vestergren *et al*., 2008). Precursor compounds (as FTOH) used in the production of fluorinated polymers may add to the exposure of PFOA; this is especially the case in high-exposure scenarioes (sum of 95th percentile values for each individual input values) (Vestergren *et al*., 2008) where precursor-based exposure to PFOA account for 48–55% of the total daily doses for adults.

The estimated intake of PFOA in the Norwegian population was found to be lower than what has been reported from Spain, Germany, UK, Canada and Japan (Haug *et al*., 2010b). Estimated dietary intakes of different PFCAs in the Norwegian population are given in the table below (table 7). The major PFC intake is from PFOA (31 ng/day) according to Haug *et al*., 2010a. The estimated intake of PFOA from the duplicate diet study given by Fromme *et al*., 2007 is 5–6 times higher than the intake of 31 ng PFOA /day estimated by Haug *et al*, 2010a and 42 ng/day (Haug *et al*., 2010b) (see table 7 on dietary intake below). This can be due to several parameters related to e.g. the differences in consumption pattern and the different levels of PFCs in food from the different countries as well as uncertainties in estimating the consumption of different foods. According to the Norwegian data, cereals give a major contribution to the intake of PFOA (see table 6) and in the total intake of PFCs *cereals* may also contribute significantly (Haug *et al*., 2009a and 2009b). In Norway PFOA in bread was estimated to be a major source of the total intake of PFCs (Haug *et al*. 2010b).

Fish is assumed to be a major source of fluorinated substances. This was also found in a Baltic study (n = 45, age 19–62) where individuals (n = 15) who declared to have a high fish consumption (mainly Baltic fish) on average showed the highest load (in blood samples) of the fluorinated substances of: PFHxA, PFHpA, PFNA, PFDA, PFUnDA, PFDoDA and to a lesser extent PFOA (Falandysz *et al*., 2006). In a Norwegian study *fish and shellfish* were estimated to give the largest contribution of PFOA and PFUnDA for human intake (38% versus 93 %), calculated from correlation between serum PFC concentration and food consumption data. As can be seen from Table 1 in Appendix F, the levels of PFOA in fish found in the Norwegian study (Haug *et al*., 2010a) were significantly lower than the data from the UK (Clarke *et al*., 2009) and also lower compared to some other studies according to Haug *et al*., 2009. According to the author (Haug *et al*, 2009) this could be explained by the Nordic fish being caught in open sea rather than costal areas and due to different fish species

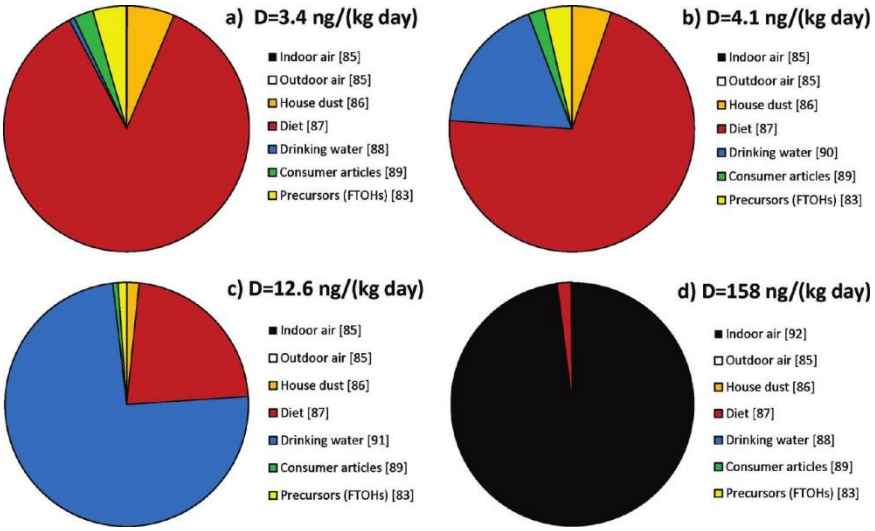
Table 7. ***Dietary intake of perfluorocarboylates, PFCA, (ng/day) for the general Norwegian population***

Reference	Food type	Country	Number of samples*	Year	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA
Haug et al., 2010a	Cereals	Norway	3		**4.3**	**3.2**	**15.0**	**2.8**	**5.2**	2.2	2.2
	Milk and dairy products		3		**1.3**	**2.0**	**4.4**	**4.4**	**2.7**	**1.4**	**2.2**
	Fish and seafood		3		0.55	0.91	**2.4**	0.44	**1.2**	**1.0**	0.36
	meat and meat product		3		0.35	1.0	**2.7**	0.94	**1.7**	0.68	0.40
	Eggs		3		0.22	0.13	0.49	0.06	0.21	0.17	0.07
	Sugar and sugar products		3		0.06	0.04	0.25	0.07	0.16	0.12	0.12
	Fats		3		0.08	0.09	0.40	0.22	0.14	0.27	0.27
	Vegetables		3		0.11	0.06	0.25	0.10	0.10	0.13	0.17
	Starchy roots and potatoes		3		0.39	0.14	0.66	0.26	0.38	0.28	0.30
	Fruits and juices		3		0.12	0.09	0.36	0.08	0.14	0.11	0.14
	Coffee, tea and cocao		3		0.18	0.25	**2.1**	0.07	0.29	0.12	0.22
	Alcoholic beverages		3		0.04	0.04	0.15	0.01	0.06	0.02	0.02
	Tap water		3		0.14	0.14	0.54	0.04	0.21	0.08	0.08
	Soft drinks		3		0.12	0.12	0.45	0.03	0.18	0.06	0.07
	Total intake				**8.0**	**8.2**	**31**	**9.5**	**13**	**6.7**	**6.7**
Number of samples*: 3 different brands/types of each food were pooled											
Note: Food consumption based on Norkost 1997 survey on 2672 adults											
Haug et al., 2010b	Mean dietary intake of a 70 kg person						42			24	
Note: Food consumption based on recent data from frequency questionnares											

The largest intake of PFOA may occur from *contaminated food included drinking water* (Trudel *et al*., 2008). According to Trudel *et al*., 2008 this is followed by the ingestion of dust and inhalation of air. The uptake of PFOA in children on a body weight basis is higher compared to adults because of a higher relative uptake from food and hand- mouth transfer from treated carpets and ingestion of dust (Trudel *et al*., 2008). In the high product scenarios the dominating pathways are found to be product- and age dependent: E.g. uptake from food contact materials is an important pathway for teenagers and adults (Trudel *et al*., 2008).

Drinking water may be a significant source of PFC, and in particular PFOA, exposure to human. In drinking water, produced from surface water in contaminated areas, PFOA was the main compound found in a German study with the level of 500–640 ng/L (Hölzer *et al*., 2008). This is in accordance with another German study reporting high levels of PFOA (519 ng/L) followed by PFHpA (23 ng/L) and PFHxA (22 ng/l) (Table 2 in Appendix F) in public water supplies produced from river water with bank filtration or artificial recharge (Skutlarek *et al*., 2006). When activated-charcoal filters were installed in the water supply, this efficiently decreased the PFC concentration in drinking water (Skutlarek *et al*., 2006). In other areas the level of PFCs in drinking water was much lower, with the sum of PFCs varying between non-detected and 27 ng/L (Skutlarek *et al*., 2006). In the Netherlands the level of PFCs in drinking water resources was found to be in the range of non-detectable to 43 ng/l (Mons *et al*., 2007). In a recent study by Haug *et al*., 2010a, three samples of tap water from different Norwegian water works in the Oslo area were analysed. The level of PFOA was 0.65–2.5 ng/L whereas the other PFCAs were below 1 ng/L as given in Table 2 in Appendix F. A review on the presence of PFCAs and PFSAs in European surface waters, ground water and drinking water was recently published as a book chapter (Eschauzier *et al*. 2012). It compared the relative importance of different sources of intake of PFCs, and showed that where raw water was affected by point contamination, e.g. by contaminated sludge, then the corresponding drinking water was the major source of human exposure. This is also shown for the intake of PFOA in the figure below (pie c) compared to different other exposure scenarios (pie a, b and d) (Vestergren *et al*., 2009).

Figure 10. Pie charts displaying a compilation of the estimated daily intakes of PFOA for male adults (D) and relative importance of exposure pathways from separate studies



Each pie chart represents an exposure scenario representative of (a) background concentrations in drinking water (1.3 ng/L); (b) elevated concentrations in drinking water (40 ng/L); (c) point sources of drinking water contamination (519 ng/L); (d) occupationally exposed individuals (indoor air concentrations 1 µg/m3). References of the individual studies are given in square brackets in the legends of each chart. (Vestergren *et al*., 2009)

In a recent study, tap water from six European cities were analysed for PFCAs. The higest level of PFCA was found for PFOA (8.6 ng/l) in water samples from Amsterdam (Ullah *et al*., 2011).

Food packaging materials

In a recent Danish study 84 different samples of food pakaging materials of paper and board were tested for contents of per-and polyfluorinate compounds by exposure to 50% ethanol. In 35 of the samples the level of PFCs were above the the limit of detection. High levels of PFCA were found in the extracts of popcorn bags (Trier *et al*., 2012).

Consumer products and cosmetics

PFCs are primarily used as processing aids in the manufacture of fluoropolymers and can be detected either as additives or residual impurities (with a content from C4 to C14) in a large variety of commercial products, including leather, carpets, paper, paint, AFFF, waterproofing agents, coated fabrics, non-stick cook ware, floor wax (dominant contributions from PFHpA, PFOA, and PFNA), ski wax and textiles and clothes (Begley *et al*., 2005; Freberg *et al*., 2010; Gewurtz *et al*., 2009;

Prevedouros *et al*., 2006; Sinclair *et al*., 2007; Trudel *et al*., 2008; Washburn *et al*., 2005). Herzke *et al*. (2012) found that none of the waterproofing agents/lubricants they analysed were free from PFCs, most abundant being PFBS, PFBA, PFNA, PFDoA, PFHxA and PFHpA (see Table 1 Appendix E). They also detected PFCAs in table cloths, presumably due to Teflon treatment, but they could not establish if the minimal levels found in paint were actually added or only result of contamination. PTFE or Teflon® is probably the most publicly well-known and most widely used fluoropolymer as a source of PFCAs (Walters and Santillo, 2006). Applications of PTFE include: electrical wire insulation, tape, waterproof membranes for garments (such as Gore-Tex), surgical implants, dental floss, engine protector additives, non-stick coatings, moulded parts and coatings for use in a wide range of chemically hostile environments (DuPont, 2012).

Consumer products like sprays and treated carpets may contribute to the consumer exposure of PFOA (Trudel *et al*., 2008) but are probably a less important source for most consumers/the general population according to Trudel *et al*., 2006. However, these sources may contribute significantly to the exposure for those consumers frequently using e.g. PFC containing sprays and who have treated carpets in their home (Trudel *et al*., 2008). Table 1 in Appendix E gives an overview of the presence of PFCs in consumer products.

Indoor air exposure

All PFCAs have been detected in indoor house dust in Norwegian houses and offices (Huber *et al*., 2011). In the latter study, PFUnA was the most abundant PFCA in houses with a median concentration in dust of 120 ng/g, followed by PFOA (38.8 ng/g) and PFHxA (10.1 ng/g). In one office reported in the same study, the pattern was different, with PFOA being the most abundant chemical (694 ng/g), followed by PFHxA (29.3 ng/g), while PFUnDA was among the least abundant, exhibiting the concentration 1.4 ng/g. Haug *et al*. (2011) also reported concentrations of PFCAs in indoor dust from Norwegian homes. In this particular study, PFHxA was the most abundant chemical in dust (33 ng/g), followed by PFNA (29 ng/g) and PFOA (20 ng/g). This was the only study that reported also detectable concentrations of PFTrDA and PFTeDA in indoor dust, suggesting that indoor air dust can be a sink for many compounds that occur in low levels in the indoor air. Finally, in a study from Sweden (Bjorklund *et al*., 2009), PFOA was studied in houses, offices and apartments and the average concentrations were 54, 93 and 70 ng/g, respectively, thus, in the same order of magnitude as in Norwegian indoor environments.

PFHxA (17.1 pg/m3), PFOA (4.4 pg/m^3) and PFNA (2.7 pg/m^3) were detected also in indoor air particles in Tromsø (Barber *et al*., 2007).

To the best of our knowledge, the only study that has quantified exposure of humans to PFOA in indoor air in the Nordic countries is the study of Haug *et al*. (2011b). In the latter study, it was shown that through indoor air dust, the uptake of PFOA through dust will range between 0.19 and 0.78 ng/kg bw/day and through air the same uptake will be between 0.002 and 0.16 ng/kg bw/day. It was shown that uptake through dust and air was particularly low compared to other exposure pathways.

7.2.2 PFSA (Perfluoroalkyl sulfonates)

Food and drinking water

Human intake of PFOS has been estimated to a wide range of 3.9–530 ng/kg bw/day (Vestergren *et al*., 2007). Precursor compounds (as PFOSA and PFOSE) used in the production of fluorinated polymers may add to the exposure due to their degradation into PFOS. The median intake of PFOS was found to be 1.4 ng/kg bw/day based on analysis of duplicate diet samples from various regions world wide (n = 214) of 31 healthy individuals (age 16–45) (Fromme *et al*., 2007). PFHxS and PFHxA could only be detected in some samples (above the LOD) with a median intake of 2.0 ng/kg bw/day and 4.3 ng/kg bw/day. The estimated intake of PFOS from the duplicate diet study given by Fromme *et al*., 2007 is about 5 times higher than the intake estimated by Haug *et al*, 2010. This can be due to several parameters related e.g. to differences in consumption pattern and the level of the PFCs in food from the different countries, to uncertainties in estimating the consumption of different foods and to uncertainties regarding the analytical test methods and analysis.

As for PFOA, the largest intake of PFOS seems to occur from *contaminated food included drinking water* (Trudel *et al*., 2008). This is followed by the ingestion of dust and inhalation of air. Consumer products like sprays, treated carpets and food contact materials may also lead to consumer exposure of PFOS (Trudel *et al*., 2008) but as for PFOA the spray sources are probably less important for most consumers/the general population according to Trudel *et al*., 2008. However, spray sources may contribute significantly to the exposure for those consumers frequently using e.g. PFC containing sprays and who have treated carpets in their home (Trudel *et al*., 2008).

A recent Norwegian study found that in general the major dietary intake of PFCs in Norway was PFOS (18 ng/day) (and from PFOA

(31 ng/day as given above)) (Haug *et al*., 2010a). The estimated intake of PFOS in this Norwegian study was found to be lower than what has been reported from Spain, Germany, UK, Canada and Japan (Haug *et al*., 2010b) and also lower than reported in another recent study of the same author (Haug *et al*., 2010b) as given in the table below. Different data on food consumption were used in the two Norwegian studies and may be one reason for the observed differences in the Norwegian PFOS intake.

In relation to age, the highest potential intakes of PFOS are estimated for infants and toddlers (Vestergren *et al*., 2007). The uptake of PFOS in children on a body weight basis tend to be higher because of a higher relative uptake from food and hand- mouth transfer from treated carpets and ingestion of dust (Trudel *et al*., 2008). In the high product scenarios the dominating pathways are found to be product- and age dependent: E.g. uptake from food contact materials is an important pathway for teenagers and adults (Trudel *et al*., 2008).

Table 8. Dietary intake of perfluoroalkyl sulfonates, PFSA (ng/day) for the general Norwegian population

Reference	Food type	Country	Number of samples*	PFBS	PFHxS	PFOS
Haug et al., 2010a	Cereals	Norway	3	0.22	0.52	5.1
	Milk and dairy products		3	0.22	0.10	4.7
	Fish and seafood		3	0.19	0.19	3.4
	meat and meat product		3	0.14	0.09	3.3
	Eggs		3	0.03	0.06	0.66
	Sugar and sugar products		3	0.01	0.005	0.05
	Fats		3	0.03	0.04	0.08
	Vegetables		3	0.01	0.006	0.06
	Starchy roots and potatoes		3	0.03	0.01	0.13
	Fruits and juices		3	0.01	0.02	0.06
	Coffee, tea and cocao		3	0.01	0.05	0.10
	Alcoholic beverages		3	0.002	0.01	0.02
	Tap water		3	0.008	0.04	0.08
	Soft drinks		3	0.007	0.03	0.06
	Total intake			**0.93**	**1.2**	**18**
Number of samples*: 3 different brands/types of each food were pooled						
Note: Food consumption based on Norkost 1997 survey on 2672 adults						
Haug et al., 2010b	Mean dietary intake of a 70 kg person					105
Note: Food consumption based on frequency questionnares						

Fish and shellfish were estimated to contribute with 81% of the total PFOS intake (Haug *et al*., 2010b). In general the level of PFOS in fish is found to be higher than the level of PFOA (Fromme *et al*., 2009). This is in accordance with a recent minor Danish study on PFOS and PFOA in fish from Danish waters where the average level of PFOS was found to be 1,8 ng/kg (n = 9) (Granby, 2012, unpublished data) whereas the level of PFOA was < 0.5 (LOD). In a German study PFOS was detected in 33 wild fish (n = 112) at a concentration up to 225 ug/kg PFOS (Schuetze *et al*., 2010).

PFOS was the PFC (of 11 PFCs analysed) most often detected in especially fish, shellfish, liver and kidney and most often at the highest concentrations in a UK study of 252 food samples (Clarke *et al*., 2009). In 70% of the samples none of the 11 analytes were present above LOD. The highest levels were 59 ug/kg PFOS and 63 ug/kg total PFCs in an eel sample followed by 40 ug/kg PFOS and 62 ug/kg total PFCs in a whitebait sample (Clarke *et al*, 2009).

Intake of *fruit and vegetables* seems to affect the level of PFOS and PFHpS. In a population of northern Norway the intake of PFOS and PFHpS was found to decrease significantly with the increased intake of fruit and vegetables (Rylander *et al*., 2009). The conclusion was based on food frequency questionnaire information from 60 adults (44 women and 16 men) in correlation to PFC levels in blood samples. Similary a study of a Danish birth cohort (n = 1,076) found a decrease in PFOS and PFOA concentrations with increased intake of fruit and vegetable (Halldorsson *et al*., 2008). In the latter study the correlation could be partly explained by a lower intake of red meat, animal fat and snacks. The authors discuss the possibility that the observed correlation between fruit/vegetables and blood PFC levels may be explained by a large number of confounding variables that characterize a healthy lifestyle (Halldorsson *et al*., 2008). It is recommended to include lifestyle factors and dietary patterns instead of single food groups in future studies (Rylander *et al*., 2009).

In tap water samples (n = 3) from the Oslo area the level of PFOS was 0.071–0.23 ng/L and the concentrations of PFBS and PFHS were below this (Haug *et al*. 2010a) as can be seen from Table 3 in Appendix F. In another recent study of PFCs in tap water from six European cities the higest levels of PFSAs were found for PFBS (18.8 ng/L) in tap water from Amsterdam and PFOS (8.8 ng/L) in tap water samples from Stockholm (Ullah *et al*., 2011).

Food packaging materials

Food contact materials may add to the human exposure of PFCs. In a recent Danish study 84 different samples of food pakaging materials of paper and board were tested for per- and polyfluorinate compunds, including PFOS and PFHxS. PFOS was not detected in any of the samples and PFHxS was only found in one sample of popcorn bag at a low level (Trier *et al*., 2012).

Consumer products
Since PFOS was banned in most industrialised countries, the appearance of alternative perfluoroalkyl sulfonates has become more obvious. According to available data these compounds appear to have a main application in fire fighting foams and carpet protection products (Huber *et al*., 2011). However, Herzke *et al*. (2012) reported the detection of PFSAs (analysed were PFOSA, PFBS, PFPS, PFHxS, PFHpS and PFDcS) in several consumer products of different brands. These included, black shoe leather, office furniture leather carpet, paint, non-stick ware, waterproofing agents and coated fabrics. Novec™ from 3M is a fluorosurfactant containing PFBS and is an ingredient in different paints and coatings (3M 2012).

Indoor air exposure
Exposure to PFSAs in the indoor environment occurs mainly through dust. In the Nordic countries, PFSAs have been reported for Norwegian homes and in an office and similarly for Sweden, again for residences and offices. PFOS is the dominant PFSA with concentrations in dust that have reached 147.7 ng/g in a Norwegian office (Huber *et al*., 2011). Very high concentrations of PFOS have been detected also in the Swedish offices analysed by Bjorklund *et al*. (2009). In homes/residences, PFOS ranged between 9.1 and 11 ng/g in Norway and between 39 and 85 in Sweden. The lower levels in residences demonstrate the higher relative importance of occupational exposure compared to exposure in private homes. Among other PFSAs, PFHxS exhibited a concentration of 27.8 ng/g in the Norwegian office (Huber *et al*., 2011), being much higher than in homes (1.4–8.4 ng/g). In indoor air particles from Tromsø, only PFDS was detected (2.6 pg/m3).

The uptake rate has been calculated for PFOS (Haug *et al*., 2011b) and based on three different scenarios, this ranged for Norwegians between 0.11 and 0.46 ng/kg bw/day, through dust, and between 0.004 and 0.36 ng/kg bw/day, through air.

7.2.3 PFAL (Perfluoro aldehydes)

Food and drinking water
No exposure data were found for exposure to humans.

Food packaging materials
No exposure data were found for exposure to humans.

Consumer products
No information on PFAL in consumer products.

7.2.4 FTOH (fluorotelomer alcohols)

Food and drinking water
Data missing.

Food packaging materials
In a recent Danish study 84 different samples of food pakaging materials of paper and board were tested for per- and polyfluorinate compounds by exposure into 50% ethanol. PFCs were found in 35 of the samples. Fluorotelomer alcohols were found in high levels in different types of packaging materials as coffee bags, popcorn bags, and paper and board for take away food and cakes (Trier *et al*., 2012).

Consumer products
A variety of fluorotelomers, including FTOHs, are used in a wide range of commercial products and in some applications, such as fire fighting foams, as well as soil, stain, and grease-resistant coatings on carpets, textiles, paper, and leather, the FTOHs are directly released into the environment (Lehmler, 2005). The manufacture of FTOHs usually results in a mixture containing six to twelve fluorinated carbon congeners and are found in materials such as (see Table 1, Appendix E) Polyfox®, Teflon® Advance carpet protector, Zonyl®, Motomaster® windshield washer and 8:2 Methacrylate (Dinglasan-Panlilio and Mabury, 2006; Herzke *et al*., 2012). Fluorotelomers are also found in Teflon® frying pans, microwave popcorn packing paper, waterproofing agents and Forafac® 1,157 fire fighting foam (Herzke *et al*., 2012; Moe *et al*., 2012; Sinclair *et al*., 2007). In addition, FTOHs are manufactured as a raw material for use in the synthesis of fluorotelomer-based surfactants and polymeric products (Dinglasan-Panlilio and Mabury, 2006).

Indoor air exposure
Exposure to FTOHs can be an important exposure path, because of the volatile nature of FTOHs. It has been shown in indoor exposure studies ([1]Report 2367/2008; [2]Haug *et al*., 2011; [3]Huber *et al*., 2011; [4]Barber *et al*., 2007; [5]Jahnke *et al*., 2007.), that FTOHs in indoor air can reach very high levels and be tens or hundreds of times higher than in the outdoor air (Table 4). Due to the fact that FTOHs have been used in many household products, the primary emissions are expected to take place directly from the indoor environment. To the best of our knowledge, there is no study estimating the uptake of FTOHs due to indoor air occupancies.

7.2.5 FTS (fluorotelomer sulfonates)

Food and drinking water
Data missing.

Food packaging materials
Data missing.

Consumer products
The FTSs are used among other fluorotelomers in fire fighting foam for their film forming properties and the ability to decrease fuel absorption. The quantities of FTSs in the foam are low, but the foam is released directly into the environment (Hagenaars *et al*., 2011a; Moe *et al*., 2012). Although most analysis for FTSs in soil samples are taken in close proximity to airports and airport fire training facilities (Hagenaars *et al*., 2011a; Moe *et al*., 2012), Huber *et al*. (2011) reported for the first time detection of FTSs in in-house dust samples.

7.2.6 PAP/di-PAP (polyfluoroalkyl phosphate esters)

Food and drinking water
Data missing.

Food packaging materials
Paper and board (n = 14) intended for food contact at high temperature were sampled from Danish retailers in 2008. Di-PAPs, tri-PAPs and S-diPAPs were detected in five of 14 samples (Xenia Trier *et al*., 2011).

In a recent Danish study 84 different samples of food pakaging materials of paper and board were analyzed for per- and polyfluorinate compounds, including mono- and di-PAPs (Trier *et al*., 2012). Mono-PAPs were only detected in a few samples and at low levels. Di-PAPs were found in several of the food contact materials tested. The highest level was found in a paper bag for flour containing several different di-PAPs (Trier *et al*., 2012).

7.2.7 Other fluorinated telomers

Food and drinking water
Data missing.

Food packaging materials
Data missing.

Consumer products (Cosmetic, Textiles)
No data were found

7.2.8 Other fluorinated compounds of interest

Food and drinking water
Perfluoroctane sulfonamides were tested in Canadian food (Tittlemier *et al*., 2006). The most frequently detected substance was N-EtPFOSA that was found in 78 of the 151 samples followed by N-MePFOSA in 25 of 51 samples. The highest levels and frequency of detection of analytes were found in fast food composites as particularly in french fries (9.7 ng/g), egg breakfast sandwiches and pizza (27.3 ng/g) (Tittlemier *et al*., 2006).

Food packaging materials
In recent years there has been a shift away from fluorotelomer surfactants towards per- and polyfluorinated polymers, such as per- and polyfluorinated polyethers (PFPEs). On the European market there is currently a shift away from telomeric PFCs to PFPEs as coatings for popcorn bags and on fastfood packaging (e.g. McDonalds) (personal communication with a czech popcorn producer and supplier for 25% or the Nordic market for microwave popcorn bags, 2012). Solvay Solexis is a major producer of PFPEs. In samples taken from Denmark, Sweden and Canada in 2009, PFPEs were found in 7 (18%) of 50 samples by measurement by 19F NMR (Trier, 2011, thesis).

Consumer products
PFPE (Perfluoropolyether, also called PFAE or PFPAE) is a clear, colourless fluorinated synthetic oil that is non-reactive, non-flammable, and long lasting. PFPE is used in greases, oils and lubricants and can be found with the trade names Fomblin® (Solvay plastics) in cosmetics, Molykote® (Dow Corning) in industrial grease, Krytox® (DuPont) in lubricants, Fluorolink® and Galden® (Solvay plastics) for miscellaneous use. Perfluorocarbon emulsions are used as artificial blood or blood substitutes (Goorha *et al*., 2003; Riess, 2002; Riess and Krafft, 1998). The

first commercially available PFC blood substitute was Fluosol® and Oxypherol® from Castro IC and comprised two PFCs, perfluorodecalin (PFD) and perfluorotrypropylamine (PFTPA). PFD is oxygenated using a bubble-through technique with 100% oxygen and infused as a red blood cell substitute (Hoang *et al*., 2009). Chosen for the second generation PFC blood substitutes were PFD, perfluorooctylbromide (PFOB) and bis(perfluorobutyl)ethylene. PFOB is known in its emulsion as Oxygent (Alliance Pharmaceutical Corp.) and is found in the products Columbian emulsion® and French emulsion® from Castor IC.

7.2.9 Conclusion on food and drinking water and consumer products

Food and drinking water

In the last years several papers have been published on PFCAs and PFSAs in food. Based on these data fish is assumed to be a major source of human exposure of PFCs from food. The levels of of PFOS and PFOA in fish from Norway were found to be significantly lower than the levels found in several other studies. According to Norwegian data, cereals (including bread) seem to be another major source to the intake of PFOA and to the total intake of PFCs.

When estimating the human intake of PFCs the intake of e.g. PFOA was found to be significantly lower in the Norwegian population than what has been reported from Spain, Germany, UK, Canada and Japan. This can be due to several parameters related to e.g. differences in consumption pattern and different levels of PFCs in food from different countries as well as uncertainties in estimating the consumption of different foods. Of course analytical uncertainties concerning PFCs have to be considered as well.

The level of PFCs in drinking water can vary a lot, and it may be a significant source of PFCs if the drinking water is produced from surface water in contaminated areas and where drinking water is affected by point contamination, as e.g. by contaminated sludge. In tap water from Stockholm and Oslo PFOS and PFOA were found at lower levels and for several other PFCs the levels are below the LODs. Drinking water seems not to be a major source of PFCs in these countries.

Only very few data are published on non-PFCA and non-PFSA in food. One reason for this is the analytical challenge in analysing these substances and therefore adequate and good performance analytical methods are a great need in this field. Several different PFCs have been found in food contact materials, including PFCA, PFSA, FTOH and PAPs. Food contact

materials may be a significant source of PFC contamination of food. At the time being more data on migration from food contact materials into food of PFCs and especially of non PFCA and non PFSA are needed, to estimate the human exposure of PFCs from food contact materials.

Consumer products

The presence of PFCs in a broad range of consumer products can give rise to a constant diffuse human exposure in the developed parts of the world. Consumer products may therefore be a significant source of PFC exposure to humans, although, estimating exposure via consumer products includes large uncertainties, e.g. brand, volume and number of usage frequency differs between individuals. In addition, the overall human exposure due to PFC treated products might be low in general, but particular sub-groups in the population may receive considerably higher doses than the rest. Direct skin exposure from a skin care product, inhalation of aerosols from an impregnation spray or the use of a blood substitute product may occasionally be important routes of exposure, but are difficult to quantify. Further, information on chemical content of different consumer products is often severely limited, especially on non-PFOS/PFOA PFCs, since the composition of technical applications and mixtures of active ingredients are mostly confidential. Consequently, there is scant knowledge of PFAS content in consumer products and as a consequence we know little about possible emissions of PFAS from consumer products (Dinglasan-Panlilio and Mabury, 2006; Fiedler *et al*., 2010; Herzke *et al*., 2012; Sinclair *et al*., 2007). Literature search gives a certain overview of consumer products both for industrial and personal use available on the market as shown in Tables 1 and 2 in Appendix D. Literature searched included analytical publications where consumer products were analysed and certain PFCs were screened for, as well as the available producer information accessible online. Patents were not included in this overview as they do not necessarily indicate a usage, rather than the compound merely existing. The main products include non-stick cooking ware, coated textiles, footwear, food packaging material, cosmetics, repellent and impregnation products, etc. Final assessment of content of non-PFOS/PFOA PFCs in consumer products indicates a large gap of knowledge.

7.3 Occurrence of PFCs in humans

PFCs are ambiphilic and bind to serum proteins and proteins in cell membranes, and accumulate in blood and internal organ such as liver, kidneys, testes and brain (Jones *et al*., 2003).

Generally the elimination half-life[23] of PFCs in humans is enhanced with decreasing carbon-chain length: PFHxS (8.5 years), PFOS (5.4 years), PFOA (2.3–3.8 years), PFBS (1 month) and PFBA (3 days) (reviewed by (Lau, 2012)).

Most peer-reviewed literature contains reports on perfluorinated alkyl substances (PFAAs)[24] detected in blood (whole blood, plasma and serum) across the world. Blood levels of perfluorinated chemicals have been monitored in many countries and usually PFOS, PFOA, and PFHxS are detected most frequently, and PFOS is detected at the highest concentrations, followed by PFOA and PFHxS. Other PFCs detected in human tissue include PFOSA, Me-PFOSA-AcOH, Et-PFOSA-AcOH or PFOSAA, PFNA, PFDA, PFUA, PFDoA, PFPeA, PFHxA, and PFBS. The short-chain perfluorinated acids are typically not monitored in human sera analysis, but in the case of detection, the concentrations are usually below or close to the limit of quantification (LOQ).

In the following we present the monitoring data found for the Nordic countries (see Tables 9, 10 11 and 12).

Levels in blood

In most studies blood serum is the material analyzed but some studies analyze whole blood or blood plasma. PFC levels in serum and plasma are comparable (1:1) regarding PFOS, PFOA and PFHxS, but levels in whole blood are 2–3 times lower than serum (Ehresman *et al*., 2007).

[23] Half-life is a characteristic parameter of a substance's persistence. If a substance has a half-life greater than two months in water or six month in soil or sediment it is considered as persistent (Annex D , Stockholm Convention).

[24] See appendix A.

7.3.1 PFCA (Perfluorocarboxylic acids) in humans

In the following the monitoring data for PFCA are described for each Nordic country (see Table 9). No biomonitoring data were found for Iceland and Finland.

Norway (NO)

The concentration and time trends of 19 different PFCs for the general Norwegian population were investigated in a cross sectional study by Haug and coworkers (Haug *et al*., 2009). Archived sera from men of age 40–50 years sampled from different county hospitals in Norway during 1976–2007 were pooled (n≥20) and analyzed.

The concentration of PFOA was 2.7 ng/ml in 2006. In most samples, PFNA, PFDA, PFUnDA and PFTrDA (median range: PFNA 0.55 to PFTrDA 0.06 ng /ml) were detected, while PFPeA, PFHpA and PFDoDA were found less frequently. PFBA, PFHxA, PFTeDA were not observed above LOQ in any of the samples.

Trends: The pooled serum samples showed an increase of PFOA (9-fold), PFNA, PFDA, and PFUnDA from 1976 to the mid-1990s where the concentrations stabilized. During 2000 to 2006, the PFOA level decreased approximately by 50%. For PFPeA, PFHpA, PFDoDA and PFTrDA, the concentrations in the serum pools showed no obvious tendencies for change over time; however, the concentrations of these PFCs were close to the LOQ. The authors conclude that the observed increase in PFOA serum concentrations until the mid-1990s are in accordance with the increasing use of products containing PFCs, while the decreasing concentrations observed the past few years are consistent with the phase-out of these compounds (Haug *et al*., 2009).

The median concentration of PFOA among 900 Norwegian pregnant women who were part of the Norwegian Mother and Child Cohort Study (enrolled 2003–2004) was 2.2 ng/mL (Whitworth *et al*., 2012).

In a study of 60 participants in northern Norway (Andøya Island) during 2005, the relationship between dietary intake and PFC concentrations were investigated (Rylander *et al*., 2009). Higher concentrations of PFOA (female: 3.4 ng/ml; male: 5.1 ng/ml) and PFNA (female: 0.77 ng/ml; male: 0.94 ng/ml) were detected in this population which could be attributable to geographical differences (coastal areas) or dietary habits (Rylander *et al*., 2009). PFHpA had more than 95% of the observations below LOD. Higher concentrations of PFOA were found in men. PFNA correlated highly to PFOS.

Rylander and co-workers (Rylander *et al*., 2010), found a range of PFCAs in 315 middle-aged Norwegian women (48–62 years of age);

PFOA (4.4 ng/mL) and PFNA (0.81 ng/mL) were detected in more than 90% of the plasma samples. The concentrations of PFCAs in this study were slightly lower than levels reported from northern Norway (Rylander *et al*., 2009).

The most recent Norwegian study investigated 19 PFCs in serum from 123 pregnant women collected at the Oslo University hospital during 2007 to 2008 from a sub-cohort of the Norwegian Mother and Child Cohort Study (Gutzkow *et al*., 2012). Five PFCAs were detected: PFOA (median = 1.12 ng /ml); PFNA (0.34 ng /ml); PFDA (0.07 ng /ml); PFUnDA (0.16 ng /ml); PFTrDA (0.04 ng /ml). Highly significant correlations ($r > 0.60$, $p < 0.001$) between most of the PFCs were found with the exception of PFUnDA. The levels of PFCAs in this study were very similar to those reported for 41 female volunteers from the Oslo area in Norway in 2008 (Haug *et al*., 2011) (Table 9).

The PFC concentrations from 2007–2008 are the lowest reported in Norwegians and lower than those reported in men in 2006 (Haug *et al*., 2009), probably due to temporal decline in serum levels for many PFCs observed after around year 2000 (Haug *et al*., 2009) or gender differences, or probably different exposure patterns.

Sweden (SE)

Several studies have reported the PFAAs level in the general population in Sweden (Glynn *et al*., 2012; Karrman *et al*., 2007a; Karrman *et al*., 2007b; Karrman *et al*., 2006). The details and PFCA levels are presented in Table 9.

