From PFASs working group

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27th Meeting of Competent Authorities for REACH and CLP Competent Authorities' Session

27 June 2018

Concerns: **Short-chain poly- and perfluorinated substances (SC-PFASs): new concerns and way forward**

Agenda point: ***Point 7***

Action requested: **Member State Competent authorities are invited to:**

(1) Reflect on the priority of working on SC-PFASs and ensuring the resources needed for the further work, in particular, on subgroups of SC-PFASs not yet addressed. MSCAs so far not involved in the PFASs work are invited to contribute.

(2) Reflect on potential additional or other approaches to risk management.

(3) Reflect whether minimisation of emissions and releases would be warranted when a substance group or substance has properties described in the document (given these are duly justified in a detailed assessment document).

(4) Written comments to this document and indications of interest to work on short-chain PFASs should be sent to [redacted]@echa.europa.eu by August 30.

For discussion and information

Disclaimer: this document does not present formal views of the Competent Authorities of the Member States, the European Commission or ECHA.

Contents

1. Introduction

Per- and polyfluorinated substances (PFASs) make up a large family of more than 4700 man-made chemical substances that are widely used in society (OECD 2018[1]). Due to their unique properties, including repellency to water, grease and dirt as well as high temperature resistance and film-formation these substances are used in applications such as impregnation of textiles, coatings for paper and packaging, fire-fighting foams and production of fluoropolymers. A significant increase of proposed uses into new technical areas is predicted (Swedish Chemicals Agency, 2015). For a rough overview of various PFASs groups as described by OECD (2017) and Wang et al (2017), please, see Appendix 1.

All PFASs have several concerns in common. These concerns are shared widely and need urgent action in order to avoid a situation where the damage caused would be irreversible. In e.g. the Madrid Statement, signed by approximately 200 leading scientists and professionals, PFASs as a group are pointed out as substances of concern that are detected globally, even in remote locations (Blum et al. 2015).

"Short-chain PFASs", a subgroup of the PFASs family (see description at the end of this section) have also been highlighted as substances of concern at several workshops such as the International Workshop for Authorities on the Assessment of Risks of Short-Chain PFASs (German Environment Agency, 2016), the Nordic Workshop on Joint Strategies for PFASs (Borg et al. 2017) and international PFASs workshop of International Panel for Chemical Pollution[2]. They have gained more and more attention by authorities and media around the globe, similar to the attention for long-chain PFASs (LC-PFASs) which gradually are being phased out in the EU and elsewhere.

Authorities under the REACH and CLP-Regulations have been working on assessment and risk management of several PFASs subgroups since 2013 having a wide coverage of individual substances on the market. An overview of these activities is provided in Appendix 2.

This document is one deliverable of the common work of PFASs informal working group. It started in 2014 and has participants from AT, BE, DE, DK, NL, NO, SE, DG-ENV, DG-GROW and ECHA aiming at better coordination and coherent approaches in assessment and regulatory risk management of PFASs.

The Risk Management Expert Meeting (RiME) 01/2018 discussed this document and the potential regulatory management options (RMOs) for these substances in February 2018. The Member State Committee was introduced to the concerns described in this document in its 59th meeting in April 2018 and was asked to comment in written procedure. Comments were received from the UK, FI and from FluoroCouncil/CEFIC. The comments received are related to the hazard assessment and will be taken into account in the preparations of the planned (SVHC/restriction) dossiers. Further discussion on the regulatory risk management options, where necessary, will take place in RiME+.

Purpose and scope

This paper focusses on the short-chain perfluorinated substances (here denoted as SC-PFASs), which are a subgroup of PFASs. The main concerns and their implications for a potential risk reduction approach are discussed.

The aim of this document is to make relevant actors, including members of the MSC

[1] http://www.oecd.org/chemicalsafety/risk-management/global-database-of-per-and-polyfluoroalkyl-substances.xlsx

[2] https://www.ipcp.ch/news/international-workshop-supporting-the-dialogue-between-science-and-policy-on-pfass-at-eth-zurich

and, e.g., representatives from Member State authorities so far not involved into the work aware of the concerns of SC-PFASs even before the first documents for regulatory measures, e.g. SVHC and/or restriction dossiers, are submitted. Furthermore, the purpose of the paper is to ensure (develop) coherent understanding of the types of concerns. However, this paper should not be understood as an in-depth hazard assessment necessary to identify the hazards/concerns in a regulatory risk management process. Moreover, this paper discusses and informs about the planned way forward under REACH for this wide group of substances.

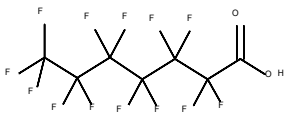
Aside from this paper, regulatory management option analyses (RMOA) are carried out for the specific SC-PFASs subgroups. This paper is also used to ensure and enhance coherence between the RMOAs to the level necessary.

Despite focussing on one (though wide) group of PFASs described below, many elements elaborated may also be directly applicable or otherwise useful for the work on other PFASs.

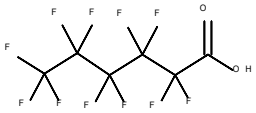
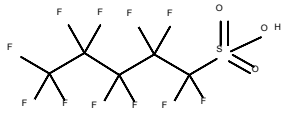
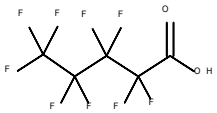
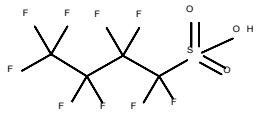
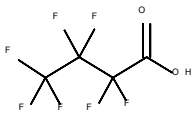
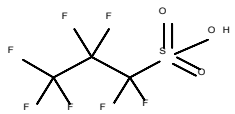
Short-chain PFASs (SC-PFASs) are the primary substitutes of long-chain PFASs (LC-PFASs) (ECHA, 2014; UNEP, 2006) and consist of short chain perfluorocarboxylic acids (SC-PFCAs), short-chain perfluorosulfonic acids (SC-PFSAs), short-chain perfluorophosphonic acids (SC-PFPAs), short-chain perfluorophosphinic acids (SC-PFPiAs) and also potentially of other similar perfluoroalkyl acids. OECD (2013) has provided the definition for distinguishing long-chain and short-chain PFASs, which is also applied in this analysis: ''long-chain'' refers to perfluoroalkyl carboxylic acids with eight or more carbons (i.e., with 7 or more perfluorinated carbons) and perfluoroalkane sulfonates with six or more carbons (i.e., with 6 or more perfluorinated carbons). In addition to the before mentioned alkyl acids, their precursors and salts are covered.

In order to be sufficiently concise, the analysis in section 2 is presented on the best known SC-PFAAs (see Table 1). They are "arrow head" representatives of the various corresponding SC-PFASs groups. The substance selection is concern based. These substances are terminal degradation products of a high number of related substances[3] (precursors) and also responsible for the hazards of SC-PFAAs salts. Precursors and salts, which also are covered by the planned regulatory risk management measures as they degrade into the respective PFAAs, are listed in separately available non-exhaustive lists. Most of the SC-PFAS on the market are precursors or salts of the "arrow head" representatives, based on ECHA's mass screenings in 2014 and 2015.

