National Institute for Public Health
and the Environment
Ministry of Health, Welfare and Sport

PFAS exposure from non-food sources compared with food and drinking water

RIVM letter report 2026-0073
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DOI 10.21945/RIVM-2026-0073

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This study was commissioned by the Ministries of Infrastructure and Water Management, of Health, Welfare and Sports, and of Agriculture, Nature and Food Quality in the context of the framework of the Programme 53 PFAS.

Published by:
National Institute for Public Health and the Environment, RIVM
PO Box 1 | 3720 BA Bilthoven
The Netherlands
www.rivm.nl/en

Synopsis

PFAS exposure from non-food sources compared with food and drinking water

PFAS exposure exceeds acceptable levels for most people in the Netherlands. RIVM is examining ways to reduce it. This means that all the potential sources of PFAS exposure need to be made clear. In 2023, RIVM demonstrated that food and drinking water are important contributors. In 2025, it selected several additional sources for further investigation.

RIVM has now estimated PFAS exposure from house dust, indoor air, outdoor clothing, impregnation sprays and personal care products. Scientific literature on these sources is still limited: few measurements exist and available data are difficult to compare. As a precaution, RIVM calculated the highest plausible exposures; actual exposure is lower.

Based on these estimates, PFAS intake from these sources over a lifetime is lower than intake from food and drinking water. For young children, however, house dust and indoor air can be significant sources. More research is needed to better understand the extent of PFAS exposure from house dust and indoor air.

PFAS exposure from outdoor clothing, impregnation sprays, and personal care products is generally low. According to scientific literature, PFAS have been found in some personal care products, such as certain face creams, foundations, eye makeup, rouge, body lotions and lip products. Only people who use PFAS-containing products frequently and over many years are exposed to higher levels of these substances than those who do not. The same applies to impregnation sprays. There is little exposure from outdoor clothing.

This study does not indicate the need for measures to be taken at this time. As a precaution, people can take simple steps to reduce exposure: vacuuming and ventilating helps lower exposure from house dust and indoor air. For personal care products, consumers can check ingredient labels. PFAS often appear under names containing “fluor”, “perfluoro”, or “polyfluoro”. Impregnation sprays increasingly carry labels stating “PFAS-free”.

Keywords: PFAS, food, drinking water, house dust, indoor air, outdoor clothing, impregnation spray, personal care products, exposure

Publiekssamenvatting

PFAS-blootstelling uit overige bronnen vergeleken met blootstelling uit voedsel en drinkwater

Bijna alle mensen in Nederland krijgen te veel PFAS binnen. Het RIVM onderzoekt hoe de blootstelling aan deze stoffen kan worden verminderd. Daarvoor moet duidelijk zijn via welke bronnen mensen aan PFAS kunnen blootstaan. Het RIVM toonde in 2023 aan dat voedsel en drinkwater belangrijke bronnen zijn en selecteerde in 2025 enkele bronnen om verder te onderzoeken.

Het RIVM heeft daarom nu de blootstelling geschat uit huisstof, binnenlucht, outdoor kleding, impregneersprays en persoonlijke verzorgingsproducten. Over deze bronnen blijkt nog weinig bekend te zijn in de wetenschappelijke literatuur. Er zijn weinig metingen gedaan en de gegevens die er zijn, zijn moeilijk met elkaar te vergelijken. Voor de zekerheid heeft het RIVM daarom berekeningen gemaakt met de hoogst denkbare blootstelling via deze bronnen. In werkelijkheid ligt de blootstelling lager.

Volgens deze schattingen krijgen mensen via deze bronnen tijdens hun leven minder PFAS binnen dan via voedsel en drinkwater. Huisstof en binnenlucht kunnen voor jonge kinderen wel belangrijke bronnen zijn. Meer onderzoek is nodig om beter te kunnen inschatten aan hoeveel PFAS mensen via huisstof en binnenlucht blootstaan.

Het gebruik van outdoor kleding, impregneersprays en persoonlijke verzorgingsproducten stelt de meeste mensen aan weinig PFAS bloot. PFAS zitten volgens de wetenschappelijke literatuur in enkele persoonlijke verzorgingsproducten, zoals in sommige gezichtscrèmes, foundations, oogmake-up, rouges, bodylotions en lipproducten. Alleen mensen die jarenlang vaak verzorgingsproducten met PFAS gebruiken, worden via de huid aan meer van deze stoffen blootgesteld dan mensen die dat niet doen. Dit geldt ook voor impregneersprays. Via outdoor kleding is de blootstelling klein.

Op basis van dit onderzoek is er geen noodzaak om maatregelen te nemen. Uit voorzorg kunnen mensen enkele eenvoudige maatregelen nemen om hun blootstelling te verkleinen. Stofzuigen en ventileren helpt om blootstelling via huisstof en binnenlucht te verminderen. Bij persoonlijke verzorgingsproducten kunnen mensen de ingrediënten op het etiket checken. Daarop zijn PFAS vaak te herkennen aan namen met ‘fluor’, ‘perfluoro’ of ‘polyfluoro’. Op impregneersprays staat steeds vaker ‘vrij van PFAS’ vermeld.

Kernwoorden: PFAS, voedsel, drinkwater, huisstof, binnenlucht, outdoor kleding, impregneerspray, persoonlijke verzorgingsproducten, blootstelling

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Summary

Per- and polyfluoroalkyl substances (PFAS) are a group of synthetic chemicals used in many consumer products due to their water-, grease-, and dirt-repellent properties (Affourtit & Verbruggen, 2025). However, their persistence and potential health effects have raised concerns worldwide. In the Netherlands, previous research has shown that chronic PFAS exposure via food and drinking water exceeds health-based guidance values (Schepens et al., 2023), highlighting the need to assess additional exposure from non-food sources.

This report provides a quantitative, screening-level estimate of PFAS exposure from prioritised non-food sources in the Netherlands: house dust, indoor air, outdoor (water-repellent) clothing, impregnation sprays, and personal care products. These exposure estimates are placed in context by comparing them with existing exposure estimates for food and drinking water to assess the relative importance of non-food sources. A second goal is to identify which of those sources warrant further investigation and refinement of the exposure assessment, as input for future risk assessment and risk management.

Concentration data for PFAS in house dust, indoor air, outdoor clothing, impregnation sprays and personal care products were systematically collected from European studies. For house dust, indoor air, outdoor clothing and impregnation sprays, exposure was calculated using mathematical equations based on established models from the literature, using conservative assumptions and input parameters. These calculations resulted in upper percentile estimates (typically P95 or higher) representing highly exposed individuals. For personal care products, exposure was estimated using a person-oriented, probabilistic model, the Probabilistic Aggregate Consumer Exposure Model (PACEM), which incorporates realistic product use patterns. PACEM was used to simulate an exposure distribution for the exposed subpopulation only. Taken together, the non-food exposure estimates in this report represent worst-case, screening-level exposures, whereas the food and drinking water estimates represent average long-term exposure in the general population.

For each source, only PFAS with available relative potency factors (RPFs) for oral exposure were included. Using the RPF approach, concentrations and exposures were converted to PFOA-equivalents (PEQ), allowing toxicity-weighted aggregation of multiple PFAS and comparison across sources. Since exposure routes differ between sources, all exposure estimates were further adjusted using route-specific absorption fractions, so that absorbed exposure from different routes could be compared on the same basis.

These screening estimates were compared with population-based long-term average exposure estimates for food and drinking water to place the non-food exposures in context and to evaluate their potential importance relative to exposure via food and drinking water.

Key findings:

- Food and drinking water are the primary contributors to long-term PFAS exposure across all age groups in the general population.
- House dust and indoor air can be important contributors for young children, under the conservative scenarios used in this report.
- Personal care products are relevant only for a small subgroup of users (about 2.7% of adults) and are expected to decrease further due to upcoming EU restrictions.
- Impregnation sprays can, under worst-case assumptions, lead to high exposures. The actual proportion of the population exposed to these levels is unknown, and further data on the prevalence and PFAS content of these products is needed.
- Other non-food sources, such as dermal exposure from outdoor clothing or mouthing of children on outdoor clothing, contribute minimally to total PFAS exposure.

The side-by-side comparison of PFAS exposure via food and drinking water versus non-food sources is subject to several uncertainties. These arise both from the source-specific exposure estimates and from methodological differences in how exposure was assessed. For non-food sources, available data are limited, heterogeneous, and often not specific to the Dutch context. In addition, exposure estimates for non-food sources are based on conservative scenarios and are intended as screening values rather than population averages. By contrast, exposure via food and drinking water was estimated by Schepens et al. (2023) using mean measured PFAS concentrations in food items and drinking water, combined with individual consumption data and body weights. These estimates are relatively robust, owing to standardized analytical methods, systematic sampling, and the use of population-based dietary surveys. Furthermore, differences in exposure metrics and target populations between the food and drinking water assessments and the non-food assessments affect the comparability of the results. Because of these methodological differences, a direct quantitative comparison between food and non-food PFAS exposure is not possible. The comparison should be interpreted as a screening-level indication of which non-food sources may be of concern.

This screening assessment shows that, although food and drinking water are the main sources of PFAS exposure for most people, non-food sources, particularly house dust and indoor air for young children and possibly impregnation sprays for some adults, can also be relevant. Further data collection and methodological improvements are needed to refine these estimates to better characterise their contribution and inform future risk management.

1 Introduction

Per- and polyfluoroalkyl substances (PFAS) are a group of synthetic chemicals that have raised concerns due to their persistence in the environment and potential health risks. PFAS are widely used in everyday products such as food packaging, non-stick cookware, outdoor water-repellent clothing, and impregnation sprays, thanks to their unique properties including water-, grease-, and dirt-repellence and heat resistance. However, their widespread use and environmental persistence mean that people in the Netherlands are exposed to PFAS through multiple sources and routes.

In 2023, RIVM calculated that the chronic exposure to PFAS via food and drinking water in the Netherlands exceeded the health-based guidance value derived for chronic exposure over a lifetime (Schepens et al., 2023). Long-term exposure above this value may be harmful to health, and additional exposure from other sources could further increase this risk. To date, the contribution of non-food sources to overall PFAS exposure in the Netherlands has not been quantified.

In 2025, non-food sources of PFAS exposure in the Netherlands were systematically identified and prioritised using a qualitative assessment (Affourtit & Verbruggen, 2025). The potentially most relevant sources identified were house dust, indoor air, water-repellent outdoor clothing, impregnation sprays and personal care products. The aim of the present report is to make a quantitative estimate of PFAS exposure from these non-food sources and to place these estimates in context by comparing them with existing exposure estimates for food and drinking water. Because PFAS are persistent and accumulate in the body, the focus is on contributions to long-term exposure rather than on acute, short-term peaks. This report, together with previous reports, is part of the PFAS research programme (2023–2026), which aims to gain insight into current exposure to PFAS in the Netherlands and to identify the sources contributing to this exposure with the strategic goal of reducing people's overall exposure to PFAS.[1]

First, this report collects and evaluates available PFAS concentration data and use data for the five prioritised non-food exposure sources. Using the best available methods, the exposure to PFAS from these sources was estimated for different age groups, with conservative assumptions where data are limited.

Next, these non-food exposure estimates are compared, as far as reasonably possible, with the previously calculated exposure via food and drinking water in the Netherlands. Due to the limited and heterogeneous data for non-food sources and the different methods for exposure assessments, the comparison is intended to give an indication of the possible importance of non-food sources for total PFAS exposure, rather than a realistic quantitative estimate of total PFAS exposure. Performing a full risk assessment based on combined exposure from all

[1] https://www.rivm.nl/pfas/onderzoeksprogramma

sources is outside the scope of this report. Rather, the report aims to identify which non-food sources warrant further investigation and refinement of the exposure assessment, as input for future risk assessment and risk management.

In this report, the methods used to estimate the exposure to PFAS via the five non-food sources are described in Chapter 2, including the concentration data and exposure scenarios used, as well as the input parameters. Results of the exposure estimates for non-food sources are reported in Chapter 3, followed by the comparison of these exposure estimates with those via food and drinking water in Chapter 4. Conclusions and recommendations are provided in Chapter 5.

2 Methods

2.1 Challenges and approach for exposure assessment of non-food sources

The exposure assessment of PFAS for non-food sources is less straightforward than for food and drinking water. For food and drinking water, concentration data are relatively well characterised, sampling and analytical methods are well-described, and exposure can be linked directly to individual consumption data from food surveys. This allows a relatively direct calculation of long-term, population-based exposure distributions.

When performing the exposure assessment for non-food sources, several complications arise. First, data availability and quality differ strongly between the five non-food sources considered in this report (house dust, indoor air, outdoor clothing, impregnation sprays and personal care products). Concentration data are often derived from heterogeneous studies, using different sampling strategies and analytical methods, and are typically not designed for exposure assessment in the Dutch context. Second, there is a high variability in the way people are exposed to the selected sources. Inhalation, ingestion and dermal contact all play a role, and the frequency, duration and intensity of contact depend on behaviours that are less systematically documented than food consumption. Third, the methods for estimating exposure are not the same for the five non-food sources, and they differ in their level of conservatism. For many consumer products, exposure is episodic or restricted to specific life stages, which makes long-term-averaged exposure less straightforward and shifts the focus towards typical or high exposures during use periods. In line with the overall aim of this report, we focus on the contributions of these non-food sources to long-term exposure, so that they can be placed in the same long-term exposure context as the exposure estimates via food and drinking water reported by Schepens et al. (2023). No new exposure calculations were performed for food and drinking water. For these sources, we rely entirely on the previously published exposure estimates by Schepens et al. (2023).

As a result, each non-food exposure source requires a tailored exposure assessment approach that reflects:

- the relevant exposure route(s) (i.e. oral, inhalation and/or dermal);
- the type and quality of concentration data;
- the behavioural patterns and use scenarios for different age groups; and
- the available exposure models.