A recent study investigated the temporal trends of blood serum levels of PFCs in primiparous women in the period 1996–2010 living in Uppsala County (Glynn *et al*., 2012). Among the PFCAs, PFOA, PFNA, PFDA, PFUnDA and PFHpA were detected in the pooled samples, whereas PFHxA, PFDoDA, PFTrDA and PFTeDA were below detection limits (Glynn *et al*., 2012). During the period 1996–2010 increasing levels were observed for PFNA (4.3%/year), and PFDA (3.8%/year), whereas level for PFOA decreased (3.1%/year). The study suggested that one or several sources of exposure to PFOA have been reduced or eliminated, whereas exposure to the former compounds has recently increased. The serum levels reported in this study are similar to levels found previously in Swedish blood samples by Karrman *et al*. and other European countries but somewhat lower than reported in the US (Fromme *et al*., 2009).

Similar to Sweden, increasing levels of PFNA and PFDA were observed in plasma/serum among adults in the US (NHANES) during 1999–2008 (Kato *et al*., 2011) whereas, among Norwegian men no significant temporal trends of PFNA or PFDA were observed between 1997 and 2007 (Haug *et al*., 2009). Based on these studies it is not possible to conclude if

the observed upward trend in Sweden is due to increased exposure to directly emitted PFNA and PFDA, or due to increased emissions of precursor compounds such as fluorotelomer alcohols (Glynn *et al*., 2012).

Denmark (DK)

Few biomonitoring studies have been conducted in Denmark measuring the levels of a broad range of PFCs.

A recent study reported the levels of eight different PFCs in serum from young women planning their first pregnancy (collected during 1992–1995) (Vestergaard *et al*., 2012). Among the women who got pregnant (n = 129), the concentrations of PFOA, PFNA and PFDA were 5.61, 0.51 and 0.11 ng/ml, respectively.

Another study reported the levels of PFOA (and PFOS) in 1,399 maternal blood plasma samples collected during 1996–2002 in Denmark (part of the Danish National Birth Cohort). For the first trimester the mean plasma PFOA levels was 5.6 ng/ml (Fei *et al*., 2007; Halldorsson *et al*., 2008).

Joensen *et al*. reported the PFC levels in serum samples from young adult males in Denmark (n = 105) collected in 2003 (Joensen *et al*., 2009). The level of PFOA was 4.9 ng/ml and the remaining PFCAs (PFDA, PFNA, PFHpA, PFUnA and PFDoA) were found in much lower concentrations with medians ranging from 0.9–0.08 ng/ml. The PFCA levels detected in this study were comparable to those found in Sweden.

The current concentrations of PFCs in Denmark are unknown since the latest biomonitoring data found is from 2003 (Joensen *et al*., 2009).

Faroe Islands (FO)

Serum concentrations of 4 PFCAs (PFOA, PFNA, PFDA and PFDoA) were measured in two population groups of whale meat consumers on the Faroe Islands (Weihe *et al*., 2008).

The first group included 12 mothers sampled in 2000 and their 5-year old children sampled in 2005. The second group consisted of 103 serum samples collected during 1993–1994 of 7-year old children and 79 serum samples of these children at age 14 (collected during 2000–2001) (Weihe *et al*., 2008).The 5-year old children had higher concentrations of PFOA levels compared to their mothers 5 years previously (4.5 ng/mL vs. 2.4 ng/ml) and PFNA (1.3 ng/ml vs. 0.6 ng/ml). The concentration of PFDeA was 0.3 ng/ml for both mothers and children. A decrease was found for PFOA between the 7- and 14-year old children (5 ng/ml vs. 4.4 ng/ml), but same PFNA (0.8 ng/ml) and PFDeA (0.3 ng/ml) concentrations were found. This suggested a decrease in PFOA during this time period (1993–2001) on

the Faroe Islands. PFNA concentrations correlated with the frequency of pilot whale consumptions.

On the Faroe Islands, where exposures to marine contaminants via food intake is high, the blood concentrations of PFOA in women were slightly below the average concentrations reported in Danish pregnant women during 1996 to 2002 (Fei *et al*. 2007), but comparable with those for Swedish women (Glynn *et al*., 2012).

Greenland (GRL)

A recent study investigated the level of 10 different PFCs in serum from 284 Inuit belonging to 10 different Greenlandic districts and the temporal trend of blood serum levels of PFCs in Nuuk during 1998–2005 (Long *et al*., 2012).

The detected PFCAs for Inuit women in this study were PFOA (2.57 ng/ml), PFNA (1.33 ng/ml), PFUnDA (1.23 ng/ml), PFDA (0.65 ng/ml), PFTrDA (0.26 ng/ml), PFDoDA (0.15 ng/ml) and pFHpA (0.05 ng/ml). Long *et al*. reported increasing trends for PFNA (28%), PFDA (28%), PFDoA (10%), PFTrDA (13%) during 1998–2005; however these trends disappeared upon age adjustment (Long *et al*., 2012). In this study some correlations between PFCs and legacy POPs (PCBs and organochlorine pesticides) were reported for different non-Nuuk districts. However, for Nuuk Inuit, no significant association was observed between PFCs and legacy POPs, suggesting different sources of exposure other than seafood intake. For non-Nuuk Inuit, significant correlations between serum PFCs and legacy POPs were observed suggesting that there might be common sources for the body burden of PFCs and legacy POPs in non-Nuuk Inuit e.g. marine food intake.

Bonefeld-Jørgensen *et al*. reported the PFC levels in 115 female Inuit controls from Greenland during 2000–2003, in a study investigating the association of PFCs to breast cancer (Bonefeld-Jorgensen *et al*., 2011). PFOA (1.63 ng/ml), PFUnA (1.06 ng/ml), PFNA (0.93 ng/ml), PFDA (0.56 ng/ml), PFDoA (0.15 ng/ml) and PFTrDA (0.15 ng/ml) and PFHpA (0.11ng/ml) were detected in the healthy control samples.

Another study reported the PFC levels in Greenlandic Inuit men (n = 196) from Greenlandic districts, during 2002–2004 (Lindh *et al*., 2012). The male median concentrations for PFOA (4.54 ng/ml), PFNA (1.74 ng/ml), PFUnDA (1.28 ng/ml), PFDA (0.87 ng/ml) and PFDoDA (0.14 ng/ml), were comparable to the levels reported for the Inuit women during 2000–2003 (Long *et al*., 2012) and (Bonefeld-Jorgensen *et al*., 2011) although the PFOA level was lower for women.

These studies show that the Greenlandic Inuit population is highly exposed to several other, more recently industrially introduced PFCs,

such as PFNA, PFDA, PFUnA and PFDoA. This indicates a fast distribution of these compounds to the Arctic area. The levels of PFOA found in Inuit men were similar to the levels found in Denmark and other European countries. In contrast, the PFOA level in Inuit women was only approximately 50% compared to Danish women.

Levels in cord blood

Four studies were found for the Nordic countries where PFCs were measured in cord blood (Table 10).

In the Norwegian study 19 PFCs were investigated in 123 samples of human maternal and cord blood (2007–2008) and up to 5 different PFCAs (PFOA, PFNA, PFDA, PFUnDA and PFTrDA) were detected (Gutzkow *et al*., 2012). The median levels in cord blood had the following % compared to the maternal concentration: for PFOA 79%; for PFNA 35%; for PFDA 57% and for PFUnDA 25%, suggesting that PFOA is transferred to the fetus twice as efficiently as the longer-chained PFNA and PFUnDA. Strong correlations between maternal and cord levels of all the tested compounds were found.

A Swedish study compared the maternal levels of PFOA, and PFNA with the levels in cord blood in 19 samples (1996–1999) (Glynn *et al*., 2012). In cord blood, mean levels of PFOA (1.4 ng/g) and PFNA (estimated to 0.13 ng/g) were considerably lower than those in blood serum from the mothers. Significant positive correlations between maternal serum and cord blood levels were found. The strongest correlations between PFC levels in cord and maternal blood were found for maternal serum samples taken during the third trimester, followed by samples taken 3 weeks after delivery.

Another study from the Faroe Islands measured in year 2000 the PFCs in maternal blood, cord blood and breast milk (Needham *et al*., 2011). Like the other studies they found lower PFC concentrations in cord serum than in maternal serum. PFOA, PFNA, and PFDA revealed good correlation between maternal and cord serum concentrations, with ratios (cord/maternal) of 0.72, 0.50, and 0.29, respectively. The cord/maternal ratio suggested that the length of PFC chain as well as the active group affected the ability to pass the placenta. PFCs with a short chain length showed higher relative cord serum concentrations than PFCs with a longer chain length. PFCs with sulfonic acid as the active group seemed to pass more easily into the fetal circulation than PFCs with carboxylic acid as the active group (Needham *et al*., 2010).

In a Danish study PFOA was analyzed in 50 cord blood plasma samples from women in Danish National Birth Cohort (1996–2002) and the results showed mean concentrations of 3.7 ng/ml for PFOA, which cor-

responded to 66% of the level in maternal serum (Fei *et al*., 2007). Concentrations in cord blood and mother's blood were highly correlated.

The consistent finding of the studies is that cord blood has lower total PFOA than maternal blood; but several PFCs are able to cross the placenta barrier to fetal blood and PFOA seems to cross the placenta most easily. The concentrations of PFOA were highest in Danish and Faroes cord blood probably because the studies are older.

Levels in breast milk

PFCs have also been found in human milk (see Table 11), but in much lower levels than in blood.

In a Norwegian study of matched samples of serum and breast milk (sampled 2007–2008) up to 11 and 2 PFCs were found in the samples of serum (n = 41) and breast milk (n = 19), respectively (Haug *et al*., 2011). Average median breast milk concentration was 0.025 ng/ml for PFOA, which corresponded to 3.8% of the serum concentrations in the mothers.

Thomsen *et al*. studied the elimination rates of PFOA in breast-milk samples from nine Norwegian mothers living in the Oslo area (Thomsen *et al*., 2010). The median concentrations of PFOA in breast milk was 0.05 ng/ml and the PFOA breast milk concentration correlated highly (correlation coefficients: 0.99) with the mothers serum concentrations. During lactation PFOA concentration in breast milk was reduced by 7.8% per month, suggesting lactation as an important route of excretion in mothers.

Kärrman *et al*. (2007) analyzed matched breast milk and serum samples (n = 12) for 7 PFCs during 2004 in Sweden. PFNA and PFOA were detected above detection limits in only one and two milk samples, respectively (Karrman *et al*., 2007a) .

Sundström *et al*. measured the concentration of PFOA in pooled human milk samples obtained in Sweden between 1972 and 2008 (Sundstrom *et al*., 2011). PFOA levels significantly increased from 1972 to 2000 and significantly decreased during 2001–2008. In 2008 the PFOA concentration in the pooled human milk was 0.074 ng/mL. The study showed that the temporal trend in PFOA concentration in pooled human milk samples is similar to the trend in serum concentrations.

A study from the Faroe Islands detected PFOA in breast milk at median concentration 0.1 ng/ml (collected in 2000) which correlated with the the maternal serum PFOA concentrations (r = 0.80) (Needham *et al*., 2011).

For comparison, a study from China (n = 19) reported the presence of PFHpA, PFDA and PFUnDA in human milk from 2004 (Tiido *et al*., 2006). The concentration of PFOA ranged from 47 to 210 ng/L. The maximum concentrations were 62 ng/L for PFNA, 15 ng/L for PFDA, and 56 ng/L for PFUnDA.

Although the levels of PFCs in human milk are relatively low compared to the mothers blood level the exposure of these chemicals to the breast fed infant may be significant because of the relatively high exposure per body weight. It is evident that lactation is an exposure pathway as well as a way for maternal excretion.

Levels in Amniotic Fluids

Only two studies were found measuring PFCAs in human amniotic fluid (Stein *et al*, 2012) (Bonefeld-Jørgensen, Unpublished data) (Table 12).

In an unpublished Danish study (Bonefeld-Jorgensen *et al*.) the concentrations of 8 PFCAs were measured in 54 amniotic fluid samples (*1995–1999*). PFOA, PFHpA and PFDoA were detected in 81.8%, 1.1% and 1.1% of the samples, respectively. The average PFOA level was 0.37–0.29 ng/ml (Bonefeld-Jorgensen *et al*. manuscript in preparation).

Using paired samples from 28 women from the US collected in 2005–2008, the concentrations of 3 PFCAs (PFOA, PFNA, and PFDeA) were measured in serum and amniotic fluid (second trimester) (*Stein, 2012*). The detected carboxylates were PFOA (0.3 ng/ml detected in 24 samples) and PFNA (0.2 ng/ml detected in 10 samples). PFOA showed weaker correlations between serum and amniotic fluid ($\rho = 0.64$). Amniotic fluid concentrations are lower than maternal blood (10–20 fold) and considerably lower than cord blood concentrations. The PFCs detected in the US were lower than those reported from Denmark, probably due to the older Danish samples (1980–1990). PFOA appeared to be more soluble than PFOS because it was detected in amniotic fluid at lower maternal serum levels than PFOS.

7.3.2 Conclusions on human biomonitoring of PFCAs

In the general population the level of PFOA has been increasing until mid-1990s and has then decreased in human serum since 2002. However, for PFPeA, PFHpA, PFDoDA, and PFTrDA no obvious tendencies have been observed (in Norway). In most studies PFOA, PFNA, PFDA, PFUnDA and PFHpA have been detected in human blood whereas PFHxA, PFDoDA, PFTrDA, PFTeDA have been below detection limits. In general, the blood levels are higher in males.

Intake of fish, shellfish, and whale were in some studies identified as determinants of serum concentrations. However, other factors, such as consumer products and indoor air (e.g., house dust in carpeted houses) were also identified to contribute to PFC exposure.

In the Faroe Islands data for 7 and 14-years children indicate a decreasing trend for PFOA during 1993–2003.

In general, comparable levels were observed for the Nordic countries although the newest and lowest levels were found for Sweden and Norway.

Of PFCAs mainly PFOA, and to some extent PFNA and PFDA, were detected in cord blood, but the concentrations are usually lower than concentrations observed in maternal serum or plasma, although the maternal and cord blood data are highly correlated. PFCAs with longer chains are transferred less efficiently to the fetus than those with shorter chain. Detection of PFCAs in cord blood means that some of the compounds can cross the placental barrier and the fetus is prenatally exposed to these compounds.

Of PFCAs only PFOA was detected in breast milk from women in Nordic countries and the concentrations in milk are 3–4% of what is found in the corresponding serum concentrations (Haug *et al.*, 2011). For comparison, in China PFNA, PFDA and PFUnDA in addition to PFOA, were also detected in some samples.

Monitoring studies of PFCAs in amniotic fluids are scarce, but a Danish study detected PFOA and PFNA in amniotic fluids at concentrations 10–20 fold lower than in maternal blood.

FTOHs are identified to be metabolized to PFOA and are thus a source of PFCAs, and an indirect exposure via fluorotelomer-based commercial products or residuals can explain continued exposure to PFOA, together with exposure to PFNA and PFDA, without similar exposure to PFOS.

Further research is therefore needed to determine whether the constant or slowly increasing concentrations of long-chain PFCAs in human serum are primarily a consequence of ongoing exposure to telomer-based precursors.

Table 9. Median concentrations (in ng/mL) of PFCAs and PFSAs in blood (serum /plasma) from Nordic countries

Reference	Coun-try	Samp. Year	n	Sex	Age	Matrix	PFBS	PFHxS	PFHpS	PFOS	PFDS	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	PFOSA	MeFOSAA	EtFOSAA
Vestergaard (2012)	DK	1992–1995	129	F	27	S		1.22		36.3				5.61	0.51	0.11					0.11	0.39	1.79
Eriksen (2011)	DK	1993–1997	652	M	55	P				34.9				6.80									
Fei * (2007)	DK	1996–2002	1,399	F	?	S				35.3				5.60									
Joensen (2009)	DK	2003	105	M	18–19	S		6.60		24.5			0.20	4.90	0.80	0.90	0.10	0.08	0–0.2		0.06		
Haug (2009)	NO	2001[#]	pool	M	40–50	S	<0.05	1.60	0.10	27.00			0.08	4.90	1.20	0.25	0.24	0.05	0.16		0.15		
Haug (2010)	NO	2003	175	M+F	57+55	P		1.70	0.42	25.0			< 0.035	3.60	0.90	0.35	0.47	0.06	0.09		0.22		
Rylander (2010)	NO	2004	326	F	56	P		1.00	0.32	20.0			nd	4.40	0.81						0.02		
Haug (2009)	NO	2004[#]	pool	M	40–50	S	<0.05	1.40	0.12	18.0	<0.05		0.14	3.40	0.78	0.31	0.18	0.06	0.11		0.05		
Rylander (2009)	NO #	2005	15	M	44	P		1.80	0.70	43.0			nd	5.10	0.94						0.11		

Reference	Coun-try	Samp. Year	n	Sex	Age	Matrix	PFBS	PFHxS	PFHpS	PFOS	PFDS	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	PFOSA	MeFOSAA	EtFOSAA
Rylander (2009)	NO #	2005	41	F	44	P		0.80	0.35	24.0			nd	3.40	0.77						0.08		
Haug (2009)	NO	2006[#]	pool	M	40–51	S	<0.05	1.40	0.06	12.0	<0.05		0.08	2.70	0.55	0.22	0.14	0.05	0.07		0.05		
Haug (2011)	NO	2007–2008	41	F	37	S		0.39	0.08	6.7			< 0.035	1.40	0.63	0.23	0.42	< 0.035	< 0.035		< 0.035		
Gützkow (2012)	NO	2007–2008	123	F		P		0.28		5.0				1.12	0.34	0.07	0.16		0.04				
Glynn A (2012)	SE	1996	pool	F	30	S	0.02	2.24		23.3	0.26	nd	0.08	2.69	0.50	0.21	0.16	nd	nd	nd	0.51		
Kärrman (2006)	SE	1997–1999	40	M		WB		1.70		17.7	Nd			2.70	0.30	0.10	0.20				2.70		
Kärrman (2006)	SE	1997–2000	26	F		WB		1.20		16.9				2.10	0.30	0.20	0.10				2.70		
Kärrman (2004)	SE	1997–2000	66	F	19–75	S		3.00		34.2				5.00									
Glynn A (2012)	SE	2000	Pool	F	30	S	<0.01	3.03		22.0	0.05	nd	0.09	2.50	0.38	0.17	0.20	nd	nd	nd	0.44		
Kärrman (2007)	SE	2004	12	F		S	-	4.00	-	18.7	1 sample	nd	nd	3.80	0.63	0.43	0.28	nd		-	0.19		

Reference	Country	Samp. Year	n	Sex	Age	Matrix	PFBS	PFHxS	PFHpS	PFOS	PFDS	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	PFOSA	MeFOSAA	EtFOSAA
Jönsson 2010	SE	2009	50	M		S		0.78		6.9				1.9	0.96	0.41	<LOD						
Glynn A (2012)	SE	2010	Pool	F	30	S	0.10	7.95		7.6	0.01	nd	0.08	1.39	0.59	0.28	0.19	nd	nd	nd	<0.040		
Nilsson H (2010)	SE (ski-wax)	2007–2008	8	M	36	WB	Nd	1.64		12.2		nd	2.80	112	14.7	7.90	0.1–2.8						
Weihe P (2008)	FO (Faroes)	1993–1994	103	Child	7	S		0.40		26.3				5.00	0.80	0.30					1.30	0.40	1.40
Needham (2010)	FO	2000	12	F		S		12.30		19.7				4.20	0.76	0.34							
Weihe P (2008)	FO	2000	12	F		S		*0.6 <LOD*		23.7				2.40	0.60	0.30					0.60	0.90	*0.2 <LOD*
Weihe P (2008)	FO	2000–2001	79	Child	14	S		2.90		31.2				4.20	0.80	0.30					0.30	0.40	1.00
Weihe P (2008)	FO	2005	12	Child	5	S		0.60		16.3				4.50	1.30	0.30					<LOD	0.30	*0.2 (<LOD)*
Long M (2012)	GRL (Nuuk)	1998–2005	5	M	65	S		8.50		74.5			0.52	7.19	5.59	3.13	7.32	0.80	1.39		0.50		
Long M (2012)	GRL (All)	1998–2005	209	F	53	S		2.50		28.5			0.05	2.57	1.33	0.65	1.23	0.15	0.26		0.20		

Reference	Coun-try	Samp. Year	n	Sex	Age	Matrix	PFBS	PFHxS	PFHpS	PFOS	PFDS	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA	PFOSA	MeFOSAA	EtFOSAA
Bonefeld-Jørgensen (2011)	GRL	2000–2003	98	F	54	S		1.66		21.9			0.11	1.63	0.93	0.56	1.06	0.15	0.15		0.13		
Lindh (2012)	GRL	2002–2004	196	M	31	S		2.18		44.7				4.54	1.74	0.87	1.28	0.14					

* Mean concentrations; # Pooled serum samples; F: female; M: male; S: serum; P: plasma; WB: whole blood; LOD: limit of detection; nd: not detecte.

Table 10. Median concentrations (in ng/mL) of PFCAs and PFSAs in cord blood

Reference	Country	Sample year	N	Matrix	PFSAs (ng/ml)					PFCAs (ng/ml)							
					PFBS	PFHxS	PFHpS	PFOS	PFDS	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA
Fei C (2007)*	Denmark	1996–2002	50	Serum				11				3.7						
Needham (2010)	Faroe Island	2000	12	Serum		9.1		6.6				3.1	0.37	0.1				
Glynn A (2012) *	Sweden	1996–2010	19	WB (ng/g)				5.3				1.4	0.13					
Gützkow (2012)	Norway	2007–2008	123	Serum	ND	0.2	ND	1.52	ND	ND	ND	0.88	0.12	0.04	0.04	ND	0.04	ND
Fromme (2010)	Germany	2008–2009	33	Serum	ND	0.2		1				1.4	<0.4	<0.4		ND		

* Mean; WB: Whole blood; ND: not detected.

Table 11. Median concentrations (in ng/ml) of PFCAs and PFSAs in breast milk

Reference	Country	Sample year	n	PFSAs (ng/ml)					PFCAs (ng/ml)							
				PFBS	PFHxS	PFHpS	PFOS	PFDS	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA
Kärrman (2007)	Sweden	2004	12		0.07		0.166				nd	2 sampl	nd	nd			
Haug (2011)	Norway	2007–2008	19	nd	nd	nd	0.087	nd	nd	nd	0.025	nd	nd	nd	nd	nd	
Thomsen (2010)	Norway	2001–2009	68		nd	nd	0.11				0.05	nd	nd				
Needham (2010)	Faroes	2000	12								0.1						
Sundström (2011)	Sweden	2008	1 pool		0.014		0.075				0.074						
Liu J (2011)	China (Jinhu)	2009	50		nd		0.042				0.121	0.019	0.017	0.024	nd		

Table 12. Median concentrations (in ng/ml) of PFCAs and PFSAs in amniotic fluids

Reference	Country	Year	n	PFSAs (ng/ml)					PFCAs (ng/ml)								
				PFBS	PFHxS	PFHpS	PFOS	PFDS	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA
Jensen (2012)	DK	1980–1996	300				1.1										
Bonefeld-Jørgensen (unpublished) [1]	DK	1995–1999	51		0.065 (4.5%)		1.44 (47%)				0.32 (82%)				0.205 (1%)		
Stein (2012)[2]	US (NY)	2005–2008	28		0.4		0.4				0.3	0.2	ND				

[1]Bonefeld-Jørgensen: PFOS, PFOSA, PFOA, PFHxS, PFHpA and PFDoA could be detected in 46.6%. 35.2%.81.8%. 4.5%. 1.1% and 1.1% of the AF samples. The rest were below detection limit.

[2]Stein: PFOA was detected in 24/28.; PFOS in 9/28 and PFDA only in 1 sample.

7.3.3 PFSAs (perfluoroalkyl sulfonates) in humans

Levels in blood

Blood levels of PFSAs have been monitored in some Nordic countries and usually PFOS and PFHxS are detected most frequently, with PFOS having the highest concentrations. Since in many studies both PFCAs and PFSAs were measured the study design of many of the studies are already mentioned under the section for PFCAs.

In the following the monitoring data for PFSAs are described for each Nordic country and data presented in Table 9.

Norway

Time trends of 19 different PFCs for the general Norwegian population were investigated in a cross sectional study by Haug and coworkers (Haug *et al*., 2009). PFOS was found in highest concentrations in all samples, followed by PFHxS. The concentrations in 2006 for PFOS and PFHxS were found to be 12 and 1.4 ng/mL serum, respectively. PFHpS were detected in most samples, while PFBS were found less frequently. PFDS were not observed above the LOQ in any of the samples (Haug *et al*., 2009).

The pooled serum samples showed an up to 9-fold increase of PFOS and PFHpS from 1976 to the mid-1990s. Between 2000 and 2006, PFOS levels decreased by approximately 50%, while PFHxS level decreased by 90%. In this study, PFOS and PFOA were significantly correlated to each other as well as to PFHxS, PFHpS, PFNA, PFDA, and PFTrDA. Correlations between the PFCAs, e.g. PFOA, and PFSAs, e.g. PFOS, indicate common sources for human exposure to these two PFC classes, as they cannot convert directly into each other (Haug *et al*., 2009).

In a study in northern Norway (Andøya Island) during 2005 slightly higher concentrations of plasma PFOS (female: 24 ng/ml; male: 43 ng/ml), PFHxS (female: 0.8 ng/ml; male: 1.8 ng/ml) and PFHpS (female: 0.35 ng/ml; male: 0.7 ng/ml) were found among coastal population compared to the general Norwegian population (Rylander *et al*., 2009). Men had higher concentrations of the PFSAs (including PFOS) but lower proportions of linear PFOS compared to women. The linear isomer of PFOS is most common in technical mixtures and also in human samples and difference in proportions could indicate different exposure sources (Karrman *et al*., 2007b; Rylander *et al*., 2009). High correlation was found between PFOS and PFHpS (r = 0.93). The PFOS and PFHpS concentrations decreased with intake of fruits and vegetables, whereas an increase was observed with intake of fatty fish (Rylander *et al*., 2009).

Intake of fatty fish and intake of fruits and vegetables were not correlated in this study, and therefore the effect of extra intake of fruit could not be fully explained by the authors (Rylander *et al*., 2009).

In the most recent Norwegian study measuring the PFCs in serum from 123 pregnant women from Oslo during 2007–2008 (Gutzkow *et al*., 2012), the PFOS and PFHxS were detected at 5 and 0.28 ng/ml, respectively, and was the lowest reported in Norwegians and other Nordic countries. This could indicate a decline in serum levels or different pattern of exposure for this sub-cohort of the Norwegian Mother and Child Cohort Study.

Sweden

Several studies have reported the PFSAs level in the general population in Sweden (Glynn *et al*., 2012; Karrman *et al*., 2007a; Karrman *et al*., 2007b; Karrman *et al*., 2006).

A study investigated the temporal trends of blood serum levels of PFCs in pregnant women during 1996–2010 living in the Uppsala County (Glynn *et al*., 2012). Increasing levels were observed for PFBS and PFHxS, whereas levels for PFOS and PFDS were decreased. The serum concentrations of the PFOS, PFHxS, PFBS and PFDS in the pooled samples from 2010 were 7.6, 7.95, 0.1 and 0.01 ng/ml respectively. In 2010, PFHxS levels did reach those of PFOS that indicates that the Swedish women have recently been exposed to increasing levels of PFHxS-related compounds from sources that are independent from PFOS exposure (Glynn *et al*., 2012).

In overall the Swedish studies show that the phase-out of PFOS-related chemicals has resulted in decreasing serum concentrations of PFOS in the blood serum of young Swedish women during the past decade. However, exposure to sulfonates with shorter carbon chains than PFOS (or their respective precursors) is currently increasing.

Denmark

For young Danish women planning their first pregnancy (serum collected during 1992–1995), the concentrations of PFOS and PFHxS were 36.3 ng/ml and 1.22 ng/ml, respectively (Vestergaard *et al*., 2012). The median serum levels were similar to most levels reported for U.S. populations during this period.

Comparable PFOS plasma concentrations of 35.1 ng/ml were reported for 1,399 pregnant women during 1996–2002 in Denmark (part of the Danish National Birth Cohort) (Fei *et al*., 2007; Halldorsson *et al*., 2008).

For young adult Danish men in 2003, the median PFOS and PFHxS concentrations were 24.5 and 6.6 ng/mL, respectively (Joensen *et al*.,

2009), and PFOSA was detected in only 56 men (median 0.06 ng/ml). The PFSA levels detected in this study were comparable with those found in other countries such as Sweden (Karrman *et al*., 2007a), but lower than results from the women in Denmark (Fei *et al*., 2007).

Faroe Islands

In paired mother (sampled in 2000) and 5-year old child (sampled in 2005) samples from the whale consuming Faroe Island population, the median concentration of PFOS was 23.7 and 16.3 ng/mL in the mothers and children, respectively (Weihe *et al*., 2008). The PFHxS was detected in only 25% of the maternal samples but in all the children (median 0.6 ng/ml).

In samples from children at age 7 (1993–1994) and again at age 14 (2000–2001), the median concentration increased for PFOS from 26.3 ng/ml to 31.2 ng/ml, and for PFHxS a 3-fold increase was seen in the PFHxS ($p < 0.001$) concentrations (Weihe *et al*., 2008).

Overall, the concentrations of PFOS for the Faroe Island women are slightly below average concentrations reported in Danish pregnant women (Fei *et al*., 2007). Traditional whale meat consumption was suggested to be a major contributor to PFOS and PFNS exposures, while fish intake was associated to PFHxS concentrations (Weihe *et al*., 2008).

Greenland

Long *et al*. investigated the levels of 10 different PFCs in serum from 284 Greenlandic Inuit during 1998–2005 (Long *et al*., 2012). The median concentrations of PFOS and PFHxS in Inuit females were 28.5 and 2.5 ng/ml respectively. No time trends were found for PFSAs, but men had higher PFSA concentrations. The observed serum PFSA concentrations in Inuits corresponded to the range observed in European biomonitoring studies of the general population (Fromme *et al*., 2009).

In another study of female Greenlandic Inuit from 2000–2003, the mean concentrations of PFOS and PFHxS were measured to be 21.9 and 1.66 ng/ml respectively (Bonefeld-Jorgensen *et al*., 2011), and lower than those reported by Long and co-workers (Long *et al*., 2012). This could be due to geographical differences as district differences was observed and also that non-Nuuk Inuit women had significantly lower PFC levels than Inuit women from Nuuk.

For Greenlandic men, the median concentrations of PFOS and PFHxS in 2002–2004 were reported to be 44.7 and 2.18 ng/ml (Lindh *et al*., 2012). Seafood was one of the determinants of PFOS. The level of PFOS reported in this study was higher than those reported by Long *et al* (14.9

ng/ml). However, again geographical differences between the different districts in Greenland was observed e.g. Long *et al* found for men in Nuuk (median, 74.5 ng/ml) and non-Nuuk (median, 13.0 ng/ml).

Levels in cord blood

The studies are also described under the PFCA section above and summarized in Table 10.

In the Norwegian study of 123 samples of human maternal and cord blood PFHxS and PFOS were detected in cord blood at 0.23 and 1.52 ng/ml, respectively (Gutzkow *et al*., 2012). The median levels in cord blood had the following % compared to the maternal concentration: PFHxS 70%; for PFOS 30% suggesting that PFHxS is transferred to the fetus much more efficiently than PFOS.

The Swedish study of 19 samples collected in 1996–1999 detected PFOS (5.3 ng/g whole blood) in the cord blood, which was considerably lower than those found in blood serum from the mothers. Significantly positive correlations between maternal serum and cord blood levels were found (Glynn *et al*., 2012).

The mean concentration of PFOS in 50 cord blood plasma samples from the Danish National Birth Cohort (1996–2002) was 11 ng/ml, corresponding to 30% of the level in maternal serum (Fei *et al*., 2007).

The study from the Faroe Islands in 2000 (Needham *et al*., 2010) found PFOS and PFHxS at 6.60 and 9.10 ng/ml, respectively in the cord blood, which correlated well with the concentrations in maternal blood, with ratios (cord/maternal) of 0.34 and 0.74. The cord/maternal ratio suggested that the length of PFC chain as well as the active group affected the ability to pass the placenta. PFCs with a short chain length showed higher relative cord serum concentrations than PFCs with a longer chain length. PFCs with sulfonic acid as the active group seemed to pass more easily into the fetal circulation than PFCs with carboxylic acid as the active group (Needham *et al*., 2010).

Overall, the studies found correlations between PFSAs in maternal and cord serum, with lower PFSA concentrations in cord serum than in maternal serum. The passage to fetal blood is easier for PFHxS.

Levels in breast milk

The results are summarized in Table 11.

Kärrman *et al*. (2007) analyzed matched breast milk and serum samples (n = 12) during 2004 in Sweden. Of the eight PFASs found in the serum samples, five were detected in the matched milk samples. PFOS and PFHxS were detected in all milk samples at median concentrations of 0.166 ng/mL and 0.07 ng/mL, respectively. Similar PFC occurrence

and levels were found in the milk composite samples collected between 1996 and 2004. The mean ratios between milk and serum (M:S) concentrations were 0.01:1 for PFOS and 0.02:1 for PFHxS. The serum and milk pattern suggested that PFHxS is excreted to milk in a higher degree than PFOS. In 2008, measured concentrations of PFOS and PFHxS in pooled human milk were 0.075 ng/mL and 0.014 ng/mL, respectively showing a decline in the PFSA concentrations from 2004 to 2008.

In the Norwegian study of matched samples of serum and breast milk (sampled 2007–2008) the average median breast milk concentrations were 0.087 ng/ml for PFOS which corresponded to 1.4% of the corresponding serum concentrations for PFOS (Haug *et al*., 2011) i.e. transfer of PFOS to breast milk is lower than for PFCAs.

Elimination rates of PFOS in breast milk from nine Norwegian mothers in the Oslo area was studied by Thomsen and co-workers. The median concentration of PFOS was 0.11 ng/ml. During lactation, PFOS cocentration in breast milk was reduced by 3.8% per month, and by 37% by year.

Sundström *et al*. measured the concentration of PFOS and PFHxS in pooled human milk samples obtained in Sweden between 1972 and 2008 (Sundstrom *et al*., 2011). PFOS and PFHxS demonstrated statistically significant increasing trends in pooled human milk samples from Stockholm over the period 1972–2000. PFOS and PFHxS showed a decline between 2001 and 2008 of approximately 13% and 6 % per year, respectively. In 2008, the measured concentrations of PFOS and PFHxS were 0.075 ng/mL and 0.014 ng/mL, respectively. The study showed that the temporal trend in PFOS and PFHxS concentration in pooled human milk samples is similar to the trend in serum concentrations (Sundstrom *et al*., 2011).

For comparison, in the study of 19 breast milk samples from China (sampled 2004), PFHxS, PFOS were detected in all samples (So *et al*., 2006). Concentrations of PFOS ranged from 45 to 360 ng/L and up to 62 ng/L for PFHxS.

The PFC concentrations in breast milk are far less than those reported in human blood or serum, however there might be a potential risk to infants because of the relatively high exposure per body weight compared to adults.

Levels in amniotic fluids

Only one unpublished and two published studies were found measuring PFSAs in human amniotic fluid (Jensen *et al*., 2012; Stein *et al*., 2012) (Bonefeld-Jørgensen, Unpublished data) (Table 12). Jensen *et al*. studied 300 randomly selected second-trimester amniotic fluid samples from a Danish pregnancy-screening biobank covering 1980 through 1996 (Jen-

sen *et al*., 2012). Only PFOS was measured and the median concentration was 1,1 ng/ml (interquartile range (IQR): 0.66–1.60 ng/mL). For each later gestational week of amniocentesis the PFOS was 9.4% higher (95CI: 3.3%, 15.9%). No associations with maternal age or parity were found.

In the unpublished Danish study (Bonefeld-Jorgensen *et al*.) the concentrations of six PFSAs were measured in 54 amniotic fluid samples (1995–1999). The results showed that PFOS, PFOSA and PFHxS could be detected in 46.6%, 35.2% and 4.5% of the samples. The average PFOS level was 1.35+0.83 ng/ml.