Table 1. Main examples of short chain PFASs groups covered by the concept paper. For simplicity only linear "arrow head" representatives (SC-PFAAs) of each SC-PFASs group are shown. Branched PFAAs are also included in the concept. Similar additional groups for which this paper is applicable are perfluoroalkyl phosphinic and perfluoroalkyl phosphonic acids.

Perfluoroalkyl acids, PFAAs			
Perfluorocarboxylic acids, PFCAs		Perfluorosulfonic acids, PFSAs	
PFHpA		--	--

[3] Related substances are substances that may break down to short-chain PFASs during the product/substance life-cycle or in the environment under relevant conditions. The short-chain PFASs breakdown substance represents the end transformation product, also called an arrowhead substance, and further chemical transformations are not expected.

PFHxA			PFPeS
PFPeA			PFBS
PFBA			PFPS

Based on the terminology of OECD (2017) and Wang et al. (2017) perfluoroethercarboxylic acids, their salts and precursors as well as perfluoroethersulphonic acids, their salts and precursors belong to the group of alkylchain PFASs similar to the above mentioned substances (see Appendix 1). It should be mentioned that they have largely similar physicochemical properties and hence many of the concerns listed apply. On-going assessments may also provide insight to further concerns on these latter substance groups.

2. Overview of concerns

This section specifies concerns attributed in general to SC-PFAAs. Degradation of precursors into SC-PFAAs is not addressed in this document but complementary work is on-going in addition to the work already done in the frame of the PFOA restriction and LC-PFCA restriction proposals.

Extreme persistence

Based on the high energy of the carbon-fluorine bond (Siegemund et al., 2000), it can be assumed that SC-PFAAs are extremely persistent, similar to the persistence of LC-PFAAs (Parsons et al., 2008). They do not undergo abiotic or biotic degradation at all under environmental conditions and are considered highly stable degradation products to which several precursors ultimately degrade into (D'Agostino and Mabury, 2017; Wang et al., 2013). This extreme persistence is regarded as an incalculable concern itself, as SC-PFAAs will stay in the environment for decades to centuries (Cousins et al., 2016). All compartments and regions will unavoidably be exposed to extremely persistent substances and concentrations will increase unless releases are regulated. Even if releases would now be regulated, the mass already present in the environment will still cause exposure of the environment and humans over a very long time period, even much longer than other substances being "very persistent" according to Annex XIII to REACH. The possible risk of extremely persistent organic fluorochemicals for humans and the environment has also been emphasised by leading scientists (Blum et al., 2015; Scheringer et al., 2014).

High mobility in water and soil

SC-PFAAs have a low adsorption potential (e.g Vierke et al., 2014; Zhang et al., 2013) and they are very mobile in water and soil[4]. The physicochemical properties of SC-

[4] This mobility -concern is also addressed by the UBA proposal for implementing criteria and an assessment procedure to identify Persistent, Mobile and Toxic (PMT) and very Persistent, very Mobile (vPvM) substances registered under REACH (https://www.umweltbundesamt.de/publikationen/protecting-the-sources-of-our-drinking-water-from/). Please note that this proposal addresses all organic chemicals under REACH, not only PFASs. While SC-PFASs might fulfil the proposed PMT/vPvM criteria they are not the ideal

PFAAs[5] (Wang et al., 2011) and their environmental distribution (Ahrens, 2011; Zhao et al., 2016) show that they are more mobile compared to their long-chain homologues. Therefore, SC-PFAAs reach water bodies effectively, which is of special concern regarding human exposure: drinking water resources are highly sensitive to contamination with SC-PFAAs (Boiteux et al., 2017; Schwanz et al., 2016). SC-PFAAs are already today found in tap water (e.g. Gellrich et al. 2013; Llorca et al. 2012; Ullah et al. 2011). SC-PFAAs (PFBS, PFBA) have been detected also recently in river and drinking water in the Netherlands (Gebbink, 2017). Due to the low adsorption potential (e.g., Vierke et al., 2014), SC-PFAAs do not bind to particles and stay dissolved in the water phase.

Very difficult to remove from water (including waste water)

Once emitted, due to the low adsorption potential and due to the extreme persistence SC-PFAAs can hardly be removed from the environment, if at all (Boiteux et al., 2017; Lundgren, 2014; Rahman et al., 2014). Where LC-PFAAs can be removed with activated carbon filters, this is not effective for SC-PFAAs. For example Eschauzier et al. have demonstrated that more hydrophilic short-chain PFAAs (especially PFBA and PFBS) could not be removed by granular activated carbon (GAC) filtration although the longer chain PFASs could and their concentrations remained constant through drinking water treatment (Eschauzier et al., 2012). Thus, with SC-PFAAs contaminated drinking water reservoirs are especially problematic, as no effective measures exist to remove the substances from the water to date (exemplary shown in Appendix 3: case study "Rastatt"). The same concern applies to waste water treatment: releases of SC-PFAAs pass due to their physicochemical properties and extreme persistence to the environment via municipal sewage treatment without effective removal. Since SC-PFAAs are not expected to chemically and biologically degrade, the concentrations in contaminated regions would only decline with further spatial distribution (i.e. dilution), provided that no further emission occurs.

Potential for long-range transport

SC-PFAAs have the potential for long-range transport via air and water (DK-EPA, 2015). Due to the high mobility caused by the low adsorption potential and high aqueous solubility of SC-PFAAs, as well as slow degradation in air, they have an even higher potential for long-range transport compared to the long-chain homologues (Vierke et al., 2014). Already today, monitoring data show that SC-PFAAs are present in remote regions (Routti et al. 2017; Kirchgeorg et al., 2016; Llorca et al., 2012) and have a widespread occurrence in biotic and abiotic compartments. For non-ionic PFASs transportation via air is an important long-range transport mechanism. Short-chain non-ionic PFAS (including precursors) have higher volatility compared to their long-chain counterparts.

Protein binding

Liver and blood and thus protein rich compartments are important sinks for SC-PFAAs. Investigations with homologues series of SC-PFAAs with various proteins elicited significant protein binding potential (Bischel et al., 2011; Chen and Guo, 2009, Kerstner-Wood et al. 2003). Several proteins have been investigated: serum albumin (ATSDR, 2016, Kerstner-Wood et al. 2003), liver fatty acid binding protein (L-FABP)(Luebker et al., 2002; Zhang et al., 2013)), thyroid hormone transport proteins, transthyretin (TTR) (Ren et al., 2016), and organic anion transporters (Weaver et al.,

example case. This is because the concerns caused by SC-PFASs are much broader and include additional intrinsic substance properties of very high concern than addressed by the PMT/vPvM proposal.