In the following sections we will address these aspects.

2.2 Selection of routes per PFAS non-food source

For each of the five non-food sources, the most relevant exposure route(s) were identified:

- House dust: ingestion, because dermal contact is considered negligible compared to ingestion (Bakker et al., 2016).
- Indoor air: inhalation.
- Outdoor (water-repellent) clothing: dermal contact and ingestion via mouthing behaviour.
- Impregnation sprays: inhalation and dermal contact.
- Personal care products: dermal contact.

These routes per source form the basis for the quantitative exposure assessment. The exposure was assessed for the "general population", in the Netherlands. The assessment of occupational exposure is outside the scope of this assessment.

2.3 Selection of scenarios and age groups

The five non-food exposure sources and routes were categorised into two scenarios to distinguish typical exposure patterns from exposures driven by specific behaviours or product use:

- Baseline scenario: representing non-food exposure sources relevant for the general population.
- Subgroup scenarios: representing specific behaviours or product use that may lead to higher PFAS exposure among certain subgroups.

In the baseline scenario, the following combinations of sources and routes were included:

- house dust: ingestion;
- indoor air: inhalation and dermal contact; and
- outdoor (water repellent) clothing: dermal contact.

This scenario was calculated for four age groups for each non-food source: early childhood (1–5 years), middle childhood (6–10 years), adolescents (11–17 years), and adults (18+ years). This was done to reflect age-dependent physiology (such as larger body surface area relative to body weight) and behaviour and to allow comparisons within the same age group (e.g. young children, adolescents, and adults).

Subgroup scenarios per non-food source were calculated for the relevant age group:

- Outdoor (water repellent) clothing: ingestion via mouthing for children aged 1–6 years, as they are more likely to mouth or chew on clothing.
- Impregnation sprays: inhalation and dermal contact for adults (18+ years), as individuals under 18 years of age are unlikely to use these products.
- Personal care products: dermal contact. Although PFAS occur infrequently in personal care products (KEMI, 2021), if present, they may occur in significant amounts. Within the category of personal care products, PFAS occur most frequently in decorative cosmetics, though other product types such as skin care products

may also contain PFAS. Exposure is calculated only for adults, since no product use pattern data were available for adolescents.

By including both baseline and subgroup scenarios for relevant non-food sources, the report provides an assessment of typical and potential high PFAS exposure, and enables comparison with exposure via food and drinking water. Table 1 provides an overview of the included sources, the relevant exposure routes, the scenario type (baseline or subgroup), and the age groups for which exposure was assessed.

Table 1 Overview of included sources, routes and scenarios.

Source	**Route(s)**	**Scenario**	**Age group(s)**
House dust	Ingestion	Baseline	All
Indoor air	Inhalation	Baseline	All
Outdoor clothing	Dermal	Baseline	All
Outdoor clothing	Ingestion (mouthing)	Subgroup	Young children (1-6 years)
Impregnation spray	Inhalation and dermal	Subgroup	Adults (18+ years)
Personal care products	Dermal	Subgroup	Adults (18+ years)

2.4 Exposure calculation methods

Exposure calculations were performed for the two scenarios and different age groups as identified in section 2.3 (see Table 1). To estimate realistic exposures for all five non-food sources in the general population, it would be preferable to use a person-oriented, probabilistic model, such as the Probabilistic Aggregate Consumer Exposure Model (PACEM). Probabilistic models can take into account realistic patterns of contact with sources and variability in contacts within the population. However, only detailed and representative product use data are currently available and implemented in PACEM for personal care products. For the other non-food sources (house dust, indoor air, outdoor clothing and impregnation sprays), such product use or contact pattern data are not available. Therefore, two different approaches were used:

- For house dust, indoor air, outdoor clothing, and impregnation sprays, exposure was calculated using mathematical equations based on established models from the literature. These calculations combine measured concentrations with generic input parameters, such as body weight, inhalation rate, and skin surface area to account for differences in physiology between age groups. Route-specific absorption fractions were applied to adjust for oral, inhalation, and dermal exposure. Concentrations, detailed equations and input parameters for each source are provided in sections 2.5 and 2.6.
- For personal care products, exposure was estimated using PACEM. PACEM simulates a distribution of exposure based on the product use patterns from the survey of Biesterbos et al. (2013), combined with concentration distributions constructed from measurements of individual products, and aggregates exposure

across multiple product types (see Section 2.7). In this report, PACEM was used to estimate exposure for the subgroup of adults (see Section 2.3), and was restricted to the exposed-only group, i.e. adults who use one or more PFAS-containing personal care products. Consequently, the reported exposure percentiles for personal care products (e.g. P50, P95) describe exposure among exposed users only.

These two approaches reflect the current availability of product use and contact pattern data. The results of formula-based and PACEM-based calculations differ in their level of conservatism and are not directly comparable.

2.5 Concentration data for house dust, indoor air, outdoor clothing and impregnation sprays

For the four non-food exposure sources house dust, indoor air, outdoor clothing and impregnation sprays, concentration data were systematically collected from the literature using searches in PubMed and Scopus (see Appendix 1 for search strategy and included studies). Only studies with measured concentrations and conducted in Europe were included. In total, 19 studies were included for house dust, 10 for indoor air, 7 for outdoor clothing and 7 for impregnation sprays. No studies specifically describing the Dutch situation were identified via the systematic search. However, for PFAS in house dust, measurements were available from three Dutch homes reported in a consultancy report (Arcadis, 2021). In addition, European house dust data and European concentration data for the other three sources were used and considered representative for the Netherlands. For house dust and indoor air, where local factors may play a larger role, this assumption is more uncertain than for products, which are typically manufactured internationally and widely available in stores and online.

We also assumed that all selected PFAS co-existed in the four non-food sources. This approach accounts for the fact that not all studies measured the same set of PFAS, and to provide a more comprehensive estimate of total PFAS exposure. This assumption is likely more accurate for house dust and indoor air, where multiple PFAS can be present simultaneously, than for products. In products, if one PFAS is used, another may not be present due to differences in manufacturing.

As only a few studies with PFAS concentrations were available for these four non-food sources, we cannot reliably estimate concentration distributions. We therefore estimated a single conservative point value for each concentration. The P75 was chosen for the four sources as it represents a higher concentration to which people can be exposed, while minimizing the influence of extreme outliers. This approach is particularly relevant since people may be loyal to specific products and brands; if the formulation of these products does not change, or if the PFAS originate from stable sources within the home (such as treated furniture, carpets, or textiles), individuals may be exposed to the same PFAS concentrations over an extended period.

For outdoor clothing and impregnation sprays, most studies report individual measurements. Analytical biases are relatively small because the products can be sampled directly. We therefore assign equal weight to each measured product and determine the P75 directly from the individual measurements. Details of the derivation of the P75 concentrations are provided in Appendix 1.

For indoor air and house dust, significant biases can be introduced by differences in sampling protocols, analytical methods, and local conditions. We therefore give every study equal weight, irrespective of the number of residences included. Most studies report only summary statistics (typically median and maximum concentrations), and the underlying measurement distributions are not available. This makes it difficult to reliably estimate a pooled concentration distribution or directly determine a conservative concentration. To avoid strong assumptions about these distributions, we applied the following approximation: we pooled the medians and maximums of the studies and determined their P75 using the *nearest-rank method*. This nearest-rank P75 of the pooled medians and maximums is a conservative estimate of the (conventional) P75 of the pooled underlying study data. For more details about this calculation, see Appendix 1.

Via the four sources, people are exposed to multiple PFAS simultaneously, which differ in toxic potency. To sum the exposure to all these PFAS per source, we used the relative potency factor (RPF) method, applying RPFs to the measured concentrations of individual PFAS to estimate a combined exposure expressed in PFOA-equivalents (PEQ). This approach allows the combined exposure to multiple PFAS to be expressed as a single, toxicity-weighted exposure for each sources. Only PFAS with an oral RPF were included in this sum exposure (Table 2). In the underlying studies, a broader range of PFAS was measured; details on all measured PFAS per source can be found in Appendix 1.

The available oral RPFs are used to express combined, toxicity-weighted oral exposure to multiple PFAS (Bil et al., 2021; Smit et al., 2025). Route-specific RPFs for dermal and inhalation exposure are currently not available, and therefore, oral RPFs were also applied to sum PFAS exposure via the dermal and inhalation routes. This constitutes a methodological approximation and introduces uncertainty, because it assumes that the RPFs are similar across exposure routes, which may not hold if there are route-dependent differences in distribution, metabolism or excretion. To account for route-specific differences in absorption at the external exposure level, we applied route-specific absorption factions (Section 2.6.2). However, this does not remove the uncertainty related to the use of oral RPFs for non-oral routes. As a result, the PEQ-based estimates for dermal and inhalation exposure should be interpreted with caution and should primarily be used to screen potential sources for which the exposure may need to be refined further, rather than for quantitative risk assessment.

Table 2 Oral relative potency factors (RPFs) for PFAS for the assessment of oral exposure to PFAS.

PFAS name	Abbreviation	RPF	Reference
Perfluorobutane sulfonic acid	PFBS	0.001	(Bil et al., 2021)
Perfluoropentane sulfonic acid	PFPeS*	0.001≤RPF≤**0.6**	(Bil et al., 2021)
Perfluorohexane sulfonic acid	PFHxS	0.6	(Bil et al., 2021)
Perfluoroheptane sulfonic acid	PFHpS*	0.6≤RPF≤**2**	(Bil et al., 2021)
Perfluorooctane sulfonic acid	PFOS	2	(Bil et al., 2021)
Perfluorodecane sulfonic acid	PFDS	2	(Bil et al., 2021)
Perfluorobutanoic acid	PFBA	0.05	(Bil et al., 2021)
Pentafluoropropanoic acid	PFPrA	0.01	(Smit et al., 2025)
Perfluoropentanoic acid	PFPeA*	0.01≤RPF≤**0.05**	(Bil et al., 2021)
Perfluorohexanoic acid	PFHxA	0.01	(Bil et al., 2021)
Perfluoroheptanoic acid	PFHpA*	0.01≤RPF≤**1**	(Bil et al., 2021)
Perfluorooctanoic acid	PFOA	1	(Bil et al., 2021)
Perfluorononanoic acid	PFNA	10	(Bil et al., 2021)
Perfluorodecanoic acid	PFDA*	4≤RPF≤**10**	(Bil et al., 2021)
Perfluoroundecanoic acid	PFUnDA	4	(Bil et al., 2021)
Perfluorododecanoic acid	PFDoDA	3	(Bil et al., 2021)
Perfluorotridecanoic acid	PFTrDA*	0.3≤RPF≤**3**	(Bil et al., 2021)
Perfluorotetradecanoic acid	PFTeDA	0.3	(Bil et al., 2021)
Perfluorohexadecanoic acid	PFHxDA	0.02	(Bil et al., 2021)
Perfluorooctadecanoic acid	PFODA	0.02	(Bil et al., 2021)
Tetrafluoro-2-(heptafluoropropoxy)propanoic acid	HFPO-DA (GenX)	0.06	(Bil et al., 2021)
3-H-perfluoro-3-[(3-methoxy-propoxy)propanoic acid]	DONA	0.03	(Bil et al., 2021)
6:2 fluorotelomer alcohol	6:2 FTOH	0.02	(Bil et al., 2021)
8:2 fluorotelomer alcohol	8:2 FTOH	0.04	(Bil et al., 2021)
Trifluoroacetic acid	TFA	0.002	(RIVM, 2023)
Tetrafluoropropanoic acid	TeFPrA	0.01	(Smit et al., 2025)

* RPFs based on read-across; values in bold are used in this report.

The resulting total high-end estimates of the P75 concentrations (in PEQ) used in the exposure calculations for house dust and indoor air were 945 ng PEQ/g and 2.1 ng PEQ/m³, respectively. The resulting P75 concentration (in PEQ) for outdoor clothing and impregnation sprays were 10 µg PEQ/m² and 13 µg PEQ/m², respectively (see Table 3).

Table 3 Overview of included studies and calculated total PFAS concentrations per source used in exposure assessment.

Source	Number of included studies	Total concentration used in calculations (PEQ)
House dust	14	945 ng/g
Indoor air	8	2.2 ng/m³
Outdoor clothing	7	10 µg/m²
Impregnation sprays	7	13 µg/g

2.6 Exposure calculations and input parameters for house dust, indoor air, outdoor clothing and impregnation sprays

2.6.1 General physiological parameters

For the exposure calculation, the general physiological parameters as listed in Table 4 were used, unless otherwise specified. Details on the derivation of age group averages and assumptions are provided in Appendix 2.

Table 4 Age-specific general parameters used in exposure calculations.

Parameter	1-5 years	6-10 years	11-17 years	18+ years	Reference
Body weight	12.3 kg	22.3 kg	43.9 kg	68.8 kg	(Te Biesebeek et al., 2014)
Arms skin surface area	8 dm^2	13 dm^2	20 dm^2	26 dm^2	(Te Biesebeek et al., 2014)
Trunk and arms skin surface area				92 dm^2	(Te Biesebeek et al., 2014)
Hand skin surface area				5 dm^2	(Te Biesebeek et al., 2014)
Inhalation rate	11.4 m^3/day	11.0 m^3/day	12.5 m^3/day	16.7 m^3/day	(U.S. EPA, 2011b)

2.6.2 Absorption factions

To ensure comparability between sources and exposure routes (as selected in Section 2.2), all external exposure values were adjusted using route-specific absorption fractions. This approach takes into account that not all externally encountered PFAS are absorbed into the body, thereby allowing a consistent assessment of absorbed doses. Absorption was set at 100% for oral and inhalation routes. For dermal exposure, PFAS-specific dermal absorption fractions (DAFs) were applied (see Table 5 and Appendix 2 for details). For some PFAS, DAFs were based directly on available experimental data; for others, DAFs were extrapolated from structurally similar PFAS (same functional group and comparable chain length), acknowledging that experimental data are limited.

