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PFAS AND THEIR DEGRADATION IN THE ENVIRONMENT

A REVIEW ON PFAS DEGRADATION PATHWAYS IN THE ENVIRONMENT



Report

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This review has been prepared following good scientific practice and aims to provide an accurate and balanced synthesis of the available literature. Nevertheless, it may not cover all relevant studies, and interpretations may evolve as new data become available. Readers are encouraged to consult the cited references and to independently assess the information for their specific purposes.

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Executive summary

This report reviews the mostly peer-reviewed scientific literature on **environmental degradation** across a wide range of PFAS groups. Its purpose is to explain, **what** is known about whether these substances break down in the environment, **where** breakdown can occur (air, water, soils/sediments, wastewater treatment¹), **how** it may occur (abiotic vs. biotic pathways), and **what** the likely end products are. The report also highlights where evidence is strong, where findings rely on laboratory-only conditions, and where major data gaps remain.

Why degradation is a complex topic

Two kinds of complexity shape the evidence throughout the literature:

1. **Chemical diversity (substance complexity).** PFAS are not one chemical. They include fully fluorinated molecules, partially fluorinated molecules, volatile compounds, and polymers, each with different properties (functional groups, volatility, water solubility, tendency to bind to particles, and reactivity). These differences strongly affect how and where degradation could occur.
2. **Environmental diversity (system complexity).** “The environment” is not a single setting. The atmosphere is strongly oxidizing and sunlight-driven, surface waters have sunlight but also radical scavengers and complex mixtures, groundwater is dark and often slower-reacting, soils and sediments contain minerals and microbes and can vary widely in oxygen availability and wastewater treatment plants are engineered biological reactors that can transform some precursors but rarely destroy stable end products. As a result, degradation pathways observed in laboratory studies may occur at substantially different rates under environmental conditions. Many laboratory experiments employ elevated temperatures, higher reactant concentrations, optimized pH, or enhanced oxidant levels to elucidate transformation mechanisms within practical timeframes. In contrast, environmental conditions are typically less extreme and more variable, often leading to slower transformation rates. At the same time, there is growing evidence that microbial communities can influence PFAS transformation under environmentally relevant conditions, including biodegradation and precursor transformation pathways. (Ye et al., 2025)

Across many studies, an important distinction emerges between **transformation** and **mineralization/defluorination** (Smith et al., 2024; Trang et al., 2022). Transformation refers to structural modification of the parent PFAS into another (often still fluorinated) organic compound, without complete breakdown of the carbon framework. Mineralization, by contrast, denotes the conversion of the compound into inorganic constituents such as carbon dioxide, water, and inorganic ions. In the context of PFAS, fluoride release provides evidence of C–F bond cleavage (defluorination), but this alone does not demonstrate complete mineralization of the carbon backbone to CO₂. Rather, it may reflect partial defluorination and/or the formation of smaller organic intermediates, with full mineralization requiring further breakdown of the carbon

¹ Although sewage treatment plants are engineered technical systems rather than natural environmental compartments, they are commonly considered in environmental fate assessments because they represent a major interface between anthropogenic releases and the natural environment. Many substances enter the environment via wastewater, and biological wastewater treatment systems therefore constitute a key pathway linking emissions to surface waters and, through sludge application, to soils. Moreover, standardized OECD simulation tests conducted in activated sludge are designed to represent environmentally relevant microbial degradation conditions and are routinely used in regulatory persistence evaluations. If no measurable degradation occurs under these comparatively favourable biodegradation conditions, it is generally considered unlikely that substantial degradation would occur in natural surface waters or soils. For this reason, results from sewage treatment simulations are often interpreted alongside data from natural compartments when assessing environmental stability.

framework. The literature often documents transformation, while complete defluorination is less frequently expected and is commonly studied under specialized or strongly amended conditions.

Key findings by substance group

In total, this report systematically reviews **31 distinct PFAS groups**, covering both non-polymeric and polymeric substances with diverse structural features and functional groups. All identified literature (mostly peer-reviewed) was screened and assigned to these predefined groups to ensure a transparent and consistent evaluation framework. This structured categorization makes it possible not only to summarize available evidence within each group, but also to clearly identify where research efforts have been concentrated and where information remains limited or entirely absent. By organizing the literature in this way, the review highlights substance classes that have been extensively investigated (often focusing on a small number of representative compounds), as well as groups for which only a few laboratory studies exist or for which no degradation data could be identified at all. This approach allows the reader to distinguish between well-supported findings and areas characterized by significant data gaps, thereby providing a clearer understanding of both the current state of knowledge and the uncertainties that remain.

For ease of reading the conclusions related to the overall literature review are provided ahead of the details of each substance class. The results of the review show that the available literature provides important insights into degradation pathways and transformation patterns across many PFAS groups, but it also has clear limitations. The evidence base is uneven, often focused on a small number of representative substances, and in many cases derived from laboratory studies conducted under controlled or enhanced conditions that may not fully reflect environmental reality. Field data are scarce for numerous groups, and comprehensive fluorine mass balances or long-term monitoring studies are frequently lacking. As a result, current knowledge allows identification of broad trends and likely transformation routes, but uncertainties remain regarding degradation kinetics, compartment-specific behaviour, and long-term fate for many substances. These limitations should be considered when defining priorities for future research and risk assessment. Figure 1-1 summarizes the number of study entries identified for each PFAS group, distinguishing between abiotic (e.g., atmospheric reactions, hydrolysis, photolysis) and biotic (e.g., water/sediment, soil, wastewater treatment) degradation. Table 1-1 further breaks this down by environmental compartment, using a heatmap approach to visualize the relative density of available evidence. In the heatmap, red indicates that no relevant study entries were identified for the respective PFAS group and degradation pathway, whereas green indicates that multiple publications were found (with intermediate colours representing increasing numbers of study entries).

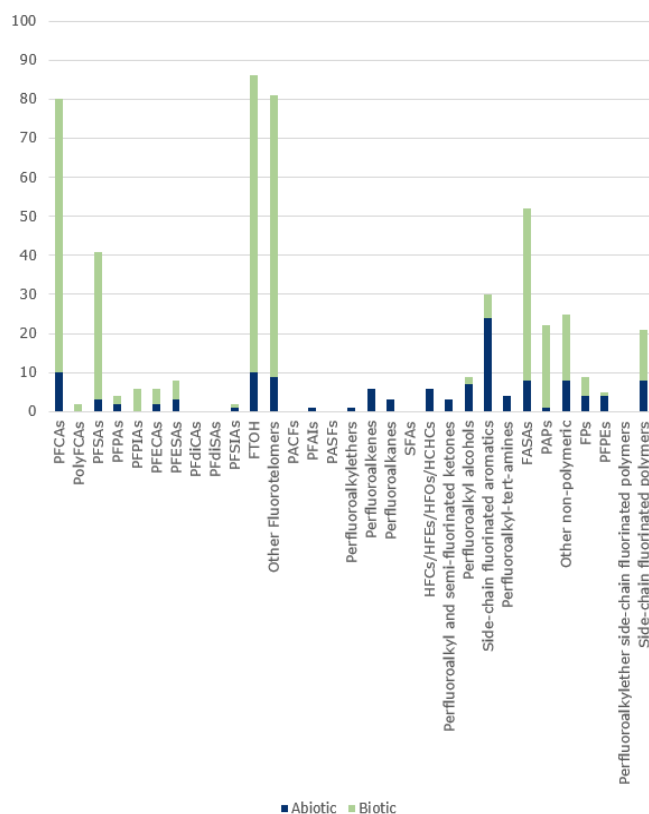


Figure 1-1 Number of study entries identified for each PFAS group split into abiotic (blue) and biotic (green) degradation

PFAS Group	Abiotic		Biotic		
	Atmosphere	Water	Water/ Sediment	Soil	WWTP
Perfluoroalkyl carboxylic acids (PFCAs)					
Polyalkyl carboxylic acids (PolyFCAs)					
Perfluoroalkane sulfonic acids (PFASs)					
Perfluoroalkyl phosphonic acids (PFPAAs)					
Perfluoroalkyl phosphinic acids (PFPIAs)					
Per- and polyfluoroalkylether carboxylic acids (PFECAs)					
Per- and polyfluoroalkylether sulfonic acids (PFESAs)					
Perfluoroalkyl dicarboxylic acids (PFdICAs)					
Perfluoroalkane disulfonic acids (PFdISAs)					
Perfluoroalkane sulfonic acids (PFdISAs)					
Fluorotelomer alcohols (FTOHs)					
Other fluorotelomers and derivatives					
Perfluoroalkyl fluorides (PACFs)					
Perfluoroalkyl iodides (PFdIs)					
Perfluoroalkane sulfonyl fluorides (PASFs)					
Perfluoroalkylethers					
Perfluoroalkenes					
Perfluoroalkanes					
Semifluorinated alkanes (SFAs)					
Hydrofluorocarbons/ -ethers / -olefins / -chlorofluorocarbons (HFCs/ HFEs/ HFOs/ HCFCs)					
Perfluoroalkyl and semi-fluorinated ketones					
Perfluoroalkyl alcohols					
Side-chain fluorinated aromatics					
Perfluoroalkyl-tert-amines					
Perfluoroalkane sulfonamides (FASAs)					
Perfluoroalkyl phosphate esters (PAPs)					
Other non-polymeric					
Fluoropolymers (FPs)					
Perfluoropolyethers (PFPEs)					
Perfluoroalkylether side-chain fluorinated polymers					
Side-chain fluorinated polymers					

Table 1-1: Number of study entries of abiotic and biotic pathways for each PFAS group

Overall, the distribution of data is uneven. Groups such as fluorotelomer alcohols, other fluorotelomer derivatives, sulfonamides, and selected carboxylic and sulfonic acids have been comparatively well studied, particularly in aerobic soils, sediments, and wastewater treatment systems. In contrast, several groups, including certain dicarboxylic acids, disulfonic acids, sulfonyl fluorides, side-chain ether polymers, and semifluorinated alkanes, have little to no published degradation data. For volatile substances such as hydrofluorocarbons (HFCs), hydrofluoroethers (HFEs), hydrofluoroolefins (HFOs), and related gases, the available studies are concentrated almost exclusively in the atmosphere, whereas studies on aquatic or biotic degradation pathways are largely absent. Conversely, for non-volatile high-molecular-weight substances such as fluoropolymers, atmospheric degradation studies are of limited relevance, and research has focused primarily on stability in soils, sediments, and sludge.

Importantly, the absence of identified studies in a specific category does not automatically indicate a true data gap. In some cases, certain degradation pathways may not be environmentally relevant for a given group due to its physicochemical properties. In other instances, information may be extrapolated from structurally related groups using read-across approaches. Furthermore, while this review aimed to be comprehensive, it is not a formal systematic review, and it is possible that despite efforts some studies were not captured.

Taken together, the evidence mapping highlights both areas where the scientific basis is relatively robust and areas where knowledge remains limited. This uneven distribution of data should be considered when interpreting conclusions for individual substance groups and when identifying priorities for future research.

In the following key findings per substance group are elaborated.

Perfluoroalkyl carboxylic acids (PFCAs)

Based on the available literature, PFCAs are generally reported to degrade very slowly under environmentally relevant conditions. In the atmosphere, shorter-chain homologues react only gradually with hydroxyl radicals. In surface waters, direct photolysis appears ineffective, and reported half-lives for PFOA are extremely long.

Biotic degradation under realistic environmental conditions has rarely been observed. Measurable transformation of saturated PFCAs, most often PFOA, has been reported mainly in laboratory or engineered systems that provide strong reducing conditions. Under such settings, partial defluorination and chain-shortening can occur over weeks to months, but complete mineralization is not demonstrated. Unsaturated analogues may show somewhat greater susceptibility under anaerobic conditions, although fluoride release remains limited.

The evidence base is heavily weighted toward studies of PFOA, with limited data for long-chain (C9–C21) or branched PFCAs, and few studies reporting full fluorine mass balances. Field data on environmental degradation are largely lacking. Overall, current evidence indicates that PFCAs are frequently observed as end products of precursor transformation in the environment, with limited evidence for substantial degradation under typical environmental conditions.

Polyfluoroalkyl carboxylic acids (PolyFCAs)

The available evidence base for this group is very limited. Only two laboratory studies were identified, both conducted in activated sludge or enrichment cultures. No studies were found addressing their behaviour in the atmosphere, natural surface waters, sediments, or soils.

The limited data suggest that certain highly fluorinated short-chain acids (e.g., propionic, butanoic, and pentanoic acids) may undergo slow co-metabolic transformation in aerobic sludge. Unsaturated C4–C5 analogues appear to be somewhat more susceptible under anaerobic conditions. Reported pathways indicate stepwise defluorination and formation of less-fluorinated acids; however, complete mineralization has not been demonstrated, and the specific microorganisms or enzymes involved remain unidentified.

Given the very small number of studies and the absence of data from environmental field settings, the environmental fate of most PolyFCAs remains uncertain. The currently available results are insufficient to support extrapolation to full-scale wastewater treatment systems or natural environments.

Perfluoroalkane sulfonic acids (PFSAs)

The available literature indicates that several well-studied PFSAs, particularly PFOS and PFHxS, show little evidence of degradation under typical environmental conditions. They resist photolysis under natural light and show no meaningful transformation in unamended waters, sediments, or conventional wastewater treatment systems.

Laboratory studies have demonstrated that cleavage of C–S and C–F bonds can occur under highly controlled or engineered conditions, including systems involving specialized microbial cultures, reactive minerals, nanoparticles, elevated pH, or tightly controlled redox environments. These studies report formation of shorter-chain sulfonates, carboxylates, and fluoride. However, such conditions are not representative of most natural environments, and field-scale evidence is lacking. In addition, most studies focus on PFOS, with limited information available for other chain lengths or structural variants, and comprehensive fluorine mass balances are rarely reported.

Based on the currently available data, certain PFSAs, particularly those that have been most extensively studied, are frequently observed as stable end points in environmental systems, although the evidence base remains uneven across the broader PFSA group.

Perfluoroalkyl phosphonic acids (PFPAs)

Only a small number of studies were identified that examine the environmental degradation of perfluoroalkyl phosphonic acids (PFPAs). In the available literature, no abiotic or biotic degradation has been reported under the tested, environmentally relevant conditions. The reviewed studies indicate that PFPAs are non-volatile and hydrolytically stable, and they show no measurable transformation in aerobic sludge systems. Where removal has been observed, it appears to be attributable to sorption to biomass rather than metabolic degradation.

A comparative assessment in the literature (Z. Wang et al., 2016) concluded that, based on the limited data available, PFPAs are likely to exhibit high environmental persistence. However, given the small number of studies and limited range of test conditions, the current evidence base remains narrow.

Perfluoroalkyl phosphinic acids (PFPIAs)

Available data for PFPIAs are limited to a few laboratory studies. These indicate that PFPIAs are non-volatile and show no measurable abiotic degradation under typical environmental conditions. In soils, slow aerobic microbial cleavage of the P–C bond has been reported, forming phosphonic acids, with half-lives varying by chain length and sorption.

In aquatic systems, strong sorption to sediments appears to dominate their environmental distribution, with only limited evidence of partial transformation. No data were identified for wastewater treatment. Overall, the small evidence base suggests that sorption and sequestration may be more important than rapid degradation, though field data are lacking.

Per- and polyfluoroalkylether carboxylic acids (PFECAs)

For a small set of well-known PFECAs (including GenX-type substances), studies often report resistance to hydrolysis, natural photolysis, and microbial transformation in conventional tests, with measurable abiotic loss reported mainly under specific, sometimes non-representative conditions (e.g., nitrate-rich irradiation). Some soil observations suggest transformation of certain substituted PFECAs into more oxidized analogues. Ether linkages are frequently hypothesized as potential reaction sites, but clear evidence for ether scission under environmentally relevant conditions remains limited.