[5] Physicochemical properties (according to Wang et al. 2011; including PFCAs C4 – C6, PFSAs C4): logKow ~ 3 - 4, water solubility > 20 g/L, logKoa = 6 – 6.7, pKa < 1, vapor pressure = 457 Pa – 3890 Pa

2010; Yang et al., 2010). The affinity to proteins is chain length dependent and increases up to a certain number of perfluorinated carbons, depending also on the protein (Ng & Hungerbuehler, 2014; Zhang et al., 2013, Liu et al., 2017). The relationship between structure (e.g. chain length, functional group) and affinity to proteins is complex and thus still a matter of research (Ng & Hungerbuehler, 2014). Nevertheless, some toxicokinetic observations may be explainable by binding to certain proteins. The expression of organic anion transporters is gender as well as species specific. They function either as an aid for uptake or elimination and some are responsible for reabsorption from urine to blood. The activity of human reabsorption transporters was far more pronounced than for rats (Weaver et al., 2010; Yang et al., 2010) and may explain the significant differences in the half lives of PFAAs in humans compared to laboratory rodents (see below). High protein binding renders the predictability of conventional approaches based on lipid binding uncertain in (eco)toxicity and bioaccumulation assessment. Especially long-term effects, caused by the high protein binding potential, cannot be sufficiently assessed.

Bioaccumulation potential

SC-PFAAs are to some extent bioaccumulative, as far as this has been possible to assess. Elimination half-lives, which have been recently used for LC-PFAAs as a metric to estimate bioaccumulation potential in air-breathing organisms, are shorter in comparison with long-chain homologues. Depending on the species, half-lives range from a couple of hours to several days in mammals (Chengelis et al., 2009; Gannon et al., 2011; Numata et al., 2014) and up to over a month-almost a year in humans (Nilsson et al., 2010). A high protein binding potential is of toxicokinetic relevance: Binding to transporter proteins with a sex and species dependent expression may explain observed significant differences in the half lives between sexes and species. Due to the binding to serum albumin the blood can easily distribute SC-PFAAs within the body resulting in a potential to enrich particularly in blood rich tissues. SC-PFAAs are effectively distributed to most organs in the body of mammals (Bogdanska et al. 2014, Numata et al. 2014, Perez et al., 2013, Burkemper et al., 2017). Unlike the accumulation in adipose tissue, binding to proteins and accumulation in specific organs has a higher potential to cause adverse effects. Estimation of true bioaccumulation potential of SC-PFAAs is not possible with conventional methods related to lipid adsorption. Currently no systematic approach for assessing bioaccumulation potential in particular for air-breathing species for this kind of substances is available.

Environmental levels in biota and trends in marine mammals

Some SC-PFAA have been detected in large air-breathing vertebrates (polar bears, whales, reindeer and/or seals (Ahrens et al 2016; Routti et al 2017; Gebbink et al., 2016; Lam et al., 2014). Distribution and bioaccumulation of short-chain PFAAs in ecosystems and along food chains of air-breathing animals is anticipated but the magnitude cannot currently be quantified by field measurements, most likely due to the relatively short history of SC-PFASs releases. The data also confirm that SC-PFAAs may be transferred to the fetuses (Gebbink et al., 2016). One study shows a significant increasing trend for some SC-PFAAs in the liver samples of some cetacean species from 2002 to 2014 (Lam et al., 2016), and e.g. the ratio of PFBS to PFOS is significantly increasing over time in dolphin samples.

Concentrations in human tissues

SC-PFAAs have been detected in blood/plasma from several European and non-European populations. In most studies and for most measured SC-PFAAs a quite low detection frequency has been observed whereas in other studies they were not detected (Schröter-Kermani et al., 2013, Ericson et al., 2007, Glynn et al., 2012, Gyllenhammar et al., 2013). In a German study of plasma samples in the period 1982-2010 up to 1.4 µg/l was measured for PFBA (detection rate: 4.3%), 13.7 µg/l for

PFPeA (0.78%), 1.1 µg/l for PFHxA (2.3%) and 2.3 µg/l for PFHpA (20.5%) (Schröter-Kermani et al., 2013). A Swedish study of pooled serum samples from nursing women three weeks after delivery from 1996-2010 showed PFAS levels of <0.013-0.101 ng/g for PFBS and 0.056-0.14 ng/g for PFHpA whereas PFHxA was not detected (Glynn et al., 2012), furthermore the PFBS concentrations increased with 11% per year in this period. The increasing trend of PFBS in the study by Glynn et al 2012 was later confirmed and linked to contamination of the drinking water in the city of Uppsala (Gyllenhammar et al., 2013). Moreover, in some fluorochemical production workers concentrations of PFBS in the range 7-32 µg/l in blood have been observed (Ehresman et al., 2007). These results indicate a potential for elevated blood concentrations in a situation with increasing exposure. Most SC-PFAAs have also been detected in human breast milk samples (PFBA, PFHxA, PFHpA; Antignac et al. 2013) and in urine (PFPeA, PFHxA and PFHpA; Hartmann et al., 2017).

PFASs analyses of autopsy samples detected SC-PFAAs in different human tissues. PFBA concentrations were the highest with 304 ng/g in the lung and 464 ng/g in the kidney, which are ~10 and ~200-fold higher compared to PFOA concentrations, respectively. Mean PFHxA concentrations were 35.6 ng/g in bone, 18.0 ng/g in brain and 50.1 ng/g in lung tissues (Perez et al, 2013). Although this study has been criticised due to a large variation in data it should be noted that studies with other mammals confirm the distribution of SC PFAAs to different organs in the body after dietary exposure (Bogdanska et al. 2014, Numata et al. 2014, Burkemper et al., 2017).

Enrichment into plants

SC-PFAAs are known to enrich in plants and due to their high water solubility and low adsorption potential especially in leaves and fruits (Blaine et al., 2013; Felizeter et al., 2012, 2014). This enrichment in the edible parts of plants is higher compared to LC-PFAAs and might imply an exceptional contamination route and distribution along the food chain. However, the relevance has not yet been fully investigated. However, in contaminated soil, especially concerning arable land, the accumulation in the edible parts of plants is of concern due to human exposure (example shown in Appendix 3: case study "Rastatt").

Effects

There is a data gap on the toxic properties of SC-PFAAs, following the hazard assessments on PFHxA, PFBA and PFBS (see also Lilienthal et al., 2017). It is therefore currently not possible to conclude on effects/toxicity with sufficient certainty for all relevant endpoints. Considering that closely related substances PFOA (CLH amongst others: Repr. 1B, STOT RE1) and PFOS (CLH amongst others: Repr 1B, STOT RE1, Carc 2) are toxic, and that the toxicity dataset for the SC-PFAAs provides some effects, sub-lethal long-term adverse effects cannot be excluded.

The data available on the toxicity of PFBS in animals have identified the liver, kidneys, stomach, and haematological systems as targets of toxicity (Lieder et al. 2009a; NICNAS, 2005; 3M 2001). Furthermore, there are indications that PFBS might have endocrine disrupting properties (Feng et al., 2017; Lou et al., 2013; Lieder et al. 2009b).

In general the half-lives of PFAAs in mammals decrease with decreasing chain length (DK-EPA, 2015). Significant differences in the half lives of PFAAS in humans compared to laboratory animals are well documented for long chain PFAAs. Half-lives for certain SC-PFAAs in humans are approximately 13.3 (PFBA), 130 (PFBS) and 560 (PFHxA) times higher compared to rats (OECD 2012). Therefore, effects observed in laboratory animals might lead to an underestimation of adverse effects in humans. When

compensating for differences in the elimination rate between different PFAAs, the toxic potency to induce liver effects (liver weight) in rats was similar between SC-PFAAs (e.g. PFBA and PFHxA) and LC-PFAAs (e.g. PFOA), indicating the importance of toxicokinetics when comparing the toxicity of PFAAs in laboratory animals (Gomis et al., 2018).