Table 5 Dermal absorption fractions (DAFs) for PFAS used in this report.

PFAS	DAF (%)	Reference
PFBA	49	Wang et al. 2025
PFPeA	32	Wang et al. 2025
PFHxA	23	Wang et al. 2025
HFPO-DA	21	Wang et al. 2025
PFHpA	17	Wang et al. 2025
PFOA	13	Wang et al. 2025
PFNA	10	Wang et al. 2025
PFDA	4.1	Wang et al. 2025
PFBS	49	Ragnarsdottir et al. 2024
PFPeS	44	Ragnarsdottir et al. 2024
PFHxS	22	Ragnarsdottir et al. 2024
PFHpS	6.4	Ragnarsdottir et al. 2024
PFOS	1.3	Ragnarsdottir et al. 2024
PFPrA	49	Assign DAF from PFBA (C4)
PFUnDA	4.1	Assign DAF from PFDA (C10)

PFAS	DAF (%)	Reference
PFDoDA	4.1	Assign DAF from PFDA (C10)
PFTrDA	4.1	Assign DAF from PFDA (C10)
PFTeDA	4.1	Assign DAF from PFDA (C10)
PFHxDA	4.1	Assign DAF from PFDA (C10)
PFODA	4.1	Assign DAF from PFDA (C10)
PFDS	1.3	Assign DAF from PFOS (C8)
DONA	10	Assign DAF from PFNA (C9)
6:2 FTOH	23	Assign DAF from PFHxA (C6)
8:2 FTOH	13	Assign DAF from PFOA (C8)
TFA	49	Assign DAF from PFBA (C4)
TeFPrA	32	Assign DAF from PFPeA (C5)

DAF values are rounded with two significant figures. Calculations were performed using the original values reported in Wang et al. (2025) and Ragnarsdottir et al. (2024). For some PFAS, DAFs were assigned based on structurally similar PFAS. See Appendix 2 for rationale.

2.6.3 *Exposure calculation equations and input parameters per source*

For each source and exposure route, the Estimated Daily Intake (EDI) was calculated using the equations and parameters below. Route-specific absorption fractions were applied (see Section 2.6.2), so in this report EDI represents the absorbed daily dose per kilogram body weight. All EDI values were multiplied by 7 to obtain weekly exposure, so that the non-food exposure estimates can be expressed on the same weekly timescale as the existing exposure estimates for food and drinking water.

House dust (oral)

The EDI of PFAS via ingestion of house dust was calculated using the equation from Poothong et al. (2020):

$$EDI_{oral,dust} = \frac{C_{dust} * Q_{dust} * F_{uptake-GIT}}{bw}$$

Where:

- C_{dust} = PFAS concentration in house dust (ng PEQ/g; see Table 3)
- Q_{dust} = daily dust intake (g/day; 100 mg/day for children 2-6 years, 50 mg/day for persons ≥ 7 years (Oomen et al., 2008; Otte et al., 2001))
- $F_{uptake-GIT}$ = uptake fraction of PFAS via the gastrointestinal tract (set to 100%, assuming that PFAS in house dust are fully bioaccessible and that all bioaccessible PFAS are absorbed)
- bw = body weight (kg; see Table 4)

Indoor air (inhalation)

The EDI of PFAS via inhalation of indoor air was calculated using the equation from Poothong et al. (2020):

$$EDI_{inhalation,indoor\ air} = \frac{C_{air} * Q_{inh} * F_{uptake-lung}}{bw}$$

Where:

- C_{air} = PFAS concentration in indoor air (ng PEQ/m³, see Table 3)
- Q_{inh} = daily inhalation volume (m³/day; age-specific, see Table 4)
- $F_{uptake\text{-}lung}$ = fraction of PFAS absorbed via the lungs (set to 100%)
- bw = body weight (kg; age-specific, see Table 4)

Outdoor jacket (dermal)
The EDI of PFAS via dermal contact with outdoor jackets was calculated in two steps: (1) transfer of PFAS from the treated textile to perspiration on the skin (migration rate), and (2) absorption of PFAS through the skin (Washburn et al., 2005). The calculation was performed using the equation from Xia et al. (2022):

$$EDI_{dermal,jackets} = \frac{(\Sigma[C_{PFAS,i} * F_{uptake-dermal,i}]) * SA_j * F_{mig} * T}{bw}$$

Where:

- $C_{PFAS,i}$ = concentration of $PFAS_i$ in outdoor clothing (ng PEQ/cm²; see Table 3)
- $F_{uptake\text{-}dermal,i}$ = dermal absorption fraction of $PFAS_i$ (%; Table 5)
- SA_j = skin surface area exposed to jackets (cm²; see Table 4)
- F_{mig} = migration rate of PFAS from textile to skin (0.0000417 h^{-1}, based on 0.001 day^{-1}; used as a generic migration rate for treated textiles, derived from Xia et al. (2022), who applied the BfR (2012) exposure model in a screening context, because PFAS-specific migration data are not available)
- T = contact duration (12 hours per day (Washburn et al., 2005))
- bw = body weight (kg; see Table 4)

Impregnation sprays (inhalation and dermal)
The EDI of PFAS via inhalation and dermal contact from impregnation sprays was calculated for adults only, as children are not expected to use these products.

The inhalation calculation was performed using the two-compartment (near field/far field) model from Trudel et al. (2008):

$$EDI_{inhalation,spray} = \frac{C_{spray} * f_{spray} * t_{spray} * r_{spray} * F_{resp} * Q_{inh}}{bw} * \left(\frac{t_{spray_{NF}}}{V_{NF}} + \frac{t_{spray_{FF}}}{V_{FF}}\right) * F_{uptake-lung}$$

Where:

- C_{spray} = PFAS concentration in spray (ng PEQ/g; see Table 3)
- f_{spray} = frequency of spraying (event/day; calculated as 3 events/year ÷ 365 days = 0.0082 events/day; P75 (U.S. EPA, 2011a).
- t_{spray} = spray duration per event (1.6 minutes/event or 0.026 hours/event; calculated by dividing the P75 annual product use (340.2 g/year) by the P75 frequency of use (3 events/year) to obtain the amount per event (113.4 g/event), and then dividing by the mass generation rate [72 g/minute]) (U.S. EPA, 2011a; Cieszynski et al., 2022)

- r_{spray} = mass generation rate during spraying (72 g/minute (Cieszynski et al., 2022))
- F_{resp} = fraction of product that becomes airborne and is available for inhalation (set to 100% because the RPF-weighted concentration is dominated by volatile PFAS)
- Q_{inh} = inhalation rate (light exercise 1.49 m³/h (Te Biesebeek et al., 2014))
- $t_{sprayNF}$, V_{NF} = exposure time (0.026 h) and volume (1 m³) for near field
- $t_{sprayFF}$, V_{FF} = exposure time (0.3 h; P75 (U.S. EPA, 2011b)) and volume (50 m³) for far field
- $F_{uptake\text{-}lung}$ = fraction of PFAS absorbed via the lungs (set to 100%)
- bw = body weight (kg; see Table 4)

For the dermal calculation, the model structure is analogous to the inhalation scenario, but with parameters adapted to dermal contact. The uptake fraction is PFAS-specific (see Section 2.6.2). The respirable fraction is replaced by a dermal contact fraction, and the inhalation rates, volumes, and near-/far field exposure times are omitted. The resulting model is equivalent to the dermal exposure scenario for shoe polish spray described in the ConsExpo Cleaning Products Fact Sheet (Meesters et al., 2018), though the parameters are estimated from different data.
We assume that the jacket is held with one hand during spraying and that the hand and jacket receive the same amount of spray per unit surface area. The exposure calculation was performed with the following equation:

$$EDI_{dermal,spray} = \frac{\Sigma[C_{spray,i} * F_{uptake-dermal,i}] * f_{spray} * t_{spray} * r_{spray} * F_{dermal}}{bw}$$

Where:

- $C_{spray,i}$ = concentration of $PFAS_i$ in spray (ng PEQ/g; see Table 3)
- $F_{uptake\text{-}dermal,i}$ = dermal absorption fraction of $PFAS_i$ (%; Table 5)
- f_{spray} = frequency of spraying (event/day; calculated as 3 events/year ÷ 365 days = 0.0082 events/day; P75 (U.S. EPA, 2011a).
- t_{spray} = spray duration per event (1.6 minutes/event or 0.026 hours/event; derived as described above)
- r_{spray} = mass generation rate during spraying (72 g/minute (Cieszynski et al., 2022))
- F_{dermal} = fraction of sprayed volume contacting the skin, estimated as the surface area of one adult hand relative to the combined surface area of the jacket and hand. The jacket surface area was approximated by the skin surface area of an adult's trunk and arms. This yields an F_{dermal} of 0.0466 (calculated by dividing the surface area of hand (450 cm^2) by the combined surface area of trunk, arms and hand (9200 cm^2 + 450 cm^2; see Table 4)
- bw = body weight (kg; see Table 4)

In this report, impregnation sprays are treated as an infrequent but potentially high-exposure source contributing to long-term PFAS body burden, rather than as an acute exposure scenario. Therefore, the event-based exposure calculated with the near-field/far-field model is averaged to an EDI and expressed as weekly exposure, consistent with the other sources. This approach is not suitable for acute risk assessment, but is considered appropriate for screening the contribution of impregnation sprays to the total PFAS exposure.

The near-field/far-field model used here (Trudel et al., 2008) is a simplified representation of indoor air dynamics and does not explicitly account for ventilation or deposition of aerosols and vapours. Combined with the assumption that all PFAS in product becomes airborne and available for inhalation (F_{resp} = 100%) and fixed exposure times in the near and far field, this likely results in a conservative estimate of inhalation exposure. The precise degree of conservatism is uncertain due to the limited empirical data on spray behaviour and indoor air removal processes.

We also note that while the inhalation calculation assumes all PFAS is released into the air, the dermal calculation assumes that all PFAS is deposited on the jacket and hand. Summing these exposures would therefore double-count PFAS. In reality, the total exposure will lie somewhere between the calculated inhalation and dermal exposures, but it cannot be quantified because the fraction of PFAS that evaporates versus deposits is unknown. As a conservative assumption, the highest exposure of the two can be taken as the total exposure.

Outdoor jacket (mouthing)
The EDI of PFAS via mouthing of outdoor jackets was calculated for children aged 1-5 years, as older children are not expected to exhibit mouthing behaviour (ter Burg et al., 2008). The calculation was performed using a modified equation from Washburn et al. (2005). This scenario assumes jackets that already contain PFAS at the time of purchase, and that this PFAS can be replenished by occasional treatment with impregnation sprays. We assume that all PFAS present in the mouthed areas is transferred over repeated mouthing events. As the PFAS concentration after treatment is unknown, it is assumed to be equal to the original PFAS concentration. The mouthing exposure was calculated with:

$$EDI_{mouthing,jackets} = \frac{C_{jackets} * FT * f_{spray} * A_{mouthing} * F_{uptake-GIT}}{bw}$$

Where:

- $C_{jackets}$ = concentration of PFAS in outdoor jackets (ng PEQ/cm²; see Table 3)
- FT = maximum fraction transferred via mouthing per impregnation treatment event (set to 1 event^{-1})
- f_{spray} = frequency of replenishment of the PFAS concentration through impregnation spray treatment (0.0082 events/day; P75 (U.S. EPA, 2011b))

- $A_{mouthing}$ = area of jacket relevant for mouthing (40 cm²; four areas of 10 cm² each (Van Engelen et al., 2008), i.e. both sleeve ends and two front panels near the zipper)
- $F_{uptake\text{-}GIT}$ = uptake fraction via the gastrointestinal tract (set to 100%)
- bw = body weight (kg; see Table 4)

2.7 Concentrations and exposure calculations for personal care products

2.7.1 Concentrations data approach (PACEM)

Relevant concentration data for personal care products were systematically collected from the scientific literature using searches in PubMed and Scopus (see Appendix 3 for search strategy and included studies). Only studies with measured concentrations and conducted in Europe were included. In total, 3 studies were included. As for the other four sources, European data were considered representative for the Netherlands. For the exposure assessment, the measured concentrations of PFAS in each product were converted to PEQ by multiplying each concentration by its RPF (see Section 2.5 and Table 2) and the PFAS-specific DAF (see Section 2.6.2 and Table 5). The resulting DAF-converted total PEQ concentrations were used to construct a concentration input distribution for each product category. PACEM uses these distributions as input to simulate the exposure distribution in the population. Table 6 provides an overview of the personal care product types for which concentration data were available in the literature, and shows how these were assigned to PACEM product categories for adults. Only product categories with measured concentrations of PFAS evaluated in this report were included. A detailed mapping at product level is provided in Appendix 3.

Table 6 PACEM product categories for which measured concentration data were available and therefore included in the exposure assessment for personal care products.

Reference	**PACEM personal care product category**
Schultes et al. (2018)	Face day/night cream, liquid foundation, eye pencil, eye shadow, rouge powder
Brinch et al. (2018)	Face day/night cream, body lotion milk, eye pencil, liquid foundation, rouge powder, eye shadow
KEMI (2021)	Eye shadow, liquid foundation, lip pencil

2.7.2 Exposure calculation approach (PACEM)

PFAS exposure via personal care products for adults was estimated using PACEM (see Section 2.4). Because the product concentrations were already converted to absorbed PEQ values by applying RPFs and DAFs in the data processing step (see Section 2.7.1), no additional absorption factors were applied in PACEM, and the model's exposure factors were set to 1 (see Appendix 3 for the full list of PACEM input parameters).