Per- and polyfluoroalkylether sulfonic acids (PFESAs)

Available studies on selected PFESAs, including F-53B (6:2 Cl-PFESA) and certain 2H-1:2 analogues, indicate limited evidence of degradation under environmentally relevant conditions. These substances are reported to be non-volatile and resistant to hydrolysis, natural photolysis, and biodegradation in water, soil, and conventional wastewater treatment tests.

Significant degradation has been observed only under artificially enhanced conditions, such as prolonged irradiation in nitrate-rich waters, suggesting that high radical flux is required to initiate transformation. Standard biodegradation assays and soil microcosm studies indicate that decreases in concentration are primarily driven by sorption rather than chemical or biological breakdown. Overall, the currently available data -restricted to a few representatives - suggest limited degradability under typical environmental conditions.

Perfluoroalkyl dicarboxylic acids (PFdiCAs)

Perfluoroalkyl dicarboxylic acids (PFdiCAs) are largely unstudied with respect to environmental degradation. One laboratory study suggests that the presence of two electron-withdrawing carboxyl groups may enhance defluorination under strongly reductive, non-environmental conditions. However, no studies were identified that assess their behaviour in the atmosphere, natural waters, soils, sediments, or wastewater treatment systems.

As a result, any assumptions regarding their environmental persistence are currently based on structural analogy to other fully fluorinated acids rather than on direct empirical evidence. This group therefore represents a clear data gap in the available literature.

Perfluoroalkane disulfonic acids (PFdiSAs)

No abiotic or biotic degradation studies were identified for perfluoroalkane disulfonic acids (PFdiSAs) in this review. Consequently, their environmental behaviour cannot be assessed on the basis of direct empirical evidence.

Current expectations regarding their stability are inferred from structural similarity to other fully fluorinated sulfonic acids. Based on this analogy, PFdiSAs would be anticipated to be non-volatile, resistant to hydrolysis and natural photolysis, and unlikely to undergo significant microbial degradation under typical environmental conditions. However, these assumptions are not supported by dedicated degradation studies, and this group therefore represents a notable data gap.

Perfluoroalkane sulfinic acids (PFSIAs)

Perfluoroalkane sulfinic acids (PFSIAs), such as perfluorooctane sulfinate (PFOSI), have been only sparsely investigated. Apart from an uncertain report of high-altitude photolysis, no reliable studies were identified demonstrating abiotic degradation under environmentally relevant conditions. One aerobic sludge study detected PFOSI as a transient intermediate formed during the transformation of a sulfonamide precursor to PFOS, with a reported half-life of less than one day. This finding suggests that sulfinates may be unstable under oxic, biologically active conditions and may oxidize further to more stable sulfonates.

Given the limited evidence, PFSIAs are best regarded as transformation intermediates that can convert into more persistent sulfonic acids, rather than as stable end products in environmental systems.

Fluorotelomer alcohols (FTOHs)

Fluorotelomer alcohols (n:2 FTOHs) are among the most extensively studied PFAS precursors. In contrast to many fully fluorinated acids, FTOHs are reported to be able to transform relatively rapidly in the environment. Because they are volatile, FTOHs readily enter the atmosphere, where they react with hydroxyl radicals and are oxidized within days, ultimately forming perfluoroalkyl carboxylic acids (PFCAs). In aerobic soils, sediments, and wastewater treatment systems, they also undergo stepwise microbial transformation via intermediate products such as FTCAs and FTUCAs, typically within days to weeks. Overall, FTOHs are best described as short-lived, mobile precursors that are efficiently converted into more stable fluorinated carboxylic acids, with limited evidence for extensive defluorination.

Other fluorotelomers and derivatives

A wide range of fluorotelomer-based precursors, including sulfonates (FTSAs), acrylates (FTAcS), iodides (FTIs), olefins (FTOs), and telomer acids (FTCAs and FTUCAs) have been reported to transform abiotically and/or biotically, ultimately forming perfluoroalkyl carboxylic acids (PFCAs). The degradation pathway depends strongly on molecular structure and functional groups. Volatile fluorotelomers (e.g., FTIs, FTOs, FTAcS) degrade mainly in the atmosphere within days to weeks via OH-initiated or photolytic reactions, forming aldehydes and reactive intermediates that yield PFCAs. Less volatile counterparts (e.g., FTSAs, FTCAs, FTUCAs) transform primarily in oxic soils, sediments, and activated sludge systems through stepwise oxidation, often with limited defluorination. Transformation rates vary widely: some compounds degrade within hours under aerobic conditions, while others degrade more slowly or persist under anaerobic conditions.

Overall, available evidence indicates that many fluorotelomer derivatives act as precursors to PFCAs. However, data remain limited for several derivatives, particularly regarding abiotic

degradation in water, behaviour under anoxic conditions, and performance in full-scale wastewater treatment systems.

Perfluoroalkanoyl fluorides (PACFs)

The available data on perfluoroalkanoyl fluorides are very limited and focus primarily on the short-chain compound $\text{CF}_3\text{C}(\text{O})\text{F}$. For this substance, photolytic lifetimes in the atmosphere are estimated to be very long, whereas hydrolysis is reported to occur on timescales of days to weeks, forming the corresponding perfluoroalkyl carboxylic acid (PFCA). No studies were identified that examine longer-chain related compounds in this class or assess abiotic or biotic degradation in water, sediments, or soils. As a result, the environmental fate of most PACFs remains insufficiently characterized.

Perfluoroalkyl iodides (PFAIs)

A small number of studies suggest that photolysis of perfluoroalkyl iodides (PFAIs) may lead to formation of perfluoroalkyl carboxylic acids (PFCAs) via cleavage of the carbon–iodine bond. However, quantitative data on reaction rates, product yields, or detailed transformation pathways are lacking. In addition, no studies were identified that investigate hydrolytic or microbial degradation. Consequently, the environmental fate of PFAIs remains poorly characterized.

Perfluoroalkane sulfonyl fluorides (PASFs)

Perfluoroalkane sulfonyl fluorides (PASFs) are produced by electrochemical fluorination and serve as precursors to perfluoroalkane sulfonamides. No laboratory or field studies were identified that directly assess their abiotic or biotic degradation.

One analytical investigation detected PFOSF and PFHxSF in soils up to two years after operations at a fluorochemical manufacturing facility had ceased, suggesting that these substances may persist under certain conditions. However, in the absence of dedicated degradation studies or identified transformation products, the environmental degradability and long-term fate of PASFs remain uncertain.

Per- and polyfluoroalkyl ethers

The limited available data on per- and polyfluoroalkyl ethers suggest low reactivity under typical atmospheric conditions. One study indicates resistance to direct photolysis and reaction with hydroxyl radicals. Another investigation examined chlorine-initiated atmospheric degradation of a methyl perfluoroalkylether and estimated lifetimes on the order of several years, with formation of intermediates such as carbonyl fluoride and perfluoroalkyl formates that may generate minor amounts of perfluoroalkyl carboxylic acids (PFCAs). No studies were identified that assess abiotic or biotic degradation of per- or polyfluoroalkyl ethers in water, sediments, or soils. As a result, information on their environmental fate outside the atmosphere remains very limited.

Perfluoroalkenes

Perfluoroalkenes include fully fluorinated alkenes such as $\text{CF}_2=\text{CF}_2$ and more complex structures containing perfluoroalkyl substituents. Available studies indicate that these substances can degrade in the atmosphere through reactions with hydroxyl radicals and chlorine atoms. These processes form intermediates such as carbonyl fluoride and perfluoroacyl fluorides, which can hydrolyse to ultra-short-chain perfluoroalkyl carboxylic acids, including trifluoroacetic acid (TFA). Reported atmospheric lifetimes vary widely, from about one day for tetrafluoroethylene to several weeks for certain perfluorobutenes, while more bulky or branched structures may persist longer. No studies were identified that assess abiotic or biotic degradation of perfluoroalkenes in water, sediments, or soils. Based on the available evidence, perfluoroalkenes are primarily characterized as volatile atmospheric precursors to ultra-short-chain acids, with limited information on their behaviour in other environmental compartments.

Perfluoroalkanes

Perfluoroalkanes such as tetrafluoromethane (CF_4), hexafluoroethane (C_2F_6), and octafluoropropane (C_3F_8) have been studied primarily in the context of atmospheric chemistry. Available laboratory kinetics and modelling studies indicate extremely long atmospheric lifetimes, ranging from thousands to tens of thousands of years. These substances show little reactivity toward common atmospheric oxidants and are not known to undergo degradation by sunlight, water, or microorganisms under environmental conditions.

Due to their high volatility and very low water solubility, environmental releases are expected to partition predominantly to the atmosphere. No studies were identified that investigate biotic degradation pathways or measurable transformation rates in other environmental compartments.

Semi-fluorinated alkanes (SFAs)

For semi-fluorinated alkanes (SFAs) with larger perfluorinated and hydrocarbon segments ($n, m > 2$), no experimental studies were identified that evaluate how their bi-segment structure influences environmental degradability. For smaller molecules ($n, m \leq 2$), classification overlaps with hydrofluorocarbons (HFCs), which are addressed separately in this report. Beyond these smaller compounds, the environmental fate and degradation behaviour of SFAs remain largely uncharacterized. This group therefore represents a clear data gap in the current understanding of PFAS degradation.

HCFCs / HFCs / HFEs / HFOs

Hydrochlorofluorocarbons (HCFCs), hydrofluorocarbons (HFCs), hydrofluoroethers (HFEs), and hydrofluoroolefins (HFOs) have been widely studied in the atmosphere. These volatile substances degrade primarily through reactions with hydroxyl radicals and, for longer-lived compounds, through stratospheric photolysis. Reported atmospheric lifetimes range from days (for many HFOs) to several decades (for some HFCs).

Atmospheric oxidation typically forms intermediates such as trifluoroacetyl fluoride, trifluoroacetaldehyde, trifluoromethanol, and carbonyl fluoride. These intermediates can hydrolyse to hydrogen fluoride (HF), carbon dioxide (CO_2), and ultra-short-chain perfluoroalkyl carboxylic acids, most notably trifluoroacetic acid (TFA). Reported TFA yields vary widely depending on molecular structure and atmospheric conditions.

Because these substances are highly volatile, atmospheric oxidation dominates their environmental fate, and no studies on biotic degradation were identified.

Perfluoroalkyl and semi-fluorinated ketones

Three studies were identified that examine the atmospheric degradation of perfluoroalkyl and semi-fluorinated ketones. For compounds such as perfluoro-2-methyl-3-pentanone, photolysis appears to be the dominant atmospheric pathway, producing intermediates including carbonyl fluoride and trifluoroacetyl fluoride within days. These intermediates subsequently hydrolyse to form short-chain perfluoroalkyl carboxylic acids, such as trifluoroacetic acid (TFA).

In aqueous systems, the fluoroketone CF-5K has been reported to hydrolyse rapidly to TFA, suggesting that abiotic processes play a key role in determining environmental fate. No studies were identified that investigate biotic degradation pathways for this group.

Per- and polyfluoroalkyl alcohols (non-FTOH)

Available evidence suggests that per- and polyfluoroalkyl alcohols can transform rapidly in the atmosphere. Some perfluoroalkyl alcohols, such as trifluoromethanol, are unstable and decompose readily, while polyfluoroalkyl alcohols (e.g., FESOH) react within days with atmospheric oxidants to form perfluoroacyl fluorides and ultimately short-chain perfluoroalkyl carboxylic acids. No studies

were identified that assess abiotic or biotic degradation in natural waters or soils. One sludge study reported partial degradation of a long-chain, partially fluorinated alcohol to a mixture of fluorinated carboxylic acids and other polyfluorinated products.

Overall, per- and polyfluoroalkyl alcohols are best described as reactive intermediates in the atmosphere, where they are converted to short-chain fluorinated acids. Evidence for biodegradation in other environmental compartments is currently very limited.

Side-chain fluorinated aromatics

Side-chain fluorinated aromatics are reported to undergo abiotic transformation in aqueous environments, including photolysis and hydrolysis. Laboratory studies show that CF₃-substituted phenols can release fluoride and form trifluoroacetic acid (TFA) and other perfluoroalkyl carboxylic acids (PFCAs), with degradation generally favoured under acidic to neutral conditions and in the presence of oxidants.

Although biodegradation data are limited, some studies indicate that specialized microbial consortia or plant–soil systems can remove CF₃ groups, again yielding fluoride and short-chain PFCAs such as TFA. TFA has been identified as a predominant degradation product for several side-chain fluorinated aromatics used as pesticide active substances.

Overall, available evidence suggests that many side-chain fluorinated aromatics can transform in water and biological systems, often forming stable ultra-short-chain acids as final products. No experimental data were identified for atmospheric degradation.

Perfluoroalkyl tert-amines

Very limited information is available on the environmental degradation of perfluoroalkyl-tert-amines. One kinetic study suggests that even the smallest homologues have extremely long atmospheric lifetimes because they are not efficiently removed by solar UV radiation or reactions with excited oxygen atoms.

No laboratory or field studies were identified that assess degradation in water, soils, or biological systems. Based on their fully fluorinated carbon skeleton and sterically shielded tertiary amine structure, biotic cleavage appears unlikely. Current evidence therefore suggests that these substances behave as highly stable, predominantly volatile compounds, with removal occurring primarily through very slow atmospheric processes.

Perfluoroalkane sulfonamides (FASAs)

In contrast to many other PFAS groups, several studies demonstrate that perfluoroalkane sulfonamides (FASAs) can undergo environmental transformation, ultimately forming stable perfluoroalkyl acids (PFAAs). In the atmosphere, FASAs typically persist for several weeks (approximately 20–50 days) before reacting with hydroxyl radicals. Oxidation produces sulfonyl ketones and aldehydes that fragment to ultra-short-chain PFCAs, with limited evidence for significant direct formation of sulfonic acids. In oxic surface waters, similar hydroxyl radical-driven processes occur. In marine sediments, microbial degradation of compounds such as EtFOSE has been observed, with reported half-lives ranging from weeks to months depending on temperature. In soils and plant–soil systems, FASAs undergo stepwise microbial transformation including N-alkyl oxidation, dealkylation, and sulfonamide hydrolysis, ultimately forming perfluoroalkane sulfonic acids (PFSA), such as PFOS (C8) or PFHxS (C6). Reported half-lives range from weeks to years, depending on environmental conditions. In wastewater treatment systems, particularly under warm and oxic conditions, FASAs is reported to be transformed within days to weeks. In contrast, anoxic or methanogenic conditions slow or inhibit this process.

Perfluoroalkyl phosphate esters (PAPs)

Available data indicate resistance to abiotic hydrolysis in water, with reported half-lives of years to decades, and no information was identified for atmospheric degradation. Under oxic biological conditions, however, PAPs are reported to be transformed. In activated sludge and aerobic soils, mono- and di-PAPs (e.g., 6:2 and 8:2) hydrolyse to fluorotelomer alcohols, which are further oxidized to form PFCAs such as PFHxA and PFOA. Transformation rates vary by chain length and environmental setting. Studies in wastewater treatment systems confirm that PAPs can act as precursors to PFCAs. Overall, evidence suggests that PAPs are stable under abiotic conditions but can serve as biologically mediated sources of fluorotelomer alcohols and PFCAs in oxic environments.

Other non-polymeric PFAS

The category of “other” non-polymeric PFAS includes a diverse range of substances, such as perfluoro-aldehydes, polyfluoro-amides, fluorinated acrylates and methacrylates, and certain fluorinated gases. In the atmosphere, many of these compounds are relatively short-lived (hours to weeks) and undergo oxidation or photolysis, often forming ultra-short-chain PFCAs such as TFA, PFPrA, and PFBA. In aquatic and soil systems, available data are limited and highly compound-specific. Some polyfluoro-amides and trifluoromethyl acrylates have been reported to degrade slowly to PFCAs, while certain small sulfonates may undergo more extensive microbial defluorination under specialized conditions. In aerobic wastewater sludge, fluorinated ether or thio-ether alcohols are reported to be able to be transformed to intermediate ether-PFCAs that appear relatively stable.