It is possible that SC-PFAAs cause hazards similar to the well demonstrated hazards of several LC-PFAAs. Effects on the immune system are well documented for long chain PFAAs; PFOA and PFOS are presumed to be hazardous for the immune system of humans (NTP, 2016) and this could be expected based on *in vitro* data on SC-PFAAs as well: certain effects on the immune system have been shown to be peroxisome proliferator-activated receptor alpha (PPARα) dependent. the liver toxicity and peroxisome proliferation potency of PFAS increase with the carbon chain length until C9, and the activity was higher in response to carboxylates compared to sulfonates. PFHxA can cause peroxisome proliferation and elicits higher human PPARα activity in a transiently transfected COS-1 cell assay than PFBS, PFHxS, PFOS and PFBA but a lower PPARα activity than PFOA (e.g., Wolf et al. 2008). A recent review describes associations of childhood asthma with certain PFAAs and also with PFBS exposure (Rapazzo et al., 2017). PFBA and PFBS have shown to activate the thyroid receptor and reduce serum thyroxin in male rats (Buttenhoff et al., 2012, Feng et al., 2017).

Developmental effects (effects on birth weight and gestational size) are effects of concern related to long-chain PFAAs exposure. Effects on birth weight and gestational size have been observed in a (mixed short-chain and long-chain) PFAAs contaminated area in Italy (WHO, 2015) and were discussed in a variety of epidemiological studies. In animal experiments developmental effects of PFBA concerning litter resorption, delayed eye opening and onset of puberty in mice were reported by Das et al. (2008). There are also indications that prenatal PFBS exposure may cause deficits in perinatal growth, pubertal onset, and reproductive organ development in female mice (Feng et al., 2017).

Summing up, the available data suggests that SC-PFAAs may cause similar type of adverse effects on human health as long-chain PFAAs. Also activation and interference with estrogen and thyroid receptors and developmental effects observed in a study with combined PFAAs exposure are reasons for concern. Sub lethal long-term health effects on wildlife and for the human population cannot be excluded given the irreversible long term exposure to SC-PFAAs.

Concerns related to the combination of the properties listed above

The combination of high mobility in water, low adsorption potential to organic matter, extreme persistence and enrichment in plants pose a concern that once the environment is exposed, the most essential resources of human nutrition (drinking water and edible plants) remain contaminated over very long time periods, even over several generations. Due to the extreme persistence and mobility, such exposure is very difficult to reverse. Such continuous exposure causes in combination with the potential for long-range transport, the substance to reach remote areas and vulnerable receptors such as drinking water and sensitive ecosystems. A recent study by Routti et al. 2017 detected PFBS in all the polar bear samples investigated (n = 70) and PFHxA in 71% of the samples. Together with the inefficiency in removing them from drinking water, continuing releases will lead to poorly reversible ubiquitous background concentrations of SC-PFAAs.

The very long-term exposure and current lack of approaches to estimate bioaccumulation potential in air-breathing organisms render high uncertainties for quantitative estimation of exposures for relevant time periods and hence also risks for food web, society and humans cannot be reliably quantified.

The combination of extreme persistence with high protein binding potential, implying at least low to medium level (but by no means negligible) bioaccumulation potential in specific organs of air breathing organisms, causes the concern that the exposure in humans and the environment may increase over the lifespan in specific organs to levels where the likelihood of effects is relevant.

PFHpA

It should be noted that PFHpa may have, due to its chain length, also PBT/vPvB properties in addition to the concerns and properties listed above. Whether Annex XIII criteria of REACH are fulfilled cannot, however, be yet concluded.

3. Use and exposure

Based on the rough mass screenings carried out in 2014, 2015 and 2017, the REACH registration database contains registrations on > 50 substances which are considered as precursors of the SC-PFAA and hence sources of SC-PFAA in the environment. In a mass screening of 2014, >> 100 SC-PFASs were contained in the CLP notification database even though none of them have harmonised classification. Detailed information on use and exposure of PFBS and PFHxA can be found in the related RMOAs[6] and [7] (and of PFBA in a draft RMOA) and is summarised below for PFBS and PFHxA. Many of the known uses result in incorporation of substances into articles and import of these substances in articles is anticipated or known to take place (depending on the subgroup). It should be highlighted that the releases can be assumed to cause a combined long-term exposure of the SC-PFAAs in the environment and humans mainly due to the gradual degradation of the precursors to SC-PFAAs.

In the period 2011 – 2015 the total global production of PFBS increased from 23 to 27 tonnes/year. In the same period, the production of perfluorobutane sulfonyl fluoride (PBSF) a key intermediate for production of PFBS-related substances, increased from 287 to 317 tonnes/year. Hence, the global manufacture of PBSF is about ten times higher compared to PFBS. Of the 317 tonnes of PBSF manufactured globally in 2015, 299 tonnes (94%) were manufactured in China. The volumes registered under REACH of PFBS-related substances are in the range 22 – 211 t/year. One specific PFBS-related substance (MeFBSE) is registered under REACH in the tonnage 10-100 t/year for use as an intermediate in the manufacture of PFBS-related polymers used as repellent agents and as tile and grout additive.

The main application area for PFBS-related substances is water and stain repellent protection[8] for leather, textiles and carpets and porous hard surfaces. Minor application areas include surfactants for inks, paints, waxes, etc.; flame retardants for polycarbonate; mist suppressants for metal plating, and surfactants for fluxes for production of electronics.

PFHxA is not registered under REACH but more than 40 potential PFHxA precursors are registered. No direct uses of PFHxA are known. PFHxA is anticipated to be mostly present in articles as impurity or degradation product. PFOA is increasingly substituted with PFHxA and its precursors (e.g., POPRC, 2016) and is expected to be reflected as changes in the registered tonnages and as new registrations.

PFHxA-related substances can be used as surfactants or as monomers for the production of side-chain fluorinated polymers. The former use is related to the hydrophobic properties of the perfluoroalkyl tail, and hydrophilic functional group:

[6] https://echa.europa.eu/pact/-/substance-rev/11911/term
[7] https://echa.europa.eu/pact/-/substance-rev/12934/term
[8] The most well-known PFBS-based water and stain repellent product line is Scotchgard™ by 3M

they are able to decrease the surface tension. Side-chain fluorinated polymers are mainly used for achieving water, stain and soil repellence of e.g. textile/leather, paper, glass, or ceramic material. Furthermore, PFHxA-related substances are for example used by the semiconductor and (opto)electronic industry, by the firefighting industry and for metal surface treatment.