2.7.3 Regulatory scenarios

Since the concentrations used in this report for personal care products were collected between 2018 and 2021, several EU regulations have entered into force that limit the PFAS concentrations allowed in these

products. Calculations based on these original concentrations will therefore primarily reflect historical exposure.

As the main aim of this report is to assess the potential contribution of non-food sources relative to food and drinking water to the total exposure to PFAS, we considered it useful to evaluate PFAS exposure from personal care products under two sets of concentration conditions: (i) a historical scenario, based on the measured concentrations, and (ii) an indicative present-day scenario in which the concentrations in personal care products were limited at the current regulatory limits (as of May 2026). In this alternative scenario, measured concentrations exceeding these limits were adjusted downwards to comply with them. This indicative present-day scenario thus provides a conservative approximation of present-day exposure from personal care products under current regulation (details of the regulatory-cap adjustments are provided in Appendix 3).

2.7.4 PACEM input parameters

Based on the DAF-converted total PEQ concentrations in personal care products belonging to the same product category, we calculated a concentration input distribution for PACEM. If multiple non-zero concentrations were available, a truncated log-normal distribution was used; in case of only one non-zero concentration, a trapezoidal distribution was applied. Occurrence percentages were based on measured PFAS concentrations and general occurrence data from KEMI (2021). Only dermal exposure was considered, as all products were leave-on (intended to remain on the skin and not to be rinsed off). Full input parameters, distribution calculations, and concentrations are provided in Appendix 3.

PACEM was run using 1000 iterations for both sets of concentration conditions (see Section 2.7.3). The exposed fraction of the adult population was calculated based on sample sizes from exposed-only and exposed-and-non-exposed analyses. Results were multiplied by 7 days/week and converted from mg PEQ to ng PEQ.

2.8 Qualitative comparison exposure estimates of non-food sources with food and drinking water

In this report, exposure estimates from non-food exposure sources (baseline and subgroup scenarios) are compared with the previously calculated exposure via food and drinking water (Schepens et al., 2023). No new exposure calculations were performed for food and drinking water.

Given the differences in data availability, modelling approaches and underlying assumptions, this comparison is qualitative rather than quantitative. The non-food exposure estimates are intended as conservative screening values to identify sources that warrant further investigation and refinement of the exposure assessment, rather than to support strict quantitative ranking of sources. The methodological differences and associated uncertainties relevant for this comparison are discussed in more detail in Chapter 4.

3 Exposure from non-food sources

3.1 Baseline scenario

The baseline scenario considers exposure to PFAS from sources relevant to the general population: house dust (oral), indoor air (inhalation), and dermal contact with outdoor clothing (see Table 1). Exposure estimates reflect only PFAS for which a RPF was available (see Section 2.5). Table 7 summarizes the estimated weekly exposure (ng PEQ/kg bw/week) for each source and age group. These exposure estimates reflect a highly exposed individual based on conservative assumptions (see Section 2.5 and 2.6).

Within the current screening approach, ingestion of house dust emerges as the most prominent non-food source of PFAS exposure for all age groups, especially for young children (see Table 7). The exposure estimate decreases with age, which reflects both the assumed age-dependent dust ingestion rates and lower body weights in young children. Dust ingestion rates include total dust intake, encompassing ingestion, inhalation, and hand-to-mouth contact (Oomen et al., 2008). In the exposure estimate, PFAS in house dust is assumed to be fully bioaccessible and fully absorbed (see Section 2.6.3), which represents a worst-case assumption and likely overestimates exposure. Additionally, the assumption that all PFAS co-existed at the high-end of the P75 concentrations may further overestimate exposure, because in reality not all PFAS need to be present in the same sample and, if present, they are unlikely to all occur simultaneously at their high-end of the P75 concentrations. These assumptions characterise a highly exposed individual rather than the average population. Typical exposures are therefore expected to be lower than the estimates presented here.

Exposure via indoor air is lower but still relevant, particularly for young children (14 ng PEQ/kg bw/week). Adolescents and adults are exposed to slightly less (4.3 ng and 3.7 ng PEQ/kg bw/week, respectively). The estimate for indoor air is based on conservative assumptions regarding time spent indoors (100% of the time) and daily average inhalation rate. These assumptions are likely to overestimate exposure. However, only a subset of volatile PFAS (6:2 FTOH and 8:2 FTOH) were included in the calculations, meaning actual exposure may be higher if exposure to additional volatile PFAS, such as 10:2 FTOH or MeFOSE, occurs.

Dermal exposure from wearing outdoor jackets contributes minimally to total PFAS exposure across all age groups (ranging from 0.016 ng PEQ/kg bw/week in young children to 0.009 ng PEQ/kg bw/week in adults).

Table 7 Estimated PFAS exposure (ng PEQ/kg bw/week) for baseline scenario by age group and exposure source.

Source	Exposure route	Child (1-5)	Child (6-10)	Adolescent (11-17)	Adult (18+)
House dust	Oral	54	29.7	7.5	4.8
Indoor air	Inhalation	14	7.4	4.3	3.7
Outdoor jacket	Dermal	0.016	0.014	0.011	0.009

3.2 Subgroup scenarios

Subgroup scenarios focus on specific behaviours or product use that may lead to higher PFAS exposure for limited fractions of the population (see Table 1).

3.2.1 Impregnation sprays and mouthing of outdoor jackets

For impregnation sprays and mouthing of outdoor jackets, the resulting exposure estimates reflect upper-bound scenarios for individuals engaging in these specific behaviours with products that contain PFAS. They do not represent the average or P95 exposure for the general population. Table 8 summarizes the estimated weekly exposure for impregnation sprays and mouthing of outdoor clothing for relevant age groups.

During use of impregnation sprays, adults may be exposed to PFAS via inhalation of the spray or dermally through the hand holding the jacket to be sprayed. Exposure via inhalation was estimated at 59 ng PEQ/kg bw/week, and via dermal contact at 9.4 ng PEQ/kg bw/week (see Table 8). These estimates overestimate actual exposure, primarily due to the assumption that all PFAS in the sprayed volume becomes airborne and is inhaled, as well as additional conservative assumptions (see Section 2.6), such as prolonged time indoors after spraying and lack of ventilation. In reality, many people apply sprays outdoors or ventilate after use. The dermal scenario assumes maximal hand coverage with spray residue, which is unlikely in typical use. Adding these two exposure estimates would double-count the same PFAS, because both scenarios are based on the full sprayed PFAS mass. For the qualitative comparisons in this report, we therefore use the most conservative assumption and take the higher of the two estimates, i.e. the inhalation exposure, as the total exposure from impregnation sprays. It should be noted, however, that only two volatile PFAS with available RPF (6:2 FTOH and 8:2 FTOH) were included in the calculations. Other volatile PFAS detected in impregnation sprays (such as 10:2 FTOH or MeFOSE; see Appendix 1 and Table A5) may contribute to actual exposure but cannot yet be quantified in PEQ terms due to the lack of RPFs for these PFAS.

For young children (1–5 years), oral exposure from mouthing outdoor jackets was estimated at 0.3 ng PEQ/kg bw/week. This scenario assumes jackets that already contain PFAS at the time of purchase and that the PFAS load is fully transferred and ingested from the original treatment and after each subsequent impregnation spraying event. This likely overestimates actual exposure, as only a fraction of the PFAS

present in the textile will be transferred and ingested per mouthing event. In reality, mouthing behaviour varies widely between children, so actual exposure may be lower or higher for individuals with less or more intensive mouthing behaviour.

Table 8 Estimated PFAS exposure (ng PEQ/kg bw/week) for subgroup scenarios by age group and exposure source.

Source	Exposure route	Child (1-5)	Child (6-10)	Adolescent (11-17)	Adult (18+)
Impregnation sprays	Inhalation	-	-	-	59
Impregnation sprays	Dermal	-	-	-	9.4
Outdoor jacket	Oral	0.3	-	-	-

3.2.2 *Personal care products*

For personal care products, exposure is estimated using PACEM (see Section 2.4). In this report, personal care products are treated as an additional subgroup scenario, and the model output is restricted to adults using PFAS-containing personal care products. Only a limited set of product categories for which measured PFAS concentrations were available was included (face creams, liquid foundations, eye pencils, eye shadows, rouge powders, body lotions and lip pencils; see Table 6). Within these categories, not all products contain PFAS (KEMI, 2021). Consequently, the reported exposure percentiles (e.g. P50, P95) describe exposure among exposed users only and are higher than corresponding population-wide percentiles, making them suitable as screening values, especially in the context of brand loyalty, i.e. long-term use of the same PFAS-containing products.

Table 9 presents the weekly exposure estimates for personal care products, including the exposed fraction of the adult population and, for the exposed group, the median (P50) and high-end (P95) values for males, females, and all adults. Exposure was estimated for two sets of concentration data: historical and limited (see Section 2.7.3).

For adults using PFAS-containing personal care products, exposure was highest among male users (P95: 10-16 ng PEQ/kg bw/week, depending on limited or historical concentrations). However, only a small fraction of adults is exposed via such products (about 0.15% of men and 4.6% of women; 2.7% of all adults). These results therefore point to a potential concern for a small, specific subgroup of adult users, and indicate that personal care products are unlikely to play a substantial role in PFAS exposure at the population level.

It is important to note that these estimates reflect more historical exposure, as recent EU regulations limiting PFAS concentrations in personal care products are not fully reflected in the available concentration data. The cosmetics industry has committed to phasing out PFAS in personal care products by the end of 2025.[2] After this, PFAS may still be present in products already on the market or at people's

[2] https://www.ncv-cosmetica.nl/actueel/actieprogramma-pfas-cosmetica

homes, but exposure from this source is expected to further decrease in the near future.

Table 9 PFAS exposure via personal care products of exposed persons (ng PEQ/kg bw/week) calculated with historical and limited concentrations.

Population	**Exposed fraction (%)**	**P50**	**P95**
	Historical concentrations		
Male adults	0.15	0.16	16
Female adults	4.6	0.0025	13
All adults	2.7	0.0029	11
	Limited concentrations		
Male adults	0.15	0.14	10
Female adults	4.6	0.0012	3.1
All adults	2.7	0.0013	3.2

4 Comparison of non-food sources to food and drinking water

4.1 PFAS exposure from food and drinking water per age group

Exposure estimates for food and drinking water were obtained from Schepens et al. (2023). In that study, long-term, population-based exposures were calculated for the Dutch population aged 1–79 years by combining mean PFAS concentrations (in PEQ) in foods and drinking water with individual consumption data and body weights. This resulted in population-based distributions of average long-term intakes. To address analytical uncertainty regarding concentrations below the limit of detection (LOD) or quantification (LOQ), Schepens et al. (2023) considered a lower bound (LB) and upper bound (UB) scenario. In the LB scenario, concentrations reported below LOD/LOQ were assumed to be zero, and such concentrations were set at the LOD or LOQ in the UB scenario. The LB scenario is expected to underestimate the actual exposure to PFAS via food and drinking water, whereas the UB scenario is expected to overestimate the actual exposure. Based on an uncertainty analysis performed by Schepens et al. (2023), the LB estimates were considered to be closer to the actual exposure to PFAS than the UB estimates.

The LB estimates were used in the current report to compare the exposure via food and drinking water with the exposure via the non-food sources. These LB estimates are presented by age group in Table 10 to allow comparisons within the same age group. Schepens et al. (2023) also differentiated between drinking water made from groundwater or from surface water due to differences in PFAS concentrations.

Table 10 PFAS exposure via food and drinking water made from groundwater or from surface water (ng PEQ/kg bw/week).

Scenario	**PFAS exposure (ng PEQ/kg bw/week)**			
	Child (1-5)	**Child (6-10)**	**Adolescent (10-17)**	**Adult (18+)**
Food and drinking water from groundwater				
Median (P50)	5.8	3.9	2.3	3.2
High (P95)	18	12	7.3	12
Food and drinking water from surface water				
Median (P50)	7.3	5.4	3.5	4.5
High (P95)	20	13	8.2	13

4.2 Side-by-side comparison of exposure to PFAS via food and drinking water with non-food sources

For most non-food sources, scenario-based upper percentile estimates (typically P95 or higher) were calculated to represent highly exposed individuals (see Section 3.1 and 3.2.1). For personal care products, PACEM was used to simulate an exposure distribution for the exposed subpopulation only (see Section 3.2.2). In the comparison with food and drinking water, non-food exposure estimates should therefore be viewed as conservative screening values, while the food and drinking water

estimates represent average long-term exposure in the general population (see Section 4.1).

For the side-by-side comparison in Table 11, we use the LB P95 estimates for food and drinking water (see Table 10), i.e. the upper tail of the food and drinking water exposure distribution for the general population, rather than the median (P50). This is because the non-food scenarios represent upper-bound or high-exposure situations and are therefore more comparable to the upper tail than to median exposure levels. The robustness and limitations of this comparison are discussed in Section 4.3.

Table 11 Overview of high-end PFAS exposure estimates (ng PEQ/kg bw/week) from food and drinking water and non-food sources by age group (food and drinking water: LB P95 from Schepens et al., 2023; non-food: scenario-based high-end estimates, see Sections 3.1–3.2).

Source / scenario	**Child (1–5)**	**Child (6–10)**	**Adolescent (10–17)**	**Adult (18+)**
Long-term, population-based exposure				
Food + drinking water (P95)	18-20	12-13	7.3-8.2	12-13
Conservative, high-end screening exposure*				
House dust (oral, baseline)	54	29.7	7.5	4.8
Indoor air (inhalation, baseline)	14	7.4	4.3	3.7
Outdoor clothing (dermal, baseline)	0.016	0.014	0.011	0.009
Impregnation sprays (inhalation, subgroup)	–	–	–	59
Outdoor clothing (mouthing, subgroup)	0.3	–	–	–
Personal care products (dermal; subgroup) (P95, exposed)	–	–	–	3.2-11

* Conservative, high-end screening exposure: upper-bound estimates for individuals with relatively high exposure within the relevant scenario or subgroup, intended for screening and not as average population exposure.