Given the chemical diversity within this group, general conclusions on environmental degradability and transformation pathways are not currently possible.

Fluoropolymers

Fluoropolymers such as PTFE, PVDF, FEP, ETFE, and PFA are generally reported to be resistant to degradation under normal environmental conditions. Due to their high molecular weight and non-volatile nature, they show little reactivity toward photolysis, hydrolysis, oxidation, or microbial processes. Laboratory studies conducted under OECD test protocols, particularly for PTFE, have shown no measurable degradation or fluoride release in water, soils, sediments, or sewage sludge under both aerobic and anaerobic conditions. Degradation of the polymer backbone has been observed only under extreme thermal conditions, producing small perfluorinated fragments.

Based on the available evidence, fluoropolymers are considered chemically stable in environmental compartments relevant to their release and occurrence, including surface waters, groundwater, soils, sediments, sewage treatment systems, and the atmosphere under ambient conditions. This conclusion refers to chemical degradation processes and does not address potential physical weathering, mechanical fragmentation, or particle formation.

Perfluoropolyethers (PFPEs)

Available evidence suggests that perfluoropolyethers (PFPEs) are resistant to hydrolysis, photolysis, oxidation, and both aerobic and anaerobic biodegradation. Commercial PFPE variants (e.g., PFPE-D, PFPE-K, PFPE-Y, PFPE-Z) are also thermally stable at temperatures far exceeding typical environmental conditions. However, the number of dedicated environmental degradation studies remains limited. Based on available data and structural considerations, including comparisons with related perfluoroether-based substances, PFPEs are generally regarded as highly resistant to abiotic and biotic degradation under environmental conditions. Nonetheless, the empirical evidence base for this group is still relatively narrow.

Perfluoroalkylether side-chain fluorinated polymers

No experimental studies were identified that investigate the environmental degradation of perfluoroalkylether side-chain fluorinated polymers. Based on analogy with other side-chain fluorinated polymers, cleavage of the perfluoroalkylether side chain from the polymer backbone

may be possible under certain environmental conditions, potentially releasing the corresponding perfluoroalkylether moiety. However, in the absence of dedicated studies, the degradability and environmental fate of this group remain uncertain and represent a clear data gap.

Side-chain fluorinated polymers

Side-chain fluorinated polymers (SCFPs), such as fluorinated acrylate, methacrylate, urethane, and oxetane polymers, are reported to potentially release per- and polyfluorinated side chains under environmental conditions. Studies show that polymers containing ester linkages may slowly hydrolyse over long timescales, forming fluorotelomer alcohols and subsequently PFCAs.

In soils, microbial activity and other biological factors are expected to accelerate this process, but estimated timescales for substantial side-chain removal range from years to centuries. SCFPs may therefore act as long-term, low-level sources of PFCAs. No data were identified for atmospheric degradation or for biotic degradation in aquatic or sediment environments.

Abbreviations

Abbreviation	Explanation
2-Cl-4-TFMP	2-chloro-4-(trifluoromethyl)phenol
2-TFMP	2-(trifluoromethyl)phenol
3-OH-5:3 FTCA	3-hydroxy-5:3 saturated fluorotelomer carboxylate
3-OH-7:3 FTCA	3-hydroxy-7:3 saturated fluorotelomer carboxylate
3-TFMP	3-(trifluoromethyl)phenol
4-TFMP	4-(trifluoromethyl)phenol
5:2 sFTOH	5:2 secondary fluorotelomer alcohol
5:3 acid	5:3 fluorotelomer carboxylic acid
6:2 diPAP	6:2 fluorotelomer phosphate diester
6:2 FTAB	fluorotelomer sulfonamidoalkyl betaine
6:2 FTCA	6:2 fluorotelomer saturated carboxylic acid
6:2 FTNO	6:2 fluorotelomer sulfonamide amine oxide
6:2 FtS	6:2 fluorotelomer sulfonate
6:2 FTSA	6:2 fluorotelomer sulfonate
6:2 FtSiAoS	6:2 fluorotelomer sulfinyl amido sulfonate
6:2 FtTAoS	6:2 fluorotelomer thioether amido sulfonate
6:2 FtTP	6:2 fluorotelomer thioether propionate
6:2 FtTPIA	6:2 fluorotelomer thioether propanoylalaninate
6:2 FtTPIAA	6:2 fluorotelomer thioether propanoylanylalaninate
6:2 FtTPoP	6:2 fluorotelomer thioether propanoyloxy propanoate
6:2 FTUA	6:2 fluorotelomer unsaturated carboxylic acid
6:2 FTUCA	6:2 fluorotelomer unsaturated carboxylic acid
6:2 monoPAP	6:2 fluorotelomer phosphate monoester
7:3 TA	7:3 taurine amide
7:3 U Amide	7:3 unsaturated amide
8:2 diPAP	8:2 perfluoroalkyl phosphate diester
8:2 FTAC	8:2 fluorotelomer acrylate
8:2 FTMAC	8:2 fluorotelomer methacrylate
ABTS	2,2'-azino-bis(3-ethylbenzothiazoline-6-sulfonic acid)
AD	anaerobic digestion
ADONA	ammonium dodecafluoro-3H-4,8-dioxanonoate
AFFF	aqueous film forming foam (firefighting foam)
AFTP	acrylate fluorotelomer-based polymer

Abbreviation	Explanation
AmPr-FAAd	N-dimethyl ammonio propyl perfluoroalkane amide
AmPr-FASA	N-dimethyl ammonio propyl perfluoroalkane sulfonamide
AmPr-FASA-PrA or TAmPr-FASA-AA	N-dimethyl ammonio propyl perfluoroalkane sulfonamido propanoic acid or n-trimethyl ammonio propyl perfluoroalkane sulfonamide acetic acid
AOB	ammonia-oxidizing bacteria
AQDS	9,10-anthraquinone-2,6-disulfonic acid disodium
ASW	artificial surface water
BOD	biological oxygen demand
BTEX	benzene, toluene, ethylbenzene and xylene
BTFMA-AS	ω -(bis(trifluoromethyl)amino)alkane-1-sulfonates
BTP	2-bromo-3,3,3-trifluoropropene
C4-FN	heptafluoroisobutyronitrile
CAS	Chemical Abstracts Service
CBT	closed bottle test
CEt-AmPr-FAAd	N-carboxy ethyl dimethyl ammonio propyl perfluoroalkane amide
CEt-AmPr-FASA-PrA	N-carboxy ethyl dimethyl ammonio propyl perfluoroalkane sulfonamido propanoic acid
CI-PFPECAs	chloro-perfluoropolyether carboxylates
CW(s)	constructed wetland(s)
CW-MFC	constructed wetland coupled microbial fuel cell
CVOC	chlorinated volatile organic compound
DEMNUM	carboxylic acid derivatives of PFPE-D
DFA	difluoro acetic acid
DFBD	2,2-fluoro-1,3-benzodioxole
DFMS	difluoromethane sulfonate
DGBE	diethyl glycol monobutyl ether
DOC	dissolved organic carbon
DT₅₀	half life
DTFA	degradable telomer fluoroalcohol
ECF	electrochemical fluorination
ECHA	European Chemical Agency
ECOHR(s)	enzyme-catalysed oxidative humification reaction(s)
EFSA	European Food Safety Authority
EMPA	Eidgenössische Materialprüfungs- und Forschungsanstalt

Abbreviation	Explanation
ETFE	ethylene tetrafluoroethylene
EtFOSA	N-ethyl perfluorooctane sulfonamide
EtFOSAA	N-ethyl perfluorooctane sulfonamidoacetate
EtFOSE	N-ethyl perfluorooctane sulfonamidoethanol
EU	European Union
F-53B	chlorinated polyfluoroalkyl ether sulfonates
FASAs	Perfluoroalkane sulfonamides and their derivatives (precursors)
FBBR	fixed-bed bioreactor
FEP	fluorinated ethylene propylene
FESCA	polyfluoroalkyl ether thioether carboxylic acid
FESOH	polyfluoroalkyl ether thiol alcohol
FESOS	polyfluoroether thioether(S)-oate) sulfonate
FHEA	2-perfluorohexyl ethanoic acid
FIA-HRMS	high-resolution mass spectrometry in flow injection analysis mode
FLX	fluoxetine
FOSA	perfluorooctane sulfonamide
FOSAA	perfluorooctane sulfonamide acetate
FOSE	fluor octane sulfonamidoethanol
FPePA	3-perfluoropentyl propanoic acid
FPs	fluoropolymers
FTAC	fluorotelomer acrylate
FTAL(s)	fluorotelomer aldehyde(s)
FtCA	5:3 fluorotelomer carboxylic acid, fluorotelomer carboxylates
FTCA(s)	fluorotelomer carboxylic acid(s)
FTEO	fluorotelomer ethoxylate
FTEOC	FTEO carboxylates
FTIs	fluorotelomer iodides
FTMAC	fluorotelomer metacrylate
FTMS	direct-infusion fourier transform mass spectrometry with Orbitrap MS
FTNO(s)	fluorotelomer sulfonamide amine oxide(s)
FTO	fluorotelomer Olefin
FTOH(s)	fluorotelomer alcohol(s)
FTOH-Gluc	fluorotelomer alcohol glucuronide
FTOHs	fluorotelomer alcohols

Abbreviation	Explanation
FTOH-Sulf	fluorotelomer alcohol sulfate
FTP	fluorotelomer-based polymer
FTP 3	side-chain fluorotelomer acrylate polymer (45-60% C8, 25-35% C10, 7-15% C12)
FTP 4	side-chain fluorotelomer acrylate polymer (50% C8, 30% C10, 10% C12)
FTSA(s)	fluorotelomer sulfonate(s)
FtSiP	6:2 fluorotelomer sulfinyl propanoate
FTU	toluene-2,4-di(8:2fluorotelomerurethane), fluorotelomer urethane
FTUA	fluorotelomer unsaturated acid
FTUAA	fluorotelomer unsaturated acetic acid
FTUAL	fluorotelomer unsaturated aldehydes
FTUAL-GSH	unsaturated fluorotelomer aldehyde glutathione conjugate
FTUCA	fluorotelomer unsaturated carboxylic acid
FTUCA-GSH	unsaturated fluorotelomer carboxylate glutathione conjugate
FTUSA	fluorotelomer unsaturated sulfonic acid
GC-MS	gas chromatography mass spectrometry
GLP	good laboratory practice
HBA	hydroxybenzoic acid
HBT	1-hydroxybenzotriazole
HCFCs	hydrofluorochlorofluorocarbons
HCFE	hydrochlorofluoroether
HCFOs	hydrochlorofluorocarbons
HET	heterotrophic
HF	hydrogen fluoride
HFCs	hydrofluorocarbons
HFEs	hydrofluoroethers
HFOs	hydrofluoroolefins
HFPO-DA	perfluoro-2- methyl-3-oxahexanoic acid, hexafluoropropylene Oxide Dimer Acid
HMU	hexamethylene-1,6-di(8:2fluorotelomerurethane)
H-PFCA	hydrogen-substituted perfluoroalkyl carboxylic acid
HPFOA	heptafluoropropylene Oxide Dimer Acid
HRMS	high resolution mass spectrometry
HRP	horseradish peroxidase
HRT	hydraulic retention time (e.g. in wastewater treatment)

Abbreviation	Explanation
LC-MS	liquid chromatography mass spectrometry
L-EtFOSA	linear-N-ethyl-perfluorooctanesulfonamide
L-FOSA	linear-perfluorooctanesulfonamide
LMWS	low molecular weight substances
L-PFOS	linear-perfluorooctanesulfonic acid
M-AmPr-FASA	N-methyl ammonio propyl perfluoroalkane sulfonamide
MEC	microbial electrolysis cell
MES	microbial electrosynthesis system
MFA	monofluoroacetate
n:2 diPAP	n:2 perfluoroalkyl phosphate diester
n:2 FTAA	n:2 fluorotelomer acetic acid
n:2 FTAB	n:2 fluorotelomer sulfonamidoalkyl betaine
n:2 FTAc	n:2 fluorotelomer acrylate
n:2 FTOH	fluorotelomer alcohol
n:2 Ketone	fluorotelomer ketone
n:2 monoPAP	n:2 fluorotelomer phosphate monoester
n:2 sFTOH	fluorotelomer secondary alcohol
na	not applicable
N-EtFASAA	N-ethyl perfluoroalkane sulfonamido acetic acid
NFLX	norfluoxetine
N-MeFASAA	N-methyl perfluoroalkane sulfonamido acetic acid
NMR	Nuclear Magnetic Resonance
NOB	nitrite-oxidizing bacteria
NOM	natural organic matter
nZVI	nano zero valent iron
OAmPr-FAAd	N-oxide dimethyl ammonio propyl perfluoroalkane amide
OAmPr-FASA	N-oxide dimethyl ammonio propyl perfluoroalkane sulfonamide
OECD	Organization for Economic Co-operation and Development
OPPTS	Office of Prevention, Pesticides and Toxic Substances
PAA	perfluoroalkyl Acid
PACFs	perfluoroalkanoyl fluorides
PAPs	polyfluoroalkyl phosphates
PASFs	perfluoroalkane sulfonyl fluorides
PFA	perfluoroalkoxy (alkane)

Abbreviation	Explanation
PFAA	perfluoroalkyl acids
PFAI(s)	perfluoroalkyl iodide(s)
PFAS	perfluoroalkyl substances
PFBA	perfluorobutanoic acid
PFBS	perfluorobutane sulfonic acid
PFBSI	perfluorobutane sulfonimide
PFCA	perfluorocarboxylic acids
PFCAs	perfluoroalkyl carboxylic acids
PFDA	perfluorodecanoic acid
PFdiCAs	perfluoroalkyl dicarboxylic acids
PFdiSAs	perfluoroalkane disulfonic acids
PFDPA	perfluorodecylphosphonic acid
PFECAs	per- and polyfluoroalkylether carboxylic acids
PFESAs	per- and polyfluoroalkylether sulfonic acids
PFHpA	perfluoroheptanoic acid
PFHpS	perfluoroheptansulfonic acid
PFHxA	perfluorohexanoic acid
PFHxS	perfluorohexansulfonic acid
PFHxSA	perfluorohexane sulfonamide
PFHxSAm	perfluorohexane sulfonamido propyl tertiary amine
PFHxSAmS	perfluorohexane sulfonamido propyl quaternary amine
PFNA	perfluorononanoic acid
PFOA	perfluorooctanoic acid
PFOAAm	perfluorooctanamide
PFOAAm	3-dimethylamino perfluorooctaneamide
PFOAB	perfluorooctane amido betaine
PFOAB	perfluorooctane amido betaine
PFOANO	perfluorooctanoyl N-oxide
PFOPA	perfluorooctylphosphonic acid
PFOS	perfluorooctansulfonic acid
PFOSA	perfluorooctane sulfonamide
PFOSAm	3-dimethyl amino perfluorooctanesulfonamide
PFOSB	perfluorooctane sulfonamido betaine
PFOSF	perfluorooctane sulfonyl fluoride
PFOSI	perfluorooctane sulfonimide
PFOSNO	perfluorooctane sulfonamide N-oxide