PFHxA was found in surface water from the global oceans (Ahrens et al., 2010) as well as rivers (Takemine et al., 2014) and lakes (Sun et al., 2011), in effluent and sludge of WWTPs (Ahrens et al., 2009b), landfill leachates (Busch et al., 2010), groundwater (Eschauzier et al., 2013), soil and sediment (Klif, 2010), tap water and raw water (Llorca et al., 2012a), snow of remote regions (Kirchgeorg et al., 2013), atmosphere (Jahnke et al., 2007), house dust and air (Shoeib et al., 2011). PFHxA and several precursors have also been detected in the various aquatic and terrestrial organisms from natural (e.g. Llorca et al., 2012b) and rather exposed sites (see Annex I, table C) including wildlife (Ahrens et al 2016; Routti et al 2017; Gebbink et al., 2016; Lam et al., 2014; Lam et al., 2016).

It has been shown for example by Boiteux et al. (2017) or Eriksson and Karrman (2015) that concentrations of precursors and PFHxA correlate with each other. This further emphasises the importance of precursor substances and their distribution in the environment. Furthermore, the decrease of environmental concentrations of PFOA has been observed to be accompanied by an increase in PFHxA concentrations (for example by Shiwaku et al., 2016). As PFOA is increasingly substituted with PFHxA and its precursors (POPRC, 2016), this trend has to be expected and environmental concentrations of PFHxA will very likely increase in the near future. Additionally, even if emissions would now stop, a large mass of fluorinated polymers which already can be expected to be present in the environment will likely act as a future emission source (Wang et al., 2014. Global emission inventories for C4-C14 perfluoroalkyl acid (PFCA) homologues from 1951 to 2030, part II: the remaining pieces of the puzzle).

Of the SC-PFASs, PFBS is the dominating PFAS in river and/or sea water in several studies of European waters (Zhao et al. 2015; Möller et al. 2010; Eschauzier et al. 2010). A European monitoring study by Zhao et al. 2015 also showed decreasing trends from 2006 to 2011 in PFOA and PFOS concentrations in the River Elbe at locations where marine water did not intrude. PFHpA and PFBS were among the most frequently detected PFASs in a study of perfluorinated compounds in drinking waters from Brazil, France and Spain by Schwanz et al., 2016. In a study by Eschauzier et al. (2012) PFBA, PFBS, PFOS and PFOA were the most abundant PFAAs in intake water that was further processed to become drinking water. Concentrations of PFBA and PFBS remained more or less constant throughout the treatment process and were detected in highest concentrations of the PFAAs in the finished drinking water (Eschauzier et al., 2012). PFBS has been detected in deep Arctic waters, and oceans are likely to be the sink (Yamashita 2008). SC-PFAA have been shown to be already in the environment and in e.g. drinking water (Filipovic et al, 2015; Filipovic and Berger, 2015).

4. Summary on the hazards and concerns

The concerns described for the SC-PFAAs in sections 2 and 3 can be divided into two parts:

Concerns and properties which based on available evidence can be concluded to exist with a sufficient level of certainty

1. SC-PFAAs are extremely persistent, therefore exposure already caused is very difficult to reverse. If releases are not reduced, the levels in the environment

and humans will increase.

2. Due to their fate properties, once released, SC-PFAAs cannot be readily removed from waste water, surface water, ground water or drinking water with currently available (or: best available) techniques: exposure cannot be removed by currently available removal techniques .

3. SC-PFAAs have high potential for long-range transport, mostly due to transport via water.

4. SC-PFAAs bioaccumulate in biota and in humans, although the level is uncertain due to lack of an appropriate approach to assess this. The current understanding is that the accumulation potential varies depending on the SC-PFAA and target species/ gender/age.

5. SC-PFAAs have been observed to be transferred via human milk.

6. SC-PFAAs are known to enrich in edible plants.

7. Tonnage of SC-PFAAs and related substances are increasing in the EU due to their use as substitutes for long-chain PFASs. It is also expected that exposures will increase and in some datasets this trend has been observed.

8. SC-PFAAs as a group, constitute a source of combined exposure. Furthermore, the SC-PFAA precursors can be considered already now as a very long-term repository for combined exposure to SC-PFAAs both of humans and the environment. Finally, also the combined exposure of SC-PFAAs and LC-PFAAs needs to be considered .

Concerns which cannot be adequately addressed by current standard test and assessment approaches, including quantitative risk assessment approaches

9. Quantitative exposure assessment tools cannot reliably predict the distribution of SC-PFAAs over the extremely long time periods for which these substances persist. Neither can the slow formation of SC-PFAAs from their precursors be taken sufficiently into account in exposure assessment. Even if reliable tools might (or might not) be developed in the future, the development seems currently not feasible.

10. Level and importance of bioaccumulation of SC-PFAAs as an additional intrinsic property, potentially causing thereby high internal exposure in air-breathing organisms (including humans), cannot currently be reliably estimated due to lack of experimental data and lack of understanding of the influence of protein binding on bioaccumulation and of the link between protein binding and effects. Based on current information, bioaccumulation potential is not negligible.

11. Long-term toxicity cannot currently be reliably estimated due to lack of experimental data for most SC-PFAAs. However, based on available data on SC-PFAAs *in vitro* and in *vivo*, the likelihood to cause similar effects as LC-PFASs is a reason for concern.

12. High protein binding capacity, bioaccumulation potential and toxicity are known to be connected, however, toxicity can currently not be predicted from protein binding capacity.

13. A specific challenge for estimating risks is caused by the combined exposure from several subgroups of SC-PFAAs, as well as the high number of the

precursors as long-term sources of SC-PFAAs. Also, combined exposure of SC-PFAAs and LC-PFAAs needs to be taken into account as an additional concern.

The properties listed in points 1-8 should trigger consideration of appropriate risk management measures to avoid potential future damage.

Many of the concern elements above match or are analogous to the elements of the PBT/vPvB concern. For comparison, the elements of the PBT/vPvB concern are provided in the introductory section of ECHA's Guidance on PBT/vPvB assessment . Furthermore, many of the concern elements in the past SVHC-cases identifying a substance based on Art. 57(f) of REACH can be found in the summary above.

In summary, the concerns described above cannot be readily quantified with a sufficient level of certainty using the currently available information and risk assessment methods. Neither can exposures be effectively reduced, once SC-PFASs have been released in the environment. Due to these reasons no save level of releases can be defined. Due to the nature of the concerns identified, minimisation of releases for diverse sources of SC-PFAAs appears to be warranted, without delay. Due to the formation of SC-PFAAs from a vast array of SC-PFASs precursors (assessment not presented in this document), the need to minimise releases and exposures would also need to cover these precursors.

5. Possible regulatory management options and planned next steps

In the EU several regulatory tools might be relevant for managing risks arising from SC-PFASs. Here only the instruments of the REACH and CLP Regulations are discussed in more in detail. For the subgroups PFBS and PFHxA RMOAs are available[9]. For PFBA a draft RMOA is available.

As described in section 4, no reliable exposure estimation neither safe thresholds can currently be derived which could sufficiently address the combination of the concerns on SC-PFAAs, as provided in the previous section. Due to the **difficulty to reverse the exposure** once caused and the anticipated **increase in exposure,** regulatory risk management measures should be imposed on SC-PFASs (consisting of SC-PFAAs, their salts and precursors) **without undue delay**. The impossibility to derive reliable quantitative risk estimates in **analogy to the PBT/vPvB concern** indicates a need for **minimisation** of the releases and exposures.