In the US study (sampled in 2005–2008), the concentrations of 2 PFSAs (PFHxS, PFOS) were measured in serum and amniotic fluid (second trimester) (Stein CR 2012). PFSAs in amniotic fluid was detected at concentrations approximately 10–20-fold lower than in maternal serum. PFOS was detected in 24 and PFHxS in 4 of the amniotic samples at median concentrations 0.4 ng/ml. For the detected sulfonates strong correlations were seen between serum and amniotic fluid (ρ = 0.76–0.80).

PFOA appeared to be commonly detected in amniotic fluid if the serum concentration exceeded approximately 1.5 ng/mL whereas PFOS was rarely detected in amniotic fluid until the serum concentration was about 5.5 ng/mL.

Amniotic fluid PFC concentrations are lower than in maternal blood (10–20 fold) and considerably lower than cord blood concentrations. The PFOS levels detected in the US study were lower than those reported from Denmark, probably due to the older Danish samples (1980–1990).

7.3.4 Conclusions on human biomonitoring of PFSAs

Blood levels of perfluorinated chemicals have been monitored in several Nordic countries including in the Arctic, and among PFSAs, usually PFOS and PFHxS are detected most frequently. PFHpS and PFBS have also been measured in the human blood, although not in all samples. PFDS were only observed above the LOQ in the Swedish study. Time trend analyses indicate decreasing tendency of PFOS and PFHxS since 2000, however, in Sweden an increasing tendency was observed for PFHxS from 1996 to 2010. Overall the Swedish studies show that the phase-out of PFOS-related chemicals has resulted in decreasing serum concentrations of PFOS in the blood serum of young Swedish women during the past decade. However, exposure to sulfonates with shorter carbon chains than PFOS (or their respective precursors), such as PFBS, is currently increasing.

Correlations have been seen between PFOS and PFOA[25] which indicate common sources for human exposures. In general, the blood level is higher for males than females.

High intake of fruit suggests lower blood level of PFOS and PFHpS, but seem to be caused by lower intake of fish and meat.

In summary, the studies found a correlation between the maternal and cord blood levels; and lower PFSA concentrations in cord serum than in maternal serum. PFOS and PFHxS were detected in cord blood and the studies showed that PFHxS was passed more easily through the placenta. PFCs with a short chain length seemed to pass the placenta more efficiently than PFCs with a longer chain length. PFCs with sulfonic acid as the active group seemed to pass more easily into the fetal circulation than PFCs with carboxylic acid as the active group.

Among PFSAs, only PFOS has been detected in breast milk in concentrations approximately 1–2% of the corresponding concentrations in serum. Only in the Swedish studies PFHxS was detected.

Few data on PFSAs (PFOS) in amniotic fluid is reported until now. Amniotic fluid PFC concentrations are considerably lower than maternal blood (10–20 fold) and also lower than cord blood concentrations. PFOS was more rarely detected in amniotic fluid until the serum concentration was about 5.5 ng/mL; probably because PFOS appeared to be less soluble in amniotic fluid. For the detected sulfonates strong correlations were seen between serum and amniotic fluid.

7.3.5 PFAL (Perfluoro aldehydes) in humans

No biomonitoring data were found for exposure in humans.

7.3.6 FTOH (fluorotelomer alcohols) in humans

FTOHs are precursor compounds that are known to degrade to PFCAs. The measurement and human exposure to FTOHs has not yet been established but it has been shown that FTOHs (8:2-FTOH and 10:2-FTOH) are metabolized to PFCAs *in vivo* and *in vitro* studies (Dinglasan *et al*., 2004).

Therefore, PFCA can give som indication of the FTOH exposure. Below are some studies on PFCAs and FTOHs in air and associations to PFCA in serum are given.

[25] The correlations were not limited to only PFOS and PFOA.

Studies from Norway and Sweden showed elevated levels of some PFCAs indicators of the FTOH exposure, with carbon chain lengths from C4 to C11 in whole blood from ski technicians using fluorinated ski wax (Freberg *et al*., 2010; Nilsson *et al*., 2010a; Nilsson *et al*., 2010b). The median blood level of PFOA was 112 ng/mL in the Swedish study and 50 ng/ml in the Norwegian study, which is 25–50 times the concentration in the general population in these two countries. For the first time the PFTeDA was found in human serum (Freberg *et al*., 2010). PFNA was the second most abundant carboxylate in blood from wax technicians (Nilsson *et al*., 2001). The levels of the other PFCAs were also much higher than general population except for PFUnDA and PFTrDA (Nilsson *et al*., 2010a).

The PFC measurements of the ski wax working place in Sweden showed that the most dominating compound in the air samples was the 8:2 FTOH (range = 830–250,000 ng/m3), followed by PFHxA (range = 57–14,000 ng/m3) and PFOA (range = 80–4,900 ng/m3) (Nilsson *et al*., 2010a). The levels of the PFCAs found in the serum correlated well with those detected in air samples collected during ski waxing, supporting that inhalation is a major route of occupational exposure. The levels of PFSAs in ski waxes were comparable with those in the Norwegian general population. The authors of the Swedish study suggest that the internal exposure to PFOA was more likely indirect through biotransformation of 8:2 FTOH to PFOA and PFNA in humans (Nilsson *et al*., 2010a).

Another study aimed to investigate the role of indoor office air on exposure to PFCs among office workers (n = 31; 5 men) (Fraser *et al*., 2012). In the newly constructed building the air samples contained mainly FTOHs (8:2 FTOH (9,920 pg/m3); 10:2 FTOH (2,850 pg/m3); 6:2 FTOH (1,320 pg/m3)) and MeFOSE (289 pg/m3) and to a lesser extent FOSAs. In the serum samples they detected PFOS (11 ng/mL), PFOA (3.7 ng/mL), PFNA (1.6 ng/mL), PFHxS (1.5 ng/mL), and PFDeA (0.36 ng/mL). PFUA, NMeFOSAA, N-EtFOSAA, and PFDoA were not detected in all samples and the concentrations were low. They reported a strong positive association between FTOH concentrations in office air and serum PFOA concentrations and weakly between FTOHs in office air and PFNA in serum. Evaluation: the authors suggested that inhalation of FTOH is an important exposure pathway to PFCAs.

7.3.7 FTS (fluorotelomer sulfonates) in humans

No studies concerning biomonitoring for fluorotelomer sulfonates was found for the Nordic countries.

Exposure to legacy and current commercial fluorinated chemicals was investigated by analyzing fifty human sera samples collected in 2009 from the United States (Lee and Mabury, 2011).

The 8:2 FTS was the dominant congener observed in human sera (<LOD (0.005 μg/L)–0.231 μg/L; >95% of the samples), followed by 6:2 FTS (<LOD (0.005 μg/L)–0.047 μg/L; >54%) and 4:2 FTS (<LOD (0.005 μg/L)–0.018 μg/L; <20%). The observation of different perfluoroalkyl chain lengths of FTS in human sera here is consistent with exposure to fluorotelomer-based products. The sources of this contamination may include exposure to commercial products containing the FTS themselves, or to other fluorotelomerthiol-based products, such as FTMAPs.

7.3.8 PAP/di-PAP (polyfluoroalkyl phosphate esters) in humans

No studies concerning biomonitoring for polyfluoroalkyl phosphate esters (PAP/di-PAP) was found for the Nordic countries.

D'eon, JC *et al*. examined pooled human serum samples collected in 2004–2005 (n = 10) and 2008 (n = 10) from the Midwestern US for the 4:2 through 10:2 PAP diesters (diPAPs) (D'Eon *et al*., 2009). The serum samples from 2004 and 2005 contained 4.5 μg/L total diPAPs, with the 6:2 diPAP dominating the congener profile at 1.9±0.4 μg/L. As diPAPs have been shown to degrade to PFCAs *in vivo*, our observation of diPAPs in human sera may be a direct connection between the legacy of human PFCA contamination and PAPs applications.

In serum samples from 2009 lower diPAP concentrations (0.035–0.136 μg/L) for the more dominant congeners (6:2, 6:2/8:2, 8:2) were detected (Lee and Mabury, 2011).

7.3.9 Other fluorinated telomers in humans

No biomonitoring data were found for exposure in humans.

7.3.10 Other fluorinated compounds of interest in humans

Lee H *et al*. investigated the exposure to current commercial fluorinated chemicals by analyzing 50 human sera samples collected in 2009 from the United States for forty fluorinated analytes that included the one fluorotelomer mercaptoalkyl phosphate diester congener (FTMAP), perfluorophosphonates (PFPAs), and perfluorophosphinates (PFPiAs) (Lee and Mabury, 2011). The 6:2 FTMAP were not detected. PFPiAs were detected for the first time in human sera, with C6/C6 and C6/C8 PFPiAs as the dominant congeners, observed in >50% of the samples. Unlike the PFCAs and PFSAs, there are no known PFPiA precursors in production. Therefore, the observation of PFPiAs in human sera gives evidence of human exposure to these chemicals.

No data for fluorotelomer mercaptoalkyl phosphate diester congener (FTMAP) was found for the Nordic countries.

7.3.11 Conclusion on PFAL, FTOH, FTS, PAP/di-PAP and other PFC telomers in humans

No data for PFAL (perfluoroaldehyde) in humans was found.

Data on FTOH in humans is not found since the measurement and human exposure and thus monitoring to FTOHs has not been established. However, it has been shown that FTOHs (8:2-FTOH and 10:2-FTOH) are metabolized to PFCAs in *in vivo* and *in vitro* studies. Strong correlation between PFOA and FTOH are observed, and therefore PFOA serum levels can give an estimation of FTOH exposure and serum levels. Studies in Norway and Sweden showed elevated PFCAs with carbon chain length of C4–C11 in whole blood of occupational ski wax technicians with a level being up to 25–50 times that of the general population.

For office workers in a new building a strong correlation between air FTOH and PFOA serum concentration (more weakly for PFNA) was found.

No data on FTS (fluorotelomer sulfonates) for the Nordic countries was found. Whereas a US study reported that the 8:2 FTS was dominant in human sera, and that the sources may include exposure to commercial products containing FTS or FTMAPs (fluorotelomer mercaptoalkyl phosphate diester).

No data on PAP/di-PAP in humans has been reported for the Nordic countries. Whereas a US study (2004–2008) found 6:2 diPAP to be the dominating compound with decreasing levels observed in samples from 2009. Since diPAPs degrade to PFCA *in vivo* the diPAP serum level can also be indicative of PFCA exposure.

A US study reported in 2009 data on human serum concentration of PFTMAP congeners. The C6/C6 and C6/C8 PFPiAs were the dominating congeners found in more than >50% of the samples. Since there are no PFPiA precursors in production the data give evidence of human exposure to the chemicals.

7.4 Suggested priority list of substances

As a result of the findings concerning occurrence in the Nordic environment and in humans the following priority list of PFCs was agreed upon with KLIF/NORAP:

1. PFCAs, C4 and higher homologues with very low focus on C8.
2. FTOH, C4 and higher.
3. PFSA, C4 and higher with very low focus on C8.
4. FTS, 6.2 and 8:2 (mayby 10:2).
5. diPAP/monoPAP.
6. PFPE.

7.5 Overall conclusion for the human biomonitoring data on PFCA, PFSA and other PFC telomers

For both PFCAs and PFSAs human biomonitoring data for the Nordic countries during the period from 1992 to 2010 are found with most and newest data from Norway and Sweden, and fewest from Denmark. No human data were found for Iceland and Finland.

Overall, a decreasing tendency has been observed for PFOA and PFOS since 2002, whereas, in Sweden it was found that sulfonates with shorter carbon chains than e.g. PFOS or respective precursors is currently increasing.

Some Nordic studies (n = 4) also show that PFAAs can be transferred to the fetus (cord blood) and most efficiently for shorter chain length.

Although the longer carbon chains are known to be more biologically persistent than the shorter carbon chain compounds the daily exposure (including mixture) from several sources must be taken into consideration in risk assessment.

Only one Nordic study from Denmark on PFSAs and PFCAs in amniotic fluids has been published and another DK study has not yet been pub-

lished. PFOS, PFHxS and PFOA were determined in a level being 10–20 folds below the serum level.

Five Nordic studies also show that the PFSAs and PFCAs can be transferred to human breast milk being in the concentration range of 1–2% and 3–4% of the corresponding serum concentration.

7.6 Future work

Thus Nordic studies are needed to follow up on the increase of sulfonates with shorter carbon chains and respective precursors in as well human/maternal blood and cord blood.

Further Nordic studies on different PFSA and PFCA congeners in amniotic fluids and breast milk are needed to explore the real pre-term and post term exposure of the fetus and newborn child.

Studies on the correlation between exposure to FTOH and PFCA at the serum level are needed to explore the general as well as occupational exposure to FTOHs.

Studies are needed for the Nordic countries on PFAL (perflouroaldehyde), FTS (fluorotelomer sulfonates), PAP/di-PAP and FTMAPs (fluorotelomer mercaptoalkyl phosphate diester) since no data are found.

8. Human health effects and related animal toxicity of per- and polyfluorinated substances

Most data found on human and animal toxicology and effects of PFCs are on PFOS and PFOA, probably because existing laboratory procedures in the past did not allow analyses of other PFCs that in general exist in lower concentrations and below detection limits; but also because PFOS and PFOA are the most abundant PFCs in the human matrix. Main epidemiological and medical surveillance studies have been conducted primarily in the United States on workers occupationally exposed to PFOS-based fluorochemicals (e.g. 3M) or populations exposed to PFOA-contaminated drinking water. These studies specifically examined PFOS or PFOA exposures and possible adverse outcomes such as mortality and cancer incidence studies. Further studies have been reported on potential endocrine effects, associations between primarily PFOS and/or PFOA serum concentrations and hematology, hormonal and clinical chemistry parameters. The results of the human studies are summarized in Table 13 and 14.

8.1 PFCA (Perfluoroalkyl carboxylates)

8.1.1 PFBA (C4)

Animal experimental studies

Several studies have been published on the potential biological responses of exposure to PFBA in experimental animals (Chang *et al*., 2008; Das *et al*., 2008; Permadi *et al*., 1992; Takagi *et al*., 1991) and in studies conducted in primary rat hepatocytes (Intrasuksri and Feller, 1991; Intrasuksri *et al*., 1998; Vanden Heuvel *et al*., 1991). In general, PFBA was effective in inducing peroxisome proliferation in these rodent studies but the *in vivo* potency was lower than for PFOA at same doses. *In vitro*,

at the molecular level, PFBA were shown to activate both the human and mouse isoforms of peroxisome proliferator-activated receptor α (PPARα) in a transfection assay system in COS-1 cells (Wolf *et al*., 2008), but again with lower response compared to PFOA. Treatment of mice with PFBA during gestation did not result in any developmental effects as seen for PFOA (Das *et al*., 2008). This difference may be due to lower potency of PFBA to activate PPARα, or due to rapid elimination of PFBA (Chang *et al*., 2008).

A very recent study evaluated the toxicity of PFBA in male and female rats (28-day and 90-day) (Butenhoff *et al*., 2012). Effects in males included: increased liver weight, slight to minimal hepatocellular hypertrophy; decreased serum total cholesterol; and reduced serum thyroxin with no change in serum thyrotropin; no effects were seen in females. No effect on the endocrine system was found.

In summary, rodent exposures to PFBA induce peroxisome proliferation but at a lower level than PFOA. Studies suggested liver toxicity and some effect on the thyroid system. The reported studies suggest minimal developmental and endocrine effects from PFBA exposures.

Human studies
No data found.

8.1.2 PFHxA (C6)

Animal experimental studies
A 90-day repeated dose oral study in rats investigated the possible toxic effects of PFHxA at levels up to 200 mg/kg/day (functional observational battery and motor activity) (Chengelis *et al*., 2009). The observed changes included: lower body weight gains in the 10, 50 and 200 mg/kg/day group males; lower red blood cell parameters; higher reticulocyte counts and lower globulin in the 200 mg/kg/day group of both males and females, higher liver enzymes in males at 50 and 200 mg/kg/day and lower cholesterol, calcium in males at 200 mg/kg/day. Based on liver histopathology and liver weight changes, the NOAEL for oral administration was 50 mg/kg/day for males and 200 mg/kg/day for females. PFHxA was a poor peroxisomal inducer. No reproductive and developmental toxicity was reported for PFHxA (Loveless *et al*., 2009).

In summary, studies suggest that PFHxA exposures of rats affect body and liver weight, changes in liver and hematologic parameters, and that PFHxA is a poor peroxisomal inducer.

Human studies
No data found

8.1.3 PFNA (C9)

Animal experimental studies
A 90-day rat PFNA feeding study with a 60-day recovery period suggested that the liver was the main target organ, with effects on serum clinical chemistry, higher liver weights, and evidence of induced peroxisome proliferation in both males and females. At the end of the recovery period most of the affected parameters had partially or completely returned to normal (Mertens *et al*., 2010).

PFNA has been shown to induce developmental toxicity and liver enlargement in mice when administered throughout the gestational period (Wolf *et al*., 2010). Using a PPAR knockout mouse model Wolf and coworkers showed that the effects of PFNA on survival rate and development of prenatally exposed mice was dependent on expression of PPAR (Wolf *et al*., 2010).

Immune toxic effects have also been observed for PFNA. Exposure of mice to PFNA for 14-days led to a decreased weight of the lymphoid organs, and the study suggested immune toxic effects on lymphoid organs and T cell (Fang *et al*., 2008). Cell apoptosis in rats can be caused by PFNA (Fang *et al*., 2010). Recently, PFNA was suggested to disrupt the hepatic glucose metabolism by increasing the levels of rat serum glucose and hepatic glycogen via altering the expression of the genes related to glucose metabolism and suppressing the hepatic insulin pathway (Fang *et al*., 2012).

In summary, studies suggest that PFNA exposures of rodents are related to liver toxicity including disrupting glucose metabolism; PPAR dependent effects on development and survival upon *in utero* exposure and PFNA induced immune toxicity. A proposal to classify PFNA as a reproductiv toxicant in the EU has been submitted by Sweden (ECHA, 2012a).

Human studies: Occupational
One occupational study of exposures to a PFNA surfactant blend was undertaken. The PFNA study examined liver enzymes and blood lipid levels (Mundt *et al*., 2007) in a cohort consisting of 630 employees at a U.S. polymer production facility using PFNA between 1989 and 2003. The exposure was assessed by work history and not by measuring the PFNA concentrations in the serum. Seven parameters of liver function (total cholesterol, Gamma glutamyl transpeptidase, transminases, alkaline phosphatase, bili-

rubin, and triglycerides) were examined and they showed no significant effect. This study did not either find any effect on thyroid function as assessed by serum levels of TSH, T4, fT4 and T3 uptake.

The authors concluded that based on laboratory measures assessed over more than a decade; no adverse clinical effects were detected from occupational exposure to a PFNA blend.

Human studies: general exposures (PFNA)

- *Lipid metabolism and cholesterol*
  In a cross-sectional study of general US population (NHANES data; 2003–2004) Nelson *et al*. investigated the relationship between exposure to PFNA (also PFOA, PFOS and PFHxS), and cholesterol levels, obesity and insulin resistance (Nelson *et al*., 2010). Total cholesterol and non-HDL were positively associated with PFNA levels.
  Another study from NHANES data (1999–2000 and 2003–2004) examined the relationship between PFCs and components of the metabolic syndrome in participants (Lin *et al*., 2009). They found that in adolescents, increased serum PFNA concentrations were associated with decreased blood insulin and clinical hyperglycaemia, increased serum HDL cholesterol, and was inversely associated with the prevalence of metabolic syndrome.
- *Thyroid function*
  In a study of New York State anglers (n = 31), potential associations were investigated between serum concentrations of 5 measured carboxylates (PFDA, PFNA, PFHpA, PFOA, PFUnA) and levels of TSH and free T4 (Bloom *et al*., 2010). No statistically significant associations were found for PFNA (mean concentration 0.79 ng/mL (0.66–0.96)).
- *Reproductive effects*
  A recent study evaluated the possible association between PFC exposure and biomarkers of male reproductive health in a larger population of almost 600 men from Greenland, Poland and Ukraine (Toft *et al*., 2012). PFNA (median: 1.2 ng/mL) was not associated with sperm concentration, volume, total count or percent motile sperm. However, PFNA was associated with a non-significant lower proportion of normal sperm at higher exposure.
  A study evaluated the association of female serum concentrations of different PFCs (PFOA, PFNA, PFDA) with time to pregnancy (TTP) in 222 Danish first-time pregnancy planners (prospective design) (Vestergaard *et al*., 2012). No clear association between PFNA (median concentration 0.45 ng/mL) and TTP were observed.

- *Developmental effects*
  In a small Canadian study (n = 101; sampling date 2004–2005) no association was found between birth weight and maternal PFNA levels (Monroy *et al*., 2008).
  A US study examined the association between PFC exposure and ADHD among children (Hoffman *et al*., 2010). Data from this NHANES study (n = 571; age 12–17; from 1999–2000 and 2003–2004) showed increased odds of ADHD disease with higher serum PFNA level (Odds ratio OR 1.32).
- *Immune system*
  A study investigating childhood infection (immunoglobulin E) found no significant correlation between cord blood PFNA and serum total IgE or cord blood IgE in 244 2-year old Taiwanese children (Wang *et al*., 2011).
  In summary, there are very few data on the suspected health effects of PFNA and further studies are needed to explore the adverse effects of PFNA in human.

8.1.4 PFDA (or PFDeA) (C10)

Animal experimental studies

Animal studies have identified PFDA as toxic to the liver. Administration of single gavage doses of ≥20 mg/kg PFDA to female C57BL/6N mice resulted in significant and dose-related increases in relative liver weight, assessed 30 days after dosing (Harris and Birnbaum, 1989; Harris *et al*., 1989). Significant elevations in liver weight (69%) were seen in mice 2 days after treatment with 40 mg/kg PFDA (Brewster and Birnbaum, 1989), which also significantly increased hepatic acyl-CoA oxidase activity and hepatic lipids. Kawashima *et al*. (Kawashima *et al*., 1995) compared the effects of lower dietary doses of PFDA (1.2–9.5 mg/kg/day) and PFOA (2.4–38 mg/kg/day) on hepatic effects in male rats in a 7-day dietary study. PFDA was considerably more potent than PFOA in reducing body weight gain, food consumption, causing hepatomegaly, and inducing biochemical markers of peroxisome proliferation.

Other reported animal effects of PFDA: increasing serum cholesterol in male rats exposed to 10 mg/kg/day (Shi *et al*., 2007); 2-and 4-fold increases in serum T3 and T4, respectively, 30 days after a single dose of 80 mg/kg PFDA to female C57BL/6N mice (Harris *et al*., 1989); reduction of body weight (33%) in male C57BL/6 mice (78 mg/kg/day) (Permadi *et al*., 1992); potent inducer of the estrogen-responsive biomarker protein vitellogenin (Vtg) in juvenile rainbow trout (Benninghoff *et al*.,

2011), decreased testicular androgen production (plasma testosterone and dihydrotestosterone levels) in male rats (Bookstaff *et al*., 1990).

In summary, the animal studies have demonstrated that the liver is the primary target for PFDA toxicity including inducement of the peroxisome proliferation and acyl-CoA oxidase activity; the PFDA had a higher toxic potency than PFOA. Furthermore, effects on the thyroid and testicular androgen level were observed in mice and rats, respectively. A proposal to classify PFNA as a reproductiv toxicant in the EU has been submitted by Sweden ((ECHA), 2012b).

Human studies: general exposures (PFDA)

- *Thyroid function*
  In the study of New York State anglers (n = 31), PFDA was above the LOD for 65% of the samples (mean 0.21 ng/ml (0.18–0.26)). No associations were found between serum concentrations of PFDA and levels of TSH and free T4 (Bloom *et al*., 2010). However the authors suggested that there was a possibility of weak positive associations between FT4 and PFDA by use of a larger sample size (Bloom *et al*., 2010).
- *Reproductive effects*
  A study evaluated the association of female serum concentrations of different PFCs (PFDA, PFOA, PFNA) with time to pregnancy (TTP) in 222 Danish first-time pregnancy planners (prospective design) (Vestergaard *et al*., 2012). No clear association between PFDA (median: 0.10 ng/mL) and TTP was observed.
- *Developmental outcome*
  No associations were found between PFDA exposure and birth weight in a Canadian study (Monroy *et al*., 2008) or between PFDA exposure and ADHD among children from US NHANES study (Hoffman *et al*., 2010).
- *Immune system*
  A study investigating childhood infection (immunoglobulin E) found no significant correlation between cord blood PFDA and serum total IgE or cord blood IgE in 244 2-year old Taiwanese children (Wang *et al*., 2011).
  In summary, there are very few data on the suspected health effects of PFDA and further studies are needed to explore the adverse effects of PFDAs in human.

8.1.5 PFUnDA/ PFUnA (C11)

Animal experimental studies
In vitro, PFUnA could at low level activate the mouse PPARα in transiently transfected COS-1 cells (Wolf *et al*., 2012).
No further animal experimental toxicity data were found.

Human studies (PFUnA)
In the study of New York State anglers (n = 31; PFOA mean conc. 1.33 ng/mL), no associations were found between serum concentrations of among others PFUnA and levels of TSH and free T4 (Bloom *et al*., 2010). However the authors suggested that there was a possibility of weak positive associations between FT4 and PFUnA by use of a larger sample size (Bloom *et al*., 2010).

In two Korean studies, no correlations were found between PFUnA exposure and TSH and T4 levels (Ji *et al*., 2012) in the general population and neither between PFUnA in maternal blood serum and T3/T4/TSH of fetal cord blood serum (Kim *et al*., 2011).

Very few studies have measured and evaluated the PFUnA in the human studies, and they found no correlations to the assessed health effects.

8.1.6 PFDoA or PFDoDA (C12)

Animal experimental studies
Treatment of male rats with PFDoA by gavage for 14 days significantly induced an increase in total serum cholesterol (10 mg/kg/day) and reduction in body weight (5 mg/kg/day). PFDoA exposure at 5 mg/kg/day or 10 mg/kg/day resulted in testis cell (Leydig and Sertoli) apoptosis and a decline in serum estradiol and testosterone levels (Shi *et al*., 2007). Another study showed that 110 days of PFDoA exposure led to significantly decreased testosterone and expressional changes of testicular steroidogenic genes in rats (Shi *et al*., 2009). The authors suggested that testosterone decline may be involved in the pathway of cholesterol transportation and steroidogenesis, and that these pathways were disrupted in testes following PFDoA exposure (Shi *et al*., 2007).

In summary, rodent studies suggest that PFDoA exposures decreases body weight, causes liver and testes toxicity, including changes in an array of parameters such as serum cholesterol and testes hormone level.

Human studies
In the few studies that measured the PFDoA, there was not found any association with thyroid hormone, semen quality or lipid metabolism (Joensen *et al*., 2009; Kim *et al*., 2011) (Halldorsson *et al*., 2012). In some of these studies PFDoA was found in few samples above LOD.

8.1.7 PFTrDA (C13)

Animal experimental studies
No data on the toxicity of PFTrDA were found.

Human studies
In the Korean studies, investigating the exposure levels of 13 PFCs, PFTrDA were detected in >90% of maternal serum samples (0.39 ng/mL (0.27–0.57). One study observed that only PFTrDA concentrations were positively correlated with TSH levels, but negatively with total T4 concentrations among the female population (Ji *et al*., 2012). A positive association was detected between PFTrDA in maternal blood serum and T3 and T4 of fetal cord blood serum and between PFOA and cord serum TSH (Kim *et al*., 2011).

In another study where PFTrDA was measured, PFTrDA was under LOD in many of the samples (Joensen, 2009).

8.1.8 PFOA (C8)

Animal experimental studies
There is a considerable amount of animal data on the health effects of PFOA, which has been reviewed (Lau *et al*., 2007). The relevance of animal data for humans is controversial because of a much shorter half-life in rodents (measured in days) and the possible dependence of some animal toxicity on a peroxisome proliferation mechanism that is likely to be less important in humans (Steenland *et al*., 2010a).

In its draft risk assessment, the U.S. EPA (2005) concluded that evidence was suggestive that PFOA is carcinogenic in humans. In its review of that risk assessment, three of the four members of the EPA scientific advisory board concluded more strongly that PFOA was "likely to be carcinogenic in humans" (U.S. EPA 2006). There has been submitted a proposal for harmonised classification and labelling of PFOA/APFO in EU by Norway. In December 2011 the Risk Assessment Committee of the ECHA came to the conclusion that classification according to regulation EC No. 1272/2008 for PFOA is Repr. 1B and STOT RE 1 (ECHA, 2011).

- *Tumor induction*
  PFOA was not mutagenic or genotoxic in the classic battery of test for genotoxicity and mutations detected by the Ames test and structural chromosome damage (Fernandez Freire *et al*., 2008). However, in human HepG2 cells this compound exerted genotoxic effects as a consequence of oxidative stress (Yao and Zhong, 2005).
  In rodents dietary intake of PFOA induced tumors of the testicles, liver, and pancreas (Biegel *et al*., 2001; Sibinski, 1987). A 2-year study in rats reported a statistically significant increase in mammary fibroadenomas and Leydig cell adenomas suggesting impact of PFOA on reproductive tissues (Sibinski, 1987). In 2007 White and coworkers reported gestational exposure to PFOA in mice was associated with altered mammary gland development in dams and female offspring (White *et al*., 2007).
- *Hepatotoxicity*
  The liver is the primary target organ by the exposure of animals to PFOA (and PFOS). PFOA has been shown to induce peroxisome (microbodies) proliferation in mouse and rat liver, and causes hepatomegaly. This proliferation has been shown to alter lipids, liver enzymes, and liver size (Kennedy *et al*., 2004; Lau *et al*., 2007; Takagi *et al*., 1992). Peroxisome induced proliferation and the resulting activation of a nuclear receptor peroxisome proliferator-activated receptor α (PPARα) have also been proposed as a mechanism for tumor induction and for the immune and hormonal changes seen in rodents (Lau *et al*., 2007). Other proposed mechanism for tumor promotion is by inhibition of gap-junctional intercellular communication (GJIC)(Upham *et al*., 2009). GJIC plays a vital role in maintaining tissue homeostasis, and disruption of gap junction function can lead to diseased states such as tumorigenesis. PFOA was shown to decrease GJIC activity in the liver of rats treated for both acute and long-term dosing (Upham *et al*., 2009).
- *Developmental toxicity*
  The developmental toxicity of PFOA (and PFOS) has been examined in rats and mice (Butenhoff *et al*., 2004; Hinderliter *et al*., 2005; Lau *et al*., 2006). Generally, the developmental toxicity induced by exposure to PFOA throughout gestation included increased neonatal mortality, reduced postnatal survival, delayed eye opening, growth deficits, and sex-specific alterations in pubertal maturation (Lau *et al*., 2006). A cross-foster study indicated that the postnatal effects on survival, eye opening, and weight gain were a consequence of gestational exposure and that exposure via lactation was not a major

factor (Wolf *et al*., 2007). Using a PPARα- knockout mouse model it was shown that PFOA developmental toxicity was dependent on expression of PPARα (Abbott *et al*., 2007).

- *Immunotoxicity*
  In rodents PFOA decreases the B-cell and T-cell immune responses and results in atrophy of the spleen and thymus (Yang *et al*., 2002), causes hepatomegaly (Takagi *et al*., 1992), and decreases levels of cholesterol (Kennedy *et al*., 2004).
- *Endocrine disruption*
  Several experimental studies have reported that PFCs impair thyroid hormone homeostasis. Effects of PFOA on thyroid hormones are not as well characterized as those of PFOS. Depression of serum triiodothyronine (T3) and/or thyroxine (T4) in PFOA exposed rats and monkeys has been reported (Butenhoff *et al*., 2002; Lau *et al*., 2007), but without an expected corresponding elevation of thyroid-stimulating hormone (TSH) through feedback stimulation of the hypothalamic–pituitary–thyroid axis. Increases in estradiol and decreases in testosterone with PFOA exposure have also been observed in rodents (Lau *et al*., 2007). In addition to thyroid hormone disruption, changes in sex steroid hormone biosynthesis have been reported. Administration of PFOA to adult male rats for 14 days led to a decrease in serum and testicular testosterone and an increase in serum estradiol levels (Biegel *et al*., 1995), which was suggested to be associated with aromatase induction in the liver.
- *Neurotoxicity*
  Not much data was found for neurotoxicity of PFCAs including PFOA. Neonatal exposure of mice to PFOA (and PFOS) affected the proteins involved in neurogenesis and synaptogenesis in the developing mouse brain, which were accompanied by neurobehavioral defects in adulthood (Johansson *et al*., 2008). Also in an avian model PFOA was reported to be a developmental neurotoxicant (Pinkas *et al*., 2010).
  In summary, PFOA exposures in rodent indicate carcinogenic responses in liver, pancreas, testes and mammary glands. Furthermore, PFOA exposure in animals affect the development and reproduction, thyroid and immune system negatively; induces peroxisome proliferation believed to be a mechanism behind tumor initiation and effect on immune- and hormone systems.

Human studies: Occupational exposures

The first retrospective cohort study on mortality of employees of the PFOA-producing factory (3M) demonstrated a significant association between prostate cancer mortality and employment duration (Gilliland and

Mandel, 1993). In an updated study the previously found association between prostate cancer and time of employment could not be confirmed.

However, in a later study of exposed workers in the 3 M factory in Minnesota, ammonium perfluorooctanoate (APFO) exposure was presumably associated with prostate cancer, cerebral vascular disease, and diabetes mellitus, but not with liver, pancreas, or testicular cancer (Lundin *et al*., 2009).

Based on the occupational exposures there were reported no significant associations between serum PFOA and reproductive hormones in men (Olsen *et al*., 1998). In the study of Olsen and Zobel (2007) PFOA was not statistically significantly associated with total cholesterol or low-density lipoproteins. High-density lipoprotein (HDL) and free T4 were negatively associated with the PFOA level, whereas triglycerides and T3 tended to be positively associated (Olsen and Zobel, 2007). Several studies have investigated the relation to liver toxicity. The liver enzymes, transaminase levels, were positively associated with PFOA serum concentrations in some studies (Olsen and Zobel, 2007) but not in others (Sakr *et al*., 2007), indicating controversial hepatotoxic effects.

Human studies: Communities with high exposed population

In the United States two communities in Minnesota and West Virginia/Ohio have been exposed via water contamination coming from adjacent industrial plants (water mean levels for PFOA: 15 ng/ml in Minnesota and 82 ng/ml in Ohio in 2005, respectively). Several studies describe the health effects of the PFOA in the "C8 Health Project" (http://publichealth.hsc.wvu.edu/c8/) which has data on 69,030 persons, and provide an opportunity to examine a very large population with high exposure median (26.6 ng/mL of serum vs. 4ng/mL in the general U.S. population). The results are summarized in Table 13.

The following associations have been seen for PFOA in the "C8 Health Project": increased total cholesterol and low-density lipoprotein in children and adolescents (Frisbee *et al*., 2010); increased blood lipid levels in relation to elevated PFOA (and PFOS) concentrations in the blood (Steenland *et al*., 2009); no associations to HDL cholesterol; positive associations with serum and liver enzymes (transaminase; a marker of hepatocellular damage) indicating hepatotoxic effect in humans (Gallo *et al*., 2012); positive association to serum uric acid (Steenland *et al*., 2010); no association between PFOA and TSH (n = 371) (Emmett *et al*., 2006); significant positive elevation in serum T4 and a significant reduction in T3 uptake in adults (Knox *et al*., 2011a); no associations with preterm birth and fetal growth restrictions (Savitz *et al*., 2012), positive association with hypothyroidism in children (Odds ratio (OR): 1.54; 95%

confidence interval (CI): 1.00, 2.37) (Lopez-Espinosa *et al*., 2012); associations with Attention Deficit Hyperactivity Disorder (ADHD) in children 5–18 years of age, with a small increase in prevalence for the second quartile of exposure and a decrease for the highest versus lowest quartile (Stein and Savitz, 2011); more likely to have experienced menopause among perimenopausal women with higher level of PFOA (and PFOS), suggesting endocrine disrupting effects (Knox *et al*., 2011b); association with lower serum concentrations of IgA and IgE (for IgE in females only) (C8 Science Panel, 2009).

The C8 Science Panel has concluded its work in determining whether there is a probable link between exposure to C8 (PFOA) and a range of human diseases (http://www.c8sciencepanel.org/prob_link.html). The Science Panel did find a Probable Link between exposure to C8 and medically-diagnosed high cholesterol, and thyroid disease, testicular cancer and kidney cancer, and pregnancy-induced hypertension (elevated blood pressure in pregnancy).