Abbreviation	Explanation
PFPAs	perfluoroalkyl phosphonic acids
PFPE(s)	perfluoropolyether(s)
PFPeA	perfluoropentanoic acid
PFPeS	perfluoropentansulfonic acid
PFPIAs	perfluoroalkyl phosphinic acids
PFSA	perfluorosulfonic acid
PFSAs	perfluoroalkane sulfonic acids
PFSIAs	perfluoroalkane sulfinic acids
PFUnDA	perfluoroundecanoic acid
PolyFCAs	polyalkyl carboxylic acids
POP	persistent organic pollutant
PP	polypropylene
PTFE	polytetrafluoroethylene
PVDF	polyvinylidene fluoride
RAC	Risk Assessment Committee
SAmPAP	perfluorooctane sulfonamidoethanol based phosphate diester
SEAC	Committee for Socio-Economic Analysis
SFA	sulfur-free acetate medium
SFAs	semifluorinated alkanes
SPE	solid phase extraction
SRM	selected reaction monitoring
TAmPr-FASA	N-trimethyl ammonio propyl perfluoroalkane
TCE	trichloroethylene
TDAEE	toluene-2,4-dicarbamicaciddiethylester
TES	2,2,2-trifluoroethane sulfonate
TFA	trifluoroacetic acid
TFM	3-trifluoromethyl-4-nitrophenol
TFMAA	2-(trifluoromethyl)acrylic acid
TFMC	trifluoromethyl carbonate
TFMHxA	6-trifluoromethoxyhexanoic acid
TFMHxOH	6-(trifluoromethoxy)-hexan-1-ol
TFMP	trifluoromethylphenol
TFMPROH	3-(trifluoromethoxy)-propan-1-ol
TFMS	trifluoromethane sulfonate
ThOD	theoretical oxygen demand
TOP	total oxidizable precursor

Abbreviation	Explanation
TP	transformation product
uFTCA-Scys	unsaturated fluorotelomer carboxylate cysteine conjugate
uFTCA-SCysGly	unsaturated fluorotelomer carboxylate cysteinylglycine conjugate
uFTCA-SCysNAcetyl	unsaturated fluorotelomer carboxylate N-acetylcysteine conjugate
uFTOH-GSH	unsaturated fluorotelomer alcohol glutathione conjugate
uFTOH-Scys	unsaturated fluorotelomer alcohol cysteine conjugate
uFTOH-SCysGly	unsaturated fluorotelomer alcohol cysteinylglycine conjugate
uFTOH-SCysNAcetyl	unsaturated fluorotelomer alcohol N-acetylcysteine conjugate
UFTP	urethane fluorotelomer-based polymer
UHPLC-HRMS	ultra-high-performance liquid chromatography coupled to high-resolution accurate mass Orbitrap mass spectrometry
UK	United Kingdom
UNEP	United Nations Environment Programme
U-PFCA	unsaturated perfluoroalkyl carboxylic acid
UPFOA	unsaturated PFOA
US	United States
USEPA	United States Environmental Protection Agency
UV	ultraviolet
VFCW	vertical flow constructed wetlands
WWTP	waste water treatment plant

Glossary

Within the scientific literature, terms related to degradation of substances in the environment are not always used unambiguously or precisely. As an example, the term “removal” is apparently sometimes used synonymously with “degradation”, while in other cases, different removal processes (e.g. abiotic and biotic degradation, sorption and volatilisation) are specified and accounted for separately.

To the extent possible and based on the information provided in literature sources analysed, the degradation related terms have been harmonised and are used consistently in the current report. Thus, in the current report, terms are used with the following meanings:

Absorption is the process by which a substance is taken up in bulk phase of another material (e.g., uptake into biological tissues or organic matter). Absorption means three-dimensional incorporation rather than surface association.

Adsorption means the accumulation of substances at the surface of a solid or liquid phase.

Aerobic describes conditions or processes occurring in the presence of molecular oxygen (O₂).

Anaerobic means conditions or processes occurring in the absence of molecular oxygen (O₂).

Abiotic refers to non-biological processes, including chemical, physical, or photochemical processes.

Bioavailability is the rate and extent to which a substance can be taken up by an organism and is available for metabolism or interaction with biologically significant receptors. Bioavailability involves both release from a medium (if present) and absorption by an organism.

Biodegradation is degradation mediated by biological activity of microorganism or other biota. Biodegradation typically implies enzymatic processes. Biodegradation does not imply complete degradation.

Biotic refers to processes or influences involving living organisms, including microorganisms, plants, fungi, or other biota.

Biotransformation means a biologically mediated modification of a substance, which can be a biodegradation or another biologically mediated conversion into another structure.

Decarboxylation is a chemical reaction involving the removal of a carboxyl group (–COO[–]) from a molecule, typically releasing carbon dioxide (CO₂).

Decomposition is a general term describing the breakdown of a compound into simpler substances, without specifying the mechanisms or pathways.

Degradation is a chemical transformation that involves structural breakdown of a substance, resulting in loss of its original molecular identity.

Defluorination is the cleavage of carbon–fluorine (C–F) bonds, resulting in the release of fluoride ions. Defluorination is a critical indicator of destruction for fluorinated organic compounds due to the high stability of C–F bonds.

Hydration is the chemical reaction involving the addition of water to a molecule without bond cleavage by water.

Hydrolysis is the chemical reaction in which water cleaves susceptible chemical bonds, resulting in bond scission and formation of new functional groups.

Hydrogenation is a reduction reaction involving the addition of hydrogen (H_2) to a molecule. Hydrogenation can alter unsaturation and, under specialized conditions, promote bond weakening or cleavage.

Mineralization refers to the degradation of an organic molecule into inorganic end products. In the scientific literature, the term “mineralization” is sometimes used to describe both partial and complete breakdown processes. In some cases, it may refer to the release of inorganic constituents (e.g., fluoride ions) while organic intermediates are still present.

In this report, however, the term **mineralization** is used to denote **complete (full) mineralization**, i.e., the full breakdown of the organic molecule into inorganic end products with no residual organic structure remaining. In the case of PFAS, complete mineralization means that no PFAS or other fluorinated organic compounds remain as degradation products. Instead, only inorganic end products are formed, such as carbon dioxide (CO_2), hydrogen fluoride (HF), fluoride ions (F^-), and, where applicable, water and other simple inorganic species.

Ozonolysis is an abiotic oxidative reaction involving ozone (O_3) that cleaves reactive functional groups or unsaturated bonds.

Photochemical transformations are chemical modification initiated by light-induced reactions, either through direct photon absorption or via photochemically generated reactive species. Photochemical transformation does not necessarily result in degradation.

Photocatalysts are materials that accelerate photochemical reactions by generating reactive species upon light absorption. Photocatalysts enable reaction pathways that are otherwise energetically unfavourable under ambient conditions.

Photolysis is the direct cleavage of chemical bonds by photon absorption, typically involving ultraviolet or visible radiation. Photolysis requires that the compound absorbs light at relevant wavelengths.

Removal usually means the physical elimination of a substance from a given medium without chemical alteration, such as through filtration, adsorption, or phase separation. However, in several of the degradation studies reviewed in this study, ‘removal’ of the parent substances was observed, and degradation was mentioned among possible removal pathways. Therefore, the term removal in this report barely describes the loss of the parent compound, without further specifying potential removal pathways such as sorption, transformation or volatilization.

Sorption is a collective term covering both adsorption and absorption, describing the partitioning of substances between phases without chemical modification.

Sequestration is the physical isolation of a substance within a matrix, making the substance less bioavailable and less mobile.

Transformation is any chemical modification of a substance that converts it into another chemical form. While transformation involves structural changes at the molecular level, it does not necessarily imply extensive fragmentation or complete degradation to inorganic constituents. The resulting product may retain substantial portions of the original molecular framework.

Volatilization is the transfer of a substance from a liquid or solid phase into the gas phase.

1. About this report

Widespread detection of various PFAS in water, soil, air, wildlife, and human biomonitoring studies has prompted an important scientific question: **to what extent do PFAS degrade in the natural environment, if at all, and under what conditions?**

This report is written to address this question through a careful and balanced evaluation of the peer-reviewed literature with the aim to critically assess what is known about environmental degradation processes, what mechanisms have been demonstrated, and where uncertainties remain.

By gathering and analysing data from various sources, this report summarizes the current state of knowledge on the degradation processes relevant for different PFAS groups in natural contexts. While commercial or laboratory-scale destruction techniques designed to destroy PFAS in optimized and artificially enhanced conditions (e.g., high temperature, advanced oxidation systems) were excluded from the scope of this assessment, degradation processes occurring in biological wastewater treatment plants (WWTPs) were included. Although sewage treatment plants are engineered technical systems rather than natural environmental compartments, they are included in this assessment because they represent a key interface between anthropogenic releases and the natural environment. Many substances enter environmental compartments via wastewater, and biological wastewater treatment systems therefore constitute an important pathway linking emissions to surface waters and, through sludge application, to soils. If no measurable degradation occurs under these comparatively favourable biodegradation conditions, it is generally considered unlikely that substantial degradation would occur in natural surface waters or soils. For these reasons, results from wastewater treatment simulations are interpreted alongside data from natural compartments when assessing environmental stability. It is, however, important to note that a clear distinction between naturally occurring degradation processes and artificially induced methods is not always straight forward.

Chapter 2 provides the scientific background in a condensed and easy to understand way. This chapter is particularly important for understanding the inherent complexity of the topic. It explains that the substances covered in this report represent a highly diverse set of chemistries, varying in chain length, degree of fluorination, functional groups, and molecular architecture (e.g., acids, sulfonates, alcohols, amides, ethers, polymers). It also emphasizes that degradation itself is not a single, uniform process. By outlining both substance complexity and environmental complexity, Chapter 2 prepares the reader to interpret the subsequent chapters with appropriate nuance. It clarifies why consistent categorization (abiotic versus biotic degradation, and subdivision by environmental compartment) is necessary to navigate the literature.

Chapter 3 describes the methodology for data collection, including how substances were defined and grouped. This allows the reader to understand how peer-reviewed studies were selected, categorized, and evaluated.

Chapter 4 presents key findings and highlights data availability and data gaps, providing a synthesis before moving into detailed evaluations.

Chapters 5 - 35 constitute the core of the report. Each chapter addresses a specific substance group and follows a consistent structure, distinguishing between abiotic and biotic degradation pathways and further subdividing these into environmental compartments such as the atmosphere, water, soil, sediments, and wastewater treatment plants (WWTPs). This standardized format allows readers to compare degradation behavior across chemical classes and environmental media. Readers seeking information on a specific group can navigate directly to the relevant chapter, while those interested in a particular degradation process or environmental compartment can consult corresponding subsections across multiple chapters.

2. Background of the project

2.1 The PFAS universe and the accompanying challenges

Per- and polyfluoroalkyl substances (PFAS) comprise a broad and structurally diverse group of synthetic organofluorine compounds characterised by the presence of highly stable carbon–fluorine (C–F) bonds. These bonds confer high thermal stability, chemical resistance, and surface-active properties, which have led to widespread use of PFAS in industrial applications and consumer products, including firefighting foams, surface coatings, textiles, food packaging, electronics, and certain pharmaceutical and agrochemical formulations.

Regulatory scrutiny

Over recent decades, PFAS have become a major focus of regulatory and scientific concern. The stability that makes these substances technically valuable also renders some of them highly persistent in the environment. Certain PFAS resist degradation under typical environmental conditions, can accumulate in living organisms, and may undergo long-range transport via air and water. Some members of this group have been associated with adverse effects on human health and wildlife, including developmental, immunological, and endocrine outcomes.

Regulatory frameworks in multiple jurisdictions have therefore begun to address PFAS either individually or as classes. For example, substances such as Perfluorooctanoic acid and Perfluorooctane sulfonic acid have been listed under the Stockholm Convention on Persistent Organic Pollutants due to their persistence, bioaccumulation, toxicity, and long-range environmental transport potential. In the European Union, PFAS are subject to increasing restrictions under REACH, including an ongoing proposal for a broad group restriction.

Given this regulatory momentum, understanding PFAS degradation, including transformation pathways, rates, and products is central to environmental risk assessment and management.

Chemical Diversity and Classification of PFAS

PFAS do not represent a single chemical entity but rather a vast chemical universe encompassing thousands of individual substances. They can be broadly divided into **non-polymeric** and **polymeric** PFAS, and occur in gaseous, liquid, and solid forms.

Non-polymeric PFAS include perfluoroalkyl acids and a wide range of neutral and precursor compounds. Important substance groups include:

- Perfluoroalkyl carboxylic acids (PFCAs)
- Polyalkyl carboxylic acids (PolyFCAs)
- Perfluoroalkane sulfonic acids (PFSAAs)
- Perfluoroalkyl phosphonic acids (PFPAAs)
- Perfluoroalkyl phosphinic acids (PFPIAs)
- Per- and polyfluoroalkylether carboxylic acids (PFECAs)
- Per- and polyfluoroalkylether sulfonic acids (PFESAAs)
- Perfluoroalkyl dicarboxylic acids (PFdiCAs)
- Perfluoroalkane disulfonic acids (PFdiSAs)
- Perfluoroalkane sulfinic acids (PFSIAs)
- Fluorotelomer alcohols (FTOHs)
- Other Fluorotelomers and derivatives
- Perfluoroalkanoyl fluorides (PACFs)
- Perfluoroalkyl iodides (PFAIs)
- Perfluoroalkane sulfonyl fluorides (PASFs)
- Per- and polyfluoroalkylethers
- Perfluoroalkenes
- Perfluoroalkanes
- Semifluorinated alkanes (SFAs)

- Hydrofluorocarbons/ -ethers / -olefins / -chlorofluorocarbons (HFCs/ HFEs/ HFOs/ HCFCs)
- Perfluoroalkyl and semi-fluorinated ketones
- Per- and polyfluoroalkyl alcohols
- Side-chain fluorinated aromatics
- Perfluoroalkyl-tert-amines
- Perfluoroalkane sulfonamides (FASAs)
- Perfluoroalkyl phosphate esters (PAPs)
- Other non-polymeric

These groups differ markedly in physicochemical properties such as solubility, volatility, sorption behaviour, and reactivity. Importantly, many neutral PFAS (e.g., fluorotelomers and sulfonamides) act as **precursors**, capable of degrading into potentially more persistent PFAS.

Polymeric PFAS include:

- Fluoropolymers (FPs)
- Perfluoropolyethers (PFPEs)
- Perfluoroalkylether side-chain fluorinated polymers
- Side-chain fluorinated polymers

Polymeric PFAS, such as fluoropolymers, are generally characterized by their high molecular weight, chemical stability, and low mobility, which are expected to limit their bioavailability under typical use conditions.

Degradation in the environment is governed by a complex interplay of abiotic and biotic processes, including hydrolysis, photolysis, oxidation–reduction reactions, microbial metabolism, and indirect radical-mediated pathways. Environmental compartments such as soil, sediment, groundwater, surface water, and the atmosphere differ substantially in temperature, light availability, redox conditions, microbial activity, and sorption dynamics.

These variables influence not only degradation rates but also transformation pathways and product formation. In many cases, partial transformation may generate stable intermediates that persist longer than the parent compound.

The following chapter provides a general framework describing environmental degradation processes and the factors influencing transformation behaviour.

2.2 The complexity of degradation in the environment

Environmental degradation is best understood not as a single process but as a network of potential reactions unfolding across diverse compartments. Each environmental medium imposes its own chemical constraints, while multiple abiotic and biotic mechanisms compete or interact. The outcome tells us less about an inherent property of a substance in isolation and more about the conditions to which it is exposed.

Degradation in the environment is often spoken of as though it were a single, uniform process. In reality, it is neither singular nor uniform. To speak of “environmental degradation” is to describe a complex interplay between chemical structure, physical conditions, biological activity, and time. The fate of a substance released into the environment depends not only on its intrinsic reactivity but also on the medium into which it is introduced and the mechanisms available for transformation within that medium.

Understanding degradation therefore requires attention to two intersecting dimensions of complexity. On the one hand, the environment is composed of multiple media like air, surface waters, groundwater, soils, sediments, and living organisms, each governed by distinct physical

and chemical regimes. On the other hand, degradation itself can proceed through fundamentally different pathways, broadly categorized as abiotic and biotic processes. These two dimensions interact in ways that make generalized statements about “persistence” or “degradability” inherently difficult.