Currently PFAS (e.g., CnF2n+1−R) are listed with limit values of 0.1 µg/l for individual substances and 0.5 µg/l for the sum of all PFAS compounds in the Revision of the Drinking Water Directive. EQSs may also be developed in the future under the WFD (to be confirmed).

Substances having the set of properties and associated concerns as listed in section 4 have so far not been subject to regulatory risk management. Advantages and disadvantages of the different available options for regulatory risk management need to be considered from several perspectives taking into account the concerns, in particular, irreversibility of exposure to the arrow head substances due to their extreme persistence. Furthermore the regulatory instruments likely need a similar type of degradation assessment to justify that the precursors degrade into the arrow

[9] https://echa.europa.eu/pact/-/substance-rev/11911/term
[9] https://echa.europa.eu/pact/-/substance-rev/12934/term

head substances. A further example of common challenges regardless of the risk management option selected is the question of which substance(s) the dossiers should address. I.e., should only (each of) the arrow head substances (SC-PFAAs) and their salts (e.g., PFDA on the Candidate List[10]) be subject to regulatory risk management or should their respective precursors and hence very large subgroup -entries (e.g., PFOA entry in restriction[11]) be subject to regulation, together with the arrow head substances? A wide substance scope (large group(s)) is challenging in any risk management process with regard to transparency of the measure and enforcement.

Confirmation of the hazard properties of SC-PFAS

Harmonised classification (CLH) and inclusion in the Candidate List can be used to confirm the hazard properties of substances and by that provide a basis for actions under other regulatory instruments. Furthermore, CLH and inclusion in the Candidate List directly trigger important obligations on the registrants and downstream users of the substances.

CLH

In case one or more of the SC-PFAAs would fulfil criteria for harmonised classification as CMR -substance, CLH might be a valuable complementary activity for SC-PFASs, especially from the international perspective (e.g., Stockholm Convention). Such activity should, however, not delay other regulatory measures which are necessary to address the concerns in the EU context.

The need for harmonised classification as toxic to reproduction is currently being assessed for one of the subgroup's arrow head (PFBA) by DK. Thereby the possibility to read across a set of effects from PFHxA, PFOA and PFNA is under consideration. For two other arrow heads (PFBS, PFHxA) such classification based on read across has been explored (by NO and DE) but with inconclusive/negative results so far. The judgement on whether read across of effects between PFOA, PFNA and the SC-PFAAs is possible will need to be coherently justified for the arrow head representatives of the SC-PFASs. Work exploring whether such coherent justification would be possible is ongoing?.

SVHC-identification based on Article 57(f) of REACH

The inclusion of substances in the Candidate List is an obligatory first step of the authorisation process. In addition, SVHC identification and inclusion in the Candidate List can be used to support the restriction process by clarifying the relevant hazard properties upfront.[12]

As described in section 4, no thresholds can be derived with sufficient certainty for estimating the level of long-term exposure and resulting risks caused by the releases of SC-PFASs. Although these concerns resemble the PBT/vPvB properties, it is clear that with the current set of information it would not be possible to identify the SC-PFAAs (or SC-PFASs) as PBT/vPvBs (potentially except PFHpA, which is under evaluation by BE and NL). Furthermore, if one or more the substances would be classified as toxic for reproduction Cat 1B (see previous subsection), there would be a possibility for SVHC-identification under Art. 57(c).

There seems to be sufficient evidence available on SC-PFAAs to justify SVHC

[10] https://echa.europa.eu/substance-information/-/substanceinfo/100.244.606
[11] https://echa.europa.eu/substance-information/-/substanceinfo/other/c0c60093c2b2ab901621da907500ba84907767c90b4ae546d4c5b0ba2fcb0e7b
[12] This is only relevant for properties for which there is no criteria in the CLP Regulation.

identification based on Art. 57 (f). The concerns resemble those of PBT/vPvB substances. Few concern elements are even the same as those of typical PBT/vPvB substances[13] but SC-PFAAs also have additional properties beyond those that PBT/vPvB substances normally have.

The general approaches developed to support the equivalent level of concern (ELoC) assessment and experience gained from the other article 57(f) cases can be used to support the preparation of a SVHC dossier for SC-PFASs. It is, however, acknowledged that ELoC cases so far have covered limited types of properties (ED, STOT RE, respiratory sensitisers).

SVHC identification is a clear signal to industry and other stakeholders that the substances are of very high concern. Candidate Listing immediately triggers obligations in the supply chain communication and seems to directly trigger substitution. For SC-PFASs this is relevant if the entry would also cover precursor substances which are on the market.

Authorisation requirement

The main source of emissions to the environment and hence for exposure is the use of the SC-PFASs (in particular of the precursors) in articles, although highest concentrations found in the environment have been sites contaminated by firefighting foams and landfill leachates. For articles, authorisation is normally not the most appropriate regulatory instrument. Therefore it is not further reflected in this document. However, authorisation requirement can be considered after the (potential) restriction(s) on SC-PFASs in articles are set up, in particular, if more information on other uses became available.

Restriction

Based on the use information of the SC-PFASs, restriction appears to be the relevant instrument under REACH in order to avoid or minimise releases and exposures to these substances. This is as only a restriction can ensure that the releases of SC-PFASs from articles, irrespective of whether they are produced in EU or the rest of the world, would be reduced. Such an approach has been the basis for the regulation of PFOA under REACH, Commission Regulation (EU) 2017/1000) and for addressing LC-PFCAs for which a restriction proposal from DE/SE is under discussion.

A restriction can cover a wide range of substances and uses and by that reduce and avoid further widespread contamination as well as risks related to combined exposure and the unwanted substitution with other SC-PFASs.

The main question appears to be whether SVHC-identification would be useful to be carried out before restriction. SVHC-identification was carried out as the step before restriction for LC-PFASs. These SVHC-dossiers covered arrow heads and their salts only whereas the restriction(s) then covered the whole group including precursors. However, the difference to SC-PFASs is that LC-PFASs were identified under 57 (d) or (e) (PFOA, PFNA and PFDA as PBTs, C11-C14-PFCAs as vPvBs). Hence, without first including the LC-PFASs in the Candidate List, RAC would directly have taken over from MSC the identification of PBT/vPvB properties. For SC-PFASs a restriction could cover hazards which cannot be addressed in SVHC-identification. Section 0.10 of Annex I to REACH could be useful to argue for the identification of a risk due to a particular effect which is not a toxicological effect but effect due to the impairment/contamination of natural resources. Due to the variety of the concerns of SC-PFASs, this could be used

[13] High persistence; bioaccumulation potential, although it seems not as pronounced as for PBT/vPvB substances.

if SVHC-identification would fail. However, hazard identification in the restriction process instead of the SVHC –process would still benefit from some comparisons to the PBT/vPvB concern.

Drafting a restriction proposal and supporting the consequent restriction process is a high workload and demanding for the authority. The six months pre-defined processing time to discuss in RAC both hazards/concerns and exposure aspects as well as to adapt the SEAC approach accordingly is a challenge. Having the SVHC identification first makes the restriction process lighter, because a discussion on the hazard is not anymore needed within the restriction process. To reduce the overall time needed before a restriction is in place, it could be considered to launch the restriction process while SVHC-identification is ongoing.