The Science Panel found no Probable Link between C8 and the following diseases: Parkinson's disease, non-malignant liver disease, non-malignant kidney disease, osteoarthritis, coronary artery disease or high blood pressure, adult onset diabetes, chronic obstructive pulmonary disease, asthma, childhood and adult infections such as influenza, neurodevelopmental disorders in children, stroke, and five other autoimmune diseases (lupus, rheumatoid arthritis, Type 1 (juvenile) diabetes, Type II (adult-onset) diabetes, Crohn's disease, and multiple sclerosis, risk of pregnancy loss, either miscarriage or stillbirth, preterm or low birth weight infants, measures of prematurity.

In summary, the Community high exposed population studies support the occcupational studies on hepatoxic and thyroid toxic effects of PFOA. In addition, association with ADHD in children, endocrine and immunotoxic effects and testis and kidney cancers was reported.

Human studies: General population

Few epidemiological studies exist with data from the general population. Below are summarized the adverse health effects observed for PFCAs (Table 14).

- *Lipid metabolism and cholesterol*
  The discovery that PFCs bind to the PPARs (nuclear receptors that play a key role in lipid metabolism) have raised the concern that the PFCs may disrupt lipid and weight regulation. The occupational studies have suggested a positive association between PFOA and levels of cholesterol in humans.

In a cross-sectional study of general US population (NHANES data; 2003–2004) Nelson *et al*. investigated the relationship between exposure to PFOA and PFNA (and also PFOS and PFHxS), and cholesterol levels, obesity and insulin resistance (Nelson *et al*., 2010). Total cholesterol and non-HDL was positively associated with PFOA (and PFNA and PFOS) level. The median serum concentration in NHANES was 4 ng/mL.

Another study from NHANES data (1999–2000 and 2003–2004) examined the relationship between PFCs and components of the metabolic syndrome in participants (Lin *et al*., 2009). They found no correlation to PFOA, but in adolescents, increased serum PFNA concentrations were associated with decreased blood insulin and clinical hyperglycaemia, increased serum HDL cholesterol, and was inversely associated with the prevalence of metabolic syndrome.

In a Danish study mothers who were overweight or obese before pregnancy had higher plasma levels of PFOA (and PFOS) (Fei *et al*., 2007), suggesting a relation between BMI and PFOA levels. In addition, a recent prospective study of pregnant women and their children 20 years later (n = 665; maternal serum from 1988–1989) showed that *in utero* exposure to PFOA was positively associated with the prevalence of overweight and a high waist circumference at 20 years in female but not male offspring (Halldorsson *et al*., 2012).

In summary, the present studies on human exposure to PFCAs suggest some liver toxicity with changes in parameters involved in metabolic syndrome such as lipids, cholesterol levels and non-HDL, with the risk of obesity and insulin resistance.

- *Cardiovascular diseases*

  Melzer *et al*. found no trend in self-reported history of heart disease in adults from the NHANES, after dividing PFOA serum levels into quartiles (Melzer *et al*., 2010). However, in a recent study from NHANES, higher serum PFOA levels were positively associated with self-reported cardiovascular diseases including coronary heart disease and stroke, and objective peripheral arterial disease (Shankar *et al*., 2012).

- *Cancer*

  A follow-up study of the general population in Denmark (55,053 Danish adults; 50–65 years of age; 1993–2006) found no clear differences in incidence rate ratios for bladder and liver cancers in relation to plasma concentrations of PFOA; although a modest positive associations were reported for prostate and pancreas cancers (Eriksen *et al*., 2009). A recent case-control study of Greenlandic Inuit (31 cases, 115 controls; collected in 2000–2003)

showed a significant association between serum PFOA (sum of PFCs) and risk of breast cancer (Bonefeld-Jorgensen *et al*., 2011).
In summary, PFCAs might be risk factors in prostate, pancreas and breast cancers, but further studies are needed.

- *Thyroid function*
  In a study of New York State anglers (n = 31; PFOA mean conc. = 1.33 ng/mL), potential associations were investigated between serum concentrations of 5 measured carboxylates (PFDA, PFNA, PFHpA, PFOA, PFUnA) and levels of TSH and free T4 (Bloom *et al*., 2010). No statistically significant associations were found for any of the PFCs or the sum of them. A recent cross-sectional analysis of self-reported thyroid disease in the NHANES (n = 3,974 adults) reported a significant association for PFOA and thyroid disease in general US population and particularly in females (Melzer *et al*., 2010). More women with blood concentrations of ≥5.7 ng PFOA/mL were found to have currently treated thyroid disease compared with women having ≤ 4.0 ng/mL of blood levels.
  In a Korean study, investigating the exposure levels of 13 PFCs, they observed no association to PFOA, but only PFTrDA concentrations was positively correlated with TSH level, but negatively with total T4 concentration among the female population (Ji *et al*., 2012). In summary, possible association for PFOA and thyroid diseases is suggested. Further studies are needed before any conclusion can be taken.
- *Reproductive effects*
  A Danish study (n = 105) showed an association between PFOA (and PFOS) exposure and the proportion of morphologically normal sperm cells (Joensen *et al*., 2009). However, an American study evaluated the semen quality among 256 infertility patients in relation to PFOS and PFOA in serum and semen and found no association between PFOS or PFOA levels and sperm concentration, volume or motility (Raymer *et al*., 2012).
  A recent study evaluated the possible association between PFC exposure and biomarkers of male reproductive health in a larger population of almost 600 men from Greenland, Poland and Ukraine (Toft *et al*., 2012). Sperm concentration, total sperm count and semen volume was not consistently associated with PFOA.
  Couple fecundity and time to pregnancy (TTP) were evaluated in the Danish National Birth Cohort study (n = 1,240) (Fei *et al*., 2009). The evaluation of TTP showed increased risk of irregular menstrual cycles in the upper three quartiles of PFOA relative to the lowest quartile and an increase in mean PFOA with longer TTP. The odds of

infertility (≥ 12 months without conception) were elevated in the upper three quartiles of PFOA. Similar patterns were reported for PFOS (Fei *et al*., 2009). Among Norwegian pregnant women having given birth before, increased odds of sub-fecundity were associated with high PFOA and PFOS (Whitworth *et al*., 2012b). Among nulliparous women, higher PFC plasma level was associated with a decreased odd of subfecundity (Whitworth *et al*., 2012b).
Another study evaluated the association of female serum concentrations of eight different PFCs (PFOS, PFOA, PFHxS, PFNA, PFDA, MeFOSAA, EtFOSAA and FOSA) with TTP in 222 Danish first-time pregnancy planners (prospective design) (Vestergaard *et al*., 2012). Inconsistently with earlier observations no clear association between any of the measured PFCs and TTP were observed, which might be related to study design and the parity status of the women.
In summary, controversial data on PFCA effects on human semen quality is reported, further studies must evaluate whether the compounds such as PFOA or PFNA affect the sperm output. Some studies suggest that PFCAs (PFOA, PFOS) affect the time to pregnancy and fecundity of females.

- *Developmental outcome*
  The most extensive set of studies has examined foetal growth, birth weight, duration of gestation, and related indices of *in utero* development (Apelberg *et al*., 2007; Fei *et al*., 2007, 2008; Hamm *et al*., 2010; Hoffman *et al*., 2010; Monroy *et al*., 2008; So *et al*., 2006; Washino *et al*., 2009; Maisonet *et al*., 2012).
  In the Danish National Birth Cohort study (DNBC) (n = 1,400 mother/child), Fei *et al*. investigated the possible correlations between the concentration of PFOA (and PFOS) in the maternal blood during the first and second trimesters of pregnancy and the birth weight and risk of premature birth (Fei *et al*., 2007). Only PFOA levels were inversely associated with birth weight. Gestational length was unaffected by PFOA concentrations. A statistically non-significant inverse association was also observed between PFOA and head circumference, and a positive association with newborn ponderal index (like BMI for babies) (Fei *et al*., 2008). Also another study from the US (n = 293) reported levels of cord blood PFOA (and PFOS) to be inversely associated with birth weight, new-born head circumference, crown–heel length, and ponderal index (Apelberg *et al*., 2007).
  Data from a subgroup of the Norwegian Mother and Child Cohort study (n = 901 women; 2003–2004) showed lower birth weight among infants born to mothers in the highest quartiles of PFOA (and PFOS)

compared to lowest quartiles (Whitworth *et al*., 2012a).
Also a British study found that girls born to mothers with maternal serum concentrations of PFOA in the upper tertile weighed less (130 g (95%CI: –237, –30)) at birth compared with girls born to mothers with serum concentrations in the lower tertile (Maisonet *et al*., 2012).
Contrary to the above findings, 2 smaller Canadian studies found no evidence of decreased birth weight and maternal PFOA levels (Monroy *et al*., 2008;Hamm *et al*., 2010).
In summary, in general Nordic and British studies suggest that PFOA affects the foetal growth negatively, whereas smaller Canadian studies did not see this effect.
Developmental milestones of children were examined in the sub-study of the Danish National Birth Cohort (n = 1,400), where early pregnancy plasma PFOA levels were unrelated to motor or mental development through 18 months of age (Fei *et al*., 2008). Beside the study from "C8 Health project" in USA (Stein and Savitz, 2011), another study did examine the association between PFC exposure and ADHD among children (Hoffman *et al*., 2010). Data from this NHANES study from 1999–2000 and 2003–2004 (n = 571; age 12–17) showed increased odds of ADHD disease with higher serum PFOA and PFNA levels (OR1.12 and 1.32, respectively).
Further studies are needed to explore the suspected effects of PFCAs on CNS and child development including behavior and ADHD.

- *Immune system*
  There are few epidemiological studies on the effects of PFOA on immune function related to infectious disease. A Danish study examined the prenatal exposure to PFOA (and PFOS) and the association with the infectious diseases in children (Fei *et al*., 2010) in the Danish National Birth Cohort. Hospitalizations for infection of the children were not associated with prenatal exposure to PFOA and PFOS. The authors concluded that their data did not support the hypothesis that prenatal PFOS and PFOA exposures decrease resistance to childhood infections.
  Wang *et al*. (2011) found that cord blood PFOA was positively associated with cord blood serum IgE in 2-year old Taiwanese boys, whereas Okada *et al*. (2012) found that maternal serum PFOA was negatively associated with cord blood IgE in newborn Japanese girls However, in the latter study no relationship was found between maternal PFOA levels and infant allergies and infectious diseases at age in 18 months (Okada *et al*., 2012).
  Recently, another prospective study of birth cohort from the National

Hospital in the Faroe Islands (n = 656) reported a negative correlation with antibody response to tetanus and diphtheria booster immunizations at age 5 years with increasing serum PFOA (and PFOS) (Grandjean *et al*., 2012). The conclusion was that elevated exposures to PFAAs were associated with reduced immune response to vaccination.

Further studies are needed to reject or document the few studies on the suspected effect on the human immune system.

8.1.9 Summary findings for PFCAs

Overall summary of animal toxicity upon PFCA exposure

PFBA (C4) exposures in rodents induce peroxisome proliferation but at a lower level than PFOA. Study suggested liver toxicity and some effects on the thyroid system. The reported studies suggest minimal developmental and endocrine effects for PFBA exposures.

PFHxA (C6) exposures in rats affect body and liver weight, changes in liver and hematologic parameters, and PFHxA is a poor peroxisomal inducer.

PFNA (C9) exposures in rodents are related to liver toxicity including glucose metabolism; immune toxicity, and PPAR dependent effects on development and survival upon *in utero* exposure in knock out mouse. A proposal to classify PFNA as a reproductiv toxicant in the EU has been submitted by Sweden

PFOA (C8) exposure in rodent causes carcinogenic responses in liver, pancreas, testes and mammary glands. Furthermore, PFOA exposure in animals affect the development and reproduction, thyroid and immune system negatively; induces peroxisome proliferation believed to be a mechanism behind tumor initiation and effect on immune- and hormone systems.

PFDA (C10) exposures in animal studies have demonstrated that the liver is the primary target including inducement of the peroxisome proliferation and acyl-CoA oxidase activity; the PFDA had a higher toxic potency than PFOA. Furthermore, effects on the thyroid and testicular androgen level were observed in mice and rats, respectively. A proposal to classify PFDA as a reproductiv toxicant in the EU has been submitted by Sweden.

No animal experimental toxicity data were found for PFUnA (C11), however *in vitro*, at a low level PFUnA (C11) activated the mouse PPARα in transiently transfected COS-1 cells.

PFDoA (C12) exposures in rodents decrease the body weight, causes liver and testes toxicity, including changes in an array of parameters such as serum cholesterol and testes hormone level.

Overall summary for human health effects upon PFCA exposure

Data on *occupational exposures* to PFOA (C8) are controversial but suggest possible relation to the risk of prostate cancer, cerebral vascular disease, and diabetes mellitus, but not liver, pancreas, or testicular cancer. Moreover, there might be effects on steroid and thyroid hormone levels, whereas the relation to liver toxicity is controversial. Based on laboratory measurements assessed over more than one decade; no adverse clinical effects were detected from occupational exposure to PFNA (C9).

Data on *community high-exposed population* studies support the occcupational studies on hepatoxic and thyroid toxic effects of PFOA. In addition, association with ADHD in children, endocrine and immunotoxic effects, testis and kidney cancers and pregnancy-induced hypertension was reported.

For the *general population* the present studies on human exposure to PFCAs suggest some liver toxicity with changes in parameters involved in metabolic syndrome such as lipids, cholesterol levels and non-HDL, with the risk of obesity and insulin resistance.

Higher serum PFOA levels were found positively associated with self-reported cardiovascular diseases including coronary heart disease and stroke, and objective peripheral arterial disease. Moreover, PFCAs might be risk factors in prostate, pancreas and breast cancers, but further studies are needed.

Although further studies are needed, a possible association between PFOA and PFTrDA and *thyroid diseases* as well as weak positive relations between thyroid effects and FT4, PFDA and PFUnDA was suggested.

Controversial data on *reproductive factors* have been reported such as the effect of PFCA (PFOA, PFNA) on human semen quality and sperm output and need further studies. Some studies suggest that PFCAs (PFOA) affect the time to pregnancy and fecundity of females.

Some Nordic and British studies suggest that PFOA affects the *foetal growth* negatively, whereas some Canadian studies did not see this effect. Moreover, one Danish study on *developmental milestones* up to 18 month did not find any association to plasma PFOA levels; whereas one US study found an increased odds of ADHD disease with higher serum PFOA and PFNA levels at age 12 to 17. Further studies

are needed to explore the suspected effects of PFCAs on CNS and child development including behavior and ADHD.

Further studies are needed to reject or document the few studies on the suspected effect on the human immune system.

8.2 PFSA (Perfluoroalkyl sulfonates)

The epidemiological studies of general population are summarized in Table 14.

8.2.1 PFBS (C4)

Animal experimental studies

PFBS (K+PFBS) was assessed for developmental and reproductive effects in a two- generational rat study (Sprague Dawley) (Lieder *et al*., 2009b). The study showed that maternal exposure to PFBS did not adversely affect the reproductive function in Sprague Dawley rats at doses as high as 1,000 mg/kg/day or developmental outcomes at doses as high as 300 mg/kg/day. In both the parental and F1-generation male, there were increased liver weight and histological changes (increased cell proliferation) in kidneys in the 300 and 1,000 mg/kg/day dose group rats. Similar kidney effects were reported in the 90-day study (Lieder *et al*., 2009a), and the authors discussed in that article that these changes likely were due to high concentrations of PFBS, a strong surface active compound, passing through the kidney, as urine is the major excretory route for PFBS (Olsen *et al*., 2009). In the 90-day study other observed effects included decreased red blood cell count, hemoglobin, and hematocrit at 200 and 600 mg/kg (Lieder *et al*., 2009a).

In summary, mild effects at the liver, kidney and blood parameters have been observed in rat studies upon exposure to PFBS (C4) at relatively high doses.

Human studies (PFBS)

No data found.

8.2.2 PFHxS (C6)

Animal experimental studies

Only few published studies were found regarding potential toxicological properties of PFHxS in experimental animals.

A reproductive and developmental toxicity study of PFHxS was conducted in rats by York (York *et al*., 2010) (Butenhoff *et al*., 2009). In parental males there was a significant reduction in cholesterol already at doses 0.3 mg/kg/day, and hepatotoxicity at doses 3 mg/kg/day. No treatment-related effect was reported on the fertility and reproductive outcomes or on viability and growth of the offspring at doses as high as 10 mg/kg/day. A NOAEL of 10 mg/kg/day was therefore estimated for the developmental effects of PFHxS.

Human studies (PFHxS)

- *Thyroid function*
  Among Koreans, PFHxS were not correlated to total T4 or TSH levels (Ji *et al*., 2012; Kim *et al*., 2011). In a recent study of self-reported ADHD in children, increasing PFHxS levels were associated with increasing prevalence of ADHD (adjusted odds ratio of 1.59) (Stein and Savitz, 2011).
- *Reproduction*
  A recent study evaluated the possible association between PFC exposure and biomarkers of male reproductive health in a larger cross country population including 588 men from Greenland (n = 106), Poland (n = 189) and Ukraine (n = 203) (Toft *et al*., 2012). For PFHxS a 35% (95% CI: 1; 70%) lower proportion of normal sperm were found at the highest tertile compared with the first, and a non-significant decrease in the proportion of normal sperm was also observed at the second tertile.
- *Lipid metabolism and cholesterol*
  In the cross-sectional study with NHANES data (Nelson *et al*., 2010) an inverse relationship between exposure to PFHxS and total cholesterol levels were observed. Another study from NHANES data (1999–2000 and 2003–2004) examined the relationship between PFCs and components of the metabolic syndrome in participants (Lin *et al*., 2009). They found no associations for PFHxS in adults.
- *Developmental effects*
  A British study found that girls born to mothers with maternal serum concentrations of PFHxS in the upper tertile weighed less (-108 g; 95% CI: –206 to –10) at birth compared with girls born to mothers with serum concentrations in the lower tertile (Maisonet *et al*., 2012).

Two other smaller Canadian studies did not find any associations between maternal PFHxS levels and fetal weight and length of gestation (Hamm *et al*., 2010; Monroy *et al*., 2008).
A study examined the association between PFCs and ADHD among children in the US (Hoffman *et al*., 2010). Data from this NHANES study from 1999–2000 and 2003–2004 (n = 571; age 12–17) showed significant increased odds of ADHD disease with higher serum PFHxS level (OR 1.06).

- *Immune system*
  Two studies investigated the childhood infection (immunoglobulin E) and found no association to cord blood PFHxS in 2-year old Taiwanese boys (n = (Wang *et al*., 2011) and newborn Japanese infants (Okada *et al*., 2012).
  A recent prospective study of a birth cohort from the National Hospital in the Faroe Islands (n = 656) reported a small negative correlation with antibody response to tetanus and diphtheria booster immunizations at age 5 years with increasing serum PFHxS (Grandjean *et al*., 2012).

8.2.3 PFHpS (C7), PFNS (C8) and PFDS (C10)

No toxicology studies were found in animals or human.

8.2.4 PFOS (C8)

Animal experimental studies

PFOS is classified as Repr. 1B and Carc.2 (Harmonised classification – Annex VI of Regulation (EC) No 1272/2008 (CLP Regulation)) (ECHA, 2008) and it was added to Annex B of the Stockholm Convention on Persistent Organic Pollutants in May 2009.

The toxicity of PFOS has been extensively studied, and numerous studies have been conducted including toxicological studies in multiple species. The results have been reviewed in an OECD report (2002) and Lau *et al*. (Lau *et al*., 2007; Lau *et al*., 2004). A summary of the important findings are given here.

Repeated-dose studies in rats and nonhuman primates have shown decreased body weight, hepatotoxicity, reduced cholesterol, and a steep dose-response curve for mortality. In a study performed by 3M, refereed in the OECD-report, male and female rats were exposed to PFOS in diet for 104 weeks (0.5 ppm–20 ppm). The study showed that PFOS induced a small increase in the incident of tumors in liver, and thyroid and

mammary glands (OECD, 2006). The NOAEL for male and female was considered to be 0.5 ppm and 2 ppm in diet respectively, which corresponds to approximately 0.03 mg/kg/day and 0.15 mg/kg/day. PFOS has not been shown mutagenic in a variety of assays. Two-generation reproductive toxicity studies in rats showed neonatal mortality (Lau *et al*., 2004; Luebker *et al*., 2005a).

The maternal and developmental studies of PFOS in rats and mice showed both maternal and developmental toxicity. Pregnant Sprague-Dawley rats and CD-1 mice were given 1–20 mg/ kg/day from gestation day (GD) 2 to GD 20 and GD 1 to GD 17 respectively (Thibodeaux *et al*., 2003;Lau *et al*., 2003). The major findings on the mothers were a reduction in serum T4 and T3, without effects on TSH. The mice dam experienced a reduction in serum triglycerides and an elevation in liver weight at a dose of 1 mg/kg/day. 50% of the newborn rats and mice died within 24 hours when prenatally exposed to 3 mg/kg/day and 10 mg/kg/day respectively. Serum T4 levels were suppressed in the PFOS-treated rat pups, although T3 and TSH levels were not altered. Delays in growth and development were observed (Lau *et al*., 2003). Luebker *et al*. (2005b) showed that maternal exposure up to 1.6 mg/kg/day was a critical dose leading to approximately 50% mortality among prenatally exposed pups within 4 days after delivery.

A two-generation reproduction toxicity study in rats showed no effects on reproduction in F0 females or their fetuses (on mating, estrous cycling and fertility) (Luebker *et al*., 2005a). However, reproductive outcome, such as decreased length of gestation, number of implantation sites, and increased numbers of dams with stillborn pups or with all pups dying on LDs 1–4, was affected at 3.2 mg/(kg day) and neonatal toxicity, as demonstrated by reduced survival and body-weight gain, occurred at a maternal dose of 1.6 mg/kg/day (Luebker *et al*., 2005a).

In a series of studies pregnant mice have been exposed to PFOS in order to evaluate the behavioral effects on the offspring. The studies are summarized by Mariussen (2012). Neonatal exposure of PFOS at specific time points, at the period of high neuronal growth, was shown to induce behaviour effects in adult mice. The exposure appeared to involve an effect on the development of the cholinergic system.

In summary, in rodents PFOS is related to decreased body weight, liver toxicity, induction of PPAR-alpha, and has adverse reproductive effects at the fetal and at the neonatal level. PFOS increases developmental mortality and affectsa the thyroid system and may induce neurobehavioral effects, particularly in developmentally exposed animals.

Human studies (PFOS)

Except for PFOS, there are very few data on the health effects of other PFSAs, probably due to lower concentrations in human blood. Since in most studies there was a high correlation between PFOS and PFOA, many of the health effects in the epidemiological studies that were observed and summarized under the PFOA section, is also mentioned below.

Data on liver function, serum cholesterol and thyroid hormone levels have been collected and associated with levels of PFOS in serum of occupationally exposed workers. However, we will not focus any further on this.

The epidemiological studies of general population are summarized in Table 14.

- *Cancer*
  Grice *et al*. were unable to detect an association between occupational PFOS exposure and the occurrence of skin, breast, prostate, or intestinal cancer in workers at a PFC-producing company (Grice *et al*., 2007).
- *Thyroid hormone*
  In Inuit adults (n = 623), PFOS concentrations were negatively associated with TSH and total T3 and positively with free T4 concentrations (Dallaire *et al*., 2009). Maternal PFOS correlated negatively with fetal T3 (Kim *et al*., 2011). PFOS was shown to compete with T4 for binding sites on human transthyretin (Weiss *et al*., 2009), which may also lead to a reduction in total thyroid hormone concentrations in the blood. Also in children serum PFOS concentrations were positively correlated with total T4 concentration (Lopez-Espinosa *et al*., 2012). An earlier study of 15 mother–infant pairs in Japan reported no association between a median 2.5 ng/mL cord blood concentration of PFOS, approximately 13% of the current study median, and newborn TSH or FT_4 levels (Inoue *et al*., 2004). A recent cross-sectional analysis of self-reported thyroid disease in the NHANES (n = 3,974 adults) reported a significant association for PFOS (PFOS ≥ 36.8 ng/mL) and current treated thyroid disease in men (Melzer *et al*., 2010).
  In summary, although the data are controversial, reported studies suggest negative effects of PFOS on the thyroid system and on the risk of ADHD.
- *Reproduction*
  A recent study evaluated the possible association between PFC exposure and biomarkers of male reproductive health in a larger cross country population including 588 men from Greenland

(n = 106), Poland (n = 189) and Ukraine (n = 203) (Toft *et al*., 2012). Higher PFC blood levels were found in Greenlandic Inuit. Cross country for the three populations and for the two European populations alone an increase in sperm cell morphology defects at increasing PFOS exposure was found, but not analyzing the Inuit population from Greenland alone. Cross country the proportion of morphological normal cells was 35% lower [95% confidence interval (CI): 4–66%) for the third tertile of PFOS exposure as compared with the first. Thus, the study suggests a concentration dependent effect and maybe diet/race dependent effect of PFOS on semen quality. The results of the present study are supported by the findings of effects of PFCs on sperm morphology reported in a previous smaller Danish study (Joensen *et al*., 2009).

- *Lipid metabolism and cholesterol*
  No explicit changes in liver enzymes, cholesterol, or lipoproteins in serum could be detected in the serum of workers with PFOS concentrations below 6 mg/L (Olsen *et al*., 1999).
  A study from NHANES data (1999–2000 and 2003–2004) examined the relationship between PFCs and components of the metabolic syndrome in participants (Lin *et al*., 2009). They found that in adults, increased serum PFOS concentrations were associated with increased blood insulin, insulin resistance status, beta-cell function, increased serum HDL cholesterol.
- *Developmental effects*
  Some studies have reported inverse associations between PFOS and birth weight. In a study from US (n = 293) cord blood PFOS levels was significantly associated with decreases in birth weight and size but not newborn length and gestational age (Apelberg *et al*., 2007).
  Data from a subgroup of the Norwegian Mother and Child Cohort study (n = 901 women; 2003–2004) showed slightly lower birth weight among infants born to mothers with the highest plasma levels of PFOS (Whitworth *et al*., 2012a).
  A British study found that girls born to mothers with maternal serum concentrations of PFOS in the upper tertile weighed less (-140 g (95%CI: −238 to −42)) at birth compared with girls born to mothers with serum concentration in the lower tertile (Maisonet *et al*., 2012).
  Also in a study from Japan (n = 428), maternal PFOS levels correlated negatively with birth weight (-148.8 g (95%CI: −297.0 to −0.5),particularly in female infants. (Washino *et al*, 2009).
  In contrast, in the Danish National Birth Cohort study (DNBC) (n = 1,400 mother/child), maternal PFOS was not associated with birth

weight and risk of premature birth (Fei *et al*., 2007), and suggested that the associations might be related to PFOA.
Two other smaller Canadian studies did also not find any association between maternal PFOS and fetal weight and length of gestation (Hamm *et al*., 2010; Monroy *et al*., 2008).
Developmental milestones of children were examined in the sub-study of the Danish National Birth Cohort (n = 1,400). No convincing associations were found beside that children who were born to mothers with higher PFOS levels were slightly more likely to start sitting without support at a later age (Fei *et al*., 2008). Another study did examine the association between PFCs and ADHD among children (Hoffman *et al*., 2010). Data from this NHANES study from 1999–2000 and 2003–2004 (n = 571; age 12–17) showed significant increase odds of ADHD disease with higher serum PFOS (OR 1.03).

- *Immune system*
  A Danish study examined the prenatal exposure to PFOA (and PFOS) and the association with the infectious diseases in children (Fei *et al*., 2010) in the Danish National Birth Cohort (n = 1,400).
  Hospitalizations for infection of the children were not associated with prenatal exposure to PFOS.
  Two studies investigated the childhood infection (immunoglobulin E): cord blood PFOS was positively associated with cord blood serum IgE in 2-year old Taiwanese boys (n = (Wang *et al*., 2011), but not in newborn Japanese infants (Okada *et al*., 2012).
  A recent prospective study of birth cohort from the National Hospital in the Faroe Islands (n = 656) reported a negative correlation with antibody response to tetanus and diphtheria booster immunizations at age 5 years with increasing serum PFOS (and to a lesser extend PFHxS) (Grandjean *et al*., 2012). The conclusion was that elevated exposures to PFAAs were associated with reduced immune response to vaccination.

8.2.5 Summary for PFSAs

Summary for animal toxicity upon PFSAs exposure
Most studies reported are on PFOS exposures, and very few studies on effects by short chain substances have been conducted.

In rodents PFOS (C8) is related to decreased body weight, liver toxicity and is a PPAR-alpha inducer. PFOS has adverse reproductive effects at the fetal and the neonatal level. PFOS increases developmental mortality and affects the thyroid system.

Upon exposure to PFBS (C4) at relatively high doses, mild effects at the liver, kidney and blood parameters were observed in rat studies.

Few studies on PFHxS (C6) toxicity in experimental animals have been conducted and need further research. A single study found hepatoxicity in rats, whereas no reproductive effect on fertility and offspring outcome was observed at relatively high PFHxS exposure.

Summary for human health effects of PFSA

Except for PFOS, only some Nordic and in general few data on the health effects of other PFSAs are found.

Occupational studies found an association between PFOS exposure and liver function, serum cholesterol, thyroid hormone levels and skin, breast, prostate, or intestinal cancer in workers at a PFC-producing company.

General population studies: A cross country study on European and Inuit populations found an increase in sperm cell morphology defects at increasing PFOS and PFHxS exposure, but not for the Inuit population from Greenland alone. This observation is supported by a smaller Danish study.

Reproductive and developmental effects: An inverse association between maternal PFOS (and for some studies PFHxS) and birth weight were reported for studies in US, Norway and England, but not significantly for the Danish birth cohort and the smaller studies in Canada. Concerning developmental milestones weak association was found in the Danish National birth cohort. In the arctic Inuit adult PFOS exposure was positively related to changes in thyroid factors but not for PFHxS. However, PFHxS levels were positively related to ADHD prevalence in children. Thus, although controversial data, the reported studies suggest negative effects of PFOS on the thyroid system and PFHxS as a risk factor in development of ADHD. In support to the Inuit study a US study found a relationship between the serum PFOS and PFHxS levels at age 12–17 and the risk of ADHD. However, further studies are needed to elucidate the possible risks.

A single study reported a link between PFOS exposure and *metabolic syndrome with effects* such as increased blood insulin, insulin resistance status, beta-cell function, increased serum HDL cholesterol. In addition, a cross-sectional study in the US found an inverse relationship between exposure to PFHxS and total cholesterol levels.

A few studies indicated an effect of PFOS on the immune system: In the Faroe Islands, a negative effect on the vaccination response in children at 5–7 year; and a Japanese study found an association between childhood infection (immunoglobulin E) and cord blood PFOS levels in 2-year old boys. However, no significant association between prenatal exposure to PFOS and infectious diseases in children in the Danish National Birth Cohort was found.

8.3 FTOH (fluorotelomer alcohols)

Limited information is currently available on the toxicological effects and the risks of FTOHs in experimental animals and humans. However, these compounds are metabolically converted to PFCAs including PFOA and therefore may be associated with the induction of hepatic peroxisome proliferation and acyl-CoA oxidase (ACOX) activity. There is a proposed classification under CLP regulations for 8:2 FTOH by Norway in 2012 as reproductive toxicant.

8.3.1 Toxicity of FTOH in experimental animals

Fluorotelomer alcohols have been shown to metabolize into PFCAs in rodents (Fasano *et al*., 2009; Kudo *et al*., 2005). Therefore, they may potentially have the same health effects as PFOA and some other PFCAs (PFNA and PFHpA).

Dietary administration of 8:2 FTOH to mice (7–28 days) resulted in liver enlargement in a dose-dependent manner. Peroxisomal acyl-CoA oxidase was induced by these treatments (Kudo *et al*., 2005). Five metabolites (PFOA, PFNA, 8-2 telomer acid and two unidentified) were present in the liver and serum of the treated mice. The concentration of PFOA positively correlated to the activity of peroxisomal acyl-CoA oxidase in the liver of mice, which suggest that rather PFOA than 8:2 FTOH itself produced the effects.

In a 90-day oral repeated dose toxicity study (Ladics *et al*., 2008) the liver and kidney were target organs. Doses of 25 mg/kg or greater increased relative liver weight, hepatic β-oxidation (females only at 25 mg/kg, both sexes at 125 mg/kg), and induced focal hepatic necrosis. No evidence of a neurotoxic response was reported. The NOAEL in this study was 5 mg/kg.

In a reproductive toxicity study with FTOH mixture [F(CF2)xC2H4OH, x = 3–6; (27% 8:2 FTOH)] by oral gavage at 0, 25, 100 or 250 mg/kg/day, pup weights were reduced at ≥100 mg/kg/day (Mylchreest *et al*., 2005a); in the developmental part of the study (0, 50, 200, or 500 mg/kg/day), maternal bodyweight was reduced and there was an increase in fetal skeletal alterations at 200 mg/kg/day. There were no effects on oestrous cycle parameters sperm morphology and motility or epididymal sperm counts in the P1 generation. NOAEL for the FTOH mixture (27% 8:2 FTOH) reproductive toxicity was 25 mg/kg/day based on litter size reduction. In another study Mylchreest *et al*. (2005b) investigated the developmental effects of 8:2 FTOH at the same doses as above. Increases in skeletal vari-

ations were reported from 200 mg/kg bw/day as for the FTOH mixture. Severe maternal toxicity was reported at 500 mg/kg bw/day including mortality. The NOAEL for both maternal and developmental toxicity was sat to be 200 mg/kg/day.

Overall, the liver appears to be the most sensitive target organ for 8:2 FTOH toxicity based on the available studies. However, no carcinogenic studies were available and the reproductive and developmental studies are insufficient to draw any conclusions. Based on the toxicity effects of PFOA (major serum metabolite of FTOH), there is proposed a classification for 8:2 FTOH equal to PFOA (http://echa.europa.eu/).

In an *in vitro* study investigating oxidative damage that may contribute to testicular toxicity, primary rat testicular cells were exposed to 8:2 FTOH and 6:2 FTOH. The study showed no significant increase in oxidative DNA damage (Lindeman *et al*., 2012).

In summary, fluorotelomer alcohols and/or there metabolites (e.g.PFOA) can induce liver toxicity in rodent, and the potency of endocrine disruption are demonstrated in *in vitro* cell culture and *in vivo* fish studies.

Gap of knowledge: Fluorotelomer unsaturated aldehydes (FTUALs) and acids (FTUCAs) are intermediate metabolites that form from the degradation of FTOHs. Their potential for toxicity is not yet defined and may be more significant compared to PFCAs. These intermediates can form adducts with glutathione (GSH) (Rand and Mabury, 2012).

8.3.2 Human health effects of FTOH

In vitro studies using human cells have reported following endocrine effects for fluorotelomer alchohols: 8:2 FTOH and 6:2 FTOH had potential estrogenic effects by inducing the proliferation of MCF-7 breast cancer cells in E-screen assay; by up-regulating of the estrogen receptor (Maras M *et al*., 2006); by interacting with the human estrogen receptor alpha and beta *in vitro* (Ishibashi H *et al*., 2007). Moreover, 8:2 FTOH showed the capacity to decrease testosterone levels in the human adrenal corticocarcinoma cell line (Liu *et al*., 2010b) but did not interfere with androgen receptor activation (Rosenmai *et al*., 2012). The authors assumed that the effect was due to FTOH and not the metabolite PFOA, as the hormone profile of these two compounds did not uniformly match.

8.3.3 Summary on FTOH

FTOH Toxicity in experimental animals
Few data on FTOH health effects in animal and humans are found.

In rodents the liver appears to be the most sensitive target organ for 8:2 FTOH toxicity. Futher studies on carcinogenesis, repoductive and developmental effects are needed.

The potency of endocrine disruption is demonstrated in *in vitro* studies. *In vitro*, oxidative damage were found in rat testicular cells exposed to 8:2 FTOH and 6:2 FTOH.