Various environmental compartments

Each environmental compartment constitutes a unique reaction environment. The **atmosphere** is dynamic and oxidizing. It is characterized by low concentrations of reactants but high reactivity due to the presence of radicals such as hydroxyl, nitrate, and ozone. Sunlight drives photochemical cycles, generating transient reactive species capable of initiating oxidation. Reactions in the atmosphere are often rapid but selective, governed by radical kinetics and diffusion-limited encounters. Volatility, vapor pressure, and partitioning behavior strongly influence whether a compound will even be exposed to these processes. **Aquatic systems** encompass both surface waters and groundwater, but the conditions governing degradation differ substantially between them. In surface waters, sunlight drives photochemical processes, although dissolved organic matter, inorganic ions, and suspended particles compete for reactive intermediates and often limit reaction efficiency. Indirect photochemistry, where light excites natural organic matter to generate radicals or reactive electrons, frequently plays a larger role than direct photolysis. In addition, hydrolysis may occur depending on molecular structure and environmental conditions such as pH and temperature, although its relevance varies widely among substance groups. Reaction rates are further influenced by oxygen levels, pH, temperature, and hydrodynamic mixing. In contrast, groundwater systems receive little to no sunlight, and photochemical reactions are largely absent. Instead, redox conditions dominate, ranging from oxic to strongly reducing depending on microbial and geochemical factors. Under such conditions, hydrolytic and reductive transformation processes may contribute to degradation, depending on substance-specific reactivity. Lower temperatures, limited energy input, and reduced mass transport typically result in slower transformation rates. **Soils and sediments** are heterogeneous matrices where mineral surfaces, organic matter, pore water, and microbial communities coexist. Sorption processes can limit mobility while simultaneously reducing bioavailability. Microenvironments within soil aggregates may differ dramatically in oxygen content and redox potential. Hydrolysis may occur in pore water, influenced by pH, moisture content, and mineral catalysis. Degradation in such systems is rarely uniform; it may proceed in localized niches while remaining negligible elsewhere.

Biological systems introduce yet another layer of complexity. Organisms possess enzymatic machinery capable of highly specific transformations. However, enzymes evolved to metabolize naturally occurring substrates may or may not interact efficiently with synthetic or structurally unusual compounds. In addition, the diversity and abundance of microbial communities play a critical role in determining degradation pathways and rates. The types of organisms present in a given environment, their collective enzymatic capabilities, and their capacity to adapt or evolve in response to exposure to a specific chemical all significantly influence degradation processes. Microbial consortia may act synergistically, enabling sequential or complementary transformations that would not occur in isolated organisms. Consequently, metabolism may result in detoxification, transformation into more persistent products, or in some cases negligible change.

Mechanistic diversity

Across these environmental compartments, degradation pathways can be broadly grouped into two categories: abiotic and biotic processes. While this classification is conceptually useful, it is not absolute as many environmental transformations involve coupled chemical and biological steps.

Abiotic Processes

Abiotic degradation encompasses chemical and physical transformations that occur without direct biological mediation. These include:

- Photolysis, where absorption of light energy leads to bond cleavage or excited-state reactions.
- Radical-driven oxidation, often involving highly reactive transient chemical species such as hydroxyl radicals.
- Reductive processes, in which electrons are transferred to a compound, potentially weakening or breaking strong bonds.
- Hydrolysis, particularly relevant for functional groups susceptible to nucleophilic attack.
- Thermal processes, which may occur under elevated temperatures in certain natural or engineered settings.

The effectiveness of these processes depends strongly on environmental conditions. For example, radical species are highly reactive but short-lived, their concentration and lifetime are controlled by competing reactions. Similarly, light-driven reactions depend on wavelength, intensity, and the compound's absorption spectrum.

Biotic Processes

Biotic degradation involves living organisms, typically microorganisms, that transform compounds through enzymatic reactions.

These processes may include:

- Direct enzymatic metabolism, where a compound serves as a substrate.
- Cometabolic transformation, where a compound is incidentally modified during metabolism of another substrate.
- Microbially mediated redox reactions, where electron transfer processes alter chemical structure indirectly

Biological degradation is influenced by nutrient availability, temperature, oxygen levels, microbial community composition, and evolutionary adaptation. Some transformations or degradation processes may occur only under very specific ecological conditions.

Transformation vs mineralization

An additional source of complexity lies in the distinction between transformation and mineralization.

Transformation refers to structural modification. A compound may be oxidized, reduced, hydrolyzed, or rearranged into a new molecule while retaining much of its original framework. Mineralization, by contrast, implies breakdown into inorganic constituents such as carbon dioxide, water, or inorganic ions.

Many environmental processes lead to transformation rather than complete mineralization. This distinction may be important because transformation products may retain persistence, mobility, or biological activity even if the parent compound is degraded. Thus, degradation does not necessarily equate to disappearance or detoxification.

Interdependence of Media and Mechanism

The complexity of environmental degradation emerges most clearly when considering the interaction between environmental medium and mechanistic pathway. A compound stable in groundwater may degrade more rapidly in sunlit surface waters. A transformation that is thermodynamically favourable may be kinetically inhibited under ambient conditions. Microbial communities may enable pathways unavailable in sterile systems.

Moreover, environmental systems are not static. Seasonal changes, hydrological events, temperature fluctuations, and biological succession can all alter degradation dynamics over time.

Analytical and Conceptual Challenges

Studying environmental degradation presents analytical challenges as well. Detecting low concentrations, identifying transient intermediates, and achieving mass balance closure require sophisticated methods. Incomplete detection of transformation products can lead to underestimation or misinterpretation of degradation pathways.

Conceptually, the temptation to categorize compounds as either “persistent” or “degradable” often oversimplifies a continuum governed by rates, conditions, and mechanisms.

2.3 PFAS degradation in the environment

Uneven distribution of scientific knowledge

A central challenge in the assessment of PFAS is the highly uneven distribution of scientific knowledge across the different substance groups. A small number of so-called “legacy” PFAS, most prominently perfluorooctanoic acid (PFOA) and perfluorooctane sulfonic acid (PFOS), have been the subject of extensive investigation over the past two decades. Their environmental occurrence, physicochemical properties, toxicology, bioaccumulation behaviour, and long-range transport potential are comparatively well characterised. For these substances, degradation pathways, environmental half-lives, and major transformation products have been studied in laboratory systems and, to a certain extent, under field conditions.

In contrast, the knowledge base for many other PFAS is fragmentary or, in some cases, virtually non-existent. This is particularly true for numerous replacement substances introduced following regulatory restrictions on long-chain PFAS. Short-chain analogues, ether-based PFAS (e.g., per- and polyfluoroalkylether carboxylic and sulfonic acids), novel fluorinated surfactants, and many polymeric PFAS often lack publicly available data on environmental degradation rates, transformation mechanisms, or stable end products.

In some instances, no peer-reviewed degradation studies could be found within the present project at all. Information may be limited to proprietary industry reports, registration dossiers with restricted access, or predictive modelling outputs that have not been empirically validated. As a result, the potential environmental persistence and transformation behaviour of many PFAS remain uncertain.

The term “PFAS” encompasses substances with fundamentally different chemical architectures and reactivity profiles. Some PFAS may act primarily as terminal, highly persistent end products, while others serve as precursors that transform into more stable forms. Degradation pathways may vary substantially between environmental compartments and may produce transformation products with greater mobility or persistence than the parent compound. A differentiated, substance-group-specific assessment approach is therefore essential. Without such differentiation, the complexity of PFAS chemistry risks being oversimplified, potentially leading to either underestimation or overestimation of environmental persistence and long-term impact.

Increasing efforts to generate knowledge

In recent years, there has been a growing emphasis on research dedicated to understanding the degradation pathways of PFAS in the environment. For example, recently the envPath-PFAS² package was published (Rich et al., 2025). It is part of the envPath platform, an online resource designed to compile and predict microbial biotransformation pathways of chemical contaminants based on curated experimental data. Originally developed to catalogue environmental degradation reactions for a wide range of organic pollutants, envPath has been extended through dedicated

² envPath is an AI-driven system for predicting microbial biodegradation pathways of chemicals, built on curated datasets from soil, sludge, and xenobiotic studies (Eawag-BBD, Eawag-Soil, and Eawag-Sludge) and validated by peer-reviewed research. [Link to envPath-PFAS](#)

efforts to consolidate PFAS-specific transformation data into a single, publicly accessible dataset and algorithmic framework. This PFAS-focused package brings together biotransformation pathways, associated metadata, and experimental conditions from peer-reviewed studies to support prediction of PFAS degradation routes and products under different environmental scenarios. By training and refining transformation rules and machine-learning models specifically on PFAS data, the package aims to improve the reliability of predicted pathways for these structurally distinct substances addressing a gap in conventional biodegradation prediction tools that were not developed with PFAS in mind.

The understanding of PFAS degradation pathways may be important for assessing human and environmental exposure to the various classes of PFAS and their potential transformation products and developing effective strategies to mitigate PFAS contamination. The currently available scientific knowledge is focused on exploring the degradation potential of various PFAS groups, including non-polymeric PFAS precursors and intermediates, side-chain fluorinated polymers, and fluoropolymers. These efforts aim to identify viable pathways and approaches for understanding, managing and remediating PFAS contamination in diverse environmental contexts. By bringing together the knowledge in these areas, this project seeks to contribute to the understanding of PFAS degradation.

3. Methodology for data collection

To develop a thorough review of the research topic while ensuring efficiency, a step-wise approach was adopted (Figure 3-1).

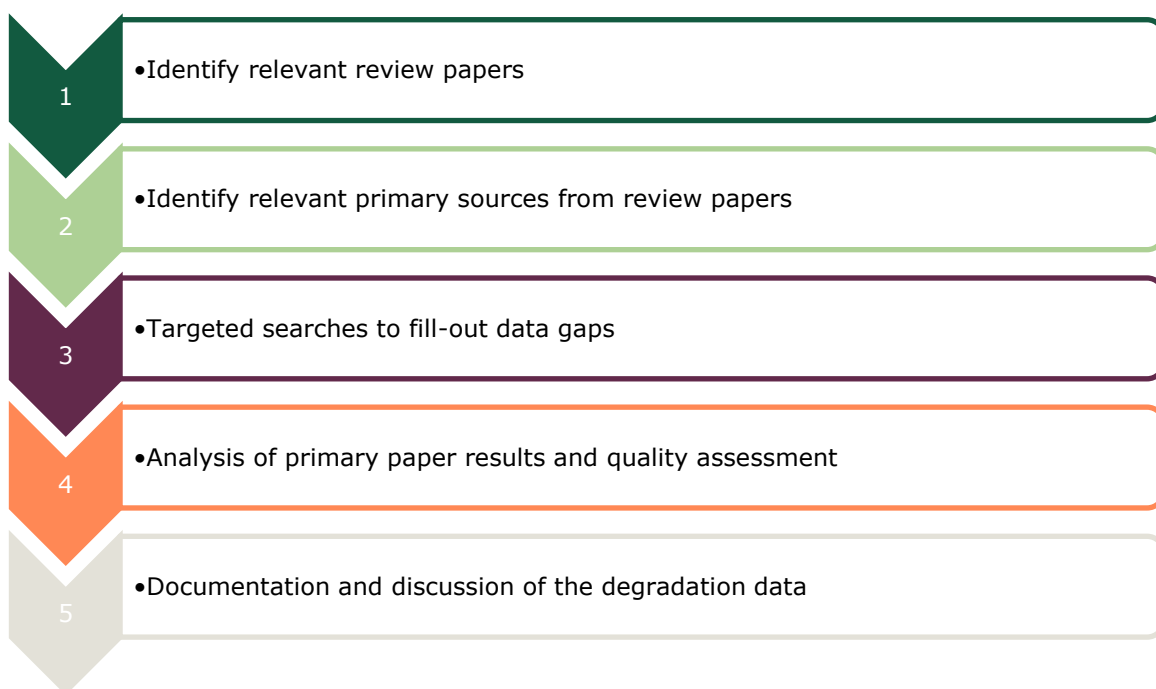


Figure 3-1. Schematic summary on step-wise approach used to prepare this review.

Step 1: Identify relevant review papers

The first step involved conducting a systematic literature review to identify review articles on PFAS degradation. Papers were searched in Pubmed and SCOPUS³ using specific search algorithms outlined in the Appendix. Initially, 309 review papers were identified.

These papers were then screened based on their titles and abstracts to assess their relevance with the following criteria:

Relevance Criteria:

- The paper explicitly discusses PFAS degradation rather than general PFAS contamination or occurrence.
- It includes chemical, biological, or physical degradation methods relevant to environmental degradation.
- It makes a clear distinction between different PFAS structures.

Papers not meeting these criteria were excluded from further consideration. It should be noted that a substantial portion of the published literature addresses engineered remediation or treatment technologies, such as advanced oxidation processes, electrochemical systems, plasma treatment, or other high-energy destruction methods. While scientifically valuable, these studies are typically designed to evaluate removal or destruction under controlled, technology-driven conditions that do not reflect natural environmental processes. Because the objective of this review was to assess degradation under environmentally relevant conditions, i.e., in air, surface waters, soils, sediments, and wastewater treatment systems, studies focused solely on engineered remediation were not considered further unless they provided insight into mechanisms that may also occur naturally.

Through this screening process, 29 review papers were identified as focusing specifically on environmentally relevant degradation. In addition, CEFIC provided one relevant review paper that was not identified in the search. Therefore, a total of 30 review papers were selected for subsequent steps.

Step 2: Identify relevant primary sources from review papers

From the 30 prioritized review papers focusing on environmental PFAS degradation, a comprehensive list of all cited primary sources was compiled using a semi-automatic approach. This process created a database of approximately 3,500 publications, including duplicates. Many of these 3,500 publications were, however, not related to PFAS degradation. Therefore, a more meticulous approach was adopted.

Experts manually screened the 30 review papers and identified primary sources pertinent to degradation studies and extracted relevant information concerning PFAS degradation such as PFAS group and degradation pathway directly from the review papers. Through this detailed screening, 207 potentially relevant primary papers were identified and categorized based on the degradation pathway and PFAS groups under investigation.

Step 3: Targeted Searches to fill-out data gaps

While for some PFAS classes several publications were relevant, other PFAS classes were scarcely addressed in the available literature. To address gaps in such underrepresented PFAS classes from the review papers, additional targeted searches were conducted. These searches were more specific and tailored to focus especially on specific PFAS groups identified as missing or under-represented

³ SCOPUS is one of the largest abstract and citation database of peer-reviewed research literature, as of April 2025, SCOPUS includes 83+ million records and over 25,000 active, including 6,000 Open Access journals from more than 5,000 international publishers and is therefore is largest abstract and citation database of peer-reviewed research literature according to its publisher (Source: [SCOPUS Website](https://scopus.com), last accessed September 2025).

in the initial review. Specific keywords were utilized to concentrate on degradation pathways and environmental compartments. Sources considered in this step included:

- Targeted searches in databases like SCOPUS using specific keywords related to the identified data gaps (see Appendix) and general desk research
- Input from CEFIC members
- The degradation chapter from the background document of the updated ECHA restriction proposal⁴

The envPath-PFAS package (as referenced in the chapter 2.1) was released only at a late stage of the present project. As a result, it could not be systematically integrated into the literature screening and evaluation process applied in this review. While it represents a potentially valuable and evolving resource, its current literature collection comprises information from 39 scientific publications and is not yet exhaustive (Rich et al., 2025). Consequently, although acknowledged as relevant, it was not possible to incorporate it comprehensively within the scope and timeline of this study. However, the comprising 39 publications represents a significantly fewer number than the number of publications considered in this review. A cross-check revealed that 34 out of the 39 envPath publications were already included in the present review, while 5 had not been identified by the applied methodology. These 5 papers were primarily not identified because they focused on PFAS compounds from groups not prioritized in the targeted searches due to the abundance of other available publications (e.g., FASAs, other fluorotelomers). These 5 papers were subsequently added to the data extraction sheet for the present project, but no thorough assessment was conducted.