In relation to the workload the restriction process for SC-PFASs can likely be simplified. Due to the similarity of the concerns on SC-PFASs with PBT/vPvB substances and, in particular, due to the understanding that risks caused by SC-PFASs cannot be quantified with sufficient reliability with conventional quantitative risk assessment approaches, it would be worthwhile to consider similar approach for assessing socio-economic impacts for SC-PFASs as is applied for PBT/vPvB substances. For PBT/vPvB substances the approach has focused on quantitative estimation of emission reduction and related costs (€ per kg of emission reduction) supported with qualitative description of the case. The same approach, adopted by SEAC (link) is used for PBT/vPvB substances in both restriction and authorisation route, and has been presented by SEAC (link) to be possible approach also for endocrine disrupting substances for the environment.

International activities

Global risk management can be achieved for substances which are covered by the Stockholm convention on POPs. However, evidence of relevant effects is necessary for a substance in order to include it in the work program under the Stockholm convention. For this purpose more information on (eco)toxicological effects may be necessary (experimental data, read-across, AOPs) for SC-PFASs. Toxicological testing is currently under planning in third countries (USA). It is noted that in case the planned CLH proposal of PFBA would be successful, it would be useful to assess the feasibility and usefulness of this international risk management route more in detail.

The current activity on PFASs under the umbrella of OECD/UNEP global PFC group (coordinated by OECD) also warrants to be mentioned in this context. This group has not focussed on SC-PFASs, but. e.g., developed a global inventory on PFASs in general, thereby supporting the work of the regulatory authorities in identifying groups of PFASs relevant for further work. This substance –identity –related work will continue. The group is launching in 2018 a project exploring the alternatives to PFASs. The group has also mapped in 2016 the existing risk management instruments on PFASs in the member countries. Several EU Member States participate in the PFC group and ECHA is also supporting its activities.

Conclusions on the risk management options

The choice of the risk management route in the REACH context is mainly a choice between whether SVHC identification should be carried out first and then be followed by restriction or whether restriction could as sole instrument be efficient and effective. Authorisation as a primary regulatory instrument does not seem to be relevant for SC-PFASs as the main concern relates to the presence of these substances in articles. Authorisation requirement could potentially be considered as supplementary measure later on, provided that the relevant substances are included in the Candidate List.

Based on the considerations presented above, it seems pertinent to carry out SVHC-identification first (restriction may be initiated in parallel). It is recognised that if SVHC-identification is used, the only option currently seems to be to identify the substances based on Article 57(f) of REACH. Identification using a combination of hazards as listed in this document has never taken place before. The substance scope of individual SVHC-dossiers and restriction dossiers should still be further considered in detail.

Due to the concerns summarised in section 4, CLH (and potential further toxicity testing, if necessary) would be useful mainly as complementary activity. However, it is likely more important if it is decided at the EU level to advocate risk management of SC-PFASs at global level (e.g., OECD, Stockholm convention).

Next steps planned by MSCAs for regulatory risk management of SC-PFASs

RMOAs of PFBS, PFBA and PFHxA have already been published/drafted and flag the need for SVHC-identification followed by restriction. With current work planning of NO, DE and DK, the work flow for the upcoming 1.5 years for the envisaged activities is provided below.

Q1-Q2/2018:

- Preparation of SVHC-dossiers of PFBS (NO) and PFHxA (DE). Consultation of PBT Expert Group, where necessary.
- Preparation of CLH dossier of PFBA (DK; submission by the end of Q3/2018)
- DE and NO further scrutinise their possibilities for read across of data (for PFBS and PFHxA) analogous to PFBA CLH proposal of DK.

Q3-Q4/2018:

- SVHC-identification of PFHxA (DE)
- Conformity check and resubmission (if needed) of CLH dossier of PFBA (DK)

Q1-Q2/2019:

- SVHC-identification of PFBS (NO)
- Preparation of the SC-PFASs group restriction proposal/restriction proposals of PFBA and PFHxA (submitters and the scope of proposals to be discussed in the PFASs working group; timeline for restriction proposal of PFBS to be defined at later stage)
- RAC opinion development for PFBA CLH

Q3-Q4/2019:

- Submission and conformity check of the restriction proposal(s)
- Depending on PFBA CLH outcome, potentially preparation and submission of CLH for other SC-PFAAs

It seems also to be necessary to trigger preparation of risk management for the remaining SC-PFASs groups at least for those for which registrations are available.

6. Conclusions

The concerns and risks summarised in section 4 cannot be quantified by current conventional risk assessment methods with sufficient certainty. No reliable threshold can be set for an acceptable amount of releases. Furthermore, exposure to SC-PFAAs is only very difficult to reduce and very long-term exposures cannot be avoided due to the extreme persistence and slow formation of SC-PFAAs from the precursor SC-

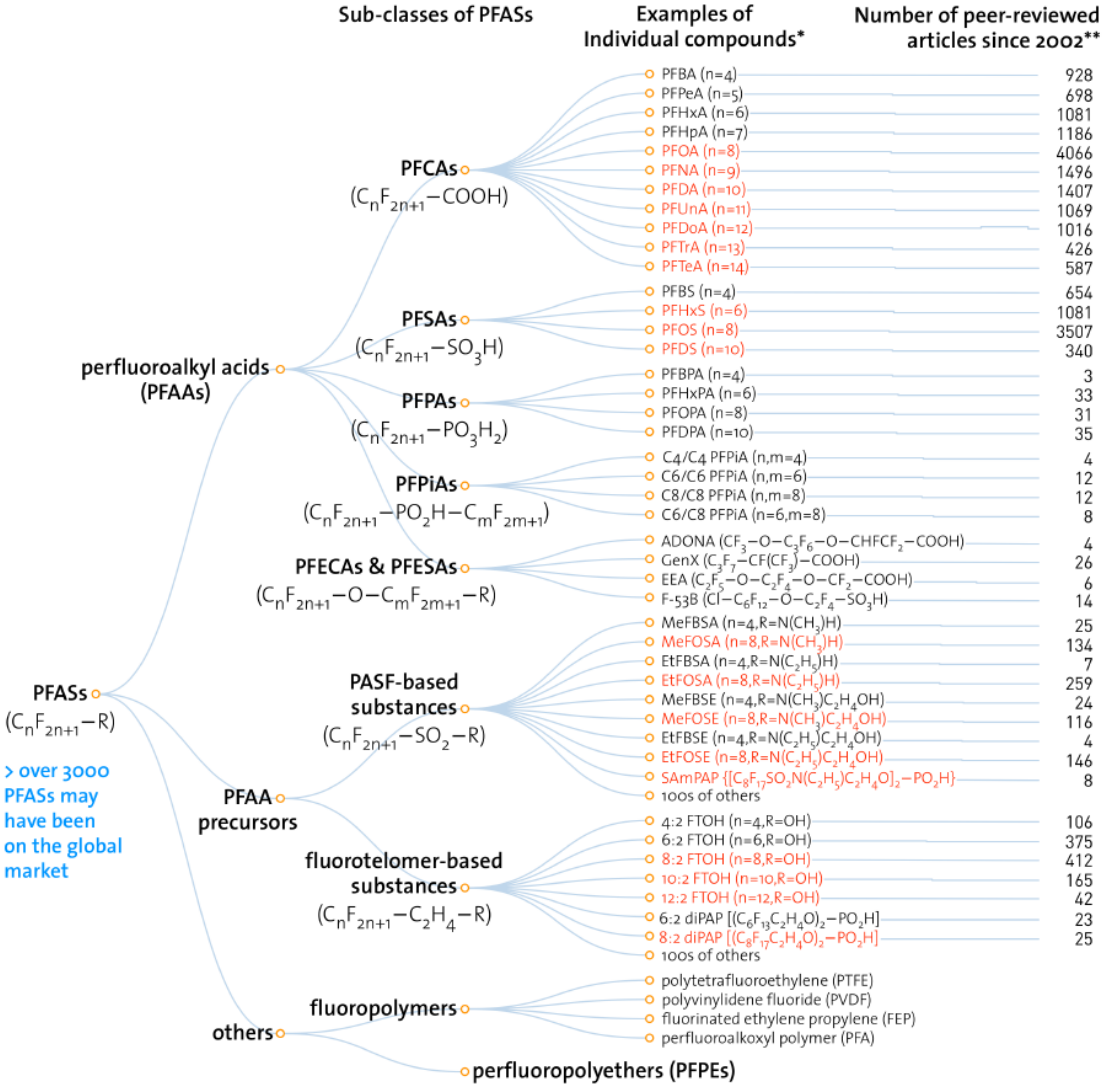
PFAS. SC-PFAAs and their precursors cause combined exposure among themselves and with the LC-PFAAs. Many of the concerns pertaining to the SC-PFAAs are comparable with elements constituting the PBT/vPvB concern but the SC-PFAAs elicit additional concerns and therefore these substances seem to be of Equivalent Level of Concern to PBT/vPvBs. Hence prompt action and minimisation of releases and exposures is warranted for the sources of SC-PFAAs. A further reason for not delaying the action is that removal of SC-PFAAs is currently not feasible from drinking water, other water resources and waste water.