There are gaps of data for the intermediate metabolites of FTOH (fluorotelomer unsaturated aldehydes (FTUALs) and acids (FTUCAs)) that form from the degradation of FTOHs. Their potential for toxicity is not yet defined and may be more significant compared to PFCAs.

Human health effects of FTOH
No available human data for FTOH are found but *in vitro* studies using human cells have reported endocrine effects for the fluorotelomer alchohols 8:2 FTOH and 6:2 FTOH.

8.4 FTS (fluorotelomer sulfonates).

Acute and repeated-dose mammalian and aquatic toxicity has been reportefor 6:2 FTS.[26]

8.5 PAP/di-PAP (polyfluoroalkyl phosphate esters)

8.5.1 Toxicity in experimental animals

In rats biotransformation of monoPAP and diPAP congeners to FTOH and PFCAs have been observed (D'Eon J and Mabury, 2011). The animals were dosed with a mixture of 4:2, 6:2, 8:2, and 10:2 monoPAP or diPAP chain lengths. Concentrations of the PAPs and PFCAs, as well as metabolic intermediates and phase II metabolites, were monitored over time in blood, urine, and feces. The diPAPs were bioavailable, with bioavailability de-

[26] UNEP/POPS/POPRC.8/INF/17

creasing as the chain length increased from 4 to 10 perfluorinated carbons. The monoPAPs were not absorbed from the gut. However, the PAP dosing concentrations at 50 mg/kg used in the studies (D'Eon J and Mabury, 2011; D'Eon and Mabury, 2007), where not toxic for the animals.

No other toxicity data were found for PAP/di-PAP.

8.5.2 Human health effects

No epidemiological data regarding health effects were found for PAP/di-PAP.

In vitro assays using human cell cultures (H295R human adrenal cortico-carcinoma cells) showed that 8:2 diPAPS and 8:2 monoPAPS gave rise to decreases in androgens (testosterone, dehydroepiandrosterone, and androstenedione) in the steroidogenic pathway, indicating an affect of steroidogenesis (Rosenmai *et al*., 2012). 8:2 triPAPS, 10:2 diPAPS showed a less marked effect on androgens in the steroidogenesis assay and did not disrupt the binding to androgen receptor.

8.6 Perfluoropolyethers (PEFPs)

The general toxicity profile of perfluoropolyethers is reported low. A study evaluating the safety of PEFPs, tested the representative Fomblin HC in experimental animals for acute, subacute toxicity, *in vitro* mutagenicity and irritancy or sensitization on human skin (Malinverno *et al*., 1996). Daily oral administration (1,000 mg/kg/day) to rats over 28 days produced no significant reaction.

No other toxicity data were found.

8.6.1 Summary for PAP/di-PAP (polyfluoroalkyl phosphate esters) and perfluoropolyethers (PFPEs)

Toxicity in experimental animals

MonoPAP and di-PAP congeners are bioavailable in rats and biotransformed to FTOH and PFCAs. PAP doses up to 50 mg/kg was not toxic in the rat study. No other toxicity data were found for PAP/di-PAP.

Human health effects

No epidemiological data regarding health effects were found for PAP/di-PAP.

In vitro assays using human cell cultures showed some endocrine disrupting potential by affecting the steroidogenesis.

Few data on PFPEs are reported. The general toxicity profile of PFPEs is low and no significant reaction in experimental animals was found for acute, subacute toxicity, *in vitro* mutagenicity and irritancy or sensitization on human skin.

No other toxicity data were found.

8.7 Summary

8.7.1 Animal toxicity

In general PFCAs and PFSAs affect the development, reproduction and immune system negatively; they decrease body weight, induce hepatoxicity, affect the endocrine system negatively including the sex hormone and thyroid hormone system; induces peroxisome proliferation believed to be one of the mechanisms behind tumor initiation and affects the immune- and hormone systems.

Hepatotoxicity: Hepatocytic hypertrophy effect in laboratory animals were reported for PFOS, PFHxS, PFBS, PFDA, PFNA, PFOA, PFHpA, PFHxA, and PFBA and is most likely associated with induced peroxisome proliferation.

Developmental Toxicity: Early pregnancy loss was noted with PFOA or PFBA exposure but only at very high doses, and the etiology of this effect is not clear. No fetal toxicity was observed after gestational exposure to PFBA or PFDA. Compared to long-chain PFAAs (≥C8), the short-chain chemicals are much less toxic to the developing animal, in part due to their faster rate of clearance. Thus, even at very high doses of PFBA (350 mg/kg, intended to match the body burden of PFOA), neither neonatal survival nor postnatal growth was compromised, although maternal hepatomegaly was detected (indicating the effectiveness of the PFBA dose regimen) and neonatal liver weight was transiently elevated. A similar lack of reproductive and developmental toxicity has been reported for PFHxA, PFBS and PFHxS.

Immunotoxic: In general adverse immunological outcomes were reported from exposure to PFOS, PFHxS, PFOA and PFNA.

Endocrine disruption: In general, alterations of thyroid hormones and sex steroid hormones have been shown after exposure to primarily PFOS and PFOA, although PFDA induced reductions of thyroid hormones have also been reported. PFOA has been shown to decrease serum and testicular testosterone and increase serum estradiol in male rats, presumably via induction of the hepatic aromatase. PFOS, PFOA, and telomer alcohols have been shown to exhibit estrogenic activity *in vivo* models and to inhibit testicular steroidogenic enzymes. In addition, PFDoA has recently been shown to decrease testosterone synthesis in male rats and to decrease serum estradiol and gene expression of estrogen receptors in the female rats, possibly through oxidative stress pathways.

8.7.2 Human epidemiological studies

An array of studies, mainly cross sectional design in US has been conducted with the main focus on PFOS and PFOA (Table 13). The overall observations were an association between PFCAs, PFSAs and effects on liver parameters such as lipid profile, the reproductive system (e.g. menopause), the thyroid hormone system, and an increased risk of ADHD (PFHxS). Serum PFAAs were associated with altered glucose homeostasis, indicators of metabolic syndrome, and elevated liver enzymes; a positive association between serum PFOS, PFOA and PFNA and cholesterol level; a significant association of PFOS and PFOA with thyroid related diseases.

Also an array of population studies on developmental effects have been conducted of which only 4 out of 12 are Nordic studies (Table 14). An inverse relationship between *in utero* exposure to PFOS and PFOA and birth weight was reported. However, these data call for further studies.

Follow-up evaluations of infants and children in the Danish National Birth Cohort indicated no associations between prenatal exposure to PFAAs and risk of infectious diseases, developmental milestones, and behavioral and motor coordination problems. Whereas a study on the Faroe Islands birth cohort study showed that PFC levels inversely correlated to the vaccination response at age 5. These data call for further studies.

Some reports suggest a relationship between PFOS, PFOA and/or PFHxS exposure and the risk of ADHD, but again these data call for further research.

A single Danish study found that *in utero* exposure to PFOA was positively associated with BMI at age 20.

A recent British cohort study did not find an association between maternal PFAA exposure and altered age at menarche of their offspring.

Five Nordic (out of seven) studies on PFCA and PFSA effects on reproductive factors have been conducted. The overall observations were: Time-to-pregnancy in pregnant women in the Danish National Birth Cohort was suggested to be associated with PFOA and PFOS exposure and cause a reduction of fecundity; high PFAA levels and reduced normal sperm in Danish men; fewer normal sperm in a cross sectional study (Poland, Sweden, Ukraine, Greenland) upon high PFAA (PFOS); no effect on age of menarche in a single UK study.

Few data on the population studies calls for further research on reproductive factors as TTP, fecundity / fertility and the mechanisms behind.

Two Nordic studies have focused on the effect of PFCA and PFSA on the immune system and found controtradictory data: in Denmark no association to hospitalization and exposure was found for PFOS and PFOA, whereas in the Faroe Islands a negative effect on children vaccination response was reported. Two other studies (Taiwan, Japan) have suggested a correlation between PFOS and PFOA exposure and effect on the immune system such as changes in IgE levels in infants and cord blood, whereas no relationship between allergy in infant and maternal exposure was found.

In a Danish study including women and men no association was found on PFOA and PFOS exposure and the risk of prostate, bladder, or liver cancer. However, a recent study showed a significant association between serum PFOA (sum of PFCs) and risk of breast cancer among Inuit.

Very few studies (no Nordic) suggesting a relationship between PFCA and PFSA exposure and lipid profile changes and cardiovascular deceases calls for further research.

Table13. Results of human studies from exposure to PFOA-contaminated water in Ohio/West Virginia communities (C8 Health Project)

1st. Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Emmett, 2006 (Emmett *et al*., 2006)	USA/ Ohio	53% female; age 2,5–89 (median 50)	cross-sectional	?	371	PFOA	health parameters	PFOA: 354 ng/ml (median)	No associations between PFOA and liver function, cholesterol, TSH, red cell/white cell or platelet counts.
Steenland, 2009 (Steenland *et al*., 2009)	USA/West Virginia + Ohio (C8)	adults (<18)	cross-sectional	2005–2006	46,294	PFOS, PFOA	serum lipids	PFOS: 20 ; PFOA: 27 ng/ml (median)	Positive associations– No association to HDL
MacNiel, 2009 (MacNeil *et al*., 2009)	USA/ Ohio	adults	cross-sectional	2005–2006	54,468	PFOA	diabetes II	PFOA: 28 ng/ml	No association between PFOA and typeII diabetes
Frisbee, 2010 (Frisbee *et al*., 2010)	USA/Ohio (C8)	children and adolescents (1–17.9)	cross-sectional	2005–2006	12,476	PFOS, PFOA	Serum lipids (total, HDL, and LDL cholesterol, and fasting triglycerides	PFOS: 20.7; PFOA: 32.6 ng/ml (medi-an) 1–12years	PFOA associated with increased total and LDL cholesterol, and PFOS associated with increased total, HDL, and LDL cholesterol.
Steenland, 2010 (Steenland *et al.*, 2010b)	USA/ West Virginia + Ohio (C8)	adults ≥ 20 years	cross–sectional	2005–2006	54,951	PFOS, PFOA	uric acid	PFOS: 20.2; PFOA: 27.9 ng/ml	positive association with hyperuricemia
Stein, 2011 (Stein and Savitz, 2011)	USA/Ohio (C8)	children (5–18)	cross-sectional	2005–2006	10,546	PFOS, PFOA, PFHxS, PFNA (detectable)	ADHD and PFCs	PFOS: 20.2 ng/ml; PFOA 28.2 ng/mL (median)	positive association for PFHxS

1st. Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Knox, 2011 (Knox *et al*., 2011a)	USA/Ohio (C8)	adults (M+F)	cross-sectional	2005–2006	52,296	PFOS, PFOA	Thyroid function	Female: PFOS: 17.3; PFOA: 52.6 (mean)- Male: PFOS: 24.8; PFOA: 91 (mean)	PFOA and PFOS associated with elevations in serum thyroxine and reduction in T3 uptake.
Knox, 2011 (Knox *et al*., 2011b)	USA/Ohio (C8)	women (18–65)	cross-sectional	2005–2006	25,957	PFOS, PFOA	menopause	PFOS: 15-21.5 ; PFOA: 17.6-32.5 ng/ml (median)	perimenopausal women: experience menopause if they have high serum concentrations of PFOS and PFOA
Gallo, 2012 (Gallo *et al*., 2012)	USA/Ohio (C8)	adults	cross-sectional	2005–2006	47,092	PFOS, PFOA	Liver function (enzymes)	PFOS: 20.3; PFOA: 28 ng/ml (median)	positive association between PFOA and PFOS concentrations and serum ALT level
Lopez-Espinosa, 2012 (Lopez-Espinosa *et al*., 2012)	US/ Ohio	children (1–17)	cross-sectional	2005–2006	10.725	PFOA, PFOS, and PFNA	TSH, TT4	PFOS:20; PFOA:29,3 ng/ml	Associations of serum PFOS and PFNA with thyroid hormone levels and of serum PFOA and hypothyroidism.

Table14. Epidemiological studies on adverse human health effects related to PFCA and PFSA exposures (general population)

Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Developmental outcome									
Fei, 2007 (Fei *et al*., 2007)	Denmark	Pregnant/ infant (DNBC)	prospective follow-up	1996–2002	1,400	PFOS, PFOA (measured)	Fetal growth	PFOS: 35.3 PFOA: 5.6 (mean)	PFOA inversely associated with BW
Fei, 2008 (Fei *et al*., 2008)	Denmark (DNBC)	Mother/ child	prospective	1996–2002	1,400	PFOS, PFOA	Developmental milestones	PFOS: 34.4 PFOA: 5.4	Higher PFOS in mothers associated to childs later start at sitting
Fei and Olsen, 2011 (Fei and Olsen, 2011)	Denmark (DNBC)	Mother/ child	prospective	1996–2002	1,400	PFOS, PFOA	ADHD	PFOS: 34.4 PFOA: 5.4	No association between higher SDQ scores and maternal levels of PFOS or PFOA; no association with motor coordination disorders.
Whitworth, 2012 (Whitworth *et al*., 2012a)	Norway	pregnant women (MoBa)		2003–2004	910	PFOA, PFOS	BW	PFOS: 19.3 PFOA: 3.7 (median)	Lower adjusted BW among infants born to women with the highest plasma levels of PFOA and PFOS
Inoue, 2004 (Inoue *et al*., 2004)	Japan/ Hokkaido	Pregnant (17–37 age)	cross-sectional	2003	15	PFOS, PFOA, PFOSA	BW/thyroid function (T4, TSH)	PFOS: 8.1 PFOA: 0.7	No association
Apelberg, 2007 (Apelberg *et al*., 2007)	US/ Maryland	pregnant /infants	cross-sectional	2004–2005	293	PFOA, PFOS	Fetal growth (BW and size)	PFOS: 5 PFOA: 1.6	Neg. association with BW, ponderal index, head circumference
Monroy, 2008 (Monroy *et al*., 2008)	Canada	Pregnant	cross-sectional	2004–2005	101	PFOA, PFOS, PFDeA, PFHxS, PFHpA, PFNA,	BW	PFOS: 16.6 PFOA: 2.13	No association with BW

Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Hamm, 2010 (Hamm *et al.*, 2010)	Canada	Pregnant	cross-sectional	2005–2006	252	PFOA, PFOS, PFHxS	fetal growt	PFOS: 35 PFOA: 1.5	No correlations with BW, GSA, and other birth parameters
Washino, 2009 (Washino *et al*., 2009)	Japan	Pregnant	prospective	2002–2005	428	PFOA, PFOS	BW and size	PFOS: 5.2 PFOA: 1.3	PFOS negatively correlated with BW (girls); no association with PFOA
Hoffman, 2010 (Hoffman *et al*., 2010)	US (NHANES)	children (12–15)	cross-sectional	1999–2000; 2003–2004	571	PFOA, PFNA, PFOS, PFHxS	ADHD	PFOS: 22.6 PFOA: 4.4	Positive relationship between parent-reported ADHD and serum PFOS, PFOA, and PFHxS
Gump, 2011 (Gump *et al*., 2011)	USA	Child	cross sectional	?	83	11 PFCs: PFOS, PFHxS, PFBS, PFDS, PFOSA,PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA	impaired response inhibition	PFOS: 8.79 PFOA: 3.28	Higher levels of blood PFOS, PFNA, PFDA, PFHxS, and PFOSA associated with shorter IRTs (children's impulsivity)
Chen, 2012 (Chen *et al*., 2012)	Taiwan	mother-infant (TBPS)		2004–2005	429	PFOA, PFOS, PFNA, and PFUA	GSA, BW, head circumference	PFOS: 5.94 PFOA: 1.84	PFOS inversely associated with GSA, birth weight, and head circumference
Maisonet, 2012 (Maisonet *et al*., 2012)	UK	singelton girls (ALSPA cohort)		1991–1992	447	PFOS, PFOA, PFHxS	BW, weight for age.	PFOS: 19.6 PFOA: 3.7	PFOS negatively associated with girls weight at birth; No associations with PFOA and PFHxS

Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Thyroid function									
Dallaire, 2009 (Dallaire *et al*., 2009)	Nunavik	Inuit adults	cross-sectional	2004	623	PFOS	Thyroid function	PFOS: 18.28 (GM)	PFOS concentrations were negatively associated with TSH, tT3 and TBG and positively with fT4 concentrations
Bloom, 2010 (Bloom *et al*., 2010)	USA/ NY	sportfish anglers	prospec-tive	1995–1997	31	PFDA, PFNA, PFHpA, PFHxS, PFOA, PFOS, PFOSA, PFUnDA	T4. TSH	PFOS: 19.6 PFOA: 1.3	No associations between PFCs exposures and thyroid function
Melzer, 2010 (Melzer *et al*., 2010)	USA	men and women		1999–2006	3966 (52% women)	PFOA, PFOS	Thyroid diseases	PFOS: 19.14 PFOA: 3.77 (Female) PFOS: 25.08 PFOA: 4.91 (men) (GM)	Higher concentrations of serum PFOA and PFOS are associated with current thyroid disease
Kim, 2011 (Kim *et al*., 2011)	Korea/ soul	Mother/ child	cross sectional	2008–2009	44	13 PFCs: PFHxS, PFHpS, PFOS, PFDS, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA PFTe-DA, MePFOSAA, EtPFOSAA.	Thyroid hormone; Bith weight	PFOS: 2.72 PFOA: 1.46	Fetal PFOS, PFOA, PFTrDA and maternal PFTrDA were correlated with fetal total T4 concentrations (before adjustment). After adjustment: negative correlations be-tween maternal PFOS and fetal T3, and maternal PFTrDA and fetal T4 and T3.

Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Ji, 2012 (Ji *et al*., 2012)	Korea/ Siheung	>12 years	cross-sectional	2008	633 (59% female)	13 PFCs: PFHxS, PFHpS, PFOS, PFDS, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, PFTeDA, MePFOSAA, EtPFOSAA.	Thyroid function (T4. TSH)	PFOS: 7.96 PFOA: 2.74	PFTrDA negatively correlated with TT4 and positively with TSH
Reproduction									
Joensen, 2009 (Joensen *et al*., 2009)	Denmark	men (19)	cross sectional	2003	105	10 PFCs (C6 to C13); PFHxS. PFHpA. PFOA. PFOS.PFOSA. PFNA. PFDA. PFUnA. PFDoA. PFTrA	Reproductive hormones and semen	PFOS: 24.5 PFOA: 4.9	High PFAA levels were associated with fewer normal sperm.
Fei, 2009 (Fei *et al*., 2009)	Denmark	pregnant women	prospec-tive	1996–2002	1,240	PFOA. PFOS	TTP; Fecundity	PFOS: 33.7 PFOA: 5.3	Pos association to TTP and reduced fecundity
Vestergaard, 2012 (Vestergaard *et al*., 2012)	Denmark	Women (18–35)	prospec-tive study	1992–1995	222	8 PFCs: PFOS. PFOA.PFHxS. PFNA. PFDA. MeFOSAA.EtFOSAA and FOSA	Fecundity	PFOS: 36 PFOA: 5.6	No clear association between PFCs and TTP

Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Whitworth, 2012 (Whitworth *et al*., 2012b)	Norway	pregnant women (MoBa)	case-control	2003–2004	910 (416 cases)	PFOA. PFOS	TTP	PFOS: 13 PFOA: 2.2	PFOA and PFOS exposure at plasma levels seen in the general population may reduce fecundity;
Toft, 2012 (Toft *et al*., 2012)	GRL/PL/ukraine	Men	cross sectional	2002–2004	196	PFHxS. PFOS. PFOA. PFNA. PFDA. PFUnDA and PFDoDA	Semen quality	PFOS: 44.7 PFOA: 4.5 (GRL) PFOS: 18.5 PFOA: 4.8 (PL) PFOS:7.6 PFOA: 1.3 (Ukraine)	Negative associations between PFOS exposure and sperm morphology
Christensen, 2011 (Christensen *et al*., 2011)	UK	ALSPA cohort	nested case control	1991–1992	218 cases; 230 controls	8 PFCs: incl. PFOS. PFOA	Age of menarche	PFOS: 19.8 PFOA: 3.7	No associations with age at menarche
Raymer, 2011 (Raymer *et al*., 2012)	US	Men	cross-sectional	2002–2005	256	PFOA. PFOS	Semen quality/reproductive hormones	PFOS: 32.3 PFOA: 5.2	No association with sperm parameters. pos correlation with LH
Immune system									
Fei, 2010 (Fei *et al*., 2010)	Denmark (DNBC)	Mother/child	prospective	1996–2002	1,400	PFOS. PFOA	Hospitalization/infection	PFOS: 34.4 PFOA: 5.4	No association between hospitalizaion and PFCs
Grandjean, 2012 (Grandjean *et al*., 2012)	Faroe Iceland	Mother/child	prospective	1997–2000	587	PFOS. PFOA. PFHxS. PFNA. PFDA	Antibody conc. after vaccination	PFOS: 27.3 PFOA: 3.20 (GM)	PFOS and PFOA associated with lower antibody responses to childhood immunizations at age 5. PFDA positively associated with the antibody concentration in blood

Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Wang, 2011 (Wang *et al*., 2011)	Taiwan	children	birth cohort study	2004	244	12 PFCs: detected PFOA. PFOS. PFNA. and PFHxS	Pediatric atopi	PFOA:1.71 PFOS: 5.50	PFOA and PFOS levels positively correlated with cord blood IgE levels
Okada, 2012 (Okada *et al*., 2012)	Japan / Sapporo	Pregnant women/ infant	Cross-sectional	2002–2005	343	PFOS, PFOA	Allergies and infectious diseases	PFOS: 5.2 PFOA: 1.3	Among female infants, cord blood IgE levels decreased significantly with high maternal PFOA serum levels. No relationship was found between maternal PFOS and PFOA serum levels and infant allergies and infectious diseases at 18 months of age.
Cancer									
Eriksen, 2009 (Eriksen *et al*., 2009)	Denmark	men and women	prospective	1993–1997	1,240 (90% women)	PFOA. PFOS	Cancer	PFOS: 35 PFOA: 6.8	No association to bladder, pancreatic or liver cancer. Small increase in risk for prostate cancer
Bonefeld-Jørgense, 2011 (Bonefeld-Jorgensen *et al*., 2011)	Greenland	Inuit women	case-control	2000–2003	31 cases 115 controls	PFOS. PFOA. PFHxS. PFOSA. PFHpA. PFNA. PFDA. PFUnA. PFDoA. PFTrDA	Breast Cancer	PFOS: 45.6 PFOA: 2.5	Higher PFC levels in breast cancer cases; Pos odds ratio
Lipid metabolisme /biochemicalparameters									
Halldorsson TI, 2012 (Halldorsson *et al*., 2012)	Denmark	Mother/ child (Aarhus BC)	prospective cohort	1988–1989	665	19 PFCs	BMI at 20 years	PFOS: 21.5; PFOA: 3.7	*in utero* exposure to PFOA positively associated with anthropometry at 20 years in female but not male offspring

Author, Year, Ref.	Location	Population	Design	Sampling date	No.	PFCs measured	Outcome	Median PFOS/PFOA conc. (ng/ml)	Results
Lin, 2009 (Lin *et al*., 2009)	USA/ NHANES	474adolesve nts + 696 adults	cross-sectional	1999–2000 and 2003–2004	1,443	PFOA. PFNA. PFOS. PFHxS	Glucose homeosta-tis. metabolic syndrome	PFOS: 24.3 PFOA: 4.3	Increased serum PFNA conc. associated with hyperglycemia; serum HDL choles-terol; inversely with prevalence of meta-bolic syndrome
Nelson, 2010 (Nelson *et al*., 2010)	USA/ NHANES	≥ 12 years	cross-sectional	2003–2004	2,094	PFOA. PFNA. PFOS. PFHxS	Lipid and weight outcomes	PFOS:19.9 PFOA: 3.8	A positive association between concentra-tions of PFOS, PFOA, and PFNA and total and non-high-density cholesterol
Cardiovascular diseases									
Shankar, 2012 (Shankar *et al*., 2012)	USA/ NHANES	Men and women above 40 years	Cross-sectional	1999–2000 and 2003–2004	1,216 (51.2% wom-en)	PFOA	Cardiovascular disease and peripherial arterial disease	Not reported	Exposure to PFOA is associated with cardiovascular disease and peripherial arterial disease, independent of tradi-tional cardiovascular risk factors

BW: Birth weight; GSA: Gestational age.

9. Environmental effects of per- and polyfluorinated substances

The general observed trend of PFC toxicity is the linear relationship found between increasing chain-length and decreasing EC_{50} (Hoke *et al*., 2012; Latała *et al*., 2009; Mitchell *et al*., 2011; Mulkiewicz *et al*., 2007; Nobels *et al*., 2010). This in combination with the longer half-lives and elimination rate of the longer chain PFCs should be recognised as a great health and environmental concern (Mulkiewicz *et al*., 2007). It has been shown that PFCs may affect the thyroid system (Weiss *et al*., 2009), influence the calcium homeostasis, protein kinase C, synaptic plasticity, cellular differentiation, induce neurobehavioral effects and induce peroxisome proliferation (Ishibashi *et al*., 2008; Mariussen, 2012). *In vitro* assessment of environmental fate and ecotoxicologial effects of individual PFCs in some cases demonstrate that their current environmental levels do not pose a threat to ecosystems (O'Brien *et al*., 2009). On the other hand, in the environment, exposure is rarely limited to one PFC, but to a mixture of various PFCs and other environmental pollutants. Toxic effects may occur as a result of interactions between hazardous chemicals and co-exposure may cause additive or synergistic effects (Eriksen *et al*., 2010). Latała *et al*. (2009) therefore suggest that future studies of PFCs ecotoxicity should mainly focus on the effects of mixtures of PFCs and their derivatives. Another factor that should be considered when PFC levels in wildlife are evaluated is the substantial differences in PFC concentrations found among life history stages. In a study on porpoises, the highest concentrations of PFCs were found in neonates, suckling juveniles and lactating females which is of concern as PFCs are known to cause toxic effects on the development of the central nervous system and reproductive organs (Galatius *et al*., 2011).

Some mechanistic studies on PFC toxicity have revealed that there are species specific differences. However, this was not the outcome when PFOS, PFOSA, PFHxA and PFBS were studied (Hu *et al*., 2002) in both a rat liver epithelial cell line (WB-F344) and a dolphin kidney epithelial cell line (CDK) for GJIC interfering effects. Gap junctional intercel-

lular communication (GJIC) is the major pathway of intercellular signal transduction, and is therefore important for normal cell growth and function. Inhibition of GJIC was found in a dose-dependent manner for all compounds except for PFBS and the inhibitory effects were neither species- nor tissue specific. A Specific Absorption Rate (SAR) was also established among the four compounds tested, and the effect was determined by the length of the fluorinated tail and not by the functional groups. This is contradicted by a study made by Hagenaars *et al*. (2011a), comparing the potential effects of the four different PFCs; PFOS, PFBS, PFOA and PFBA on embryonic development in zebrafish (Danio rerio). The authors conclude that sulfonated and carboxylated PFCs act by different processes and that the exact mechanisms of the potential effects of PFOS, PFOA and PFBS on endothelial cells or vasoactive substances of the heart will need further investigation.

In order to evaluate the potential reproductive effects of PFCs, a study was conducted on the toxic effects of the four PFCs; PFOA, PFNA, PFOS and FC-807 (perfluoro alkyl phosphate) on zebrafish embryos. Oedemas and spine malformations occurred throughout the duration of the study and all the tested PFCs caused lethality in a concentration-dependent fashion. Based on the LC_{50} values the toxic potency was in the order of PFOS>FC807>PFNA>PFOA. Although all four compounds caused malformations, FC-807, with the larger ester molecule, caused more yolk-sac and pericardial-sac oedemas than the other PFCs. The authors therefore propose that FC-807 might more easily disturb the water barrier around the embryos and disturb heart functions, causing oedemas (Zheng *et al*., 2012).

9.1 Perfluoro carboxylates (PFCAs)

PFCA toxicity has been tested in different organisms. Blue-green algae and cyanobacteria were shown to be very sensitive to PFCAs (Latała *et al*., 2009). *E. coli* bacteria were exposed to selected PFCs which resulted in clear indications of markers for oxidative stress and DNA damage with the most severe DNA damage observed after exposure to PFDA, PFUnDA and PFDoDA (Nobels *et al*., 2010). The structure-activity results from the *E. coli* study (Nobels *et al*., 2010) indicated that the carboxylic acids with long carbon tails induced several stress genes. Other studies have made similar findings; ROS and DNA damage was observed in HepG2 cells after exposure to PFCAs (Eriksen *et al*., 2010), cytotoxicity was found against primary rat testicular cells, where PFNA attributed as most DNA damag-

ing (Lindeman *et al*., 2012). *In vitro* cytotoxicity of PFCAs was determined in human colon carcinoma (HCT116) cells (Kleszczyński *et al*., 2007; Kleszczyński and Składanowski, 2009) and treatment with PFCAs caused cell apoptosis. It seems that the PFCAs are not acutely toxic, but the cell viability inhibition is intensified after longer exposure. The estimated EC_{50} values decreased with elongation of the fluorocarbon chain (PFHxA > PFHpA > PFOA > PFNA > PFDA > PFDoA > PFTeDA), although, chain lengths above C16 and C18 did not increase the effect. PFCA effects on hatching success of birds was investigated by O´Brien *et al*. (2009) and in their results, only PFUDA caused statistically significant increases in gene transcription at the highest dose applied (10 µg/g).

PFCAs are also potent inducers of the oestrogen-responsive biomarker protein vitellogenin (Vtg). This effect was investigated *in vivo* in rainbow trout by Benninghoff *et al*. (2011). They found that all PFCAs tested (PFOA, PFNA, PFDA and PFUnDA) bound weakly to trout liver ER with IC_{50} values of 15.2–289 mM, whereas, 8 to 10 fluorinated carbons and a –COOH end group were optimal for Vtg induction. The same authors suggest that multiple PFCAs can promote hepatic tumorigenesis in trout in a manner similar to that of 17β-estradiol (Benninghoff *et al*., 2012).

9.2 Perfluoroalkyl sulfonates (PFSAs)

The perfluoroalkyl sulfonates contain a sulfonate group instead of a carboxyl group, but as mentioned earlier, there still seems to be no clear understanding of the relative importance of chain length and functional groups for the toxic effects of PFCs.

Some of the bigger PFC producers, such as 3M, have already phased out the production of PFOS (C8) and replaced it with the production of PFBS (perfluorobutane sulfonate, C4) as an alternative. Due to its shorter chain length it is expected to be less bioaccumulative. As a result, the production volumes of PFBS have expanded during the last decade and PFBS has already been detected in environmental samples. The general lack of information on its toxicological mode of action and the potential hazard it may pose against ecological species, has therefore directed a great deal of focus on PFBS during the last years (Hagenaars *et al*., 2011b). According to the obtained ecotoxicological information on PFBS, its direct toxicity seems to be relatively low. Flow cytometric measurements on some membrane systems of the freshwater alga species Scenedesmus obliquus revealed that PFBS did not inhibit algal growth within the test concentration ranges (Liu *et al*., 2008). PFBS was exam-

ined by Rosal *et al*. (2010) against the marine bacterium Vibrio fischeri, the cyanobacterial recombinant strain Anabaena CPB4337 and the algae Pseudokirchneriella subcapitata and the toxicity against all organisms tested was very low (EC50 > 8,000 ml/L). In the study mentioned above on effects of PFBS on embryonic development in zebrafish (Danio rerio), it was not possible to determine its LC50 value due to the low mortality rate of the compound even at high exposure concentrations (500 and 3,000 mg/L). The results however, demonstrated significantly altered heart rates in the embryos as well as malformations of their heads after exposure to PFBS (Hagenaars *et al*., 2011b). In juvenile mallards and northern bobwhite quail, PFBS was found to affect the body weight gains of quail exposed to 5,620 or 10,000 mg PFBS/kg feed, which were statistically less than that of unexposed controls (Newsted *et al*., 2008).

What appears to be of more concern regarding the short-chained PFCs is their elicited transcriptional responses. Long-chained PFCs bind more strongly to extracellular proteins than short-chained PFCs, which may render their availability for uptake into cells. Hence, short-chained PFCs may be more bioavailable and may cause more genetic and neural damage (Vongphachan *et al*., 2011).

Slotkin *et al*. (2008) used PC12 cells, (a standard in vitro model for neuronal development) to characterise essential features of the developmental neurotoxicity of PFBS. PFBS demonstrated a unique effect on differentiation of both dopamine (DA) and acetylcholine (ACh) neurotransmitter phenotypes; displaying a concentration-dependent suppression in both the expression of TH (Tyrosine Hydroxylase) and ChAT (Choline AcetylTransferase). PFBS did not evoke any effect on DNA synthesis, although it did produce significant cell enlargement due to an increase in the total protein/DNA ratio (Slotkin *et al*., 2008).

The short-chained PFCs, PFBS and PFHxS were shown to significantly alter the messenger RNA (mRNA) of TH-responsive genes in primary cultures of avian (chicken embryonic and herring gull embryonic) neuronal cells. Exposure to these compounds resulted in e.g. induction of D3 mRNA, RC3 mRNA and down-regulation of TTR mRNA (Vongphachan *et al*., 2011). These effects could affect e.g. TH-dependent processes as well as changes in RC3 expression could have consequences in synaptic plasticity, associative learning and memory (Iniguez *et al*., 1993; Iniguez *et al*., 1996; Vongphachan *et al*., 2011). Likewise, an earlier in vitro study on CEH (Chicken Embryo Hepatocyte) cultures exposed to PFHxS, PFHpS and PFDS demonstrated alterations of transcriptional responses. The shorter-chained PFHxS and PFHpS induced CYP1A4 or CYP1A5 mRNA, while the longer-chained PFDS repressed CYP1A4 mRNA (Hickey *et al*., 2009).

These reported significant effects of PFHxS treatment on messenger RNA (mRNA) levels of thyroid hormone (TH)-responsive genes in chicken embryonic neuronal cells initiated the determination of in ovo effects of PFHxS exposure (maximum dose = 38,000 ng/g egg). The previous in vitro results were successfully validated, since plasma TH levels of chicken embryos were reduced in a concentration-dependent manner following PFHxS exposure. In addition, pipping success was significantly reduced, the tarsus length and embryo mass decreased and D2 and D3 and cytochrome P450 3A37 mRNA levels were induced in the liver tissue, whereas D2, RC3 and octamer motif binding factor 1 mRNA levels were up-regulated in cerebral cortex. PFHxS accumulation could be seen in the three tissue compartments analysed: yolk sac > liver > cerebral cortex (Cassone *et al*., 2012).

In an in vivo study (Nøst *et al*., 2012), plasma concentrations of a wide range of halogenated organic contaminants, including PFHxS and PFHpS, and their correlations with circulating thyroid hormones (TH) in developing Arctic seabirds was assessed. Plasma from chicks of black-legged kittiwake (Rissa tridactyla) and northern fulmar (Fulmarus glacialis) was taken and analysed for thyroid hormones. A positive association was found between total thyroxin (TT4) and PFHpS in both species and PFHxS was negatively correlated to the TT3:FT3 ratio. Since the disruption of TH homeostasis may cause developmental effects in young birds, the correlations between the relatively low plasma levels of PFCs and THs found in the study may be of concern on the health related effects of these compounds in seabird fledglings (Nøst *et al*., 2012).

The C10 PFSA, PFDS, seems to be less industrially utilised and has accordingly attained less attention, which is reflected by its appearance in the scientific literature. But since Hickey *et al*. (2009) identified the responsiveness of genes exposed to PFDS in in vitro cultured chicken embryonic hepatocytes (CEH) the in ovo toxicity of PFDS was assessed by injection into white leghorn chicken (Gallus gallus domesticus) eggs (O'Brien *et al*., 2009). The compound was detected in livers of the chicken embryos, at levels in agreement with concentrations (up to 10 µg/g) injected, indicating that PFDS was efficiently taken up by the developing embryos. Despite this, transcriptional activity for CYP1A4, CYP1A5, CYP4B1 or L-FABP mRNA in the chicken embryo liver tissue was not significant altered and the pipping success was not affected (O'Brien *et al*., 2009).