Overall, Table 11 illustrates that, for the general population in the Netherlands, food and drinking water contribute most to long-term PFAS exposure, even when conservative, high-end scenarios for non-food sources are considered.

For young children (1–5 years), house dust and indoor air may also contribute substantially to PFAS exposure under the conservative exposure scenarios used in this report. As these are worst-case estimates rather than averages, further refinement of exposure estimates for these sources is warranted.

For adults, exposure to PFAS via impregnation sprays and personal care products could be relevant for users of PFAS-containing products. For personal care products this concerns only a small fraction of the population, as reflected in the modelled exposed fraction of about 2.7%

of adults (see Section 3.2.2). For impregnation sprays, no information is available on the size of the affected population or the market share of PFAS-containing sprays, making it unclear what proportion of adults is actually exposed. For the large majority of adults, other sources, such as house dust, indoor air and dermal exposure from outdoor clothing are expected to contribute less to total PFAS exposure than food and drinking water.

4.3 Uncertainty regarding comparison

The side-by-side comparison between the exposure to PFAS via food and drinking water and via non-food sources is subject to several uncertainties. These uncertainties arise both from the source-specific exposure estimates and from methodological differences in how exposure was assessed for different sources. As a result, the side-by-side comparison presented in Table 11 should be regarded as a qualitative, screening-level comparison rather than as a precise quantitative apportionment of total PFAS exposure.

4.3.1 Uncertainties in source-specific exposure estimates

For food and drinking water, exposure estimates were derived by Schepens et al. (2023) from mean concentrations of measured PFAS in food items and drinking water combined with individual consumption data and body weights. These estimates are relatively robust due to standardized analytical methods, systematic sampling, and the use of population-based dietary surveys. The methodological uncertainties were analysed in detail by Schepens et al. (2023). In that study, lower-bound estimates were judged to be closest to the actual exposure and are used here for comparison (see Section 4.1).

For non-food sources, concentration data were mainly derived from European studies with heterogeneous sampling strategies and analytical methods, as no systematic Dutch monitoring data are available. For house dust, indoor air, outdoor clothing and impregnation sprays, (high-end) P75 concentrations were derived from the available studies and used as input in the exposure calculations. In addition, it was assumed that the PFAS included in this report co-exist at these high-end concentrations in each source. This assumption may overestimate exposure, particularly for consumer products where PFAS formulations may differ between items and brands.

Exposure for most non-food sources was estimated using scenario-based formulas with generic, literature-based parameters (e.g. dust ingestion rates, time spent indoors, skin surface area), reflecting upper-bound scenarios rather than average population behaviour. For personal care products, exposure distributions were simulated using PACEM for the exposed subpopulation only, so reported percentiles (e.g. P50, P95) apply only to users of PFAS-containing products, not to the total population.

In summary, the non-food exposure estimates in this report are conservative, designed as screening values for highly exposed individuals. For most people, actual exposure is expected to be lower than these scenario-based estimates.

4.3.2 *Methodological uncertainties specific to the comparison*

Beyond source-specific uncertainties, there are methodological differences between the exposure assessments for food and drinking water and for non-food sources that affect the comparison itself, due to different exposure metrics and populations. For food and drinking water, exposure was assessed using population-based distributions of long-term average intake, derived from individual consumption data and mean concentrations. The P95 values used in Table 11 represent the upper tail of average long-term exposure in the general population.

For most non-food sources, exposure was assessed using scenario-based upper-bound calculations that describe high-exposure situations for individuals within the relevant scenario (baseline or subgroup), rather than population-wide exposure distributions. For personal care products, exposure distributions were modelled only for the exposed subpopulation, so the reported P50 and P95 values refer to users of PFAS-containing products, not the total population. These percentiles are therefore not directly comparable to those for food and drinking water.

To achieve a conservative comparison, the LB P95 values for food and drinking water were compared with high-end scenario estimates for non-food sources. However, the high-end estimates for non-food sources may exceed the P95 for the general population and are not based on fitted distributions. For personal care products, the P95 applies only to the exposed subpopulation. Consequently, Table 11 presents a side-by-side view of a population-based upper tail (food and drinking water) and conservative upper-bound scenarios for non-food sources, rather than directly comparable percentiles.

To combine exposure to multiple PFAS and enable comparison across sources, the RPF approach was used to express concentrations and exposures in PEQ. This introduces additional uncertainty as RPFs are available for a subset of PFAS and some are based on read-across rather than direct experimental data. The RPFs used in this report were derived for oral exposure. In the absence of route-specific RPFs for dermal and inhalation routes, oral RPFs were applied, assuming route-independent relative potency. This is a methodological approximation and may not fully capture route-specific differences in toxicokinetics and toxicity. As such, PEQ-based exposure estimates for dermal and inhalation routes should primarily be interpreted as indicative, toxicity-weighted screening values.

Finally, temporal patterns of exposure differ: food and drinking water typically contribute to continuous, daily exposure over long periods. Non-food sources may involve both continuous (house dust, indoor air) and episodic exposures (impregnation sprays, mouthing of outdoor clothing, personal care products) exposures. All exposures were converted to a weekly timescale for comparability, but the underlying temporal patterns and variability differ between sources.

5 Discussion, conclusions and recommendations

5.1 Discussion

5.1.1 Overall patterns and comparison with previous studies

Our results are broadly consistent with previous studies. For individuals who are not occupationally exposed or living near point sources of PFAS emissions, food and drinking water are generally considered to contribute more to PFAS exposure than house dust, indoor air, or consumer products (Consortium et al., 2023; de la Torre et al., 2019; De Silva et al., 2021; Poothong et al., 2020; Serrano et al., 2021).

For example, de la Torre et al. (2019) analysed house dust from Belgium, Italy and Spain, and reported that in a worst-case scenario, PFOA exposure via house dust ingestion could reach around half of dietary exposure for toddlers, whereas for adults dust ingestion contributed only a small fraction of dietary exposure. Poothong et al. (2020) found that, on average, about 90% of total PFAS exposure in Norwegian adults came from food and drinks, with house dust, indoor air and dermal contact making smaller but sometimes non-negligible contributions for some individuals. De Silva et al. (2021) similarly concluded that diet is the dominant exposure route for PFAS, but noted that multiple pathways, including indoor environment and consumer products, can be important in particular contexts.

Compared with these previous studies, the present report provides a first screening of exposure to multiple PFAS in a Dutch context using a single, toxicity-weighted concentration (PEQ) across sources and routes for prioritised non-food sources, and comparing these with existing exposure estimates for food and drinking water. The results confirm that food and drinking water are the main contributors to long-term PFAS exposure in the general population, while non-food sources can be relevant under specific conditions, especially for young children or specific user subgroups.

5.1.2 Interpretation of age-specific exposure and other sources

Because PFAS are persistent and accumulate in the body, the age-specific exposure estimates in this report should be interpreted as the contribution of each life stage to cumulative, long-term exposure rather than as separate short-term exposure situations. In this context, the relatively high exposures at young ages, for example via house dust and indoor air for toddlers under the conservative scenarios used in this report, can contribute to cumulative body burdens over the lifespan, even if exposures later in life are lower. This pattern is reflected in Table 11 where exposure via house dust is highest for young children and decreases with age, whereas exposure via food and drinking water remains the dominant and more stable contributor to long-term PFAS exposure across age groups.

The present report focuses on five prioritised non-food sources identified by Affourtit & Verbruggen (2025). Other potential exposure sources are summarised in that study. These sources were not prioritised mainly

because PFAS in those applications are typically tightly bound and hardly become available, concentrations are very low, there is little or no direct contact with the general population, or insufficient data are available to reliably assess their contribution.

5.1.3 *Methodological considerations and differences between sources*

Different methodological approaches were used for the various sources assessed in this report, and these differences affect how the results should be interpreted. For most non-food sources, exposure estimates are based on scenario-based upper-bound calculations using conservative input parameters (such as P75 concentrations and worst-case assumptions). These estimates represent the upper end of potential exposure for individuals within the relevant scenario, rather than population averages. For personal care products, an exposure distribution for the exposed subpopulation was simulated using PACEM, based on realistic product use patterns; the reported percentiles refer specifically to exposed users, not to the general population.

For food and drinking water, Schepens et al. (2023) combined mean PFAS concentrations in foods and drinking water with detailed individual consumption data from a dietary survey, resulting in population-based exposure distributions rather than scenario-based upper bounds. In the present report, the LB P95 values from Schepens et al. (2023) were used for comparison, as they reflect the upper tail of long-term average population exposure.

As described in Section 4.3, these methodological differences, together with limitations in the non-food concentration data, mean that the side-by-side comparison in Table 11 should be interpreted as a qualitative, screening-level comparison rather than as a precise quantitative apportionment of total PFAS exposure. The non-food exposure estimates are conservative and likely to overestimate exposure for most individuals; they are intended to identify which non-food sources warrant further data collection and refinement of exposure assessment.

5.1.4 *Future developments and implications*

Market developments and regulations are important factors influencing future PFAS exposure from non-food sources. Recent and upcoming EU restrictions on PFAS in consumer products, especially personal care products, are not yet fully reflected in the available concentration data in this report. Thus, the exposure estimates for personal care products presented here primarily reflect historical levels, and actual present-day exposure is likely lower and expected to further decrease. As PFAS use is further restricted or banned, exposure from non-food sources is expected to decline over time. This will also be the case for house dust and indoor air as fewer PFAS-containing products will be present in homes. The extent of exposure from the indoor environment is also influenced by household behaviour, such as cleaning frequency and ventilation. This report does not quantify the impact of such behaviours on PFAS exposure.

5.2 Conclusions

Based on the results and considerations discussed above, the main conclusions are as follows:

- Food and drinking water are the primary contributors to long-term PFAS exposure across all age groups in the general population.
- For young children, house dust and indoor air can be important contributors to long-term PFAS exposure. Under the conservative scenarios used in this report, estimated exposure via house dust and indoor air can reach or exceed exposure via food and drinking water. Further refinement of the exposure estimates for house dust and indoor air is needed for a more robust comparison with exposure via food and drinking water.
- For adults using specific personal care products, exposure from these products can contribute to long-term PFAS exposure. In this report, PFAS exposure from personal care products was assessed only for those product categories where measured PFAS concentrations were available (face creams, liquid foundations, eye pencils, eye shadows, rouge powders, body lotions and lip pencils). Within these categories, not all products contain PFAS. Given that this concerns a small exposed fraction of the population and that PFAS in personal care products are being phased out, this source does not warrant priority for further refinement of exposure estimates at the population level.
- For impregnation sprays, high-end scenario estimates indicate that exposure could contribute to the total PFAS exposure. However, the proportion of the adult population exposed at such levels and the market share of PFAS-containing sprays remain unknown. Further data on the prevalence of use and PFAS content in these products is warranted before firm conclusions can be drawn about their overall contribution to population exposure.
- Outdoor clothing is expected to contribute only minimally to total PFAS exposure across all age groups, and, based on the current results, do not warrant prioritisation for further refinement of exposure estimates.
- Uncertainties remain in both concentration data and exposure parameters, but the use of conservative assumptions to estimate the exposure via the non-food sources means that, for the scenarios considered, actual exposures via these sources are unlikely to be higher than estimated here.

5.3 Recommendations

This report provides a first, screening-level assessment of PFAS exposure from selected non-food sources and their contribution relative to food and drinking water. Because of limited data and the exploratory nature of the assessment, no firm conclusions can be drawn about the necessity of additional risk management measures for these non-food sources. Nevertheless, the findings and identified knowledge gaps support the following recommendations, grouped by their relevance for refining exposure estimates, improving methods, and supporting risk management.

Refinement of exposure estimates:

- Prioritise the collection of PFAS concentrations in house dust and indoor air in the Netherlands. Given the potential importance of house dust and indoor air as a source of PFAS exposure for young children and the current reliance on European data, targeted measurement campaigns in Dutch homes are warranted to refine exposure estimates for these two sources. Furthermore, refinement of ingestion rates (e.g. age-specific dust intake), exposure durations and other key parameters is needed to obtain more realistic, population-representative exposure estimates.
- Collect data on the prevalence of use, PFAS content, and user behaviour related to impregnation sprays. Given the high-end exposure estimates for this source, further research is needed to determine the actual number of exposed individuals and the typical levels of exposure in the general population. This will allow for a more accurate risk assessment and prioritisation.

Methodological improvements:

- Improve methods for integrating exposure estimates across multiple sources. The current report shows that exposure assessment methods and input data differ substantially between food and drinking water and non-food sources, making a direct quantitative comparison difficult. Further methodological development is recommended to harmonise exposure modelling across sources (e.g. food, drinking water, indoor environment, and consumer products) and routes (i.e. oral, dermal and inhalation), including probabilistic approaches for non-food sources.
- Conduct further research on dermal and inhalation absorption fractions for a broader range of PFAS and product types to reduce uncertainty in exposure assessments.
- Develop RPFs for dermal and inhalation exposure routes, and extend the set of oral RPFs for PFAS that are toxicologically relevant or frequently detected in important exposure sources. This would allow more comprehensive and harmonised mixture exposure assessments across sources and routes.

Regulation and measurements:

- Support and monitor implementation of existing and upcoming regulatory measures such as the EU restriction on PFAS in personal care products and the proposed broad European PFAS restriction under REACH.
- Continue monitoring of PFAS in consumer products and indoor environments in order to determine the impact of these measures on the contribution of these sources to PFAS exposure.
- Promote good cleaning and ventilation practices as easily applicable precautionary measures that may help reduce exposure via house dust and indoor air. The precise contribution of these exposure sources to total PFAS exposure, and the magnitude of the effect of such measures, have not yet been sufficiently quantified.
- Inform consumers that, if they wish to limit PFAS exposure, they can check whether personal care products contain PFAS by reviewing the ingredients listed on the label. PFAS in personal

care products are often recognizables by ingredient names containing “fluor”, “perfluoro” or “polyfluoro”. For impregnation sprays, label information is more limited and typically includes only substances relevant for the hazard classification of the mixture, so the presence of PFAS is not always visible on the label.