Due to formation of SC-PFAAs from SC-PFASs precursors, the need to minimise releases and exposures also covers SC-PFASs. Based on the advantages and disadvantages considered in the previous section it is recommended to first carry out the SVHC identification of the substances based on Article 57(f). Given the information on the uses, the use in articles in particular, and the possibilities to address combined exposure and the high number of substances, restriction(s) should be subsequently proposed. However, it would also be possible to carry out restriction without prior SVHC –identification.

As discussed in section 5, at least some of the SC-PFAAs may meet the specific classification criteria for human health. Depending on the final outcome of the planned CLH-activity, further targeted testing of (eco)toxicity, where needed, and carrying out harmonised classification might be of added value specifically in order to judge whether international risk management activities would be possible/necessary to pursue. Due to the extreme persistence, however, the proposed steps of SVHC-identification and restriction should not be delayed but additional concerns and risk management activities can be followed up in parallel or sequentially as complementary activities.

Finally, the knowledge on uses and the concerns associated with SC-PFASs in products and articles is unlikely comprehensive. The current, not complete, information available on effects in combination with other information on properties causing several concerns imply that the lack of full scientific certainty on risks estimated by conventional quantitative risk assessment methods should not be used as a reason for postponing cost-effective measures to prevent deterioration of the environment.

Appendix 1: Nomenclature of PFASs, as illustrated in Wang et al. (2017)



* PFASs in **RED** are those that have been restricted under national/regional/global regulatory or voluntary frameworks, with or without specific exemptions (for details, see OECD (2015), Risk reduction approaches for PFASs. http://oe.cd/1AN).

** The numbers of articles (related to all aspects of research) were retrieved from SciFinder® on Nov. 1, 2016.

Appendix 2: Overview of current and past REACH and CLP regulatory activities on PFASs

Subgroup	PFASs group (includes precursors where relevant)	RMOA	Assessment	CLH/SVHC	Restriction
Long-chain PFCAs	PFOA	√	√	√ / √	√
	PFNA	√	√	√ / √	Under preparation
	PFDA	√	√	√ / √	Under preparation
	C_{11}-C_{14}-PFCAs	√	√	- / √	Under preparation
	PFHxS	√	√	- / √	Scheduled
Short-chain PFCAs	PFHpA (C_7-PFCA)		On-going (including testing)		
	PFHxA (C_6-PFCA)	√	On-going	/ Under prep.	
	PFBA (C_4-PFCA)	On-going	On-going	Under prep. / -	
Short-chain PFSAs	PFBS (C_4-PFSA)	√	Scheduled		
Perfluoroether carboxylic acids (PFECA)	ADONA HFPOA-DA (GenX) 2 other related subst.		On-going On-going		
Perfluoropolyethers (PFPEs)	TFEE-5		On-going		

Appendix 3: Case examples

In 2016 the Swedish Chemicals Agency published a report which included case examples addressing the socio-economic impacts of PFASs due to contamination of drinking water (KEMI 2016). Swedish estimates for addressing PFAS contamination of drinking water are:

- Charcoal filtering of water in Uppsala: annual cost 1 million EUR.
- New water supply in Ronneby: 3 million EUR. Ronneby is a small city – approximately 5000 households were immediately affected when high levels of PFASs were discovered in 2013.
- Larger new water supplies for Växjö and Alvesta: 45.5 million EUR. The exploitation of the new water supply in Växjö / Alvesta had nothing to do with PFASs. Alvesta and Växjö were taken as an examples to show that the costs of replacing a water supply can be significantly larger than in the Ronneby case. This depends partly on the size of the municipality but also on access to appropriate water supplies to exploit. In Alvesta / Växjö a large part of the costs are due to the need to build a 50-100 km long water pipe.

PFOS in fire-fighting foam was prohibited from 2007, and in the Norwegian petroleum sector, PFOS-containing foam was substituted with PFOS-free foam in time for the regulation. However in 2016 chemical analysis demonstrated a PFOS-content above the limit value at several installations. Apparently, the practical procedures in the substitution process have not ensured complete emptying of the tanks when switching to PFOS-free foam. There have been considerable emissions to sea in the period 2007-2016, estimated to around 100 kg PFOS. It is important to avoid similar mistakes in the future when other PFASs are phased out.

In 2013, during a routine sampling of a drinking water well in the district of Rastatt, Germany, a contamination with PFAS has been detected. In the course of further investigations 400 hectares of PFAS contaminated agricultural crop land and a thereby caused pollution of the groundwater has been found. Concentrations of up to 1340 µg/kg (0-30 cm, solid) and 32 µg/l (aqueous eluate) for the sum of PFCs have been detected.[14] According to the current state of knowledge the input took place in the context of agricultural management. For the vast majority of contaminated areas, there are evidences that composts mixed with paper sludge have been applied. The PFCs or precursors are most likely been mixed up with paper sludge to the compost and thus to the soil.[15]

Since the contamination most likely took place more than 10 years ago, the PFCs have already been able to leach through the entire soil layer and reach the groundwater. Due to the vast extent of contaminated soil and groundwater, no reliable cost estimations in terms of remediation have been made so far.

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