9.3 FTOHs

The PFOS and PFOA use has been exchanged for shorter chain PFCs such as the fluorotelomer alcohols. Little data on the distribution and environmental fate of FTOH makes it difficult to assess the environmental risk. Available studies of their biological and ecotoxicological assessment have demonstrated the need to monitor their environmental distribution and further investigate their effects on the biota, especially for long-term exposure of environmental relevant concentrations. FTOHs have been detected in the aquatic environment and their biotic and abiotic degradation lead to a range of products, including PFCAs of various chain lengths, causing secondary pollution. PFOA and PFNA have been confirmed as metabolites of 8:2 FTOH (Ishibashi *et al*., 2008; Martin *et al*., 2005), and since PFCAs, particularly PFOA, have been reported to cause liver cancer, pancreatic tumour, and Leydig cell tumour, FTOHs might indirectly induce tumours via PFCAs (Oda *et al*., 2007).

As other PFCs, FTOHs have been characterised as xenoestrogens *in vitro* and owe their oestrogen-like effects to their structural and chemical similarities to other xenoestrogenic compounds. Fluorotelomer alcohols have been shown to exert estrogenic activity in MCF-7 cells, in a yeast two-hybrid assay and in aquatic organisms where 6:2 FTOH has been characterised as a stronger xenoestrogen than 8:2 FTOH (Ishibashi *et al*., 2008; Maras *et al*., 2006; Vanparys *et al*., 2006; Wang *et al*., 2012). According to Ishibashi *et al*. (2007) treatment with 6:2 FTOH, 8:2 FTOH and NFDH (nonadecafluoro-1-decanol) show a dose-dependent interaction with the human estrogen receptor (hER), and rank the estrogenic effects for hERα and hERβ in the descending order of 17β-estradiol>>> 6:2 FTOH > NFDH > 8:2 FTOH. Waterborne exposure of both 6:2 and 8:2 FTOH alter the plasma levels of testosterone and estradiol and 8:2 FTOH adversely impair reproductive success in the offspring of zebrafish (Liu *et al*., 2010a; Liu *et al*., 2009). In addition, it has been suggested that 8:2 FTOH has the potential to suppress stereoidgenesis (Liu *et al*., 2010b).

The fluorotelomer alcohols have been ecotoxocologically assessed through their growth impairment potential. Wang *et al*. (2010) suggested that 4:2 and 6:2 FTOH might cause apoptosis, while, Oda *et al*. (2007) suggested that they are unlikely mutagens. The fluorotelomer alcohols 8:2 FTOH and 10:2 FTOH were shown to be rapidly metabolised by rainbow trout to the fluorotelomer acids (8:2 FTCA, 10:2 FTCA) and the unsaturated acids (8:2 FTUCA, 10:2 FTUCA), respectively (Brandsma *et al*., 2011). Studies have found that these transformation products are more bioaccumulative and more acutely toxic to aquatic organisms than their

precursors (Brandsma *et al*., 2011; Hoke *et al*., 2012; Mitchell *et al*., 2011). Both Hoke *et al*. (2012) and Mitchell *et al*. (2011) suggest, however, that these fluorinated acids pose little or negligible risk to aquatic biota, since the available environmental concentrations are still well under the toxicity thresholds.

9.4 Other fluorinated compounds of interest

Polyfluorinated iodine alkanes (PFIs) are important intermediates in the synthesis of organic fluoride products. Recently, they have been detected in fluoropolymers as residual raw materials, as well as in the ambient environment. Wang *et al*. (2012) studied for the first time the estrogenic activity of PFIs, fluorinated iodine alkanes (FIAs), fluorinated telomer iodides (FTIs), and fluorinated di-iodine alkanes (FDIAs) in MCF-7 cells. They concluded that some PFIs could act on ERs and potentially cause detrimental effects on reproductive and developmental systems (Wang *et al*., 2012).

Semi-fluorinated emulsifiers derived from the dimorpholinophosphate polar head group CnF2n+1(CH2)mOP(O)[N(CH2CH2)2O]2 (FnHmDMP) allow for the preparation of stable water-in-fluorocarbon emulsions. These emulsions are being investigated as delivery systems of drugs into the lung, either by systemic or local administration. The cytotoxicity of a series of FnHmDMP was evaluated by Courrier *et al*. (2003). FnHmDMP compounds with the longest fluorinated chain length or total chain length ratio, i.e. F8H11DMP and F10H11DMP were shown to be the least toxic. Moreover, emulsions stabilised with these amphiphiles were found to be non-cytotoxic, or less cytotoxic than solutions of the same amphiphiles in fluorocarbons (Courrier *et al*., 2003).

Fluorotelomer unsaturated aldehydes (FTUALs) and acids (FTUCAs) are intermediate metabolites that form from the degradation of FTOHs. Their toxicity potential is not yet defined and may be more significant compared to PFCAs, but studies have shown that they form adducts with glutathione (GSH). Results presented by Rand and Mabury (2012) indicate that the α,β-unstaurated aldehydes react most comparatively with GSH and that the reaction is possibly influenced by the length of the fluorinated tail. They also suggest that given the low EC50 values measured for 6:2 FTUAL and 8:2 FTUAL, these compounds may exert cytotoxic influences on biological nucleophiles present in proteins as well as nucleic acids.

10. Discussion

There are considerable data gaps on the content of specific PFCs in commercial products used on the Nordic market. Some of these PFCs exhibit hazardous characteristics and therefore it is of very high concern to facilitate access to specific PFC substance information from industrial actors on the market either on a voluntary basis or if this is not possible by legal means. The current legal tools such as the EC Regulation 1272/2008 (CLP) and the EC Regulation 1907/2006 (REACH) are currently not sufficient to provide that kind of specific substance information although the information exists. For publicly available MSDS there is no legal incentive for a company to provide specific substance data and when provided to the authority this information is legally classified as confidential with no access to the public. Concerning PFCs in articles it is not possible to achieve specific PFC substance information according to REACH unless they are identified as Substances of Very High Concern (SVHC). Then there is a legal possibility to access downstream information. However, this is only possible if the concentration of the PFC (then as an SVHC) exceeds 0,1% by weight of the article in question. Since many PFCs are added in much lower concentrations in products, the SVHC approach to PFCs may be ineffective from a legal perspective. It is important to mention that there are small opportunities to get production data on specific PFCs in articles since almost all production occurs outside the EU.

There are few studies on PFCs in the Nordic environment. Therefore there is an urgent need for new data on PFCs, especially for PFCs other than PFOA and PFOS, regarding their environmental occurrence. This is necessary if we want to get a better and more complete picture of the PFC levels in different Nordic environmental compartments. This includes more in-depth knowledge of spatial and temporal distribution, and clear temporal trends.

Modeling and field monitoring are essential prerequisites for detailed environmental fate studies of PFCs. In many cases these studies are hindered by the lack of reliable (or in some cases total lack of) physical-chemical properties for many fluorinated compounds. Furthermore, there is still a lack of analytical reference standards of PFCs but lately there is an increased access to new and better reference standard sub-

stances on the market which are necessary for these kinds of environmental studies. Further resources for in depth research are thus needed.

There are few studies on biomonitoring of PFCs other than PFOA and PFOS. Therefore there is a great need for further studies. This is especially true for those with shorter carbon chains and their corresponding precursors in human/maternal blood and cord blood. There is also a need to explore the real pre-term and post-term exposure of the fetus and newborn child. For some less known PFCs such as PFAL (perflouroaldehyde), FTS (fluorotelomer sulfonates), PAP/di-PAP and FTMAPs (fluorotelomer mercaptoalkyl phosphate diester) there are no studies at all carried out and consequently no data is available. Also in this case further in-depth research is needed. Further studies concerning PFCs' impact on maternity and immunology are called for since only inconsistent data exist.

11. Conclusions

As a result of the mapping study, stage 1 of this project, carried out on more than 50 actors on the Nordic market that trade with PFC products it was concluded that there are considerable information gaps for most of the PFC chemicals regarding the exact chemical composition in commercial products, their quantities produced and uses on the Nordic market. These gaps may be a combination of lack of knowledge and/or trade secrets from the actors on the Nordic market.

In parallel with the mapping of the Nordic market a net list of specific PFCs that may be used on the market was produced. This net list was extracted from three public lists, namely one list from OECD, the REACH preregistration database, and the Nordic SPIN database. Since neither of these databases contains complete information on the market use of PFCs, the net list is necessarily incomplete and there may be other PFCs used on the Nordic market in addition to those found in the net list.

There exists only a few scientific reports on PFCs in the Nordic environment other than PFOA and PFOS that cover both biotic and abiotic samples. Regarding PFCAs, most studies report results for PFOA, PFHxA and PFNA. However other PFCA substances (C10–C13) have also been detected in a few studies. For PFSAs, PFOS and PFHxS are the most studied compounds. Observations reported in the few studies available report that the concentrations in the Nordic environment and the Arctic are much lower compared to other countries especially when compared with central European countries with high GDPs, which is to be expected as populations are smaller and there is less industry in the Nordic countries. However these substances have also been found in the Arctic, far from any sources, which shows that these substances are global contaminants.

Publications that report human biomonitoring data of PFCs (PFCAs and PFSAs) for the Nordic countries during the period from 1992 to 2010 are available. Most and most recent data are reported from Norway and Sweden, whereas fewer exist from Denmark. No human data were found for Iceland and Finland. Results from these studies report that since 2002 decreasing trends have been observed for PFOA and PFOS but not for other PFCAs and PFSAs. In Sweden, for instance, it was found that perfluorinated sulfonates with shorter carbon chains (≤6) currently show an increasing trend.

Only a few studies on PFSAs and PFCAs in amniotic fluids have been published but all show low levels that are 10–20 folds below the levels in the corresponding serum. Nordic studies show that the PFSAs and PFCAs can be transferred to human breast milk with a concentration range of 1–2% and 3–4%, respectively of the serum concentration. For other PFCs such as PFAL (perflouroaldehyde), FTS (fluorotelomer sulfonates), PAP/di-PAP and FTMAPs (fluorotelomer mercaptoalkyl phosphate diester) no studies have been carried out.

Animal studies on toxicity show that PFCAs and PFSAs affect the development, reproduction and immune system negatively by decreasing body weight, inducing hepatoxicity, affecting the endocrine system including the sex hormone and thyroid hormone system. Hepatocytic hypertrophy effect in laboratory animals were reported for PFOS, PFHxS, PFBS, PFDA, PFNA, PFOA, PFHpA, PFHxA, and PFBA and is likely associated with induced peroxisome proliferation.

Early pregnancy loss was observed in animal studies with PFOA or PFBA exposure but only at very high doses, and the etiology of this effect is not clear. No fetal toxicity was observed after gestational exposure to PFBA or PFDA. Compared to long-chain PFAAs (≥C8), the short-chain chemicals are much less toxic to the developing animal, in part due to their faster rate of clearance. A similar lack of reproductive and developmental toxicity has been reported for PFHxA, PFBS and PFHxS.

Adverse immunological outcomes have been reported from exposure to PFOS, PFHxS, PFOA and PFNA. Alterations of thyroid hormones and sex steroid hormones (endocrine disruption) have been shown after exposure to primarily PFOS and PFOA, although PFDA-induced reductions of thyroid hormones have also been reported. PFDoA has recently been shown to decrease testosterone synthesis in male rats and to decrease serum estradiol and gene expression of estrogen receptors in the female rats, possibly through oxidative stress pathways.

The overall observations on liver parameters such as lipid profile, the reproductive (e.g. menopause), the thyroid hormone system, and the risk of ADHD (PFHxS) were observed as a combined effect of PFCAs and PFSAs. Follow-up evaluations of infants and children in the Danish National Birth Cohort indicated no associations between prenatal exposure to PFAAs and risk of infectious diseases, normal developmental milestones, and behavioral and motor coordination problems. Whereas a study on the Faroe Islands birth cohort showed that PFC levels inversely correlated to the vaccination response at age 5.

A linear relationship between increasing PFC chain-length and decreasing EC_{50} has been observed. This in combination with the longer

half-lives and elimination rate of the longer chain PFCs should be recognised as a great health and environmental concern. In the environment, exposure is rarely limited to one PFC, but to a mixture of various PFCs and other environmental pollutants. Toxic effects may occur as a result of interactions between hazardous chemicals and co-exposure may cause additive or synergistic effects. Future studies of PFCs ecotoxicity should focus on the effects of mixtures of PFCs and their derivatives.

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Appendix A

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Sammanfattning

Nordiska Kemikaliegruppen (NKG) som är underställd Nordiska Minister Rådet har via Klima- og forurensningsdirektoratet (KLIF) gett i uppdrag till författarna att genomföra en Nordisk studie baserad på öppna infomationskällor samt egna marknadsstudier för att beskriva de vanligaste perfluorerade ämnena (PFC) med mindre fokus på PFOS och PFOA.

Undersökningen omfattar tre delmoment:

1. Identifiering av relevanta per- och polyfluorerade ämnen och deras användning inom olika industrisektorer på den nordiska marknaden.
2. Förekomst i industri-och konsumentprodukter och potentiella utsläpp till och i den nordiska miljön och människor av de ämnen som beskrivs i delmoment 1.
3. En sammanfattning av kunskapen om toxiska effekter på människa och miljö hos de ämnen som prioriterats i delmoment 2.

Intervjuer genomfördes med fler än 50 aktörer på den Nordiska marknaden med syftet att få information om användning och typ av av PFC-ämnen. Denna undersökning gav emellertid magert resultat. Parallellt med denna kartläggning togs därför en nettolista över PFC-ämnen fram baserat på listor (var och en för sig och tillsammans ofullständiga) från OECD, REACH förregistreringsdatabas samt den nordiska SPIN-databasen. Större delen av varuproduktionen sker idag utanför EU och dagens regelverk inte ger tillräckliga förutsättningar för att få tillräcklig information om specifika PFC ämnen som finns i importerade varor. Denna nettolista är således inte komplett varför det kan finnas avsevärt fler PFC-ämnen som används på den nordiska marknaden.

Det finns få studier om PFC-ämnens förekomst i miljön i de nordiska länderna utöver PFOA och PFOS som omfattar både biotiska (luft, mark och vatten) och abiotiska (djur och människa) data.

De flesta humandata avseende PFCA och PFSA från åren 1992 till 2010 kommer från Norge och Sverige, färre från Danmark och inga uppgifter från Island och Finland. Gällande PFCA visar de flesta studier på förekomst av PFOA, PFHxA och PFNA. Även andra PFCA ämnen (C10–C13) har också påvisats i flera studier. För PFSA är PFOS och PFHxS är

de mest studerade föreningarna. Humandata saknas helt för PFAL, FTS, PAP/di-PAP samt FTMAPs.

I jämförelse med långkedjiga PFC-ämnen (≥C8) är de kortkedjiga föreningarna bedömts vara mindre toxiska men ett antal studier visar på både ekotoxikologiska och humantoxikologiska effekter. Inom detta område är dock bristen på studier stor.

I stort iakttas minskande halter av PFOA och PFOS i miljön sedan 2002. Däremot observeras en ökande halt av kortkedjiga sulfonater i miljön. I jämförelse med andra länder är bakgrundskoncentrationen i miljön av PFOA och PFOS lägre i de nordiska länderna särskilt i jämförelse med centraleuropeiska länder, som kan förväntas pga lägre befolkningstäthet och mindre industriell verksamhet i de Nordiska länderna. . Dessa ämnen har även hittats i Arktis, långt från alla källor, vilket visar att dessa ämnen är globala föroreningar.

Ett resultat av denna översikt av förekomsten av fluorerande ämnen i miljön, är att det finns en stor informations- och kunskapsbrist om PFC utöver PFOA och PFOS. Dessutom finns generellt en stor brist på human- och miljödata kring dessa PFC-ämnen. De få data som finns indikerar viss toxisk påverkan på människa och miljö. Det krävs fler och djupare studier för att få en tydligare bild av dessa PFC ämnens innan mer långtgående slutsatser kan dras om deras toxiska egenskaper.

Bristen på fysikalisk-kemiska data för PFC-ämnen utöver PFOA och PFOS utgör ett hinder för modellberäkningar kring dessa ämnens spridning i miljön

Bristen på analytiska referenssubstanser utgör idag också ett hinder för utökade studier kring dessa ämnens förekomst i människa och miljö.

Appendix A – List of abbreviations and acronyms

Literature review terminology	OECD glossary[27]
Perfluoro- / Perfluorinated	
A general term for a substance where fluorine (F) is substituted for *all* hydrogen (H) atoms attached to carbon atoms except carbon atoms whose substitution would affect the nature of the functional group(s) present[2]. Examples: $F(CF_2)_nCHO$, $F(CF_2)_nCO_2H$, $F(CF_2)_nSO_3H$, $(CF_3)_2NH$	A fully fluorinated or perfluorinated chemical is one in which all the carbon-hydrogen bonds in a chain have been replaced by carbon-fluorine ones. All fully fluorinated chemicals are man-made. Examples include perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS).
Perfluoroalkyl Substance / Compound (PFA)	
A general term for a substance that is perfluorinated according to the definition given above, but excluding perfluorocarbons. Comments: The term has also been used to describe substances which contain a perfluoroalkyl moiety attached to other atoms that may not be perfluorinated but may have potential to transform to a perfluoroalkyl substance. Justification for the acronym PFA is given in Part 3 of this document.	A substance which bears a perfluorocarbon, also known as a perfluoroalkyl, functional group. F(CF2)n-X where n is an integer and X is not a halogen, or hydrogen.
Perfluorocarbon (PFC)	
A perfluorinated hydrocarbon, especially a perfluorinated alkane, C_nF_{2n+2}. Perfluorocarbons contain only carbon and fluorine atoms.	Perfluorinated chemicals in which all carbon-hydrogen bonds in a chain have been replaced by carbon-fluorine bonds. Examples include perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS). PFC term also refers to PFC precursors, chemicals which contain a perfluoroalkyl moiety attached to other atoms that may not be perfluorinated, and have potential to transform to produce PFCs.
Perfluorinated Surfactant / "Perfluorinated Tenside (PFT)" (in publications of German origin)	
A general term for a surface active, low molecular weight (<1,000 daltons), substance where fluorine (F) is substituted for *all* hydrogen (H) atoms attached to carbon atoms except carbon atoms whose substitution would affect the nature of the functional group(s) present[2]. Example: $F(CF_2)_6SO_3^-NH_4^+$.	A term used to describe a surface active, low molecular weight (<1,000), substance where all carbons bear fluorine in place of hydrogen; the term Fluorosurfactant is less specific but misused synonymously; a perfluorinated example is F(CF2)6SO3-NH4+; while a fluorinated surfactant might be F(CF2)4CH2SO2-NH4+.

[27] http://www.oecd.org/document/54/0,3746,en_21571361_44787844_45162486_1_1_1_1,00.html

Literature review terminology	OECD glossary[27]
Perfluoroalkyl Acid / Perfluorinated Acid (PFAA)	
A general term for a substance which contains a perfluoroalkyl, $F(CF_2)_n$-, functionality bound to an acid functionality, e.g., carboxylate, sulfonate, phosphonate.	Perfluoroalkyl acids
Perfluoroalkyl Carboxylic Acid / Perfluoroalkyl Carboxylate (PFCA)	
A general term for a substance whose chemical structure is $F(CF_2)_nCO_2H$ and its anionic form $F(CF_2)_nCO_2^-$.	Perfluorinated carboxylic acids and their salts are a series of substances whose anion has the general structure of CF3(CF2)nCOO-. Certain members of this class, including the PFCA with 8 carbons, called perfluorooctanoic acid (PFOA or C8), are manufactured as a processing aid to produce fluoropolymers.
Comments: This term may also be used to describe the salts of these acids (e.g., ammonium, sodium, potassium). Justification for the acronyms proposed to represent the various species (free acid, anion and salts) is given in Part 3 of this document.	
Examples:	
PFNA: Perfluorononanoic acid, $F(CF_2)_8CO_2H$, is a fully fluorinated, nine-carbon chain length carboxylic acid (C9) (CAS 375-95-1).	Perfluorononanoic acid is a fully fluorinated, nine-carbon chain length carboxylic acid (C9) (CAS 375-95-1).
PFOA: Perfluorooctanoic acid, $F(CF_2)_7CO_2H$, is a fully fluorinated, eight-carbon chain length carboxylic acid (C8) (CAS 335-67-1).	Perfluorooctanoic acid is a fully fluorinated, eight-carbon chain carboxylic acid (C8) (CAS 335-67-1) sometimes used to refer to the anionic salt form.
APFO: Ammonium perfluorooctanoate, $F(CF_2)_7CO_2NH_4$ (CAS 3825-26-1), is the ammonium salt of PFOA.	
PFBA: Perfluorobutanoic acid, $F(CF_2)_3CO_2H$, is a fully fluorinated, four-carbon chain length carboxylic acid (C4) (CAS 375-22-4).	
PFSA Perfluoroalkyl (or Perfluoroalkane) Sulfonic Acid / Perfluoroalkyl (or Perfluoroalkane) Sulfonate	
A generic term for a substance whose chemical structure is $F(CF_2)_nSO_3H$ and its anionic form $F(CF_2)_nSO_3^-$.	Perfluoroalkyl sulfonate is a generic term used to describe any fully fluorinated carbon chain length sulfonic acid, including higher and lower homologues as well as PFOS.
Comments: This term may also be used to describe its salts (e.g., ammonium, sodium, potassium). Justification for the acronyms proposed to represent the various species (free acid, anion and salts) is given in Part 3 of this document.	
Examples	
PFOS: Perfluorooctane sulfonic acid (CAS 1763-23-1) / sulfonate (CAS 45298-90-6) is a fully fluorinated, eight-carbon chain homologue.	Perfluorooctane sulfonic acid is a fully fluorinated, eight chain sulfonic acid (CAS 1763-23-1) sometimes used to refer to the anionic salt form.

Literature review terminology	OECD glossary[27]
PFHxS: Perfluorohexane sulfonic acid (CAS 355-46-4) / sulfonate (CAS 108427-53-8) is a fully fluorinated, six-carbon chain homologue. *PFBS:* Perfluorobutane sulfonic acid (CAS 375-73-5) / sulfonate (CAS 45187-15-3) is a fully fluorinated, four-carbon chain homologue.	
Perfluoroether A general term for a substance which contains short perfluoroalkyl moieties, typically 1–3 carbon atoms, connected to an oxygen and capped by the same type of perfluoroalkyl / perfluorocarbon functionality or/and by other non-fluorinated functionality.	
Polyfluoroether A general term for a substance which contains short fluoroalkyl moieties that are not fully fluorinated and do contain hydrogen bound to carbon, typically 1–3 carbon atoms, connected to an oxygen and capped by a fluoroalkyl / fluorocarbon functionality or/and by other non-fluorinated functionality.	
Perfluoropolyether (PFPE) A general term for a substance which contains short perfluoroalkyl moieties, 1–3 carbon atoms, connected by oxygen bridges and capped by the same type of perfluoroalkyl / perfluorocarbon functionality or/and by other non-fluorinated functionality.	
ECF & TELOMERS – TERMINOLOGY	
Electrochemical fluorination A process technology used to manufacture fluorinated chemicals where an organic raw material is dissolved in hydrogen fluoride and electrolyzed, resulting in the replacement of hydrogen with fluorine The free-radical nature of the process leads to rearrangement resulting in a product mixture of linear and branched isomers of multiple carbon chain lengths. Comment: A systematic numbering system for identifying the linear and branched congeners of several families of perfluoroalkyl substances has been proposed[3].	
Telomerisation (or Telomerisation) A process technology used to manufacture fluorinated chemicals where pentafluoroethyl iodide (telogen) is reacted with tetrafluoroethylene (TFE, taxogen) to yield a mixture of even carbon-numbered perfluoroalkyl iodides $F(CF_2CF_2)_nI$.	
Telomer (or Fluorotelomer) A general term for a substance derived from a raw material produced from the telomerisation process.	Telomer Based Product: Chemical substances that have the fluoroalkyl portion of the molecule derived from telomers manufactured from low molecular weight polymerisation of tetrafluoroethylene.

Literature review terminology	OECD glossary[27]
Examples	
Fluorotelomer alcohol (FTOH) A general term for substances with the general structure $F(CF_2CF_2)_nCH_2CH_2OH$.	
Fluorotelomer olefin (FTO) A general term for substances with the general structure $F(CF_2CF_2)_nCH=CH_2$.	

Appendix B – Illustration of mapping of SPIN- and preregistered chemicals

Aggregated PFC information of SPIN- and preregistered chemicals

There were 118 CAS numbers on the SPIN list of which 27 were polymers or not-precise defined mixtures, which are excluded from the schemes but listed in the end. 91 CAS numbers were included in the sorting. The SPIN chemicals are indicated with an asterisk (*).

There were 518 CAS numbers on the preregistration list. Of these 79 were polymers or not-precise defined mixtures which are listed at the end.

Additionally synonyms, acronyms, trade names, physical-chemical data and use data have been collected, but only included a few of these data in the tables. This can be further developed.

The applied names are as simple as possible and we have chosen to use the the ones that are easiest to understand. Those are not necessarily the most correct ones but we have made this choice to make it easier to get an overview and see homologue rows and relationships. That's also why "perfluor" and fluorotelomer names have been used where possible. Fluorotelomers with a branched fluoroalkyl chain, however, have got more systematic names.

Perfluoroalkyl sulfonic acids (PFSAs)

CAS 375-73-5	Perfluorobutane sulfonic acid,	PFBS
CAS 355-46-4	Perfluorohexanesulfonic acid,	PFHxS
CAS 375-92-8	Perfluoroheptane sulfonic acid,	PFHpS
CAS 335-77-3	Perfluorodecane sulfonic acid,	PFDS
CAS 79780-39-5	Perfluorododecane sulfonic acid	PFDoS
CAS 70259-86-8	4H-Perfluorobutane sulfonic acid	

Perfluoroalkyl sulfonates (salts)

*CAS 29420-49-3	Potassium perfluorobutane -sulfonate,	PFBS-K
*CAS 3872-25-1	Potassium perfluoropentane sulfonate,	PFPS-K
*CAS 3871-99-6	Potassium perfluorohexane -sulfonate,	PFHxS-K
*CAS 60270-55-5	Potassium perfluoroheptane-sulfonate,	PFHpS-K
CAS 85187-17-3	Potassium perfluorododecane sulfonate,	PFDS
CAS 70259-85-7	Potassium 4H-perfluorobutane sulfonate	
CAS 68259-10-9	Ammonium perfluoro-butanesulfonate	
CAS 68259-09-6	Ammonium perfluoro-pentane sulfonate	
CAS 68259-08-5	Ammonium perfluoro-hexane sulfonate	
CAS 68259-07-4	Ammonium perfluoroheptane sulfonate	
*CAS 17202-41-4	Ammonium perfluorononane sulfonate	
*CAS 67906-42-7	Ammonium perfluorodecanesulfonate	
*CAS 54950-05-9	Sodium 1,4-dioxo-1,4-bis(3,3,4,4,5,5,6,6,7,7,8,8,8-tridecafluorooctoxy)butane-2-sulfonate	
*CAS 70225-15-9	Bis(2-hydroxyethyl)ammonium perfluoroheptane sulfonate	
*CAS 70225-16-0	Bis(2-hydroxyethyl)ammonium perfluorohexane sulfonate	
*CAS 70225-17-1	Bis(2-hydroxyethyl) ammonium perfluoropentane sulfonate	
*CAS 70225-18-2	Bis(2-hydroxyethyl) ammonium perfluorobutane sulfonate	
CAS 56773-42-3	Tetraethyl ammonium heptadecafluorooctane sulfonate	Metal plating, Fumetrol 108, Fluortensid FT 248,
CAS 220689-12-3	Tetrabutyl phosphonium perfluorobutane sulfonate	Anti-Stat FC-1,wetting agent

Perfluoroalkyl sulfinic acid/sulfinates

CAS 68555-66-8	Sodium perfluoroheptane -sulfinate	
CAS 68555-67-9	sodium perfluorooctane sulfinate	C8 Chemical

Perfluorocycloalkyl sulfonic acid and derivatives

CAS 335-24-0	Potassium perfluoro-4-ethyl cyclohexane sulfonate	
*CAS 68156-01-4	Potassium perfluoro[1,2-dimethylcyclohexane] sulfonate	
*CAS 68156-07-0	Potassium perfluoro[1-methylcyclohexane] sulfonate	
CAS 355-03-3	Perfluorocyclohexane sulfonyl fluoride	
CAS 68318-34-3	Perfluoro(2-methylcyclohexane) sulfonyl fluoride	
CAS 68156-06-9	Perfluoro[4-methylcyclohexane] sulfonyl fluoride	
CAS 68156-00-3	Perfluoro[1,2-dimethylcyclohexane] sulfonyl fluoride	

Perfluoroalkyl sulfonamides (FASAs)

CAS 68298-12-4	N-Methyl perfluorobutane sulfonamide	FBSA
CAS 68298-13-5	N-Methyl perfluoropentane sulfonamide	
CAS 68259-15-4	N-Methyl perfluorohexane sulfonamide	
CAS 68259-14-3	N-Methyl perfluoroheptane sulfonamide	
*CAS 68957-62-0	N-Ethyl perfluoroheptane sulfonamide	
CAS 68298-10-2	N-(Phenylmethyl) perfluoroheptane sulfonamide	
*CAS 34449-89-3	N-Ethyl-N-(2-hydroxyethyl)perfluorobutane sulfonamide	
*CAS 34454-97-2	N-(2-hydroxyethyl)-N-methyl perfluorobutane sulfonamide/ N-Methyl perfluorobutane sulfonamidoethanol	MeFBSE
CAS 40630-65-7	N-Allyl perfluorobutane sulfonamide	AlFBSE
CAS 335-97-7	N-Allyl perfluoropentane sulfonamide	
CAS 67584-48-9	N-Allyl perfluorohexane sulfonamide	
CAS 67584-49-0	N-Allyl perfluoroheptane sulfonamide	
CAS 67906-41-6	N-Allyl -N-ethyl perfluoroheptane sulfonamide	
CAS 93894-53-2	N-(2-Hydroxyethyl)-N-methyl 4H-perfluorobutane sulfonamide	
*CAS 68555-74-8	N-(2-Hydroxyethyl)-N-methyl perfluoropentane sulfonamide	

*CAS 68555-75-9	N-(2-Hydroxyethyl)-N-methyl perfluorohexane sulfonamide
*CAS 68555-76-0	N-(2-Hydroxyethyl)-N-methyl perfluoroheptane sulfonamide
*CAS 68555-72-6	N-Ethyl-N-(2-hydroxyethyl) perfluoropentane sulfonamide
*CAS 34455-03-3	N-Ethyl-N-(2-hydroxyethyl) perfluorohexane sulfonamide
*CAS 68555-73-7	N-Ethyl-N-(2-hydroxyethyl) perfluoroheptane sulfonamide
CAS 85665-64-1	N-(2-Hydroxyethyl)-N-propyl perfluorohexane sulfonamide
CAS 68310-02-1	N-Butyl-N-(2-hydroxyethyl) perfluoroheptane sulfonamide
CAS 93894-54-3	N,N-Bis(2-hydroxyethyl) 4H-perfluorobutane sulfonamide
CAS 34455-00-0	N,N-Bis(2-hydroxyethyl) perfluorobutane sulfonamide
CAS 812-94-2	N-(4-Hydroxybutyl)-N-methyl perfluorobutane sulfonamide
CAS 68239-72-5	N-(4-Hydroxybutyl)-N-methyl perfluoropentane sulfonamide,
CAS 68239-74-7	N-(4-Hydroxybutyl)-N-methyl perfluorohexane sulfonamide
CAS 68298-89-5	N-(4-Hydroxybutyl)-N-Methyl perfluoroheptane sulfonamide
CAS 68555-77-1	N-[3-(Dimethylamino)propyl] perfluorobutane sulfonamide
CAS 68555-78-2	N-[3-(Dimethylamino)propyl] perfluoropentane sulfonamide
CAS 50598-28-2	N-[3-(Dimethylamino)propyl] perfluorohexane sulfonamide
CAS 67584-54-7	N-[3-(Dimethylamino)propyl] perfluoroheptane sulfonamide
CAS 67584-63-8	Perfluorobutane sulfonamide, N-ethyl-N-ethyl (ethyl acetate)

Perfluoroalkyl sulfonamide, quaternary ammonium salts

CAS 68957-59-5	Perfluorobutane sulfonamide, N-[3-(dimethylamino)propyl)]-, hydrochloride
CAS 68957-60-8	Perfluoropentane sulfonamide, N-[3-(dimethylamino)propyl)]-,-hydrochloride
CAS 68957-61-9	Perfluorohexane sulfonamide, N-[3-(dimethylamino)propyl)]-,-hydrochloride
CAS 67940-02-7	Perfluoroheptane sulfonamide, N-[3-(dimethylamino)propyl)]-,-hydrochloride
CAS 38850-52-1	Perfluorohexane sulfonamide ,N-carboxymethyl-N-(N'N'N'-trimethylpropanaminium
*CAS 38850-58-7	Perfluorohexane sulfonamide, N-sulfoxypropyl-N-(N',N'-dimethyl-N'-hydroxyethyl propanaminium)
CAS 38850-60-1	Perfluorohexane sulfonamide, N-sulfoxypropyl-N-(N',N'-dimethyl-propanaminium)
*CAS 52166-82-2	Perfluorohexane sulfonamide, N-(N',N',N'-trimethyl propanaminium) chloride
*CAS 53518-00-6	Perfluorobutane sulfonamide, N-(N',N',N'-trimethyl propanaminium) chloride
CAS 67939-95-1	Perfluorobutane sulfonamide, N-(N',N',N'-trimethyl propanaminium) iodide
*CAS 68957-55-1	Perfluoropentane sulfonamide N-(N',N',N'-trimethyl propanaminium) chloride
*CAS 68957-57-3	Perfluoropentane sulfonamide N-(N',N',N'-trimethyl propanaminium) iodide
*CAS 68957-58-4	Perfluorohexane sulfonamide N-(N',N',N'-trimethyl propanaminium) iodide
*CAS 68555-81-7	Perfluoroheptane sulfonamide N-(N',N',N'-trimethyl propanaminium) chloride
*CAS 67584-58-1	Perfluoroheptane sulfonamide N-(N',N',N'-trimethyl propanaminium) iodide
CAS 70225-22-8	Di[Perfluorobutane sulfonamide N-(N',N',N'-trimethyl propanaminium)] sulfate
CAS 70225-24-0	Di[Perfluoropentane sulfonamide N-(N',N',N'-trimethyl propanaminium)] sulfate
CAS 70248-52-1	Di[Perfluorohexane sulfonamide N-(N',N',N'-trimethyl propanaminium)] sulfate
CAS 70225-20-6	Di[Perfluoroheptane sulfonamide N-(N',N',N'-trimethyl propanaminium)] sulfate

Perfluoroalkyl sulfonamide acrylates (MeFASACs)

*CAS 67584-55-8	Perfluorobutane sulfonamide, N-methyl -N-ethyl acrylate/ N-Methyl perfluorobutane sulfonamidoethyl acrylate	
*CAS 67584-56-9	Perfluoropentane sulfonamide, N-methyl -N-ethyl acrylate	
*CAS 67584-57-0	Perfluorohexane sulfonamide, N-methyl -N-ethyl acrylate	
*CAS 68084-62-8	Perfluoroheptane sulfonamide, N-methyl-N-ethyl acrylate	
CAS 17329-79-2	Perfluorobutane sulfonamide, N-ethyl-N-ethyl acrylate	
CAS 68298-06-6	Perfluoropentane sulfonamide N-ethyl-N-ethyl acrylate	
CAS 1893-52-3	Perfluorohexane sulfonamide, N-ethyl-N-ethyl acrylate	
CAS 59071-10-2	Perfluoroheptane sulfonamide, N-ethyl-N-ethyl acrylate	
CAS 1492-87-1	Perfluorobutane sulfonamide, N-methyl- N-butyl acrylate	
CAS 68227-99-6	Perfluoropentane sulfonamide, N-methyl-N-butyl acrylate	
CAS 68227-98-5	Perfluorohexane sulfonamide, N-methyl-N-butyl acrylate	
*CAS 68298-60-2	Perfluoroheptane sulfonamide, N-butyl-N-ethyl acrylate	
CAS 66008-70-6	1H,1H-Perfluoroheptane sulfonamide, N-methyl-N-ethyl acrylate	
*CAS 49859-70-3	1H,1H-Perfluorooctane sulfonamide, N-methyl-N-ethyl acrylate	
CAS 66008-69-3	1H,1H-Perfluorononane sulfonamide, N-methyl-N-ethyl acrylate	
CAS 66008-68-2	1H,1H-Perfluoroundecane sulfonamide, N-methyl-N-ethyl acrylate	
*CAS 72276-05-2	1H,1H-Perfluorododecane sulfonamide, N-methyl-N-ethyl acrylate	
CAS 66008-67-1	1H,1H-Perfluorotridecane sulfonamide, N-methyl-N-ethyl acrylate	C8 Chemical
CAS 72276-06-3	1H,1H-Perfluorotetradecane sulfonamide, N-methyl-N-ethyl acrylate	
CAS 68758-55-4	1H,1H-Perfluoro-pentadecyl sulfonamide N-methyl-N-ethyl acrylate	
CAS 68758-56-5	1H,1H-Perfluoroheptadecyl sulfonamide N-methyl-N-ethyl acrylate	