Acknowledgement

We would like to thank Anouk Fransen, who contributed to the project as a trainee, for her work in mapping relevant information and providing quantitative estimates. Her efforts laid a foundation for the analyses presented in this report.

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List of acronyms

Acronym	Full Term
6:2 FTOH	6:2 Fluorotelomer alcohol
8:2 FTOH	8:2 Fluorotelomer alcohol
DAF	Dermal absorption fraction
diPAP	Polyfluoroalkyl phosphoric acid diester
DONA	3-H-perfluoro-3-[(3-methoxy-propoxy)propanoic acid]
EDI	Estimated daily intake
EtFOSE	Ethyl perfluorooctane sulfonamidoethanol
GenX (HFPO-DA)	Tetrafluoro-2-(heptafluoropropoxy)propanoic acid
HFPO-DA (GenX)	Tetrafluoro-2-(heptafluoropropoxy)propanoic acid
LB	Lower bound
LOD	Limit of detection
MeFOSA	Methyl perfluorooctane sulfonamide
monoPAP	Polyfluoroalkyl phosphoric acid monoester
PEQ	PFOA-equivalents
PFCA	Perfluoroalkyl carboxylates
PFAS	Per- and polyfluoroalkyl substances
PFBA	Perfluorobutanoic acid
PFBS	Perfluorobutane sulfonic acid
PFCA	Perfluoroalkyl carboxylates
PFDA	Perfluorodecanoic acid
PFDoDA	Perfluorododecanoic acid
PFDS	Perfluorodecane sulfonic acid
PFHpA	Perfluoroheptanoic acid
PFHpS	Perfluoroheptane sulfonic acid
PFHxA	Perfluorohexanoic acid
PFHxDA	Perfluorohexadecanoic acid
PFHxS	Perfluorohexane sulfonic acid
PFNA	Perfluorononanoic acid
PFOA	Perfluorooctanoic acid
PFODA	Perfluorooctadecanoic acid
PFOS	Perfluorooctane sulfonic acid
PFPeA	Perfluoropentanoic acid
PFPeS	Perfluoropentane sulfonic acid
PFSA	Perfluoroalkyl sulfonates
PFTrDA	Perfluorotridecanoic acid
PFTeDA	Perfluorotetradecanoic acid
PFUnDA	Perfluoroundecanoic acid
PEQ	PFOA-equivalents
RPF	Relative potency factor
TFA	Trifluoroacetic acid
TeFPrA	Tetrafluoropropionic acid
UB	Upper bound

Appendix 1 PFAS concentration data for house dust, indoor air, outdoor clothing and impregnation sprays

House dust

Concentration data for PFAS in house dust were identified through systematic searches in PubMed and Scopus using the following search string: ("house dust" OR "indoor dust" OR "domestic dust") AND (PFAS OR "perfluoroalkyl substances" OR "polyfluoroalkyl substances" OR "perfluorinated compounds" OR "perfluorinated chemicals" OR "perfluorooctane sulfonate" OR "perfluorooctanoic acid" OR PFOS OR PFOA) AND (Europe OR "European Union" OR Austria OR Belgium OR Bulgaria OR Croatia OR Cyprus OR Czech OR Denmark OR Estonia OR Finland OR France OR Germany OR Greece OR Hungary OR Ireland OR Italy OR Latvia OR Lithuania OR Luxembourg OR Malta OR Netherlands OR Poland OR Portugal OR Romania OR Slovakia OR Slovenia OR Spain OR Sweden OR Switzerland OR Norway OR UK OR "United Kingdom") AND (concentration* OR level* OR amount* OR content*).

This search identified 17 studies reporting PFAS concentrations in house dust in Europe (Barreca et al., 2023; Bjorklund et al., 2009; D'Hollander et al., 2010; de la Torre et al., 2019; Ericson Jogsten et al., 2012; Eriksson & Kärrman, 2015; Goosey & Harrad, 2011; Gustafsson et al., 2022; Harrad et al., 2019; Haug et al., 2011; Huber et al., 2011; Karaskova et al., 2016; Kato et al., 2009; Padilla-Sanchez & Haug, 2016; Weiss et al., 2021; Winkens et al., 2018; Xu et al., 2013). No studies were identified for the Netherlands through the systematic search. However, PFAS concentrations were measured in three Dutch homes (Arcadis, 2021). In addition, a separate study in Belgium investigated PFAS concentrations in household dust in 129 homes near the 3M factory (Consortium et al., 2023).

Studies that did not report both a median and a maximum were excluded, and other summary statistics (e.g. geometric means) were not used to maintain consistency across studies. As a result, Barreca et al. (2023), Consortium et al. (2023), D'Hollander et al. (2010), Gustafsson et al. (2022) and Padilla-Sánchez & Haug (2016) were not included, and 14 studies remained for deriving the concentration used in this study.

PFAS concentrations in house dust varied between studies, likely due to differences in sampling strategies and analytical methods. An additional uncertainty in the reported mean and median values was that some studies included values below the limit of detection (LOD) in the calculation, usually by substituting them with LOD/2 or LOD/√2. Median concentrations were often very low, around or even below the LOD. Other reasons for variation could include the presence of PFAS-containing products (carpets, furniture, coatings, packaging), although the contribution of these sources is often uncertain (Du et al., 2026), or the location, for example proximity to a hotspot. Notably, the Belgium study near the 3M factory, a known hotspot, did not report the highest

median concentrations, higher median concentrations were observed in other studies (Consortium et al., 2023).

The highest reported median values of all PFAS measured in the included studies were for PFOS, PFOA and PFHxS. For PFOS, the highest reported median concentration was 479,6 ng/g with a detection frequency of 74.4%, for PFOA 300 ng/g with a detection frequency of 100%, and for PFHxS 200 ng/g with detection frequency of 100% (Goosey & Harrad, 2011; Kato et al., 2009). Sulfonamide derivatives (e.g. EtFOSE and MeFOSA) and polyfluoroalkyl phosphoric acid esters (e.g. diPAPs and monoPAPs) were also reported at median concentration up to 150 ng/g (Eriksson & Kärrman, 2015; Goosey & Harrad, 2011).

For the concentration used in the calculations in this report, we extracted the reported median and maximum concentrations for each PFAS with an available RPF from the included studies. For indoor environmental media such as house dust, most studies reported only summary statistics and the underlying measurement distributions were not available. This makes it difficult to reliably estimate the pooled concentration distribution or directly determine a high percentile such as the 75th percentile of the pooled data. To avoid strong assumptions about these distributions, we applied an empirical approximation.

Each study was given equal weight, irrespective of the number of residences sampled, to limit potential bias arising from differences in sampling strategies, analytical methods and local conditions. For each PFAS, we extracted the reported median and maximum concentration from each study. By definition half of the measurements in each study lie below the median and the remainder between the median and the maximum. For n equally weighted studies, each median or maximum therefore represents (and has a concentration larger than) half of a study and contributes $1/(2n)$ to the total weight. Any value that exceeds m of the $2n$ reported medians and maximums is guaranteed to be at or above at least a fraction $m/(2n)$ of the total weight. This does not require any assumptions on the shape of the studies' concentration distributions.

We therefore selected the 75th percentile of the combined set of medians and maxima across studies using the *nearest-rank method* (the smallest value such that 75% of the medians and maxima are at or below this value – this method does not interpolate the data, as is done in the more conventional *interpolated method*). This 75th percentile of medians and maximums is therefore guaranteed to correspond to at least the 75th percentile of the underlying concentration measurements (when weighted by study), and can be interpreted as a conservative estimate of the 75th percentile of indoor concentrations. Studies that did not report both a median and a maximum were excluded, and medians below the detection limit were replaced by the detection limit (if reported) or by zero.

Table A1 presents, for each PFAS with an RPF, the approximated P75 of the study-weighted median and maximum concentrations, shown both as ng/g and as ng PEQ/g. The resulting total concentration used in the exposure calculations was 945 ng PEQ/g. Besides PFOS and PFOA, PFNA

and PFDA also contributed to the total PEQ concentration due to their high RPFs of 10, even though their concentrations were lower.

Table A1 Approximated P75 of the study-weighted median and maximum concentrations of PFAS measured in house dust, shown both as ng/g and as ng PEQ/g.

PFAS group	PFAS	P75 concentration (ng/g)	RPF	P75 concentration (ng PEQ/g)
PFSA	PFBS	12	0.001	0.012
	PFPeS	ND	0.6	ND
	PFHxS	11.3	0.6	6.78
	PFHpS	2.1	2	4.2
	PFOS	140	2	280
	PFDS	1.1	2	2.2
PFCA	PFBA	19	0.05	0.95
	PFPrA	ND	0.01	ND
	PFPeA	6	0.05	0.3
	PFHxA	28	0.01	0.28
	PFHpA	19.4	1	19.4
	PFOA	98	1	98
	PFNA	14.8	10	148
	PFDA	25.8	10	258
	PFUnDA	7.68	4	30.72
	PFDoDA	17.4	3	52.2
	PFTrDA	6.83	3	20.49
	PFTeDA	14.6	0.3	4.38
	PFHxDA	4.83	0.02	0.0966
	PFODA	0.84	0.02	0.0168
Other	HFPO-DA (genX)	63	0.06	3.78
	DONA	ND	0.03	ND
	6:2 FTOH	246	0.02	4.92
	8:2 FTOH	256	0.04	10.24
	TFA	ND	0.002	ND
	TeFPrA	ND	0.01	ND
Total				**945**

ND is not detected

Indoor air

Concentration data for PFAS in indoor air were identified through systematic searches in PubMed and Scopus using the following search string: ((PFAS OR perfluoroalkyl* OR polyfluoroalkyl* OR fluorotelomer* OR perfluorinated compound* OR perfluorooctane* OR PFOA OR PFOS OR FTOH) AND ("indoor air" OR "indoor environment") AND (household* OR home* OR residential OR "private residence*") AND (Europe OR

European OR Austria OR Belgium OR Denmark OR Finland OR France OR Germany OR Italy OR Netherlands OR Norway OR Spain OR Sweden OR Switzerland OR UK OR "United Kingdom" OR Ireland OR Portugal OR Greece OR Poland OR Czech OR Slovakia OR Hungary OR Romania OR Bulgaria OR Estonia OR Latvia OR Lithuania OR Slovenia OR Croatia OR Luxembourg)). Only household air measurements were included; occupational or commercial settings were excluded.

This search resulted in 11 studies reporting PFAS concentrations in indoor air in Europe (Barber et al., 2007; Ericson Jogsten et al., 2012; Fromme et al., 2015; Goosey & Harrad, 2012; Harrad et al., 2019; Haug et al., 2011; Huber et al., 2011; Langer et al., 2010; Padilla-Sanchez et al., 2017; Sha et al., 2018; Winkens et al., 2017). For the derivation of the concentration used in this report, studies that did not report both a median and a maximum, or only other summary statistics (e.g. geometric means), were excluded. As a result, Ericson Jogsten et al. (2012), Barber et al. (2007) and Sha et al. (2018) were not included, and 8 studies remained for deriving the concentration used in this study.

PFAS concentrations in indoor air varied within and between studies, likely due to differences in sampling strategies, analytical methods and how median, geometric mean and mean values were calculated. In most studies on indoor air, <LOD values were substituted with LOD/2 or LOD/√2 in their calculations.

In indoor air, mainly fluorotelomers and sulfonamide derivatives were measured. The highest median concentrations were measured for 8:2 FTOH (13668 pg/m^3), 10:2 FTOH (3817,5 pg/m^3), MeFOSE (2993.5 pg/m3), 6:2 FTOH (2970 pg/m^3), EtFOSE (2139 pg/m^3), EtFOSA (1475,5 pg/m^3), 4:2 FTOH (1153 pg/m^3) and MeFOSA (1105.5 pg/m^3) (Langer et al., 2010; Padilla-Sanchez et al., 2017). Median concentration for carboxylic acids, such as PFOA (56 pg/m^3), and sulfonic acids, such as PFOS (11 pg/m^3) and PFHxS (23 pg/m^3), were much lower (Goosey & Harrad, 2012; Harrad et al., 2019).

For the concentration used in the calculations in this report, we extracted the reported median and maximum concentrations for each PFAS with an available RPF from the included studies. As with house dust, most studies reported only summary statistics (typically median and maximum concentrations), and the underlying measurement distributions were not available. To avoid strong assumptions about these distributions, we applied the same empirical approximation as described for house dust (see above). In short, each study was given equal weight, and the 75th percentile of the combined set of medians and maxima across studies was determined using the nearest-rank method. For further methodological details, see the description under house dust.

Table A2 shows, for each PFAS with an RPF, this approximated P75 of the study-weighted median and maximum concentrations in indoor air, presented both as ng/m^3 and as ng PEQ/m^3. Since only 6:2 FTOH and 8:2 FTOH of the fluorotelomers and sulfonamide derivatives have an RPF, the resulting total concentration of 2.2 ng/m^3 was mainly based on these two PFAS. We also assumed that these PFAS co-existed in indoor

air. This assumption is supported by studies that measured both 6:2 FTOH and 8:2 FTOH. As the RPFs of 6:2 FTOH and 8:2 FTOH are 0.02 and 0.04, respectively, the resulting PEQ concentrations were much lower than the measured concentrations.

Table A2 Approximated P75 of the study-weighted median and maximum concentrations of PFAS measured in indoor air, shown both as ng/m^3 and as ng PEQ/m^3.