The following number of additional papers were identified through targeted searches:

Table 3-1. Number of additional primary studies identified through targeted searches.

Source	Additional relevant primary studies not identified by review papers
ECHA background document	50
Provided by CEFIC	7 (more primary paper have been provided but were already included)
enviPath	5
Other targeted searches	27

In addition to primary studies, also additional review articles and authority reports were identified that were considered for the current review. For example, CEFIC members provided additional reports from industry and authorities containing relevant information that were used to complement the discussion in the relevant chapters of the report. The aim of integrating additional publications was primarily to strengthen the work for PFAS classes with limited or no originally found information on degradation.

Step 4: Analysis of Primary Papers results and Quality Assessment

Full-text screening of primary sources was conducted to extract the high-level information typically provided in review papers and incorporate details identified from targeted searches on PFAS degradation. During this process, studies were also reassessed for relevance based on these criteria:

- Studies should provide information on PFAS degradation.

⁴ <https://echa.europa.eu/registry-of-restriction-intentions/-/dislist/details/0b0236e18663449b>

- Studies should explicitly distinguish between different PFAS structures.
- Studies should be relevant to environmental conditions, excluding artificial destruction techniques.

As a result, some primary studies initially identified were excluded based on the outlined criteria, reducing the number of relevant primary papers identified from review papers from 207 to 186. The 186 publications could be divided into two groups based on their publication date:

- **“New”**: 113 publications from 2015 or later (less than 10 years old)
- **“Old”**: 73 publications from before 2015 (more than or 10 years old)

To optimize resources, full-text analyses of primary sources was prioritized based on the following criteria:

- Focus on PFAS classes and degradation pathways with limited available data over those with a lot of data.
- Give higher priority to more recent studies published from 2015 onwards ("New" studies).

Overall, 200 relevant primary papers were analysed in full-text (see Table 3-2).

Table 3-2. Sources and number of relevant primary studies employed in the current review.

Source	Number of relevant primary studies	Number of prioritized primary studies with full-text assessment and extraction completed
Reviews	186	142
ECHA background document	50	34
Provided by CEFIC	7	3
enviPath	5	0
Other targeted searches	27	21
Total	275	200

The extracted data were compiled into an Excel®-based database, documenting detailed information on study conditions and results. For studies not adhering to specific guidelines, a quality assessment according to an adapted Klimisch evaluation was applied (see Appendix 2 for more details). In the evaluation of the evidence base, studies assigned adapted Klimisch reliability scores of 1 (reliable without restriction) or 2 (reliable with restrictions) were primarily used to support conclusions. Where publications with scores of 3 (not reliable) or 4 (not assignable) contained information considered relevant for completeness or contextual understanding, this was clearly indicated in the text and interpreted with appropriate caution.

In several instances, a study was divided into multiple entries in the Excel database due to varying experimental conditions, compartments, or different investigated PFAS groups. This decision was made on a case-by-case basis. For example, if a study investigated biotic degradation of a specific PFAS in sewage sludge (relevant for biotic degradation in WWTP) and freshwater (relevant for biotic degradation of water/sediment) with differing outcomes, separate entries were created. Similarly, if a study addressed multiple PFAS with various outcomes, separate entries were generated.

The 76 primary studies that were not prioritized for full-text assessment are also included in the database and categorized by PFAS groups and degradation pathways. As far as possible,

degradation information was extracted from review papers for those papers. However, this information was not subjected to cross checks with primary data.

All identified primary degradation studies divided into separate entries based on investigated compounds, degradation pathways, and study conditions, resulted in a total of 441 database entries from 275 primary sources.

Step 5: Documentation and Discussion of the Degradation data

A heatmap providing an overview of the data availability for various PFAS groups and degradation pathways was prepared (see chapter 4). A study could be included in the heatmap if the following criteria were met:

- The study could be assigned to at least one environmental degradation pathway.
- At least one PFAS class could be identified.

Also, studies that were not prioritised for a full evaluation of the full text were categorised using the above criteria and are therefore included in this heatmap. Consequently, the heatmap includes all primary studies on degradation that are relevant to the environment⁵ and were identified during this research. If a study was applicable to more than one PFAS class or pathway, it was counted multiple times for each individual PFAS class and/or pathway. Only primary studies are included in the heat map; review papers and other reports were not included as they refer to the same primary studies and thus would manifold and skew the number datapoints in the heatmap.

The database of primary studies formed the basis for preparing this review report. Accordingly, primary studies that underwent full-text assessment are discussed in greater detail within the report. To maintain conciseness and focus in the individual chapters, not all primary studies are elaborated upon extensively. For a comprehensive overview of all identified degradation data, please refer to the accompanying Excel database. Additionally, selected review papers and reports were used to complement and enrich the information on degradation for specific PFAS and pathways, as detailed in the respective chapters of the report.

3.1 PFAS definition and grouping

3.1.1 PFAS definition

For the purpose of this report PFAS are defined as follows:

Any substance that contains at least one fully fluorinated methyl (CF₃-) or methylene (-CF₂-) carbon atom (without any H/Cl/Br/I attached to it).

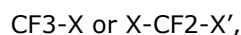
This definition is in line with the OECD definition, derived in 2021, which reads as: "PFASs are defined as fluorinated substances that contain at least one fully fluorinated methyl or methylene carbon atom (without any H/Cl/Br/I atom attached to it), i.e. with a few exceptions, any chemical with at least a perfluorinated methyl group (-CF₃) or a perfluorinated methylene group (-CF₂-) is a PFASs." (OECD, 2021) and the definition of PFAS in the universal restriction proposal (ECHA, 2025).

FAS, according to this definition, form a broad group of substances, including volatile as well as non-volatile PFAS, anionic, cationic, zwitterionic and non-ionic substances, polymers of different kinds as well as non-polymers, amphoteric liquids (surfactants), etc., with various chain-lengths and degree of fluorination.

⁵ Studies that were claimed by the authors to be relevant to the environment are also included, even if detailed assessment revealed that the conditions applied were only partially relevant to typical environmental settings. More detailed discussions of this issue are provided for the individual studies in respective chapters.

It should be further noted that OECD report states that “As PFASs are a chemical class with diverse molecular structures and physical, chemical and biological properties, it is highly recommended that such diversity be properly recognized and communicated in a clear, specific and descriptive manner. The term “PFASs” is a broad, general, non-specific term, which does not inform whether a compound is harmful or not, but only communicates that the compounds under this term share the same trait for having a fully fluorinated methyl or methylene carbon moiety.” (OECD, 2021) The OECD report further provides a “Practical Guidance on How to Use the PFAS Terminology” that should provide practical guidance to governments and other stakeholders on how to use the PFAS terminology, starting from the distinction between the general definition and user-specific working scopes of PFAS.

It is argued in the EU restriction proposal (ECHA, 2025) that PFAS (or their PFAS degradation products) share “very high persistence” as a common characteristic. In the EU restriction proposal, however, one exception on the scope of the restriction is introduced. This exception concerns certain fully degradable PFAS subgroups that only contain some specific structural elements. A substance that only contains the following structural elements is excluded from the scope of the proposed restriction:



where X = -OR or -NRR' and X' = methyl (-CH₃), methylene (-CH₂-), an aromatic group, a carbonyl group (-C(O)-), -OR'', -SR'' or -NR''R'''; and where R/R'/R''/R''' is a hydrogen (-H), methyl (-CH₃), methylene (-CH₂-), an aromatic group or a carbonyl group (-C(O)-).

In this report, PFAS are defined in line with the OECD definition (see blue box), without distinguishing between substances that have been proposed for exemption from restriction.

3.1.2 PFAS grouping

There are several approaches to group PFAS, and a comprehensive overview of PFAS groups was developed by the OECD in 2021 (see Figure 36-1). This grouping was used as a basis for the grouping of PFAS in this report with the following adaptations:

- A clearer distinction between fluoropolymers and non-polymeric PFAS
- Additional groups introduced for specific PFAA precursors not covered in the OECD-list

This leads to the following PFAS groups investigated in this study:

Table 3-3: PFAS classes addressed in the present report

Group	PFAS Group	Chapter
Non-polymeric PFAS	Perfluoroalkyl carboxylic acids (PFCAs)	5
	Polyalkyl carboxylic acids (PolyFCAs)	6
	Perfluoroalkane sulfonic acids (PFSAs)	7
	Perfluoroalkyl phosphonic acids (PFPAs)	8
	Perfluoroalkyl phosphinic acids (PFPIAs)	9
	Per- and polyfluoroalkylether carboxylic acids (PFECAs)	10
	Per- and polyfluoroalkylether sulfonic acids (PFESAs)	11
	Perfluoroalkyl dicarboxylic acids (PFdiCAs)	12
	Perfluoroalkane disulfonic acids (PFdiSAs)	13
	Perfluoroalkane sulfinic acids (PFSIAs)	14
	Fluorotelomer alcohols (FTOHs)	15

	Other Fluorotelomers and derivatives	16
	Perfluoroalkanoyl fluorides (PACFs)	17
	Perfluoroalkyl iodides (PFAIs)	18
	Perfluoroalkane sulfonyl fluorides (PASFs)	19
	Per- and polyfluoroalkylethers	20
	Perfluoroalkenes	21
	Perfluoroalkanes	22
	Semifluorinated alkanes (SFAs)	23
	Hydrofluorocarbons/ -ethers / -olefins / -chlorofluorocarbons (HFCs/ HFEs/ HFOs/ HCFCs)	24
	Perfluoroalkyl and semi-fluorinated ketones	25
	Per- and polyfluoroalkyl alcohols	26
	Side-chain fluorinated aromatics	27
	Perfluoroalkyl-tert-amines	28
	Perfluoroalkane sulfonamides (FASAs)	29
	Perfluoroalkyl phosphate esters (PAPs)	30
	Other non-polymeric	31
Polymeric PFAS	Fluoropolymers (FPs)	32
	Perfluoropolyethers (PFPEs)	33
	Perfluoroalkylether side-chain fluorinated polymers	34
	Side-chain fluorinated polymers	35

All PFAS contain at least one fully fluorinated methylene (-CF₂-) or methyl (-CF₃) group bonded to a saturated carbon atom, consistent with the OECD definition. In case of perfluoroalkyl substances the carbon chain is fully fluorinated (e.g., continuous -CF₂- or -CF₃ units) or partially fluorinated in polyfluoroalkyl substances.

The presence of different functional groups such as carboxylate (-COO⁻), sulfonate (-SO₃⁻), or phosphate, imparts hydrophilicity and influences PFAS solubility, ionization state (anionic, cationic, or zwitterionic), and environmental mobility.

4. Results on data availability

The degradation of PFAS is governed mainly by their molecular structures (functional groups and chain length) and environmental conditions such as pH, temperature, and the presence of reactive species or microbial enzymes. Functional groups specifically direct degradation pathways, for example, carboxylates favour decarboxylation while sulfonates undergo desulfonation. However, the strong C-F bonds confer substantial resistance to breakdown, meaning that degradation generally occurs only under extreme conditions. There are certain precursors that are known to degrade in various environmental matrices (e.g. FTOHs, FASAs), however potential persistency of degradation products must be considered as well.

Common degradation mechanisms involve hydrogen/fluorine exchange, chain shortening, and reactions catalysed by photochemical, hydrolytic, microbial, enzymatic, or electrochemical

pathways, depending on the context. Many studies describe microbial degradation by investigating enzymes, bacterial strains, and genetic potential (i.e. the presence of genes in the genome encoding enzymes capable of PFAS degradation) under laboratory conditions. However, even though genes for enzymes capable of degrading PFAS may be present in microorganisms (or other biota), it does not necessarily mean that such genes are expressed or that the expressed enzymes are active (at significant rates) under environmental conditions. Many of the available studies thus deliver valuable information for engineered remediation activities but often provide limited value for describing degradation under relevant environmental conditions.

In conclusion, PFAS degradation is governed by both chemical structure and environmental conditions. Functional groups, chain length, and degree of fluorination influence intrinsic reactivity, while physicochemical properties such as solubility, volatility, sorption behaviour, and physical state determine environmental mobility, bioavailability, and ultimately the prevailing degradation mechanisms.

The availability of data on the degradation of various PFAS groups varies significantly. While certain PFAS groups have been extensively researched, others have limited data availability regarding their degradation pathways.

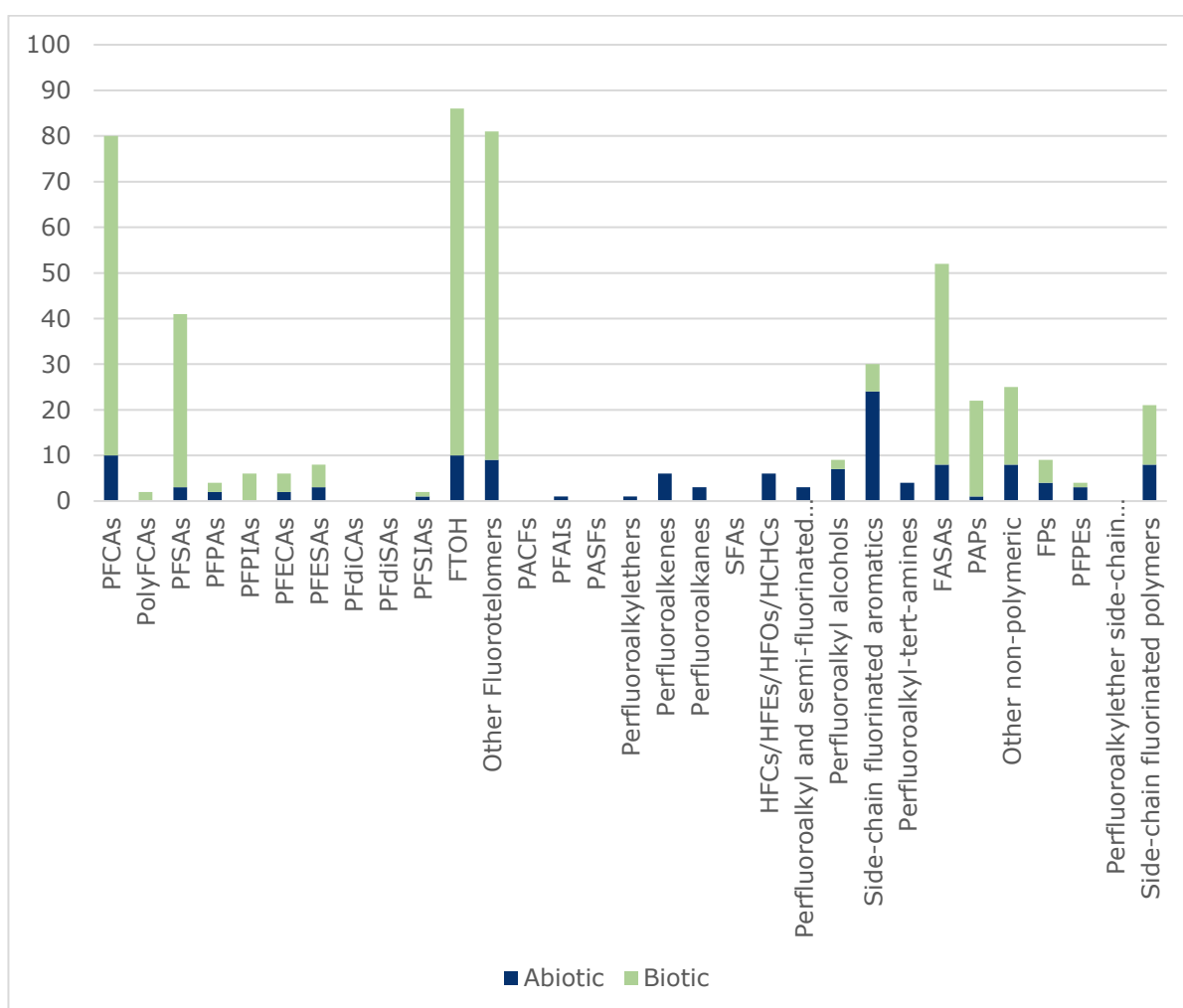


Figure 4-1 provides an overview of the study entries identified for each PFAS group, categorized by abiotic and biotic degradation. In addition, Table 4-1 provides a heatmap on data identified for this review dividing abiotic and biotic degradation in sub-categories on degradation pathways.