Perfluoroalkyl sulfonamide methacrylates

*CAS 67584-59-2	Perfluorobutane sulfonamide, N-methyl-N-ethyl methacrylate	
*CAS 67584-60-5	Perfluoropentane sulfonamide, N-methyl-N-ethyl methacrylate	
*CAS 67584-61-6	Perfluorohexane sulfonamide, N-methyl-N-ethyl methacrylate	
CAS 67939-96-2	Perfluoroheptane sulfonamide, N-methyl-N-ethyl methacrylate	
CAS 67939-33-7	Perfluorobutane sulfonamide, N-ethyl-N-ethyl methacrylate	
CAS 67906-73-4	Perfluoropentane sulfonamide, N-ethyl-N-ethyl methacrylate	
CAS 67906-70-1	Perfluorohexane sulfonamide, N-ethyl-N-ethyl methacrylate	
CAS 67939-36-0	Perfluoroheptane sulfonamide, N-ethyl-N-ethyl methacrylate	
CAS 67906-39-2	Perfluorobutane sulfonamide, N-methyl-N-butyl methacrylate	
CAS 67906-40-5	Perfluoropentane sulfonamide, N-methyl-N-butyl methacrylate	
CAS 67939-61-1	Perfluorohexane sulfonamide, N-methyl-N-butyl methacrylate	
CAS 68227-97-4	Perfluoroheptane sulfonamide, N-methyl-N-butyl methacrylate	

Perfluoroalkyl sulfonamide phosphates

CAS 67939-89-3	[Perfluorobutane sulfonamide-N-ethyl]-N-ethyl dihydrogen-phosphate	
CAS 67939-90-6	[Perfluoropentane sulfonamide-N-ethyl]-N-ethyl dihydrogen-phosphate	MonoPAP
CAS 67969-65-7	[Perfluoroheptane sulfonamide-N-ethyl]-N-ethyl dihydrogen-phosphate	MonoPAP
CAS 67923-61-9	[Perfluoroheptane sulfonamide-N-ethyl]-N,N'-diethyl dihydrogen-phosphate	DiPAP
CAS 67939-98-4	Diammonium [Perfluoroheptane sulfonamide-N-ethyl]-N,N'-diethyl dihydrogenphosphate	DiPAP
CAS 67939-91-7	Di[perfluorobutane sulfonamide N-ethyl]-N,N'-diethyl phosphate	DiPAP
CAS 67939-87-1	Di[perfluoropentane sulfonamide N-ethyl]-N,N'-diethyl phosphate	DiPAP
CAS 67939-92-8	Di[perfluorohexane sulfonamide N-ethyl]-N,N'-diethyl phosphate	DiPAP
CAS 67939-93-9	Di[perfluoroheptane sulfonamide N-ethyl]-N,N'-diethyl phosphate	DiPAP
CAS 67939-97-3	Ammonium di[perfluoroheptane sulfonamide N-ethyl]-N,N'-diethyl phosphate	DiPAP
CAS 67939-94-0	Tri[perfluoroheptane sulfonamide N-ethyl]-N,N',N"-triethyl phosphate	TriPAP

Perfluoroalkyl sulfonyl halides

CAS 375-72-4	Perfluorobutane sulfonyl fluoride
CAS 90268-45-4	Perfluorobutane sulfonyl fluoride, branched
CAS 375-81-5	Perfluoropentane sulfonyl fluoride
CAS 335-71-7	Perfluoroheptane sulfonyl fluoride
CAS 55591-23-6	Perfluorohexane sulfonyl chloride
CAS 68259-06-3	Perfluorononane sulfonyl fluoride
CAS 51947-19-4	4-Perfluoroalkenoxybenzene -sulfonyl chloride

Other polyfluoroalkyl sulfur compounds

CAS 36913-91-4	Perfluorobutane sulfonic -anhydride	
CAS 93894-55-4	4H-Perfluorobutane sulfonic -anhydride	
CAS 68957-33-5	N-Ethyl-N-perfluorobutyl sulfonyl glycine	
CAS 68957-31-3	N-Ethyl-N-perfluoropentyl -sulfonyl glycine	
CAS 68957-32-4	N-Ethyl-N-perfluorohexyl sulfonyl glycine	
CAS 68957-63-1	N-Ethyl-N-perfluoroheptyl -sulfonyl glycine	
CAS 68555-79-3	Ethyl N-ethyl-N-perfluoropentyl sulfonyl glycinate	
CAS 68957-53-9	Ethyl N-ethyl-N-perfluorohexyl sulfonyl glycinate	
CAS 68957-54-0	Ethyl N-ethyl-N-perfluoroheptyl sulfonyl glycinate	
*CAS 67584-51-4	Potassium N-ethyl-N-perfluorobutyl sulfonyl glycinate	
*CAS 67584-52-5	Potassium N-ethyl-N-perfluoropentyl sulfonyl glycinate	
*CAS 67584-53-6	Potassium N-ethyl-N-perfluorohexyl sulfonyl glycinate	
*CAS 67584-62-7	Potassium N-ethyl-N-perfluoroheptyl sulfonyl glycinate	
*CAS 68900-97-0	Chromium(III) N-ethyl-N-perfluorobutyl sulfonyl glycinate	
*CAS 68891-99-6	Chromium(III) N-ethyl-N-perfluoropentyl sulfonyl glycinate	
*CAS 68891-98-5	Chromium(III) N-ethyl-N-perfluorohexyl sulfonyl glycinate	
*CAS 68891-97-4	Chromium(III) N-ethyl-N-perfluoroheptyl sulfonyl glycinate	
CAS 68555-68-0	Sodium N-ethyl-N-perfluorobutyl sulfonyl glycinate	
CAS 68555-69-1	Sodium N-ethyl-N-perfluoropentyl sulfonyl glycinate	
CAS 68555-70-4	Sodium N-ethyl-N-perfluorohexyl sulfonyl glycinate	
CAS 68555-71-5	Sodium N-ethyl-N-perfluoroheptyl sulfonyl glycinate	
CAS 52584-45-9	4-Perfluoroalkenoxybenzene sulfonic acid	
CAS 68299-19-4	Sodium (perfluorobutylsulfonyl)aminomethyl benzene sulfonate	
CAS 68299-20-7	Sodium (perfluoropentylsulfonyl)aminomethyl benzene sulfonate	
CAS 68299-21-8	Sodium (perfluorohexylsulfonyl)-aminomethyl benzene sulfonate	
CAS 68299-29-6	Sodium (perfluoroheptylsulfonyl)-aminomethyl benzene sulfonate	
*CAS 68649-26-3	Reaction product with PFOS and PFBS derivatives	C8 chemical
*CAS 68541-01-5	Perfluoroheptane sulfonic acid ester with complex alcohol	Tetrachloro-phthalic acid derivative
*CAS 68541-02-6	Perfluoropentane sulfonic acid ester with complex alcohol	Tetrachloro-phthalic acid derivative
*CAS 68568-54-7	Perfluorobutane sulfonic acid ester with complex alcohol	Tetrachloro-phthalic acid derivative
CAS 69013-34-9	N-Methyl-4-[[4,4,5,5,5-pentafluoro-3-(1,1,2,2,2-pentafluoroethyl)-1,2,3-tris(trifluoromethyl)-1-penten-1-yl]oxy]-N-[2-(phosphonooxy)ethyl]-benzene sulfonamide	

Perfluoroalkyl carboxylic acids (PFCA)

*CAS 2706-90-3	Perfluoropentanoic acid	PFPA
*CAS 307-24-4	Perfluorohexanoic acid	PFHxA
*CAS 375-85-9	Perfluoroheptanoic acid	PFHpA
CAS 375-95-1	Perfluorononanoic acid	PFNA
CAS 15899-31-7	Perfluoroisononanoic acid	
CAS 335-76-2	Perfluorodecanoic acid	PFDA, C10
CAS 2058-94-8	Perfluoroundecanoic acid	PFuDA, C11
CAS 16486-94-5	Perfluoroisoundecanoic acid	
CAS 307-55-1	Perfluorododecanoic acid	PFDoA, C12
CAS 72629-94-8	Perfluorotridecanoic acid	C13
CAS 16486-96-7	Perfluoroisotridecanoic acid	
CAS 376-06-7	Perfluorotetradecanoic acid	C14
CAS 18024-09-4	Perfluoropentadecanoic acid	C15
CAS 18024-09-4	Perfluoroisopentadecanoic acid	
CAS 67905-19-5	Perfluorohexadecanoic acid	C16
CAS 16517-11-6	Perfluorostearic acid	C18
CAS 68310-12-3	Perfluoroeicosanoic acid	C20
CAS 336-08-3	Perfluoroadipic acid	
CAS 376-72-7	5H-Perfluoropentanoic acid	
CAS 1546-95-8	7H-Perfluoroheptanoic acid	
CAS 76-21-1	9H-perfluorononanoic acid	
CAS 1765-48-6	11H-Perfluoroundecanoic acid	

Perfluoroalkyl carboxylic salts

CAS 2706-89-0	Sodium perfluoropentanoate	PFPA
CAS 20109-59-5	Sodium perfluoroheptanoate	PFHpA
CAS 68259-11-0	Ammonium perfluoropentanoate	PFPA
CAS 21615-47-4	Ammonium perfluorohexanoate	PFHxA
CAS 6130-43-4	Ammonium perfluoroheptanoate	PFHpA
CAS 3658-62-6	Ammonium perfluoro-isononanoate	PFiNA
CAS 3108-42-7	Ammonium perfluorodecanoate	PFDA
CAS 3658-63-7	Ammonium perfluoro-isoundecanoate	
CAS 3793-74-6	Ammonium perfluorododecanoate	
CAS 22715-45-3	Ammonium 5H-perfluoropentanoate	
CAS 376-34-1	Ammonium 7H-perfluoroheptanoate	
CAS 1868-86-6	Ammonium 9H-perfluorononanoate	
CAS 307-71-1	Potassium 11H-Perfluoroundecanoate	
CAS 3658-57-9	Ammonium 7-(chlorodifluoromethyl)perfluorooctanoate	
CAS 16557-94-1	Ammonium 7-(chlorodifluoromethyl)perfluoroheptanoate	
CAS 68015-84-9	Ethylammonium perfluoro-isohexanoate	
CAS 68015-86-1	Ethylammonium perfluoro-isooctanoate	
CAS 68015-85-0	Ethylammonium perfluoro-isodecanoate	
CAS 68015-87-2	Ethylammonium perfluoro-isododecanoate	
CAS 68052-68-6	Ethylammonium perfluoro-isopentadecanoate	

Perfluoroalkyl carboxylic acid halides

CAS 375-62-2	Perfluoropentanoyl fluoride
CAS 355-38-4	Perfluorohexanoyl fluoride
CAS 375-84-8	Perfluoroheptanoyl fluoride
CAS 18017-31-7	Perfluoroisohexanoyl fluoride
CAS 15899-29-3	Perfluoroisoheptanoyl fluoride
CAS 15742-62-8	Perfluoroisononanoyl fluoride
CAS 15720-98-6	Perfluoroisoundecanoyl fluoride
CAS: 15811-52-6	Perfluoroisotridecanoyl fluoride
CAS 68025-62-7	Perfluoroisopentadecanoyl fluoride
CAS 37881-62-2	Perfluorohexanedioyl difluoride
CAS 423-95-0	9H-Perfluorononanoyl chloride
CAS 64018-26-4	1H,1H-Perfluorododecanoyl chloride

Perfluoroalkyl alcohols/ketones

CAS 355-80-6	1H,1H,5H-perfluoropentanol
CAS 375-82-6	1H,1H-Perfluoroheptanol
CAS 335-99-9	1H,1H,7H-Perfluoroheptanol
CAS 307-30-2	1H,1H-Perfluorooctanol
*CAS 376-18-1	1H,1H,9H-Perfluorononanol
CAS 307-70-0	1H,1H,11H-Perfluoroundecanol
CAS 67824-44-6	3-Perfluoroisononyl-propane-1,2-diol
CAS 94159-92-9	1-Phenoxy-3-perfluoroisononyl-2-propanol
CAS 94158-62-0	1-[2-(2-butoxyethoxy)ethoxy]-3-perfluoroisononyl-propan-2-ol
CAS 93776-07-9	32-(Perfluorodecyl)-2,5,8,11,14,17,20,23,26-decaoxatetratetracontan-31-ol
CAS 93776-06-8	32-(Perfluorododecyl)-2,5,8,11,14,17,20,23,26-decaoxatetratetracontan-31-ol
CAS 93776-09-1	32-(Perfluoroisotridecyl)-2,5,8,11,14,17,20,23,26-decaoxatetratetracontan-31-ol
CAS 93776-11-5	32-(Perfluoroisononyl)- 31-hydroxy-dotetracontane-2,5,8,11,14,17,20,23,26,29-decone
CAS 93776-10-4	32-(Perfluoroisoundecyl)- 31-hydroxy-dotetracontane-2,5,8,11,14,17,20,23,26,29-decone

Perfluoroalkyl halides

CAS 375-88-2	Perfluoroheptyl bromide	C7
CAS 307-43-7	Perfluorodecyl bromide	C10
CAS 25398-32-7	Perfluoroalkyl iodides	Zonyl TELA-N
CAS 423-39-2	Perfluorobutyl iodide	C4
CAS 638-79-9	Perfluoropentyl iodide	C5
CAS 355-43-1	Perfluorohexyl iodide	C6
CAS 335-58-0	Perfluoroheptyl iodide	C7
CAS 507-63-1	Perfluorooctyl iodide	C8 chemical
CAS 558-97-4	Perfluorononyl iodide	C9
CAS 865-77-0	Perfluoroisononyl iodide	C9
CAS 423-62-1	Perfluorodecyl iodide	C10
CAS 677-93-0	Perfluoroisoundecyl iodide	C11
CAS 307-60-8	Perfluorododecyl iodide	C12
CAS 307-63-1	Perfluorotetradecyl iodide	C14
CAS 3248-61-1	Perfluoroisotridecyl iodide	C13
CAS 3248-63-3	Perfluoroisopentadecyl iodide	C15
CAS 355-50-0	Perfluorohexadecyl iodide	C16
CAS 29809-35-6	Perfluorooctadecyl iodide	C18
CAS 29809-34-5	Perfluoroeicosyl iodide	C20
CAS 29809-36-7	Perfluorodocosanyl iodide	C22
CAS 39823-55-7	Perfluorotetracosyl iodide	C24

CAS 65975-15-7	Perfluorohexacosanyl iodide	C26
CAS 375-50-8	1,4-diiodoperfluorobutane	
CAS 375-80-4	1,6-Diiodoperfluorohexane	

Perfluoroalkyl alkyl ethers

*CAS 297730-93-9	Ethyl perfluoroisoheptyl ether	Novec Engineered Fluid HFE 7500
*CAS 163702-08-7	Methyl perfluoroisobutyl ether	3M Novec Engineered Fluid HFE-7100 (Mixture with CAS 163702-07-6.
*CAS 163702-07-6	Methyl perfluorobutyl ether	Cosmetic Fluid CF 61; 3M Novec Engineered Fluid HFE-7100 (Mixture with CAS 163702-08-7)
*CAS 163702-05-4	Ethyl perfluorobutyl ether	
CAS 66396-73-4	4H-Perfluorobutyl vinyl ether	
CAS 78971-81-0	1H,1H,7H-Perfluoroheptyl vinyl ether	
CAS 71726-31-3	1H,1H,9H-Perfluorononyl vinyl ether	
CAS 94231-58-0	1H,1H,11H-Perfluoroundecyl vinyl ether	
CAS 73928-40-2	Perfluorovinyl 5H-perfluoropentane ether	
CAS 70729-63-4	Tributyl ammonium 4-((4,4,5,5,5-pentafluoro-3-(pentafluoroethyl)-1,2,3-tris(trifluoromethyl)pent-1-enyl)oxy)benzene sulfonate	
CAS 84029-54-9	Tetratriacontafluoro-10,13,16,19-tetraoxaoctacosadiene	
CAS 93776-05-7	Bis(1-perfluoroiisononyl-4-methyl- 3-oxy-2-hexanol) ether	
CAS 93776-01-3	Bis(1-perfluorodecyl-4-methyl- 3-oxy-2-hexanol) ether	
CAS 93776-04-6	Bis(1-perfluoroisoundecyl-4-methyl- 3-oxy-2-hexanol) ether	
CAS 93776-00-2	Bis(1-perfluorododecyl-4-methyl- 3-oxy-2-hexanol) ether	
CAS 93776-03-5	Bis(1-perfluoroisotridecyl-4-methyl- 3-oxy-2-hexanol) ether	

Perfluoroalkyl amines

CAS 311-89-7	Tri(perfluorobutyl)amine	
CAS 338-84-1	Tri(perfluoropentyl)amine	
CAS 31841-41-5	N,N-bis(2-hydroxyethyl)-N-methyl ammonium iodide	C8 chemical
CAS 80909-29-1	Perfluoroisononyl 2-ethyl-propyl trimethyl ammonium iodide	
CAS 94159-78-1	N,N-Bis(2-hydroxyethyl)-N-methyl-N-[(2-hydroxy-3-perfluoroisononyl)propyl] ammonium iodide	
CAS 93776-17-1	N,N-Bis(2-hydroxyethyl)-N-methyl-N-[(2-hydroxy-3-perfluorodecyl)propyl] ammonium iodide	
CAS 94159-77-0	N,N-Bis(2-hydroxyethyl)-N-methyl-N-[(2-hydroxy-3-perfluoroisoundecyl)propyl] ammonium iodide	
CAS 93776-16-0	N,N-Bis(2-hydroxyethyl)-N-methyl-N-[(2-hydroxy-3-perfluorododecyl)propyl] ammonium iodide	
CAS 94159-76-9	N,N-Bis(2-hydroxyethyl)-N-methyl-N-[(2-hydroxy-3-perfluoroisotridodecyl)propyl] ammonium iodide	
CAS 73353-26-1	1-[[3-(Dimethylamino)propyl]amino]-3-perfluoroisononyl-2-propanol	
CAS 94159-80-5	1-[[3-(Dimethylamino)propyl]amino]-3- perfluorodecyl-2-propanol	
CAS 94159-83-8	1-[[3-(Dimethylamino)propyl]amino]-3-perfluoroisoundecyl-2-propanol	
CAS 94159-79-2	1-[[3-(Dimethylamino)propyl]amino]-3- perfluoro-dodecyl-2-propanol	
CAS 94159-82-7	1-[[3-(Dimethylamino) propyl]amino]-3-perfluoro-isotridecyl-2-propanol	

Perfluoroalkyl amino acids/salts

CAS 94159-89-4	Potassium N-methyl-N- [(3-perfluoro-isononyl-2-hydroxy)propyl] glycinate
CAS 93776-13-7	3-[Dimethyl-[3-[(3-perfluoro-decyl-2-hydroxy)amino]-propyl]ammonio] propanoate
CAS 93777-12-9	3-[Dimethyl-[3-[(3-perfluoro-isoun-decyl-2-hydroxy)amino]-propyl]-ammonio] propanoate
CAS 93776-15-9	3-[Dimethyl-[3-[(3-perfluoro-isotridecyl-2-hydroxy)amino]-propyl]ammonio] propanoate
CAS 93776-12-6	3-[Ethyl-[3-[(3-perfluorododecyl-2-hydroxy)amino]propyl-]ammonio]-propanoate
CAS 73353-25-0	N-[(2-Carboxyethyl)-3-[2-hydroxy-3-perfluoroisononyl]-propylamino] -N,N-dimethyl-propanaminium hydroxide

Perfluoroalkyl phosphates

CAS 78974-42-2	Perfluoroisononyl ethyl dihydrogen phosphate	MonoPAP, isotelomer
CAS 94200-56-3	Perfluoroisoundecyl ethyl dihydrogen phosphate	MonoPAP, isotelomer
CAS 94200-57-4	Perfluoroisotridecyl ethyl dihydrogen phosphate	MonoPAP, isotelomer
CAS 93857-42-2	Perfluoroisopentadecyl ethyl dihydrogen phosphate	MonoPAP, isotelomer
CAS 94231-59-1	Perfluoroisoheptadecyl ethyl dihydrogen phosphate	MonoPAP, isotelomer
CAS 93857-49-9	Diammonium perfluoroisononyl ethyl phosphate	MonoPAP
CAS 93857-45-5	Diammonium perfluorodecyl ethyl phosphate	MonoPAP
CAS 93857-50-2	Diammonium perfluoroisoundecyl ethyl phosphate	MonoPAP
CAS 93857-46-6	Diammonium perfluorododecyl ethyl phosphate	MonoPAP
CAS 93857-51-3	Diammonium perfluoroisotridecyl ethyl phosphate	MonoPAP
CAS 93857-47-7	Diammonium perfluorotetradecyl ethyl phosphate	MonoPAP
CAS 93857-52-4	Diammonium perfluoroisopentadecyl ethyl phosphate	MonoPAP
CAS 93857-48-8	Diammonium perfluorohexadecyl ethyl phosphate	MonoPAP
CAS 93857-43-3	Diammonium perfluoroisoheptadecyl ethyl phosphate	MonoPAP
CAS 1895-26-7	Di[2-(perfluorodecyl)ethyl] hydrogen phosphate	DiPAP
CAS 78974-41-1	Di[(2-perfluoroisononyl)ethyl] hydrogen phosphate	DiPAP
CAS 93857-55-7	Di[(2-perfluoroisoundecyl)ethyl] hydrogen phosphate	DiPAP
CAS 93857-56-8	Di[(2-perfluoroisotridecyl)ethyl] hydrogen phosphate	DiPAP
CAS 93857-53-5	Di[2-(perfluorotetradecyl)ethyl] hydrogen phosphate	DiPAP
CAS 93776-29-5	Di[(2-perfluoroisopentadecyl)ethyl] hydrogen phosphate	DiPAP
CAS 93857-54-6	Di[2-(perfluorohexadecyl)ethyl] hydrogen phosphate	DiPAP
CAS 93776-19-3	Di[(2-perfluoroisoheptadecyl)ethyl] hydrogen phosphate	DiPAP
CAS 93776-24-0	Ammonium di[(2-perfluoroisononyl)ethyl] phosphate	DiPAP
CAS 93776-21-7	Ammonium di[(2-perfluorodecyl)ethyl] phosphate	DiPAP
CAS 93776-25-1	Ammonium di[(2-perfluoro-isoundecyl)ethyl] phosphate	DiPAP
CAS 93776-22-8	Ammonium di[(2-perfluoro-dodecyl)ethyl] phosphate	DiPAP
CAS 93776-26-2	Ammonium di[(2-perfluoro-isotridecyl)ethyl] phosphate	DiPAP
CAS 93777-13-0	Ammonium di[(2-perfluoro-tetradecyl)ethyl] phosphate	DiPAP
CAS 93776-27-3	Ammonium di[(2-perfluoro-isopentadecyl)ethyl] phosphate	DiPAP
CAS 93776-23-9	Ammonium di[(2-perfluoro-hexadecyl)ethyl] phosphate	DiPAP
CAS 93776-28-4	Ammonium di[(2-perfluoroisoheptadecyl)ethyl] phosphate	DiPAP
CAS 94291-77-7	Bis(2-hydroxyethyl)ammonium di [(2-perfluoroisononyl)ethyl] phosphate	DiPAP
CAS 94291-78-8	Bis(2-hydroxyethyl)ammonium di [(2-perfluoroisoundecyl)ethyl] phosphate	DiPAP
CAS 94231-56-8	Bis(2-hydroxyethyl)ammonium di [(2-perfluoroisotridecyl)ethyl] phosphate	DiPAP
CAS 93776-30-8	Bis(2-hydroxyethyl)ammonium di [(2-perfluoroisopentadecyl)ethyl] phosphate	DiPAP

CAS 93776-31-9	Bis(2-hydroxyethyl)ammonium di [(2-perfluoroisoheptadecyl)ethyl] phosphate	DiPAP
CAS 355-86-2	Tri(1H,1H,5H-perfluoropentyl) phosphate	TriPAP
CAS 54009-73-3	(2-Hydroxy-3-perfluoroisononyl)propyl dihydrogenphosphate	MonoPAP
CAS 94158-70-0	(2-Hydroxy-3-perfluorodecyl)-propyl dihydrogenphosphate	MonoPAP
CAS 63295-27-2	(2-Hydroxy-3-perfluoroisoundecan-yl)propyl dihydrogen phosphate	MonoPAP
CAS 94200-42-7	(2-Hydroxy-3-perfluorododecyl)-propyl dihydrogen-phosphate	MonoPAP
CAS 63295-28-3	(2-Hydroxy-3-perfluoroisotridecanyl)propyl dihydrogen phosphate	MonoPAP
CAS 94200-43-8	(2-Hydroxy-3-perfluorotetradecyl)propyl dihydrogen-phosphate	MonoPAP
CAS 63295-29-4	(2-Hydroxy -3-perfluoroisopentadecanyl)propyl dihydrogen phosphate	MonoPAP
CAS 94200-44-9	(2-Hydroxy -3-perfluoroisohexadecanyl)propyl dihydrogen phosphate	MonoPAP
CAS 63295-18-1	Diammonium (2-hydroxy-3-perfluorononyl)propyl phosphate	MonoPAP
CAS 94200-46-1	Diammonium (2-hydroxy-3-perfluorodecyl)propyl phosphate	MonoPAP
CAS 94200-50-7	Diammonium (2-hydroxy-3-perfluoroisoundecyl)propyl phosphate	MonoPAP
CAS 94200-47-2	Diammonium (2-hydroxy-3-perfluorododecyl)propyl phosphate	MonoPAP
CAS 94200-51-8	Diammonium (2-hydroxy-3-per-fluoroisotridecyl)propyl phosphate	MonoPAP
CAS 94200-48-3	Diammonium (2-hydroxy-3-per-fluorotetradecyl)propyl phosphate	MonoPAP
CAS 94200-52-9	Diammonium (2-hydroxy-3-perfluoroisopentadecyl)propyl phosphate	MonoPAP
CAS 94200-49-4	Diammonium (2-hydroxy-3-per-fluorohexadecyl)propyl phosphate	MonoPAP
CAS 94200-53-0	Diammonium (2-hydroxy-3-perfluoroisoheptadecyl)propyl phosphate	MonoPAP

Perfluoroalkyl acrylates

CAS 307-98-2	1H,1H-Perfluorooctyl acrylate
CAS 4180-26-1	1H,1H,9H-Hexadecafluorononyl acrylate

Perfluoroalkyl methacrylates

*CAS 3934-23-4	1H,1H-Perfluorooctyl methacrylate
CAS 1841-46-9	1H,1H,9H-Perfluorononyl methacrylate

Other perfluoralkyl esters

CAS 376-50-1	Perfluoroadipic acid diethylester

Perfluoroalkyl heterocyclic compounds

CAS 38565-52-5	1H,1H-Perfluoroheptyl oxirane
CAS 38565-53-6	1H,1H-Perfluorononyl oxirane
CAS 47795-34-6	1H,1H-Perfluorododecyl oxirane
CAS 41925-33-1	1H-1H-Perfluoroisodecyl oxirane
CAS 54009-78-8	1H,1H-Perfluoroisotridecanyl oxirane
CAS 54009-79-9	1H,1H-Perfluoroisoheptadecanyl oxirane
CAS 54009-77-7	2H-Perfluoroisohexadecyl oxirane
CAS 356-47-8	Perfluoro-2-methyl tetrahydropyran)

CAS 40464-54-8	Perfluoro-2-butyl tetrahydrofuran	
*CAS 335-36-4	Perfluoro-2-isobutyl tetrahydrofuran	
CAS 69661-30-9	Perfluoro-[2,3,4,5-tetramethyl-3-ethyl] tetrahydrofuran	
CAS 94159-90-7	2,2-Dimethyl-4-(1H,1H-perfluoroisodecyl)-1,3-dioxolane	
CAS 359-71-7	Perfluoro-N-methyl piperidine	
CAS 564-11-4	Perfluoro-N-ethyl piperidine	
CAS 42060-64-0	Perfluorosulfolane	
CAS 71356-38-2	1-(Carboxylatomethyl)-1-(2-hydroxyethyl)-4-(perfluoro-1-oxodecyl) piperazinium	

Perfluoroalkylsilanes

CAS 375-63-3	Trichloro(1,1,2,2,3,3,4,4-octafluoro-butyl)silane	
CAS 67584-50-3	N-Ethyl-N-(3-(trichlorosilyl)-propyl)perfluoroheptane sulfonamide	
CAS 68239-75-8	N-Ethyl-N-(3-(trimethoxysilyl) propyl) perfluoroheptane sulfonamide	

Fluorotelomer alcohols

*CAS 2043-47-2	4:2 Fluorotelomer alcohol	4:2 FTOH
*CAS 647-42-7	6:2 Fluorotelomer alcohol	6:2 FTOH
*CAS 678-39-7	8:2 Fluorotelomer alcohol	8:2 FTOH, C8 chemical
*CAS 865-86-1	10:2 Fluorotelomer alcohol	10:2 FTOH
*CAS 39239-77-5	12:2 Fluorotelomer alcohol	12:2 FTOH
*CAS 60699-51-6	14:2 Fluorotelomer alcohol	14:2 FTOH
*CAS 65104-67-8	16:2 Fluorotelomer alcohol	
CAS 65104-65-6	18:2 Fluorotelomer alcohol	

Fluorotelomer halogenides

CAS 2043-55-2	4:2 Fluorotelomer iodide	
CAS 1682-31-1	5:2 Fluorotelomer iodide	
CAS 2043-57-4	6:2 Fluorotelomer iodide	Zonyl®, TELB-LN
CAS 2043-52-9	7:2 Fluorotelomer iodide	
*CAS 2043-53-0	8:2 Fluorotelomer iodide	C8 chemical, Zonyl®TELB-LN
CAS 65510-56-7	9:2 Fluorotelomer iodide	
CAS 2043-54-1	10:2 Fluorotelomer iodide	Zonyl®, TELB-LN
CAS 30046-31-2	12:2 Fluorotelomer iodide	
CAS 65104-63-4	18:2 Fluorotelomer iodide	
CAS 65510-55-6	14:2 Fluorotelomer iodide	
CAS 26650-09-9	6:2 Fluorotelomer thiocyanate	

Fluortelomer sulfonates, sulfonyl chlorides and sulfonamides

*CAS 27619-97-2	6:2 Fluorotelomer sulfonic acid	Fumetrol®21 (metal plating), Forafac 1033
CAS 59587-38-1	Potassium 6:2 fluortelomer sulfonate	Zonyl 1176, wetting agent
CAS 65702-23-0	5:2 Fluorotelomer sulfonyl chloride	
CAS 65702-24-1	9:2 Fluorotelomer sulfonyl chloride	
*CAS 72276-08-5	10:2 Fluorotelomer sulfonyl chloride	
CAS 68758-57-6	12:2 Fluorotelomer sulfonyl chloride	
*CAS 34455-29-3	6:2 Fluorotelomer sulfonamide N-propylmethyl betaine	
CAS 61798-69-4	6:2 Fluorotelomer sulfonamide, N-propylethyl betaine	
CAS 66008-71-7	6:2 Fluorotelomer sulfonamide, N-methyl-N-propyl betaine	
CAS 66008-72-8	6:2 Fluorotelomer sulfonamide, N-methyl-N-propan-aminium N'-2-carboxyethyl	
CAS 72276-07-4	14:2 Fluorotelomer sulfonamide, N-methyl-N-ethyl acrylate	

Fluorotelomer acrylates

CAS 1799-84-4	4:2 Fluorotelomer acrylate	
*CAS 17527-29-6	6:2 Fluorotelomer acrylate	Zonyl® TA-N <5%
*CAS 27905-45-9	8:2 Fluorotelomer acrylate	C8 Chemical, Zonyl® TA-N<65%
*CAS 17741-60-5	10:2 Fluorotelomer acrylate	Zonyl® TA-N <29%
*CAS 34395-24-9	12:2 Fluorotelomer acrylate	
*CAS 34362-49-7	14:2 Fluorotelomer acrylate	
*CAS 65150-93-8	16:2 Fluorotelomer acrylate	
*CAS 65104-64-5	18:2 Fluorotelomer acrylate	
CAS 15577-26-1	2-(Perfluoroisononyl) ethyl acrylate	
CAS 52956-81-7	2-(Perfluoroisoundecyl) ethyl acrylate	
CAS 52956-82-8	2-(Perfluoroisotridecyl) ethyl acrylate	
CAS 91615-22-4	2-(Perfluoroisopentadecyl) ethyl acrylate	
CAS 94158-63-1	2-(Perfluoroisoheptadecyl) ethyl acrylate	

Fluorotelomer methacrylates

*CAS 2144-53-8	6:2 Fluorotelomer methacrylate	Capstone™ 62-MA
*CAS 1996-88-9	8:2 Fluorotelomer methacrylate	C8 chemical
CAS 2144-54-9	10:2 Fluorotelomer methacrylate	
*CAS 6014-75-1	12:2 Fluorotelomer methacrylate	
CAS 4980-53-4	14:2 Fluorotelomer methacrylate	
CAS 59778-97-1	16:2 Fluorotelomer methacrylate	
CAS 65104-66-7	18:2 Fluorotelomer methacrylate	
CAS 15166-00-4	2-(Perfluoroisononyl) ethyl methacrylate	
CAS 74256-14-7	2-(Perfluoroisoundecyl) ethyl methacrylate	
CAS 74256-15-8	2-(Perfluoroisotridecyl) ethyl methacrylate	
CAS 94158-64-2	2-(Perfluoroisopentadecyl) ethyl methacrylate	
CAS 94158-65-3	2-(Perfluoroisoheptadecyl) ethyl methacrylate	

Other acrylates

CAS 24407-09-8	3-Perfluoroisononyl-2-hydroxypropyl acrylate
CAS 16083-87-7	3-Perfluoroisotridecyl-2-hydroxypropyl acrylate
CAS 16083-78-6	3-Perfluoroisoheptadecyl-2-hydroxypropyl acrylate

Fluorotelomer phosphates

CAS 94200-54-1	14:2 Fluorotelomer dihydrogen phosphate	PAP
CAS 57678-05-4	10:2 Fluorotelomer dihydrogen phosphate	PAP
CAS 57678-07-6	12:2 Fluorotelomer dihydrogen phosphate	PAP
CAS 94200-55-2	16:2 Fluorotelomer dihydrogen phosphate	PAP
CAS 101896-22-4	Di(9:2 Fluorotelomer) phosphate	diPAP
CAS 57677-98-2	Di(10:2 Fluorotelomer) hydrogen phosphate with 2,2'-iminodiethanol	diPAP
CAS 57677-99-3	Di(12:2 fluorotelomer) hydrogen phosphate	diPAP
CAS 57678-00-9	Di(12:2 Fluorotelomer) hydrogen phosphate with 2,2'-iminodiethanol	diPAP
CAS 94291-75-5	Di(14:2 fluorotelomer) hydrogen phosphate with 2,2’iminodiethanol	diPAP
CAS 94291-76-6	Di(16:2 fluorotelomer) hydrogen phosphate with 2,2’iminodiethanol	diPAP