PFAS group	PFAS	P75 of concentration (ng/m^3)	RPF	P75 of concentration (ng PEQ/m^3)
PFSA	PFBS	0.015	0.001	0.000015
	PFHxS	0.023	0.6	0.0138
	PFOS	0.208	2	0.416
PFCA	PFHxA	0.144	0.01	0.00144
	PFHpA	0.0126	1	0.0126
	PFOA	0.386	1	0.386
	PFNA	0.013	10	0.13
	PFDA	0.0296	10	0.296
	PFUnDA	0.00824	4	0.03296
	PFDoDA	0.00565	3	0.01695
Other	6:2 FTOH	5.424	0.02	0.10848
	8:2 FTOH	18.508	0.04	0.74032
Total				**2.2**

Outdoor clothing

Concentration data for PFAS in outdoor clothing were identified through systematic searches in PubMed and Scopus using the following search string: ((PFAS OR perfluoroalkyl* OR polyfluoroalkyl* OR fluorotelomer* OR perfluorinated compound* OR perfluorooctane* OR PFOA OR PFOS OR FTOH) AND ("textile" OR "clothing" OR "jackets") AND ("concentration" OR "measurements") AND (Europe OR European OR Austria OR Belgium OR Denmark OR Finland OR France OR Germany OR Italy OR Netherlands OR Norway OR Spain OR Sweden OR Switzerland OR UK OR "United Kingdom" OR Ireland OR Portugal OR Greece OR Poland OR Czech OR Slovakia OR Hungary OR Romania OR Bulgaria OR Estonia OR Latvia OR Lithuania OR Slovenia OR Croatia OR Luxembourg)).

This search resulted in seven European studies reporting PFAS concentrations in outdoor clothing (Borg & Ivarsson, 2017; Brigden et al., 2013, 2014; Gremmel et al., 2016; Kotthoff et al., 2015; Lavalle et al., 2025; van der Veen et al., 2020; van der Veen et al., 2022; van der Veen et al., 2016). PFAS concentrations in other textile were not taken into consideration. In total, 32 individual measurements of jackets, 3 ponchos, 8 (rain)coats, and 5 outdoor trousers, including boy's pants and salopettes, are available. See Table A3 for an overview of the tested items per study. Kotthoff et al. (2015) did not further specify the types of outdoor materials tested.

Table A3 Number of outdoor clothing items tested per study.

Study	**Items tested**	**N**
Gremmel et al., 2016	Jackets	14
van der Veen et al.,2022	Outdoor trousers, jackets	5
Lavalle et al., 2025	Jackets, poncho, raincoat, boy pants, salopette	12
Kotthoff et al.,2015	Outdoor textiles, gloves	8
Borg & Ivarsson,2017	Jackets, raincoat	6
Brigden et al., 2013	Coats, jackets, trousers	10
Brigden et al., 2014	Jacket	1
Total		56

For our analysis, we used the P75 of the individual concentrations. This approach was chosen because products are unlikely to differ significantly in concentrations from country to country within the EU. Furthermore, in contrast to indoor environmental media, the measurement outcome is less sensitive to the chosen sampling strategies and analytical methods. Therefore, each product is given the same weight in the P75 calculation. If a study only reported a median value and not the individual product measurements, we assumed that all products in that study had this median concentration and included it once for each product in the study. In other words, the study median was replicated for the number of products sampled, so that the total P75 was still based on individual products rather than on studies. This assumption (replicating the median for all products in a study) may underestimate the within-study variability, but was considered acceptable given the limited availability of individual product data and our focus on high-end exposure scenarios.

Concentrations reported in g/kg were converted to μg/m^2, assuming a density of 1.5 g/dm^2 (Berger & Herzke, 2006). This conversion was applied to concentrations from Lavalle et al. (2025), Kotthoff et al. (2015) and Borg & Ivarsson (2017).

Among the PFAS detected in outdoor clothing, the highest P75 values were found for 8:2 FTOH (18.4 μg/m²) and 10:2 FTOH (9.1 μg/m²). In contrast, the P75 for carboxylic acids such as PFOA (0.9 μg/m²) and sulfonic acids such as PFOS (0.2 μg/m²) were much lower.

For the exposure assessment, we used the individual concentrations for each individual PFAS with an RPF. We assumed that all selected PFAS co-existed in textile. Table A4 shows, for each PFAS with an RPF, the P75 of the reported individual concentrations across all included studies, presented both as μg/m² and as μg PEQ/m². The resulting total concentration used in the exposure calculations was 10 μg PEQ/m². Although the P75 of individual measurements of 8:2 FTOH was highest, its low RPF meant it contributed little to the total PEQ values. In contrast, PFDA and PFNA were measured at lower concentrations, but due to their high RPF, they contributed substantially to the total PEQ concentration.

Table A4 The 75th percentile (P75) of the individual concentration of PFAS measured in outdoor clothing, shown both as µg/m² and as µg PEQ/m².

PFAS group	PFAS	P75 of individual measurements (µg/m²)	RPF	P75 of individual measurements (µg PEQ/m²)
	PFBS	0.13275	0.001	0.000133
	PFHxS	0.11	0.6	0.066
	PFHpS	0.1405	2	
	PFOS	0.20975	2	0.4195
	PFDS	0.137	2	0.274
PFCA	PFBA	0.1325	0.05	0.006625
	PFPeA	0.345	0.05	0.01725
	PFHxA	0.23625	0.01	0.002363
	PFHpA	0.098	1	0.098
	PFOA	0.9	1	0.9
	PFNA	0.15	10	1.5
	PFDA	0.40125	10	4.0125
	PFUnDA	0.2	4	0.8
	PFDoDA	0.37	3	1.11
	PFTrDA	0.10275	3	0.30825
	PFTeDA	0.15	0.3	0.045
Other	6:2-FTOH	1.7475	0.02	0.03495
	8:2-FTOH	18.4	0.04	0.736
Total				**10**

Impregnation sprays

Concentration data for PFAS in impregnation sprays were identified through systematic searches in PubMed and Scopus using the following search string: ("PFAS" OR "perfluoroalkyl" OR "polyfluoroalkyl" OR "perfluorinated" OR "perfluorochemical" OR "PFOA" OR "PFOS" OR "fluorochemical") AND ("impregnation spray" OR "water repellent spray" OR "protective spray" OR "fabric spray" OR "textile spray" OR "shoe spray" OR "leather spray" OR "aerosol spray") AND ("measure*" OR "concentration" OR "analysis" OR "determination" OR "quantification" OR "content" OR "monitoring").

This search resulted in nine studies reporting PFAS concentrations in impregnation sprays in Europe (Blom & Hanssen, 2015; Borg & Ivarsson, 2017; Favreau et al., 2017; Fiedler et al., 2010; Herzke et al., 2012; Kotthoff et al., 2015; Sero et al., 2022; Trudel et al., 2008; Washburn et al., 2005). Kotthoff et al. (2015) reported median concentrations for 16 products, Favreau et al. (2017) presented median concentrations for 60 products, while Sero et al. (2022), Fiedler et al. (2010), Herzke et al. (2012), Borg & Ivarsson (2017), and Blom & Hanssen (2015) each reported individual values for 16, 9, 5, 5, and 4 impregnation sprays, respectively. The study by Trudel et al. (2008) referenced Washburn et al. (2005), but concentrations could not be identified in Washburn et al. (2005). Both studies were not included in

the concentration derived for the exposure assessment. All values were converted to ng/g, assuming a density of 1 kg/L.

For the exposure assessment, the P75 of individual concentrations was used for each PFAS. Studies that reported individual measurements were included using those measured concentrations. In studies that only reported a median concentration and not the individual product data, we assumed that all products in that study had this median concentration and included it once for each product in the study. In other words, the study median was replicated for the number of products sampled, so that the overall P75 remained based on individual products rather than on studies. This assumption (replicating the median for all products in a study) may underestimate the within-study variability, but was considered acceptable given the limited availability of individual product data and our focus on high-end exposure scenarios. Where medians were reported as <LOD, the LOD was used; if the LOD was not provided, a value of zero was assigned. The highest P75 values were found for FTOH10:2 (231000 ng/g), FTOH8:2 (223312,5 ng/g), FTOH6:2 (194000 ng/g) and 8:2 FTA (158000 ng/g).

The media of P75 value used in the exposure estimation was mainly based on FTOH6:2 en FTOH8:2 as for the other volatile PFAS no RPF is available. Table A5 shows, for each PFAS with an RPF, the P75 of the reported individual concentrations across all included studies, presented both as ng/g and as ng PEQ/g. The resulting total concentration used in the exposure calculations is 13 µg PEQ/g.

Table A5 The 75th percentile (P75) of the median or (geometric) mean concentrations of PFAS measured in impregnation sprays, shown both as ng/g and as ng PEQ/g.

PFAS group	PFAS	P75 of individual values (ng/g)	RPF	P75 of individual values (ng PEQ/g)
PFCA	**PFBA**	0.80825	0.05	0.040413
	PFHxA	6.9	0.01	0.069
	PFHpA	2.7	1	2.7
	PFOA	15.9	1	15.9
Other	**FTOH6:2**	194000	0.02	3880
	FTOH8:2	223312.5	0.04	8932.5
Total				12831.21

Appendix 2 General parameters and dermal absorption factions used in the calculations

General parameters

The following general parameters were used in exposure calculations for non-food sources, unless otherwise specified for a particular source (see Section 2.6.1):

- Body weights: For the four age groups, P25 values from Tables 17 and 18 of the General Fact Sheet (Te Biesebeek et al., 2014) were used. For 1-5 years, a weighted average of 6-12 months, 1-2 years, 2-3 years and 3-6 years (weights: 1:1:1:2) was applied. For 6-10 years, a weighted average of 2-6 years and 6-11 years (weights: 1:4) was applied. For 11-17 years, a weighted average of 6-11 years, 11-16 years and 16-18 years (weights: 1:5:1) was applied. The resulting values are: 12.3 kg (2–5 years), 22.3 kg (6–10 years), 43.9 kg (11–17 years), and 68.8 kg (18+ years). The dataset used to derive the body weight percentiles does not include persons above the age of 70. Therefore the value for 18-70 years was applied to all adults.
- Surface area: the surface area of outdoor jackets in contact with skin was derived from Tables 29 and 30 of the General Fact Sheet (Te Biesebeek et al., 2014). It was assumed that a t-shirt is worn under the jacket, so only the arms were in direct contact with the jacket. For each age group, the average surface area for boys/men and girls/women was used. The resulting surface area were 798 cm^2 (≈ 8 dm^2) for 1-5 years (weighted average of 6-12 months, 1-2 years and 2-6 years, weighted 1:1:3), 1260 cm^2 (≈ 13 dm^2) for 6-10 years (weighted average 3-6 and 6-11 years, weighted: 1:4), 2005 cm^2 (≈ 20 dm^2) for 11-17 years (weighted average of 6-11, 11–16 and 16–18 years, weighted 1:5:1), and 2600 cm^2 (≈ 26 dm^2) for adults (18+ years). For adults, trunk and arm skin surface area and hand skin surface area were 9,200 cm^2 (≈ 92 dm^2) (2,600 cm^2 arms + 6,600 cm^2 trunk) and 450 cm^2 (≈ 5 dm^2) (half of 900 cm^2 for both hands), respectively.
- Inhalation rates: Long-term inhalation rates, applicable to inhalation of indoor air, are not given directly in the General Fact Sheet (Te Biesebeek et al., 2014). Reference is made to values provided in the REACH Guidance on information requirements and chemical safety assessment, Chapter R.15: Consumer exposure estimation (2012). These values are no longer included in the current REACH guidance (2016 version) (ECHA, 2016).The US EPA Exposure Factors Handbook offers mean and P95 long-term inhalation rates, both unadjusted and adjusted for body weight (U.S. EPA, 2011b). To ensure realistic systemic dose estimates (inhaled substance per kg body weight), we avoided combining high inhalation rates with low body weights, as these rarely coincide. We used the P75 of daily inhalation rate adjusted for body weight from Tables 6-14 and 6-15 of the Exposure Factors Handbook (NHANES data). Male and female rates were averaged, and age group values were estimated as weighted averages based on years of overlap between US EPA and our age

groups. For adults, we used an upper age limit of <71 years for consistency with our body weight data. Finally, we multiplied by daily inhalation rate adjusted for body weight the P25 body weight for each age group to approximate the P25 inhalation rate.

Dermal absorption fractions

The absorption fractions were set at 100% for oral and inhalation routes. For the dermal contact with outdoor clothing and personal care products, PFAS-specific dermal absorption fractions (DAFs) were applied. Where available, these DAFs were derived from recent in vitro skin permeation studies (Ragnarsdottir et al., 2024; Wang et al., 2025). Both studies report DAFs for individual PFAS after defined exposure durations (8-36 hours), and together cover a range of chain lengths and functional groups. Wang et al. (2025) provides DAFs for those PFAS that were sufficiently frequently detected in personal care products, whereas Ragnarsdottir et al. (2024) reports DAFs for a broader set of PFAS. Both studies demonstrate that dermal absorption of PFAS decreases with increasing fluorinated carbon chain length and varies by functional group: perfluoroalkyl carboxylates (PFCAs) are absorbed more readily through the skin than perfluoroalkyl sulfonates (PFSAs). The DAFs are summarized per PFAS in Table A6.

Table A6 Experimentally determined dermal absorption fractions (DAFs) for PFAS, based on Wang et al. (2025) and Ragnarsdottir et al. (2024).