Despite our aim to provide a comprehensive review, it is important to note that this is not a fully systematic review. The methodologies employed may have resulted in overlooking some studies, potentially contributing to perceived data gaps.

Further, the presence of no study entries does not necessarily mean a data gap. Importantly, specific degradation pathways relevant for certain PFAS may be irrelevant to others. For example, biotic degradation might not be significant for gaseous PFAS like hydrofluorocarbons (HFCs), hydrofluoroethers (HFEs), hydrofluoroolefins (HFOs), and hydrochlorofluorocarbons (HCHCs), due to their different chemical properties and environmental behavior. Similarly, for non-volatile PFAS groups like fluoropolymers, abiotic studies on atmospheric degradation are considered of less importance. In some other cases, information from one PFAS group may be extrapolated to others by read-across based on structural similarities.

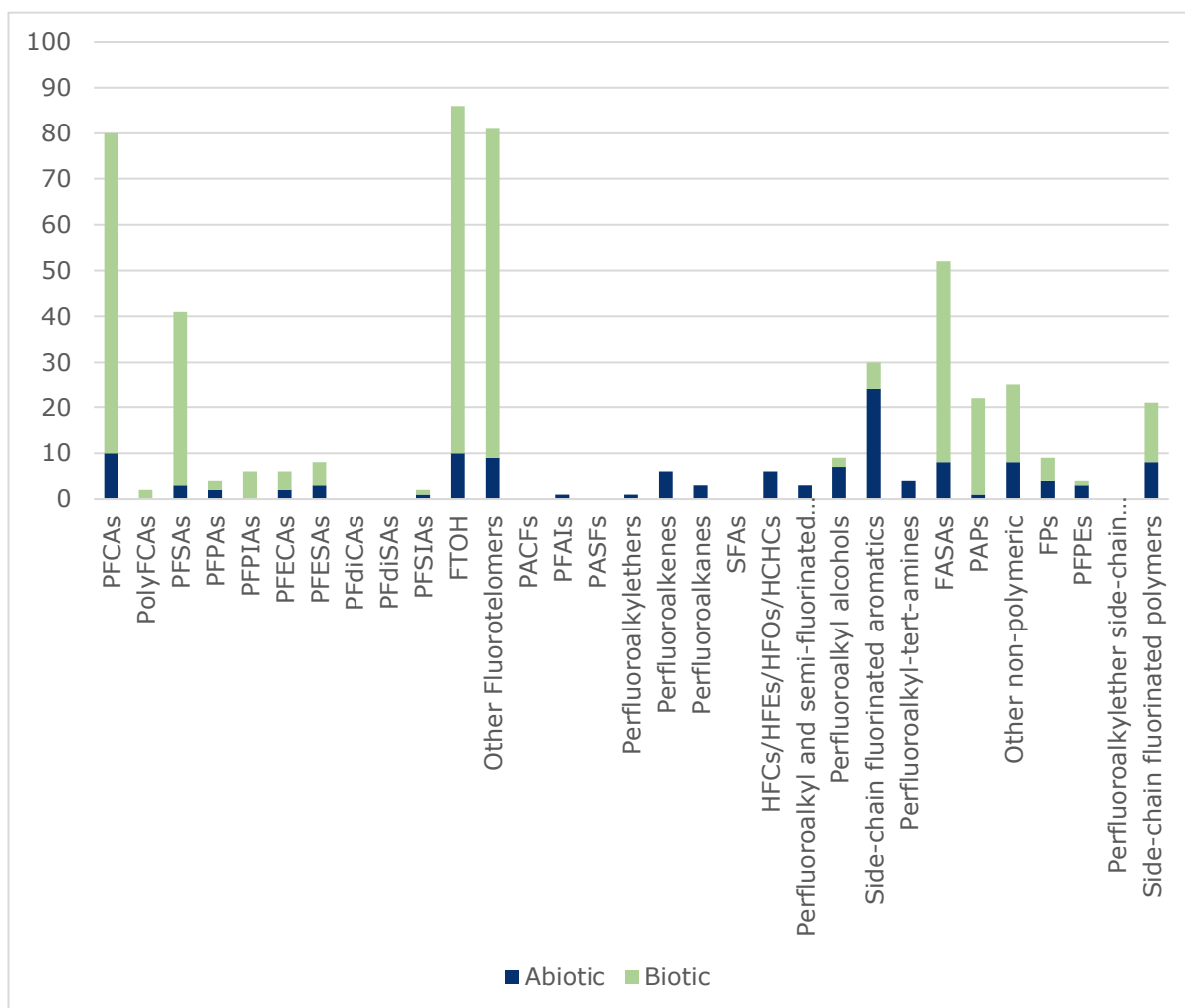


Figure 4-1. Number of studies identified for each PFAS group split into abiotic (blue) and biotic (green) degradation.

Table 4-1. Number of studies of abiotic and biotic pathways for each PFAS group.

Chapter	PFAS Group	Abiotic		Biotic		
		Atmosphere	Water	Water/ Sediment	Soil	WWTP
5	Perfluoroalkyl carboxylic acids (PFCAs)	1	9	31	12	27
6	Polyalkyl carboxylic acids (PolyFCAs)	0	0	0	0	2
7	Perfluoroalkane sulfonic acids (PFSAs)	0	3	12	9	17
8	Perfluoroalkyl phosphonic acids (PFPAAs)	0	2	0	0	2
9	Perfluoroalkyl phosphinic acids (PFPIAs)	0	0	2	3	1
10	Per- and polyfluoroalkylether carboxylic acids (PFECAs)	1	1	1	1	2
11	Per- and polyfluoroalkylether sulfonic acids (PFESAs)	1	2	0	2	3
12	Perfluoroalkyl dicarboxylic acids (PFdiCAs)	0	0	0	0	0
13	Perfluoroalkane disulfonic acids (PFdiSAs)	0	0	0	0	0
14	Perfluoroalkane sulfinic acids (PFSiAs)	0	1	0	0	1
15	Fluorotelomer alcohols (FTOHs)	7	3	22	25	29
16	Other Fluorotelomers and derivatives	6	3	22	24	26
17	Perfluoroalkanoyle fluorides (PACFs)	0	0	0	0	0
18	Perfluoroalkyl iodides (PFAIs)	1	0	0	0	0
19	Perfluoroalkane sulfonyl fluorides (PASFs)	0	0	0	0	0
20	Per- and polyfluoroalkylethers	1	0	0	0	0
21	Perfluoroalkenes	6	0	0	0	0
22	Perfluoroalkanes	3	0	0	0	0
23	Semifluorinated alkanes (SFAs)	0	0	0	0	0
24	Hydrofluorocarbons/ -ethers / -olefins / -chlorofluorocarbons (HFCs/ HFEs/ HFOs/ HCFCs)	6	0	0	0	0
25	Perfluoroalkyl and semi-fluorinated ketones	3	0	0	0	0
26	Per- and polyfluoroalkyl alcohols	7	0	0	0	2
27	Side-chain fluorinated aromatics	0	24	2	1	3
28	Perfluoroalkyl-tert-amines	4	0	0	0	0
29	Perfluoroalkane sulfonamides (FASAs)	2	6	4	20	20
30	Perfluoroalkyl phosphate esters (PAPs)	0	1	4	12	5
31	Other non-polymeric	8	0	1	7	9
32	Fluoropolymers (FPs)	2	2	2	2	1
33	Perfluoropolyethers (PFPEs)	1	2	0	0	1
34	Perfluoroalkylether side-chain fluorinated polymers	0	0	0	0	0
35	Side-chain fluorinated polymers	0	8	0	12	1

No studies (red), 1-3 study entries (orange), 4-5 study entries (yellow), 6-10 study entries (light green), >10 study entries (green)

In the following chapters degradation pathways, based on current understanding from the literature, are discussed for each PFAS group separately.

5. Perfluoroalkyl carboxylic acids (PFCAs)

Perfluoroalkyl carboxylic acids (PFCAs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, and a hydrophilic -COOH head group. PFCAs follow the general molecular formula $C_nF_{(2n+1)}COOH$, where n represents the number of carbon atoms in the perfluoroalkyl chain.

For example, when $n = 7$, the formula becomes $C_7F_{15}COOH$, corresponding to perfluorooctanoic acid (PFOA). Its full condensed structural formula can be written as $CF_3-(CF_2)_6-COOH$, which shows the fully fluorinated eight-carbon chain attached to the carboxylic acid group. Other prominent members of the PFCA homologous series include perfluorobutanoic acid (PFBA), perfluorohexanoic acid (PFHxA), and perfluorononanoic acid (PFNA), which differ in chain length and associated physicochemical properties.

Summary of review findings:

Several individual perfluoroalkyl carboxylic acids (PFCAs) are subject to regulation in the EU due to concerns about their persistence in the environment, among others. The C8 homologue perfluorooctanoic acid (PFOA), along with its salts and PFOA-related substances, has been banned under the Persistent Organic Pollutants (POPs) Regulation since 2020.⁶ Recently, long-chain perfluorocarboxylic acids (C₉-C₂₁), their salts, and related compounds have also been added to the list of POPs.⁷ In addition, perfluorohexanoic acid (PFHxA), its salts, and related substances are restricted under the REACH regulation.⁸

Perfluoroalkyl carboxylic acids are generally recognised among the most recalcitrant PFAS classes: in the atmosphere they react slowly with hydroxyl radicals, giving lifetimes of roughly half a year for C₂-C₅ homologues. In surface waters, photolysis is ineffective, with PFOA half-lives estimated to exceed two centuries.

Biotic degradation is likewise negligible under environmentally realistic conditions; measurable transformation of saturated PFCAs (usually PFOA) has been reported almost exclusively in pure-culture or engineered systems that supply strong reductants such as ferrihydrite, electrodes or redox-mediating enzymes, where partial defluorination and chain-shortening to PFHpA, PFHxA, PFPeA, PFBA, TFA and fluoride can reach 10-70 % over weeks to months. Unsaturated analogues like PFMeUPA have been shown to degrade more readily under anaerobic conditions but rarely exceed ~15 % fluoride release. Existing evidence is dominated by PFOA studies, with scant data for long-chain (C₉-C₂₁) or branched PFCAs; only a few studies report fluorine mass balances. Data on genuine field degradation are lacking.

5.1 Abiotic degradation

5.1.1 Degradation in the atmosphere

Degradation of PFCAs in the atmosphere is considered a minor pathway due to the strength of the C-F bond rendering them resistant to direct photolysis at relevant atmospheric temperatures and wavelengths, and most gas-phase oxidants (Prevedouros et al., 2006). PFCAs have been shown to undergo reaction with carbonyl oxide Criegee intermediates, which are reactive zwitterionic species

⁶ List of substances subject to POPs Regulation (ECHA): [Perfluorooctanoic acid \(PFOA\), its salts and PFOA-related substances](#)

⁷ Proposals for listing chemicals in the Stockholm Convention on POPs (ECHA): [Long-chain perfluorocarboxylic acids, their salts and related compounds](#), Status: Listed under the Stockholm Convention

⁸ ANNEX XVII TO REACH: [Entry 79 - Undecafluorohexanoic acid \(PFHxA\), its salts and PFHxA-related substances](#)

formed from ozonolysis of alkenes in the atmosphere over land masses (Taatjes et al., 2019). This reaction forms hydroperoxyfluoroesters which in turn partition to aqueous aerosols and subsequently hydrolyse through reaction with hydroxyl ($\cdot\text{OH}$) radicals to reform the PFCAs, which then undergo wet or dry deposition. The only well-quantified gas-phase sink for PFCAs is reaction with $\cdot\text{OH}$ radicals. Hurley et al. (2004) used relative-rate methods to determine that the $\cdot\text{OH}$ rate constants for short-chain PFCAs (TFA to PFPeA) resulted in atmospheric lifetimes of ~ 130 to ~ 230 days. Given that these lifetimes far exceed the timescales for wet and dry deposition, which occurs on the order of days, $\cdot\text{OH}$ oxidation represents a minor atmospheric degradation route, with PFCAs most effectively removed from the atmosphere primarily through aerosol partitioning and deposition processes rather than chemical breakdown.

5.1.2 Degradation in water

Degradation of PFCAs in water under environmentally relevant conditions involves complex mechanisms influenced by various factors. While laboratory experiments often simulate contaminated water, actual photolytic processes in the environment are more complex.

Direct photolysis under UV or visible light is limited due to the high energy requirements needed for effective degradation (F. Liu et al., 2022). PFOA does not absorb solar radiation directly but can undergo photochemical transformation through natural sensitizers like dissolved organic matter (DOM), nitrate, or ferric iron. DOM contains benzene rings, carboxyl, and hydroxyl functional groups that absorb visible light and generate reactive oxygen species such as singlet oxygen, hydroxyl radicals ($\cdot\text{OH}$), triplet excited-state (3DOM^*), and other reactive oxygen species (ROS), significantly impacting the photolysis of PFAS (H. Zhu et al., 2024). Several studies have further investigated photolysis as a remediation strategy using artificial photocatalysts, achieving high decomposition and defluorination efficiencies within shorter time frames. In aqueous systems, these processes typically rely on the generation of reactive species, particularly hydrated electrons, which promote reductive defluorination by cleaving C–F bonds. These electrons primarily target carbon atoms near the terminal functional groups in PFAS, facilitating defluorination and transforming them into more reactive groups such as $\cdot\text{C}_n\text{F}_{2n}\text{CO}^-$, $\text{C}_n\text{F}_{2n}\text{HCOO}^-$, $\cdot\text{C}_n\text{F}_{2n}\text{HSO}_3^-$, and $\cdot\text{C}_n\text{F}_{2n}\text{SO}_3^-$ (Qu et al., 2010; H. Zhu et al., 2024).

The degradation potential of PFOA under environmentally relevant conditions was investigated by Vaalgamaa et al. (2011) via indirect photolysis in aquatic solutions mimicking surface water conditions. Samples containing purified water, fulvic acid (DOM representative), nitrate, ferric iron, or Baltic Sea water were spiked with PFOA and irradiated using artificial sunlight (290–800 nm). Comparable samples were exposed to UV radiation at 254 nm for direct photolysis. While UV radiation at 254 nm decomposed PFOA into shorter-chain acids (perfluoroheptanoic, perfluorohexanoic, and perfluoropentanoic acids), the artificial sun did not affect PFOA concentration. The photochemical half-life of PFOA was estimated to be at least 256 years at the depth of 0 meters.

Taniyasu et al. (2013) conducted field and laboratory studies to investigate the photolysis of PFCAs and PFSAAs by exposing water samples to sunlight at high altitudes. They reported that long-chain PFCAs degraded into short-chain compounds. However, Zhanyun Wang, Cousins, & Scheringer (2015) questioned these findings, arguing that the reported results are unlikely to be environmentally relevant, citing inconsistencies in the experimental results. They noted missing experimental details, including UV absorption spectra, laboratory light conditions, and consideration of compound adsorption on vial surfaces. Therefore, the study results are assumed to be not reliable.

Another study by Ellis et al. (2001) showed that TFA, the shortest PFCA, was extremely persistent, with no degradation observed during a one-year field study in pond waters, although it appeared

to undergo transient partitioning within an unknown pond phase as environmental temperatures decreased.