Other fluorotelomers

CAS 94094-26-5	4:2 Fluorotelomer -1,1'-di(tetradecanoic acid)-methyl silane	
CAS 61798-68-3	8:2 Fluorotelomer pyridinium salt	C8 Chemical
*CAS 78560-45-9	6:2 Fluorotelomer trichlorosilane	
*CAS 78560-44-8	8:2 Fluorotelomer trichlorosilane	C8 chemical
CAS 83048-65-1	8:2 Fluorotelomer trimethoxysilane	C8 chemical
CAS 67846-66-6	Sodium C-ethyl [2-(sulfonato-thio)ethyl]-(3,3,4,4,5,5,6,6,7,7,8,-8,8-tridecafluorooctyl) carbamate	

Polymers, CAS numbers

479029-28-2	56372-23-7	65530-65-6
68298-79-3	68298-80-6	68298-81-7
45080-67-0 Polyfox PF-156A, polymer with C3-fluoro chain, floor polish	452080-64-7 Polyfox PF-136A, polymer with C3-fluoro chain, floor polish	65545-80-4Zonyl FSO 100, wetting agent
69991-61-3	65530-61-2 Zonyl UR	65530-60-1 Zonyl BA-N
123171-68-6 Zonyl® FSK, wetting agent[28]	135228-60-3 Zonyl 9155, carpet protector	203743-03-7 Foraperle®225, repellent
60164-51-4 polyperfluoropropyl ether, Zonyl PFPE, lubricant	65605-70-1 Zonyl Acrylate N-Li	65605-58-5 Zonyl G Fabric Protector
65530-62-3 Zonyl UR	65530-64-5 Zonyl 9027, repellant	65530-63-4 Zonyl 9027, repellant
65530-69-0 Zonyl FSA, wetting agent	6530-82-7 Zonyl TELB-L67	65530-74-7 Zonyl 9027, repellent
71215-70-8 (Zonyl PFHEI),		

Undefined mixtures, CAS numbers

68081-83-4	68140-18-1	68140-19-2	68140-20-5
68187-25-7	68187-47-3	68140-21-6	68391-08-2 (Zonyl BA-LD)
68391-09-3	68412-68-0	68412-69-1	70983-60-7
71608-60-1	74499-44-8	84238-62-0	85631-54-5
86508-42-1	90622-43-8	91032-01-8	91081-99-1
101940-12-9	161074-58-4	479029-28-2	68081-83-4
68140-18-1	68140-19-2	68140-20-5	68140-21-6
68187-24-6	68187-25-7	68187-42-8	68187-47-3
68188-12-5 (Zonyl TELB)	68333-92-6	68391-08-2	68391-09-3
68412-68-0	68412-69-1	68608-13-9	68954-01-8
70983-60-7	72968-38-8	74499-44-8	84238-62-0
85631-40-9	85631-54-5	85681-64-7	85995-90-0
85995-91-1	86508-42-1	90481-10-0	90622-43-8
90622-71-2	90622-99-4	91032-01-8	91081-09-3
91081-99-1	91648-32-7	91770-74-0	91770-94-4
92129-34-5	92332-25-7	92332-26-8	93062-53-4
93572-72-6	94095-37-1	94166-88-8	95370-51-7
97660-44-1	98219-29-5	98561-40-1	

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Appendix C – List of contacted companies/institutions

Company name	Country	Application/use
DuPont	USA	Producer
DuPont	France	Producer
3M	USA	Producer
3M	Belgium	Producer
Plastics Europe	Belgium	Trade association
Fluorocouncil	USA	
KEMI (Swedish Chemicals Agency)	Sweden	Authority
NYCO	Norway, France	Aviation hydraulic fluids
Solberg		Fire fighting foams
Dr. Sthamer	Switzerland	Fire fighting foams
Kiesow Dr. Brinkmann	Germany	Metal plating
Atotech	Sweden	Metal plating
McDermid	Sweden	Metal plating
Wolfgang Podestà	Germany	Trade association; Metal plating
Statoil	Norway	Chemically driven oil production
Clariant	Germany	Impregnation
Daikin		Impregnation
Rudolf Chemie	Germany	Impregnation
Huntsman	Germany; Sweden	Impregnation
Everest		Impregnation
Dickson	Sweden	Impregnation
Helly Hansen	Norway	Impregnation
Emballageindustrien (Dansk Industri)	Denmark	Trade association; Food contact materials
Valsemøllen	Denmark	Food contact materials
Sonax	Germany	Impregnation
Melvo	Germany	Impregnation
NixWax	UK	Impregnation
Dansk Industri	Denmark	Trade association; Coatings
STAMI	Norway	Authority
Omnova Solutions	USA	Cleaning products
Dansk Mode & Textil	Denmark	Trade associations; Textiles; Impregnation
Tukes (Finnish Safety and Chemicals Agency)	Finland	Authority
Umhverfisstofnun (The Environment Agency of Iceland)	Iceland	Authority
Plastindustrien i Danmark	Denmark	Trade asscociation; Plastics

Company name	Country	Application/use
The Icelandic Industry Association	Iceland	Trade association – industry
ITEK, Dansk Industri	Denmark	Trade association; Electronics
Finish Printing Ink Association	Finland	Trade association; Coatings
Tikkurila	Finland	Coatings
The Federation of Finnish Textiles and Clothing Industries	Finland	Trade association; Textiles
Finnish Forest Industries	Finland	Trade association; Packaging/paper
Finnish Plastics Industries Federation	Finland	Trade association; Plastics
The Federation of Finnish Technology Industries	Finland	Trade association; Electronics
Selected trade associations in Norway		Trade associations
Selected trade associations in Sweden		Trade associations
Mondi	Finland	Paper and packaging
Solvay Specialty Polymers	Italien	Producer
AGC Chemicals Europe	Netherlands	Producer
Mitsubishi International GmBH	Germany	Distributor
HOPI POPI	Czech Republic	Food paper (popcornbags)

Questionaire sent for the contacted companies

Questions – Nordic study on use and emission of per- and polyfluorinated substances

In short we would like information about the use of certain groups of per- and polyfluorinated substances (see item no. 1) used in the Nordic countries. Any information is welcome, however, the more detailed the better.

1. *Mapping of polyfluorinated chemicals in the Nordic countries*
   Please provide information about the use of these groups of fluorinated compounds within your industry. Are the substances used – yes or no?
   - PFCA (Perfluoro carboxylates)
   - PFSA (Perfluoroalkyl sulfonates)
   - PFAL (Perfluoro aldehydes)
   - FTOH (fluorotelomer alcohols)
   - FTS (fluorotelomer sulfonates)
   - Other fluorinated telomers
   - PAP/di-PAP (polyfluoroalkyl phosphate esters)
2. *Mapping of polyfluorinated chemicals in the Nordic countries – detailed information*
   Please provide more detailed information about the use of the above mentioned groups of fluorinated compounds:
   - information about quantities
   - information about application/uses

- information about producers
- information about downstream users and traders in the Nordic countries for the uses within your industry
- information about trade names for products containing any of these substances

For those uses that are not relevant for the Nordic market, please, make a remark for these uses.

3. *Identity and properties*
   Please provide the following information for the above PFCs in question that are relevant for the Nordic market:
   - Chemical name
   - CAS number
   - Trade name
   - Concentration
   - The corresponding uses
4. *Efficacy and availability*
   Please provide, if available, for these PFCs any information on:
   - Performance
   - Benefits
   - Costs and limitations
   - Availability on the Nordic market

Thank you very much for your help!

Appendix D – Commercial PFC products and brands on the market

Table 1. A selection of fire fighting foam products on the market[29]

PFC product	References
DuPont Capstone® Fluorosurfactants	http://www2.dupont.com/Capstone/en_US/assets/downloads/capstone_1157.pdf
Chemguard Fluorosurfactants	http://www.chemguard.com/fire-suppression/catalog/foam-concentrates/aqueous-film-forming-foam-afff/
Dynax Fluorosurfactants	http://www.dynaxcorp.com/resources/pdf/2009/dx5022.bul-rev0909.pdf
Solberg high hydrocarbon foaming agent concentration (Fluorine Free)	http://www.solbergfoam.com/Foam-Concentrates/RE-HEALING™-Foam.aspx

Table 2. A selection of metal plating mist suppressant products on the market[30]

PFC product	References
3M Mist Suppressants	http://solutions.3m.com/wps/portal/3M/en_US/Energy-Advanced/Materials/Products/Acid_Mist_Suppressants/
Atotech Mist Suppressants	http://www.atotech.com/products/general-metal-finishing/functional-chrome-plating/fumetrolr-21-lf.html
Enthone Mist Suppressants	http://enthone.com/en/Industries/Industrial_Finishes/Technology_Selector/Products/ENTHONE_PFOS-Free_Solutions.aspx
McDermid Mist Suppressants	http://industrial.macdermid.com/cms/engineering/hardchrome/index.shtml
Hunter Chemical Fume Suppressants	http://www.hunterchem.com/metal-finishing-Cr.html
Kiesow Dr. Brinkmann	http://www.kiesow.org/aktuelles/aktuelles/article/proquel-of-mit-grossem-erfolg/

[29] Personal information from the Fluorocouncil.
[30] Personal information from the Fluorocouncil and from producers/suppliers of mist suppressants

Table 3. A selection of dirt- and water repellent (DWR) products on the market[31]

PFC product	References
Everest Water Repellant Finishes	http://www.everest.com.tw/_english/00_site/01_edit.aspx?MID=87&SID=118&TID=125
Maflon Leather Fluorosurfactants	http://www.maflon.com/index.php/fluoropolymers.html
Perfluorobutane sulfonamido-based products Scotchgard®	http://www.scotchgard.com/wps/portal/3M/en_US/NAScotchgard/Global/
Fluorinated oxetane-based products Polyfox®	http://www.omnova.com/products/chemicals/PolyFox.aspx
Wacker Silicone-based Dirt and Water Repellants	http://www.wacker.com/cms/media/publications/downloads/6304_EN.pdf
AsahiGuard	http://www.asahiguard.jp/eng/
Short-chain fluorotelomer-based products – Finishing Agents	
Nuva® finishing agents	http://www.textiles.clariant.com/C12571C400483A78/vwWebPagesByID/ABCE5BDE71BE7555C12572AC0049E92D
Unidyne® finishing agents	http://www.daikin-america.com/products/ProductGrades/default.aspx?ApplicationID=&IndustryID=&MyDaikin=&productgradeid=73
Rudolf Finishing Agents	http://www.rudolf.de/products/details-brochure.htm?year=2010&ri=201005
Oleophobol® Finishing Agents	http://www2.dupont.com/Capstone/en_US/assets/downloads/Capstone_Oleophobol_Detail_Chart_ProductsForTextiles_K-25183_CapstoneforTeflon_FINAL_22february2011.pdf

Table 4. A selection of paper and packaging impregnation products on the market[32]

PFC product	References
DuPont Capstone® Fluorosurfactants	http://www2.dupont.com/Capstone/en_US/uses_apps/paper_packaging/paper_packaging.html
Solvay PFPE specialty polymers	http://www.solvayplastics.com/sites/solvayplastics/EN/specialty_polymers/Fluorinated_Fluids/Pages/Solvera_PFPE.aspx
AsahiGuard	http://www.asahiguard.jp/eng/

[31] Personal information from the Fluorocouncil.
[32] Personal information from the Fluorocouncil.

Table 5. A selection of coating agents on the market[33]

PFC product	References
DuPont Capstone® Fluorosurfactants	http://www2.dupont.com/Capstone/en_US/uses_apps/Fluorosurfactants/paints_coatings.html
Chemguard Fluorosurfactants	http://www.chemguard.com/specialty-chemicals/lodyne-connections.htm
Dynax Fluorosurfactants	http://www.dynaxcorp.com/technology/coating.html
3M Fluorosurfactants	http://solutions.3m.com/wps/portal/3M/en_US/Energy-Advanced/Materials/Industry_Solutions/Paints-Coatings/Novec/
Maflon Fluorosurfactants	http://www.maflon.com/index.php/fluorosurfactant-products-and-applications.html
3M Aqueous Fluorinated Polyurethane	http://multimedia.3m.com/mws/mediawebserver?mwsId=66666UF6EVsSyXTtMXf6LXfXEVtQEVs6EVs6E666666--&fn=prodinfo_src220.pdf
Byk Chemie Additives	http://www.byk.com/en/press-events/new-additives.html
Tego Siloxane based Surfactants	http://www.tego.de/sites/dc/Downloadcenter/Evonik/Product/Tego/en/Technical-Papers-Additives/article-multifuctional-siloxane-based-gemini-surfactatng-tego-twin-4000-e.pdf
Air Products Hydrocarbon Surfactants	http://www.airproducts.com/~/media/Files/PDF/industries/paints-coatings-surfynol-surfactants-multifunctional-problem-solvers-waterborne.ashx
Fluorinated oxetane-based products Polyfox®	http://www.omnova.com/products/chemicals/PolyFox.aspx
Diederich Siloxane additives	http://www.diedrichtechnologies.com/Water-Repellents-3.php

[33] Personal information from the Fluorocouncil

Appendix E – Data contributions to “Mapping of uses and applications of PFCs on the Nordic market”

Table 1. FTOHs, FTSs, PFSAs and PFOSA in comsumer products (excluding PFCAs)

Reference	Data	Country	Product type	Usage	FTOHs				FTS			Sulfonates				
Dinglasan-Panlilio MJA	A.D.	n.i.	Polyfox-L-diol	C.P.	4:2	6:2	8:2	10:2			X					
Dinglasan-Panlilio MJA	A.D.	n.i.	Teflon Advance	C.P.		6:2	8:2	10:2			X					
Dinglasan-Panlilio MJA, DuPONT 3	A.D.	DuPont	Zonyl FSO 100	C.P.		6:2	8:2	10:2								
Dinglasan-Panlilio MJA	A.D.	DuPont	Zonyl FSE	C.P.		6:2	8:2	10:2			X					
Dinglasan-Panlilio MJA	A.D.	Canada	Motomaster windshield washer	C.P.	4:2	6:2	8:2	10:2			X					
Dinglasan-Panlilio MJA	A.D.	n.i.	8:2 Methacrylate	C.P.		6:2	8:2									
Dinglasan-Panlilio MJA	A.D.	n.i.	Scotchgard	C.P.							X					
Sinclair E	A.D.	n.i.	Teflon Frying pans	C.P.		6:2	8:2									
Sinclair E	A.D.	n.i.	Microwave popcorn	C.P.		6:2	8:2									
Sinclair E	A.D.	n.i.	Microwave popcorn packing paper	C.P.		6:2	8:2									
Herzke D	A.D.	Norway	Paint	C.P.									PFHxS	PFHpS		
Herzke D	A.D.	Norway	AFFF*	I.U.		6:2	8:2	10:2	6:2	8:2		PFBS	PFHxS	PFHpS	PFDcS	PFOSA
Herzke D	A.D.	Norway	Waterproofing agents	C.P.		6:2	8:2	10:2				PFBS				
Herzke D	A.D.	Norway	PCB	I.U.									PFHxS			
Herzke D	A.D.	Norway	Coated fabrics	C.P.		6:2	8:2	10:2	6:2			PFBS	PFHxS			
Herzke D	A.D.	Norway	Non-stick ware	C.P.		6:2		10:2				PFBS	PFHxS			
Berger	A.D.	n.i.	Textile	C.P.												
Berger	A.D.	n.i.	textile	C.P.												

C.P.: Consumer products; I.U.: Industrial use; X: NMeFOSE; * Still usage of some products.

Table 2. PFCAs in consumer products

References	Data	Country	Product type	PFCAs										
Dinglasan-Panlilio MJA	A.D.	n.i.	Polyfox-L-diol											
Dinglasan-Panlilio MJA	A.D.	n.i.	Teflon Advance											
Dinglasan-Panlilio MJA, DuPONT 3	A.D.	DuPont	Zonyl FSO 100											
Dinglasan-Panlilio MJA	A.D.	DuPont	Zonyl FSE											
Dinglasan-Panlilio MJA	A.D.	Canada	Motomaster windshield washer											
Dinglasan-Panlilio MJA	A.D.	n.i.	8:2 Meth-acrylate											
Dinglasan-Panlilio MJA	A.D.	n.i.	Scotchgard											
Sinclair E	A.D.	n.i.	Teflon Frying pans											
Sinclair E	A.D.	n.i.	Microwave popcorn											
Sinclair E	A.D.	n.i.	Microwave popcorn packing paper		PFPA		PFHpA	PFNA	PFDcA	PFUnDA	PFDoDA			
Herzke D	A.D.	Norway	Paint	PFBA										
Herzke D	A.D.	Norway	AFFF*	PFBA	PFPA	PFHxA			PFDcA		PFDoA			
Herzke D	A.D.	Norway	Waterproofing agents	PFBA		PFHxA	PFHpA	PFNA	PFDcA	PFUnDA	PFDoA		PFTeA	
Herzke D	A.D.	Norway	PCB	PFBA										
Herzke D	A.D.	Norway	Coated fabrics		PFPA	PFHxA	PFHpA	PFNA	PFDcA		PFDoA			
Herzke D	A.D.	Norway	Non-stick ware	PFBA										
Berger	A.D.	n.i.	Textile			PFHxA	PFHpA	PFNA	PFDcA	PFUnDA	PFDoA	PFTriA	PFTeA	PFPDA
Berger	A.D.	n.i.	Textile	PFBA		PFHxA	PFHpA	PFNA	PFDcA	PFUnDA	PFDoA		PFTeA	

Table 3. PFCs in consumer productions, (other than stated in table 1 and 2)

Reference		Prod. Country	Product type	Usage	type	Usage still?					
Vorob'ev SI	A.D.	Japan	Fluosol-DA 20%	Blood substitutes	perfluorocarbon emulsions	No	PFD	PFTPA			
Vorob'ev SI	A.D.	Japan	Fluosol-DA 35%	Blood substitutes	perfluorocarbon emulsions	No	PFD	PFTPA			
Vorob'ev SI	A.D.	Russia	Perftoran	Blood substitutes	perfluorocarbon emulsions	No	PFD		PFMCP		
Vorob'ev SI	A.D.	Russia	Ftorosan	Blood substitutes	perfluorocarbon emulsions	Yes	PFD		PFMCP		
Vorob'ev SI; Castro IC	A.D.	USA	Oxygent	Blood substitutes	perfluorocarbon emulsions	Yes				PFOB	PFDB
3M MSDS	P.I.	n.i.	Scotchgard Carped & rug protector (1023-17N)	surfactant	n.i.	n.i.	Fluorochemical Urethane (trade secret)				
3M	P.I.	n.i.	Dyneon	Industrial processing	Polytetrafluoroethylene	n.i.					
3M	P.I.	n.i.	Fluorinet	Electonic liquid	Perfluoro compounds (>C_{15})	n.i.					
3M	P.I.	n.i.	Novec 1230	Fire extinguishing agent	CF3CF2C(=O)CF(CF3)2	n.i.					
3M	P.I.	n.i.	Novec™ Fluorosurfac-tants FC-4432	Paints and coatings	PFBS	n.i.					
Daikin	P.I.	Japan, EU, USA	Neoflon-PCTFE Flu-oropolymer	n.i.	Poly(chlorotrifluoroethylene)						
Daikin	P.I.	Japan, EU, USA	Neoflon-PFA Fluoro-telomer	n.i.	Perfluorovinylpropyl ether-tetrafluoroethylene						
Daikin	P.I.	Japan, EU, USA	OPTOOL	Prevent finger-print marking	Perfluorohexane						
Daikin	P.I.	Japan, EU, USA	UNIDYNE	Fluoro coating	n.i.						
Daikin	P.I.	Japan, EU, USA	DAIFREE	Fluoro coating	n.i.						
Daikin	P.I.	Japan, EU, USA	Polyflon PTFE-Fluoropolymer	Teflon	polytetrafluoroethylene						
Daikin	P.I.	Japan, EU, USA	Dai-El	Elastomer	vinylidenefluoride/ hexafluoropropylene copolymer						

Reference		Prod. Country	Product type	Usage	type	Usage still?			
Asahi Glass Company	P.I.		Fluon PTFR_E	Polymers	Polytetrafluoroethylene				
Dow Corning	P.I.		Molykote	Grease	PFPE, PTFE				
DuPont 1,2	P.I.	USA	Teflon Advanced	Carpet and upholstery protection	n.i.	n.i.	Fluorochemi-cal dispersion in water	Partially fluorinated aliphatic polyurethane	
DuPont	P.I.	USA	Viton	Elastomer	Hexafluoropropene polymer				
DuPont	P.I.	USA	Kalrez	Fire fighting	Perfluoroalkylpolyether, polytetrafluoroethylene				
DuPont	P.I.	USA	Krytox	Lubricant	Polyhexafluoropropylene oxide, PFPE, Perfluoroalky-lether, PTFE				
Moe M; Zaggia A	A.D.		ForaFac (DuPont)	Fire extinguishing surfactant	(CF2)nC9H19 fluorotelomer				
Key BD	P.I.	n.i.	monofluoroacetic acid	pesticide	CH2FCO2H	n.i.			
Key BD	P.I.	n.i.	Trifluoroacetic acid	Reagent	CF2CO2H	n.i.			
Key BD	P.I.	n.i.	trifluoromethanesulfo-nic acid	catalyst/reagent	CF3SO3H	n.i.			
Key BD	P.I.	n.i.	1H,1H,2H,2Hperfluoro octanesulfonic acid	surfactant	C6F13CH2CH2SO3H	n.i.			
Key BD	P.I.	n.i.	N-acetic-N-ethyl perfluorooctane sulfonamide	surfactant	C8F17SO2N(CH2COOH) (CH2CH3)	n.i.			
Key BD	P.I.	n.i.	sulfuramid	insecticide	C8F17SO2NH(CH2CH3)	n.i.			
Key BD	P.I.	n.i.	polytetrafluoroethylene	teflon	(-(CF2CF2)n-)	n.i.			
Key BD	P.I.	n.i.	Perfluoropolyether	lubricant	(-(CF(CF3)CF2O)n-)	n.i.			
Key BD	P.I.	n.i.	Zonyl alcohol	surfactant	C8F17CH2CH2OH	n.i.			8:2-FTOH
Yang Z	P.I.	n.i.	Oxycyte	Blood substitutes	perfluorocarbon emulsions	n.i.			
Castro IC	P.I.	n.i.	Fluosol	Blood substitutes	perfluorocarbon emulsions	n.i.	PFD	PFTPA	
Castro IC	P.I.	n.i.	Oxypherol	Blood substitutes	perfluorocarbon emulsions	n.i.		PFTPA	

Reference		Prod. Country	Product type	Usage	type	Usage still?				
Castro IC	P.I.	n.i.	Perftoran	Blood substitutes	perfluorocarbon emulsions	n.i.	PFD			
Castro IC	P.I.	n.i.	Oxyfluor	Blood substitutes	perfluorocarbon emulsions	n.i.			PFDCO	
Castro IC	P.I.	n.i.	Oxycyte	Blood substitutes	perfluorocarbon emulsions	n.i.				TBPCH
Castro IC	P.I.	Columbia	Columbian emulsion	Blood substitutes	perfluorocarbon emulsions	n.i.		PFOB		
Castro IC	P.I.	France	French emulsion	Blood substitutes	perfluorocarbon emulsions	n.i.		PFOB		
Gelest	P.I.	n.i.	SiBRID FCS 331	Skin care product ingredient (cosmetics)	n.i.	n.i.	tetrafluoro-ethylene	fluorinated dimethyl fluid		
Solvay plastics	P.I.	n.i.	Fomblin PFPE	Lubricants/ oils/grease/ surfactant	PFPE					
Solvay plastics	P.I.	n.i.	Fomblin HC PFPE	Personal care products	PFPE					
Solvay plastics	P.I.	n.i.	Hyflon	Wire & cable coatings	PFA/MFA					
Solvay plastics	P.I.	n.i.	Tecnoflon	Per-fluoroelastomer	Fluoropolyether derivative					
Solvay plastics	P.I.	n.i.	Algoflon	Wire & cable coatings	PTFE					
Solvay plastics	P.I.	n.i.	Fluorolink	Miscellaneous	PFPE					
Solvay plastics	P.I.	n.i.	Galden	Electronics	PFPE					
Solvay plastics	P.I.	n.i.	Solvera	Paper packaging	n.i.					
Hoechst AG	P.I.	Germany	Hostaflon	Teflon						
Hoechst AG	P.I.	Germany	Hostinert	In electronic components	Perfluorinated liquid	No				
Penwalt	P.I.	n.i.	Pentel	Water and soil repellency to fabrics	Fluorotelomer composition					
Miteni	P.I.	Italy	Perflutel RM82	Industry and science	Perfluorohexane					

Reference		Prod. Country	Product type	Usage	type	Usage still?
Miteni	P.I.	Italy	Perflutel RM57	Industry and science	Perfluoroheptane	
NearChimica	P.I.	Italy	Naiguard	Repellant finishes	PTFE	
Zaggia A	P.I.	n.i.	PolyFox	Fire-extinguishing surfactant	Hydroxyl terminated Fluoropolyether co-polymer	
OMNOVA	P.I.	n.i.	X-Cape	Repellants	Fluoropolymer	

A.D.: Analytical data, P.I.: Producer information.

Abbreviations used in Table 3

PFD	Perfluorodecalin
PFTPA	Perfluorotributylamine
PFMCP	perfluoromethylcyclohexylpiperidine
PFOB	Perfluorooctylbromide
PFDB	Perfluorodecylbromide
PFDCO	Perfluorodichlorooctane
TBPCH	Tetrabutylperfluorocyclohexane

Table 4. Geometric mean concentration in home and work related environments indicating human exposure

	Reference	Jogsten IE	Fracer AJ	Shoeib M (2005)	Shoeib M (2004)	Barber JL	Barber JL	Jahnke A	Jahnke A	Shoeib M (2007)
	Country	Spain	USA	Canada	Canada	Norway	Norway	Norway	Norway	Canada
	Matrix	Indoor dust	Office air	Indoor air/dust	Indoor air/dust	Indoor air/dust	Indoor air/dust	Indoor air/dust	Indoor air/dust	Indoor air/dust
	Year	2012	2012	2005	2004	2007	2007	2007	2007	2007
	Unit	ng/g	pg/m3	pg/m3	pg/m3	pg/m3	pg/m3	pg/m3	pg/m3	pg/m3
Telomers	**4:2 FTOH**					114	<20			
	6:2-FTOH	0.295	1,320			2,990	<40	177	248	
	8:2-FTOH	4.04	9,920			3,424	<10	853	421	2,070
	10:2-FTOH	0.76	2,850			3,559	13	898	1,660	891
	EtFOSA	<0.062	17	59		6,626	7	188	158	
	MeFOSA	<0.054	29,1			6,608	6			
	EtFOSE	3.238	18,1	1,100	770	5,755	76	305	815	
	MeFOSE	1.19	289	1,970	2,590	6,018	763	727	798	
				35	73					
	PFBS	1.028								
	PFHxS	1.073								
	PFDS	<0.002								
	PFBA	16.35								
	PFPeA	0.367								
	PFHxA	1.382								
	PFHpA	1.63								
	PFNA	6.769								
	PFDA	9.735								
	PFUnDA	3.373								
	PFDoDA	3.363								
	PFTrDA	10.9086								
	PFTDA	0.67								
	PFOcDA	<0.46								
	5:3 FTSA	<2								
	6:2 FTUCA	0.0107								
	8:2 FTUCA	0.0265								
	10:2 FTUCA	<0.006								

	Reference	Goosey E (2012)	Goosey E (2012)	Goosey E (2011)	Goosey E (2011)	Goosey E (2011)	Goosey E (2011)	Goosey E (2011)	Goosey E (2011)	Goosey E (2011)	Goosey E (2011)	Goosey E (2011)	Kim SK (2012)	Liu W (2012)
	Country	UK	UK	UK	UK	Australia	Canada	France	Germany	Kazahkstan	Thailand	USA	Korea	Japan
	Matrix	Indoor air - homes	Indoor air - offices	Indoor air /dust -homes	Indoor air /dust -offices	Indoor air /dust - homes	Indoor air /dust - homes	Indoor air /dust - homes	Indoor air /dust - homes	Indoor air /dust - homes	Indoor air /dust - homes	Indoor air /dust – homes	Indoor air	Indoor air – homes
	Year	2008–2009	2008–2009	2007–2009	2007–2009	2007–2009	2007–2009	2007–2009	2007–2009	2007–2009	2007–2009	2007–2009	2009	2008
	Unit	pg/m3	pg/m3	ng/g	ng/g	ng/g	ng/g	ng/g	ng/g	ng/g	ng/g	ng/g	pg/m3	ng/m3
Telomers	4:2 FTOH													
	6:2-FTOH													0.59
	8:2-FTOH												4,839	10.16
	10:2-FTOH												2,610	2.29
	8:2-FTOHAc													0.34
	8:2-FTOHMac													0.05
	EtFOSA	120	59	98	120	2,000	1,300	150	190	150	140	140	5.3	
	MeFOSA	<2.5	6	13	61	360	32	5.4	1.7	<0.1	1.6	15		
	EtFOSE	600	490	320	290	60	8.4	190	100	5.7	59	210	27	
	MeFOSE	950	480	230	250	84	8.4	190	84	12	14	120	8.3	
	MeFOSEA												6.9	
	FOSA	152	74	54	21	25	190	3.4	56	<0.02	13	66		
	PFBS													
	PFHxS	36	94	450	620	240	150	130	290	94	25	270		

Appendix F – Data contributions of PFCA and PFSA in food and drinking water

Table 1. PFCA in food and drinking water

Concentration, mean (range) of perfluorocarboylates, PFCA, in food (ng/kg or ug/kg) and drinking water (ng/L)															
Reference	Foodstuff	Country	Number of samples	Year	PFBA	PFPA	PFHxA	PFHpA	PFOA	PFNA	PFDA	PFUnDA	PFDoDA	PFTrDA	PFTeDA
								Data in ng/kg							
Haug et al., 2010 a	Lettuce	Norway	3	2010			0.98	0.43	1.8	< 1.0	0.78	< 1.3	1.3		
(NB: Data in ng/kg)	Carrot		3				< 1.3	< 0.89	2.0	< 2.1	< 1.4	< 2.5	< 2.4		
	Potato		3				3.1	1.1	5.3	< 4.1	3.0	2.2	< 4.8		
	Cheese		3				< 7.7	7.4	13	16	6.6	4.1	< 15		
	Margarine		3				2.5	< 5.6	12	< 13	< 8.6	< 16	< 16		
	Milk		3				1.5	< 0.87	4.7	< 2.1	4.0	< 2.5	< 2.4		
	Bread		3				14	11	51	9.5	17	< 15	< 15		
	Strawberry jam		3				< 7	< 4.7	14	3.7	8.7	< 13	< 13		
	Pork meat		3				< 4.3	2.8	15	5.5	16	< 8.2	< 8.0		
	Beef		3				< 3.3	7.6	12	15	23	< 6.4	< 6.2		
	Chicken meat		3				< 13	20	52	6.8	< 23	13	< 9.2		
	Egg		3				13	< 16	30	< 7.4	12	9.9	< 8.1		
	Fish sticks		3				< 18	21	49	< 11	17	18	< 13		
	Canned mackerel		3				< 18	< 24	24	< 11	< 31	19	2		
	Salmon		3				11	16	46	10	26	4.5	< 12		
	Cod		1				< 11	< 15	30	5.9	13	21	< 7.5		
	Cod liver		1				< 48	< 66	51	14	39	230	< 33		
								Data in ug/kg							
Clarke et al., 2009	All oily fish	UK	47	2009			< 1 - 7	< 1	1.1	< 1	< 1 -2	< 1 - 2	< 1 - 2		
	All whitefish		12				< 1	< 1	< 1	< 1	< 1	< 1	< 1		
	All shellfish		12				< 1	< 1- 1	3.3 (1-8)	<1 - 3	< 1	< 1	< 1 - 1		
	Liver, diffent animals		25					< 1	1.1 (1-3)	< 1	< 1	< 1			
	Kidney different animals		12					< 1	< 1	< 1	< 1	< 1			
	Vegetables		42					< 1	< 1	< 1	< 1	< 1			

Table 2. PFCA in food and drinking water

Concentration, mean (range) of perfluorocarboylates, PFCA, in food (ng/kg or ug/kg) and drinking water (ng/L)															
								Data in ug/kg							
Kit Granby, DTU, 2012	Herring	Denmark	6	2012					< 0.5						
	White fish		3						< 0.5						
								Data in ng/L							
Skutlarek et al., 2006	Drinking water	Germany		2006	11	5	22	23	519						
Haug et al., 2010 a	Drinking water 1	Norway		2010			0.78	0.76	2.5	< 0.22	1.0	0.35			
	Drinking water 2	Norway		2010			0.31	0.32	1.2	< 0.22	0.52	0.20			
	Drinking water 3	Norway		2010			< 0.11	< 0.12	0.65	< 0.22	0.22	0.065			

Table 3. PFSA in food and drinking water

Concentration, mean (range), of perfluoroalkyl sulfonates, PFSA, and perfluoroalkyl sulfonic Acid Amides in food (ng/kg or ug/kg) and drinking water (ng/L)												
Reference	Foodstuff	Country	Number of samples	Year	PFBS	PFHxS	PFHpS	PFOS	PFDS	PFOSA	∑ PFC's
						Data in ng/kg					
Haug et al., 2010	Lettuce	Norway	3		< 0.12	< 0.06		0.17			
	Carrot		3		< 0.25	< 0.1		0.67			
	Potato		3		< 0.48	< 0.22		1.0			
	Cheese		3		< 1.5	< 0.6		12			
	Margarine		3		< 1.6	1.3		2.			
	Milk		3		< 0.24	< 0.1		7.0			
	Bread		3		< 1.5	1.7		17			
	Strawberry jam		3		< 1.3	< 0.59		3.0			
	Pork meat		3		< 0.81	1.2		17			
	Beef		3		< 0.63	< 0.28		60			
	Chicken meat		3		3.2	2.3		21			
	Egg		3		2.0	3.5		39			
	Fish sticks		3		5.0	1.6		13			
	Canned mackerel		3		5.5	< 3		43			
	Salmon		3		2.2	5.5		55			
	Cod		1		< 3.4	2.8		100			
	Cod liver		1		< 15	< 8.2		310			
						Data in ug/kg					
Clarke et al., 2009	All oily fish	Germany	47		< 1	< 1- 1		4.8 (<1-59)		2.5 (1 - 27)	6.7 (0-63)*
	All whitefish		12		< 1	< 1		1.2 (<1-2)		1.1 (1-2)	0.8 (0-4)*
	All shellfish		12			< 1 -2		4.4 (1-13)		1.3 (1-3)	8.2 (0-20)*
	Liver, diffent animals				< 1	< 1		2.5 (1-10)		< 1	2.4 (0-14)*
	Kidney different animals				< 1	< 1		1.4 (1-3)		< 1	12 (0-5)*
	Vegetables				< 1 - 1	< 1		< 1		< 1	0.1 (0-2)*

Note ∑PFC's: * = ∑PFHxA, PFHpA, PFOA, PFNA, PFDeA, PFUnA, PFDoA, PFBS, PFHxS, PFOS and PFOSA

Concentration, mean (range), of perfluoroalkyl sulfonates, PFSA, and perfluoroalkyl sulfonic Acid Amides in food (ng/kg or ug/kg) and drinking water (ng/L)												
							Data in ug/kg					
Kit Granby, DTU, 2012	Herring	Denmark	6						1.6			
	White fish		3						2.1 (1.3 - 3.3)			
							Data in ng/L					
Haug et al., 2010 a	Drinking water 1	Norway		2010	< 0.045		0.15		0.23			
	Drinking water 2	Norway		2010	< 0.045		0.12		0.31			
	Drinking water 3	Norway		2010	< 0.045		0.045		0.071			

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Per- and polyfluorinated substances in the Nordic Countries

This Tema Nord report presents a study based on open information and custom market research to review the most common perfluorinated substances (PFC) with less focus on PFOS and PFOA.

The study includes three major parts:

1. Identification of relevant per-and polyfluorinated substances and their use in various industrial sectors in the Nordic market by interviews with major players and database information
2. Emissions to and occurence in the Nordic environment of the substances described in 1)
3. A summary of knowledge of the toxic effects on humans and the environment of substances prioritized in 2)

There is a lack of physical chemical data, analystical reference substances, human and environmental occurrence and toxicology data, as well as market information regarding PFCs other than PFOA and PFOS and the current legislation cannot enforce disclosure of specific PFC substance information.

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