PFAS	**Wang et al. 2025**		**Ragnarsdottir et al. 2024**	
	DAF (%)	**Time**	**DAF (%)**	**Time**
PFBA	31.55 / 48.61	8h / 12h	-	36h
PFPeA	20.73 / 31.74	8h / 12h	58.9	36h
PFHxA	14.81 / 22.83	8h / 12h	36.4	36h
HFPO-DA	13.7 /20.55	8h / 12h	-	-
PFHpA	10.41 / 17.38	8h / 12h	28.8	36h
PFOA	6.99 / 13.44	8h / 12h	13.5	36h
PFNA	4.03 / 10.07	8h / 12h	0.9	36h
PFDA	4.11	12h	0.1	24h
PFBS	-	-	48.7	36h
PFPeS	-	-	43.9	36h
PFHxS	-	-	21.9	36h
PFHpS	-	-	6.4	36h
PFOS	-	-	1.3	24h

N/A: not available or below limit of detection.

For the dermal exposure via clothing and personal care products, we applied PFAS-specific DAFs from Wang et al. (2025) for a 12 hours of exposure scenario, as we consider it likely that decorative cosmetics and outdoor clothing are worn for approximately 12 hours per day. If no DAF

was available from Wang et al. (2025), we used the value from Ragnarsdottir et al. (2024). For PFAS with an RPF but without an experimental DAF (namely PFDS, PFPrA, PFUnDA, PFDoDA, PFTrDA, PFTeDA, PFHxDA, PFODA, DONA, 6:2 FTOH, 8:2 FTOH, TFA, TeFPrA) we assigned a DAF by analogy, based on the closest available DAF for a PFAS with similar chain length and functional group. Specifically, for PFCAs we used the DAF of the PFCA with the closest chain length; for PFSAs, the DAF of the PFSA with the closest chain length; and for other PFAS, the DAF of the most structurally similar PFAS. Only for PFPrA and TFA, we assigned the DAF of a PFCA with a longer chain length (PFBA, C4), which may have resulted in an underestimation of dermal absorption. However, to maintain consistency with our approach, and in the absence of other data, we used this value. The DAF assigned to PFAS without experimental data are shown in Table A7.

Table A7 Extrapolated DAF for PFAS without experimental data, based on closest chain length and functional group.

PFAS	**Structural Group**	**Chain Length**	**Assign DAF from**	**DAF (%)[a]**
PFPrA	PFCA	C3	PFBA (C4)	31.55 / 48.61
PFUnDA	PFCA	C11	PFDA (C10)	4.11
PFDoDA	PFCA	C12	PFDA (C10)	4.11
PFTrDA	PFCA	C13	PFDA (C10)	4.11
PFTeDA	PFCA	C14	PFDA (C10)	4.11
PFHxDA	PFCA	C16	PFDA (C10)	4.11
PFODA	PFCA	C18	PFDA (C10)	4.11
PFDS	PFSA	C10	PFOS (C8)	1.3
DONA	PFCA (ether)	C9-ether	PFNA (C9)	4.03 / 10.07
6:2 FTOH	FTOH	C6/C2	PFHxA (C6)	14.81 / 22.83
8:2 FTOH	FTOH	C8/C2	PFOA (C8)	6.99 / 13.44
TFA	PFCA	C2	PFBA (C4)	31.55 / 48.61
TeFPrA	PFCA	C5	PFPeA (C5)	20.73 / 31.74

[a] Assigned DAF (%) is based on the value for the indicated reference compound (fourth column). For PFPrA and TFA, the DAF of PFBA (C4) was used, which may underestimate absorption due to shorter chain length.

Appendix 3 Concentration data and detailed PACEM input parameters for personal care products

This appendix provides the detailed methodology and full input tables used for PACEM exposure calculations for personal care products.
It includes:

- The mathematical approach for constructing concentration input distributions (log-normal and trapezoidal)
- Occurrence percentage calculations
- All input parameter tables for both historical and regulatory-limited scenarios
- Assumptions regarding exposure routes and absorption fractions

These details support the summary provided in section 2.7 of the main text.

Concentration data

Concentration data were identified through searches in Scopus, performed for previous work in 2023 with following search terms: (title-abs-key(pfas*) or title-abs-key(*fluoroalkyl*)) and (title-abs-key("personal care product*") or title-abs-key(pcp*) or title-abs-key(cosmetic*)). In the meantime, the results have been supplemented with new publications that we have encountered while conducting research into PFAS exposure from personal care products or that have been brought to our attention by our network. For this report, we have limited the results to studies of European personal care products. This search resulted in three studies reporting PFAS concentrations in personal care products in Europe (Brinch et al., 2018; KEMI, 2021; Schultes et al., 2018).

A significant fraction of the measured concentrations exceed the current regulatory limits and would not be allowed on the EU market today. In the regulatory-cap scenario, PFHxS, PFOS, PFHpA and PFOA were limited to 25 ng/g or prohibited from intentional use, and the sum of PFUnDA, PFDoDA, PFTeDA, PFPeA, PFNA and PFDA was limited to 25 ng/g. Concentrations above these limits were truncated or proportionally scaled such that the sum of regulated PFAS did not exceed 25 ng/g, while preserving their relative ratios. Taken together, the historical and regulatory-cap scenarios provide a range of plausible exposures that inform our screening assessment of PCPs as non-food sources.

In Table A8, we assign the products to product categories included in PACEM version 1.3 for Dutch adults, which is based on the survey of Biesterbos et al. (2013). None of the PFAS evaluated in this report were detected in any of the products in the categories ‘Hairspray’ and ‘Shaving foam/gel/oil’, therefore these product categories were excluded from further consideration.

Table A8 Assignment of products to PACEM product categories.

Ref.	Product	Category	Ref.	Product	Category
Schultes et al. (2018)	CRE01	Face cream[(a)]	Brinch et al. (2018)	Facial cream no 22	Face cream[(a)]
	CRE02	Face cream[(a)]		Facial cream no 23	Face cream[(a)]
	CRE03	Face cream[(a)]		Facial scrub no 5	—
	CRE04	Face cream[(a)]		BB cream no 7	—
	CRE05	Face cream[(a)]		Body lotion no 19	Body lotion milk
	CRE06	—[(b)]		Body lotion no 21	Body lotion milk
	CRE07	Face cream[(a)]		CC cream no 2	—
	FOUN01	—[(c)]		CC cream no 20	—
	FOUN02	Liquid foundation		Concealer no 4	—
	FOUN03	Liquid foundation		Eyeliner no 6	Eye pencil
	FOUN04	Liquid foundation		Foundation no 13	Liquid foundation
	FOUN05	Liquid foundation		Foundation no 14	Liquid foundation
	FOUN06	Liquid foundation		Foundation no 17	Liquid foundation
	FOUN07	Liquid foundation		Foundation no 8	Liquid foundation
	FOUN08	Liquid foundation		Highlighter no 10	Rouge powder
	FOUN09	Liquid foundation		Hair spray no 12	Hairspray
	PEN01	Eye pencil		Powder no 3	Rouge powder
	POW01	Eye shadow		Eye shadow no 16	Eye shadow
	POW02	Eye shadow		Body lotion no 18	Body lotion milk
	POW03	Rouge powder		Foundation no 11	Liquid foundation
	POW04	Rouge powder	KEMI (2021)	Concealer 1	—
	POW05	Rouge powder		Eye shadow 5	Eye shadow
	POW06	Rouge powder		Found./BB cream 3	Liquid foundation
	POW07	Rouge powder		Found./BB cream 4	Liquid foundation
	POW08	Rouge powder		Lip liner, pen 1	Lip pencil
	POW09	Rouge powder		Mask 1	—
	POW10	Rouge powder		Mask 2	—
	POW11	Rouge powder			
	POW12	Rouge powder			
	SHAV01	Shaving foam/gel/oil			
	SHAV02	Shaving foam/gel/oil			

—: No corresponding category in PACEM, data not used in this assessment.
(a) Applied to both 'Face day cream' and 'Face night cream'.
(b) This product is a serum and therefore not included in 'Face day/night cream'.
(c) This product is called 'Shade adjusting drops darkening' and therefore not included in 'Liquid foundation'.

PACEM input parameters

From the DAF-converted total PEQ concentrations of products belonging to the same product category, we calculated a concentration input distribution for PACEM (see Tables A9-A12 for input parameters).

If more than one non-zero concentrations were available, we assumed a truncated log-normal distribution. First, we calculated the geometric mean (GM) and estimated the upper confidence interval limit of the geometric standard deviation (GSD) at a confidence level of 95%. This GSD estimator was calculated using the equation

$$GSD_{\mathrm{P95}} = \sqrt{\frac{(n-1) \times SD^2}{\chi^2_{1-0.95,n-1}}}$$

where *n* is the number of products with non-zero concentrations, *SD* is the sample standard deviation of the log-transformed non-zero concentrations, and $\chi^2_{p,d}$ is the quantile function of the chi-squared distribution with probability *p* and with *d* degrees of freedom. We limited the GSD estimator at 30, because this is the maximum value supported by PACEM. This is not expected to significantly impact results, as above a GSD of 30 the log-normal probability density function does not change much with GSD. Finally, we truncated the log-normal distribution at a factor 10 below the lowest concentration and at a factor 10 above the highest concentration.

If only one non-zero concentration was available, we assumed a trapezoidal input distribution with a minimum at a factor 10 below the concentration, a lower mode at a factor 2 below the concentration, an upper mode at a factor 2 above the concentration, and a maximum at a factor 10 above the concentration. The input parameters for the log-normal and trapezoidal distributions, based on historical and limited concentration data, are shown in Tables A9-A12.

The occurrence percentages of the studied PFAS considered in this report were calculated based on the following factions:

- "% PEQ/PFAS" represents, within the subset of measured products, the fraction of products that contained the studied PFAS.
- "% PFAS/all" represents the occurrence of products with intentionally added PFAS among all products in that category (from KEMI, 2021).
- "% PEQ/all" combines both aspects and indicates the fraction of all products in the category that contain the studied PFAS.

The concentration measurements originated from studies that analysed personal care products known to contain intentionally added PFAS, such as PTFE. The fraction of products with positive concentration measurements therefore indicated the occurrence of the studied PFAS within the measured products (%PEQ/PFAS). To estimate the occurrence of the studied PFAS across all personal care products (% PEQ/all), we multiplied this faction by the occurrence of intentionally added PFAS in all personal care products (%PFAS/all). These latter fractions were described by KEMI (2021, table 8). The relevant fractions were: 0.78% for Skin care (Body lotion milk and Face day/night cream), and 3.67% for Decorative cosmetics (Eye pencil, Eye shadow, Lip pencil, Liquid foundation, and Rouge powder).

We assumed only dermal exposure in this scenario. As all products considered here were leave-on products, we set all dermal exposure factors to 1. All oral and inhalation exposure factors were set to 0. As the dermal absorption fraction was already applied to calculate the DAF-converted total PEQ concentrations, we left the absorption fractions in PACEM at 1, to avoid double application of the dermal absorption fractions.

Table A9 PACEM input parameters, historical concentrations, log-normal distributions.

Product	n	n >0	% PEQ/PFAS	% PFAS/all	% PEQ/all	GM (ng/g)	GSD P95 (ng/g)	Lower bound (ng/g)	Upper bound (ng/g)
Body lotion milk	3	2	67	0.78	0.5	11	30	0.81	160
Eye shadow	4	4	100	3.67	3.7	4.2	30	0.02	1288
Face day/night cream	8	2	25	0.78	0.2	4.5	30	0.20	101
Liquid foundation	15	8	53	3.67	2.0	73	30	0.48	19064
Rouge powder	12	8	67	3.67	2.4	12	30	0.27	507

Note: The percentages represent occurrence of products containing the studied PFAS; geometric mean concentrations (GM) already account for potency and dermal absorption (PEQ).

Table A10 PACEM input parameters, historical concentrations, trapezoidal distributions.

Product	n	n >0	% PEQ/PFAS	% PFAS/all	% PEQ/all	Minimum (ng/g)	Lower mode (ng/g)	Upper mode (ng/g)	Maximum (ng/g)
Eye pencil	2	1	50	3.67	1.8	0.0028	0.014	0.055	0.28
Lip pencil	1	1	100	3.67	3.7	0.014	0.072	0.29	1.4

Note: The percentages represent occurrence of products containing the studied PFAS; geometric mean concentrations (GM) already account for potency and dermal absorption (PEQ).

Table A11 PACEM input parameters, limited concentrations, log-normal distributions.

Product	n	n >0	% PEQ/PFAS	% PFAS/all	% PEQ/all	GM (ng/g)	GSD P95 (ng/g)	Lower bound (ng/g)	Upper bound (ng/g)
Body lotion milk	3	2	67	0.78	0.5	7.9	30	0.77	81
Eye Shadow	4	4	100	3.67	3.7	2.2	30	0.02	168
Face day/night cream	8	2	25	0.78	0.2	3.7	30	0.20	67
Liquid foundation	15	8	53	3.67	2.0	17	30	0.48	448
Rouge powder	12	8	67	3.67	2.4	7.1	30	0.27	127

Note: The percentages represent occurrence of products containing the studied PFAS; geometric mean concentrations (GM) already account for potency and dermal absorption (PEQ).

Table A12 PACEM input parameters, limited concentrations, trapezoidal distributions.

Product	n	n >0	% PEQ/PFAS	% PFAS/all	% PEQ/all	Minimum (ng/g)	Lower mode (ng/g)	Upper mode (ng/g)	Maximum (ng/g)
Eye pencil	2	1	50	3.67	1.8	0.0028	0.014	0.06	0.28
Lip pencil	1	1	100	3.67	3.7	0.014	0.072	0.29	1.4

Note: The percentages represent occurrence of products containing the studied PFAS; geometric mean concentrations (GM) already account for potency and dermal absorption (PEQ).

Published by

National Institute for Public Health and the Environment, RIVM
P.O. Box 1 | 3720 BA Bilthoven
The Netherlands
www.rivm.nl/en

October 2026

Committed to health and sustainability