5.2 Biotic degradation

Reports of microbial biotransformation of PFCAs such as PFOA are usually under very specific conditions with a limited relevance to real environmental conditions.

5.2.1 Degradation in water and sediments

Several studies have investigated degradation of PFCAs by extracellular enzymes, microbial species or in microbial settings related to water/sediments, however this remains mainly pure culture work with limited relevance to real environmental conditions.

Table 5-1. Biotic degradation of PFCAs in water (including pure culture) and sediment. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
PFOA (5 mg/L)	Proteobacteria and Bacteroidetes in soil-water system	Anaerobic, 22 °C, 10 months, pH 7.2	Not provided	13.52% biotransformation, 8.44% defluorination, TPs: PFHxA, PFHpA, TFA and pentafluoropropionic acid (PFA), F-	(Tang et al., 2024)
PFOA	Microbial community from industrial site sediment	Aerobic (up to 110 days) and anaerobic (up to 259 days), different mediators	Not provided	No degradation	(Liou et al., 2010)
PFOA (0.1 and 100 mg/L)	<i>Acidimicrobium</i> sp. strain A6	Anaerobic, 30 °C, pH 4.5-5, 100 days	Not provided	Up to 60% reduction, TPs: HFBA, PFPeA, PFHxA, PFHpA	(Huang & Jaffe, 2019)
PFOA (10 mg/L)	microbial - Synergetic diatom (<i>Chaetoceros muelleri</i>)	25 °C, pH 9, 24h, Synergetic diatom (<i>Chaetoceros muelleri</i>) and iron nanoparticle system (biomicroreactor) ^a	Not provided	93% defluorination, TPs: PFHpA, PFHxA, PFPA, PFPrA, TFA	(Albert et al., 2020)
PFOA (2, 60 and 600 ng/l)	Microbial - Marine, Gammaproteobacteria and Roseobacter, Flavobacteria	Aerobic, arctic water and spiked seawater, 6 days	Not provided	No PFOA removal	(Cerro-Gálvez et al., 2020)
PFOA (1 µmol/L)		Aerobic, 4 days ^a	Not provided	No PFOA transformation	(Steffens et al., 2023)
PFOA (1 mg/l)	microbial - <i>Synechocystis</i> sp. PCC 6803 (cyanobacterium)	30°C, 4 days (batch experiment), 12-15 days (continuous reactor) ^a	Not provided	28% removal (batch experiment), 37% removal (continuous reactor)	(Marchetto et al., 2021)
PFOA	Extracellular, ligninolytic enzyme, HRP Horseradish Peroxidase (Enzyme Source: <i>Armoracia rusticana</i> (Horseradish))	Room temperature, pH 7, 6h, H ₂ O ₂ as mediator and 4-methoxyphenol as co-substrate ^a	Not provided	68% transformation of PFOA; defluorination efficiency <1%	(Colosi et al., 2009)
PFOA (1.0 µM)	Enzymatic - Lacasse from bacteria and fungi (Enzyme Source: <i>Pleurotus ostreatus</i> (Oyster mushroom))	ECOHRs, 137 days, 22 °C, pH 4.5, 1-hydroxybenzotriazole as mediator ^a	157	50% degradation, Defluorination efficiency: 28%, TPs: partially fluorinated alcohols and/or aldehydes and	(Luo et al., 2015)

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
				fluoroalkyl-substituted aromatic compounds	
PFOA (1 and 100 µM)	Extracellular, ligninolytic enzyme, laccase (Enzyme Source: <i>Pleurotus ostreatus</i> (Oyster mushroom))	ECOHRs, 156 days, 22 °C, pH 4.5, 1- hydroxybenzotriazole as mediator ^a	157	47% degradation, TPs: Partially fluorinated or containing perfluoroalkyl moieties	(Luo et al., 2017)
PFOA (1.0 µM)	Extracellular, ligninolytic enzyme, laccase	ECOHRs, 36 days, 22 °C, 1- hydroxybenzotriazole as mediator ^a	Not provided	14–29% degradation, depending on the treatment and solution	(Luo et al., 2018)
PFOA (1-10 mg/l)	Laccase, lignin peroxidase, manganese peroxidase (enzyme source: <i>Trametes versicolor</i>)	Degradation of perfluorooctanoic Acid by Fungal Cultures, and by In vitro Enzymes, 28-30 days	Not provided	No degradation	(Mahendra et al., 2016)
PFOA (1 mg/l)	Microbial - <i>Acidimicrobium sp.</i> Strain A6	Anaerobic, 20 °C, pH 5.5, 40 days, Ferrihydrite (bare and PAA-coated) as mediator	Not provided	Degradation efficacy not reported. 14% defluorination efficiency, TPs: Fluoride, PFHpA, PFHxA, PFPeA, PFBA	(Park et al., 2023)
PFOA (1, 24, 30, 100 and 1000 µM)	Extracellular, ligninolytic enzyme	ECOHR in aqueous phase under various combinations of different conditions, including three different enzymes (HRP, LiP and Laccase) at different dosages, five different mediators (guaiacol, catechol, HBT, veratryl alcohol, and 4-methoxyphenol) at different concentrations, and different initial PFOA concentrations. 120 hours, room temperature.	Not provided	30% degradation; degradation observed only at low initial PFOA (1–30 µM); none at ≥ 100 µM.	(Qingguo, 2013)
PFOA (0.002 mM)	microbial - <i>Phaenerochaete chrysosporium</i> (fungi) with possible genes for laccase, lignin peroxidation and manganese peroxidase	Aqueous liquid culture system, 55 days, 25 °C, pH 3, ABTS mediator substrate ^a	Not provided	69.23% removal in 35 days; TPs: Partially fluorinated aldehydes, alcohols and aromatic rings	(L. Zhou et al., 2023)
FMeUPA (C6HF9O2) (20.7 mg/L)	KB1 enrichment culture dominated by Dehalococcoides	Anaerobic (34 °C), 63 days	Not provided	100 % biotransformation, ~10 % defluorination (>52 µM)	(Y. Yu et al., 2020, 2022)
PFMeUPA (75 µM)	Reductive defluorination by an organohalide- respiring microbial community	Anaerobic (34 °C), 150 days, Vitamin B12 as co-factor, lactate and TCE added as electron donor and receptor	Not provided	100% biotransformation, ~10 % defluorination, TPs: C6H2F8O2, C6H3F9O2, C6H4F8O2, FMePA, C6H4F6O2	(Y. Yu et al., 2020)
TFA (10 mg/L)	Proteobacteria and Bacteroidetes in soil- water system	Anaerobic, 22 °C, 10 months, pH 7.2	Not provided	8.03% biotransformation, 3.46% defluorination	(Tang et al., 2024)

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
TFA (20 mg/L)	Sediment	Aerobic, 25 °C, 63 days	Not provided	No degradation	(Alexandrino et al., 2018)
TFA (15 mg/L)	Mixed culture (found in natural water systems)	Anaerobic, 35 °C, pH 7.2, 90 weeks, engineered reactor	Not provided	TPs: DFA, MFA, F-, CH ₄	(B. R. Kim et al., 2000)

^a Limited environmental relevance

The compiled literature shows a clear imbalance in data coverage. Most experiments address PFOA, with 15 separate entries, whereas studies on longer-chain PFCAs (e.g. PFNA, PFDA) are essentially absent and only a few studies target short-chain analogues such as FMeUPA, PFMeUPA and TFA.

Aerobic vs. anaerobic conditions are represented in roughly equal numbers, yet outcomes differ markedly. Under strictly aerobic settings e.g. no measurable PFOA loss was reported (Cerro-Gálvez et al., 2020; Liou et al., 2010; Steffens et al., 2023), while a handful of enzyme-assisted systems (Colosi et al., 2009; Luo et al., 2015, 2017, 2018; Qingguo, 2013) achieved 14–68 % transformation. In contrast, several anaerobic studies reported reproducible although still modest biodegradation: 13.5 % PFOA removal over 10 months in a soil-water microcosm (Tang et al., 2024), up to 60 % in pure cultures of *Acidimicrobium* sp. strain A6 at 30 °C within 100 d (Huang & Jaffe, 2019), and 69 % removal by *P. chrysosporium* after 35 d with ABTS as mediator (L. Zhou et al., 2023). The most efficient defluorination (93 %) was observed in a phototrophic diatom/Fe-nanoparticle “biomicroreactor” that is of limited environmental relevance (Albert et al., 2020).

Recurring transformation products (TPs) include shorter PFCAs such as PFHxA, PFHpA, PFPeA, PFBA, and the ultra-short acids TFA and PFA, along with release of inorganic fluoride; these appear in most successful PFOA/TFA studies irrespective of redox regime, suggesting stepwise shortening via decarboxylation/ β -oxidation-like pathways.

Favourable degradation factors include: (i) reducing conditions (Fe(III)-rich, lactate-fed, or organohalide-respiring communities), (ii) elevated temperature (25–30 °C), (iii) presence of redox-active mediators (H₂O₂, HBT, phenolics) or Fe-bearing minerals, and (iv) unsaturated carbon-fluorine structures that facilitate reductive defluorination (e.g., PFMeUPA).

Degradation is impeded by oxic, nutrient-poor marine conditions, absence of mediators, short incubation times and high initial concentrations (>100 μ M).

Overall, while specific laboratory systems show to be able to partially degrade PFOA and a few short-chain PFAS, rates are slow and efficiencies are low under environmentally realistic conditions.

5.2.2 Degradation in soil

Some studies have investigated degradation of long-chain PFCAs by microbial species related to soil, however, this remains mainly pure culture work with limited relevance to real environmental conditions (Table 5-2).

Table 5-2. Biotic degradation of PFCAs in soil (including pure culture). Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
PFDA, PFHpA (250 mg/L)	<i>Pseudomonas mosselii</i> 5(3)	Aerobic, 28 °C, pH 5.2, 7 days	Not provided	Up to 100% reduction, TPs: Fluoride, PFHpA, PFHxA, PFHpS, PFHxS	(S. Chetverikov et al., 2023)

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
PFOA (500 mg/L)	<i>Pseudomonas parafulva</i> YAB1	Aerobic, pH 7, 30 °C, 96 days	Not provided	Up to 48.1% reduction in PFOA (transformation products not specified)	(L. B. Yi et al., 2016)
PFOA (1000 mg/L)	<i>Pseudomonas plecoglossicida</i> DD4	Aerobic, pH 6.8-7.2, 26-30 °C, 96 hours	Not provided	Complete transformation, TP: Fluoride and PFHpA	(S. P. Chetverikov & Sharipov, 2022)
PFOA (1000 mg/L)	<i>Ensifer adhaerens</i> M1	Aerobic, pH 6-8, 26-28 °C, 4 days	Not provided	Complete transformation, TP: Fluoride and PFHpA	(S. P. Chetverikov & Loginov, 2019; S. P. Chetverikov & Sharipov, 2022)
PFOA	<i>Delftia acidovorans</i> DeHa1	Aerobic, 4 hours	Not provided	Fluoride	(Harris et al., 2022)
PFOA	Mixed culture site (Industrial sediment, agricultural soil, soils from fire training, WWTP)	Anaerobic (259 days), aerobic (110 days), various co-metabolites	Not provided	No transformation	
PFOA (0.5 mg/kg)	Soil slurry	ECOHRS, 22-25 °C, 140 days, soybean meal and laccase	Not provided	Up to 40% in soil slurry, TP: primarily partially fluorinated compounds containing perfluoroalkyl moieties	(Luo et al., 2018)

Long-chain PFCAs such as PFOA are in general regarded as microbiologically inert in soil, which is supported by early microcosm work that showed no alteration of PFOA under a wide range of anaerobic redox regimes and different microbial species (Liou et al., 2010). Selected studies, however, indicate that specialised enzymatic systems attack the C–F backbone, albeit slowly and under restrictive conditions. Laccase-mediated “enzyme-catalysed oxidative humification reactions” (ECOHRS) removed 24 % of dissolved PFOA in 36 d and 40 % in a soil slurry after 140 d when soybean-meal extracts supplied phenolic mediators and multivalent metals (Luo et al., 2018). Pure-culture work has found some potential degraders: *Delftia acidovorans* from PFAS-impacted soil grew on PFOA as the sole carbon source with measurable fluoride release (Harris et al., 2022); *Ensifer adhaerens* strain M1 isolated from soil of a firefighting facility transformed both PFOA and PFOS (S. P. Chetverikov & Loginov, 2019); *Pseudomonas plecoglossicida* DD4 isolated from soil liberated 132 mg/L fluoride from 1000 mg/L PFOA (S. P. Chetverikov & Sharipov, 2022); and *P. mosselii* 5(3) from herbicide-contaminated soil completely consumed C₇–C₁₀ PFCAs within 7 d (S. Chetverikov et al., 2023). Yet reproducibility and metabolic pathways remain poorly resolved: *Pseudomonas parafulva* YAB1 isolated from soil degraded ~48 % of PFOA without detectable fluoride release or identifiable products (L. B. Yi et al., 2016). Overall, while isolated enzymes and strains demonstrate that defluorination of PFCAs is possible, turnover rates in bulk soils are still negligible, and the environmental relevance of these testing conditions is limited.

5.2.3 Degradation in WWTPs

PFCAs are generally resistant to biodegradation in wastewater treatment, as concluded by several recent reviews (e.g. Berhanu et al. 2023, Grgras et al. 2023). PFOA is recognised as most studied PFCA, which is reflected by the identified studies listed in Table 5-3 and the following paragraphs.

Table 5-3. Biotic degradation of PFCAs in water (including pure culture) and sediment. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
PFOA (0.1 and 100 mg/L)	<i>Acidimicrobium</i> sp. A6	Anaerobic, pH 4.5–5, 100 days	Not provided	HFBA, PFPeA, PFHxA, PFHpA	(Huang & Jaffe, 2019)
PFOA (max. 47 mg/L)	<i>Acidimicrobium</i> sp. A6	Electrochemical, Anaerobic, pH 5–5.5, 18 days	Not provided	PFBA, PFPeA, PFHxA, PFHpA, F–	(Ruiz-Urigüen et al., 2022)
PFOA (10, 1 and 0.1 mg/L)	<i>Pseudomonas aeruginosa</i> , <i>Pseudomonas putida</i>	Aerobic, 25 °C, incubation at 35 °C and 37 °C for bacterial cultures, pH ~7, 96 hours	Not provided	PFHxA, PFHpA, PFPeA	(Chiriac et al., 2023)
PFOA (0.2 and 10 mg/L)	<i>Acidimicrobium</i> sp. A6	Anaerobic, pH 4.5–5, 150 days	Not provided	PFBA, PFPeA, PFHxA, PFHpA, F– production	(Huang et al., 2022)
PFOA (100 ppm or 100 ppb)	Municipal WWTP sludge	Aerobic, nitrate reduction, iron reduction, sulfate reduction, and methanogenesis (with and without acetate as an electron-donor), 110 days	Not provided	None detected	(Liou et al., 2010)
PFOA (100 ppm or 100 ppb)	Municipal WWTP sludge	Anaerobic, 259 days	Not provided	None detected	(Liou et al., 2010)
PFOA (100 µg/L, 1000 µg/L)	Constructed wetlands - subsurface flow treatment wetlands	25 °C, 7 months	Not provided	Short chain PFCAs, perfluoroalkyl alcohols/aldehyde, unsaturated/saturated organic fluorine (identity of degradation products not detailed specified)	(Xiao et al., 2023)
PFOA (0.0, 0.5, 1.0, 1.5, 2.0, 3.0 mg/L)	Sludge, Bacterial consortium from anaerobic digestion	Anaerobic, pH 7, 10 days, Glucose as carbon source	Not provided	na	(Y. Yu et al., 2022)
PFMeUPA* (~75 µmol)	WWTP activated sludge, mixed anaerobic defluorinating enrichment culture	Anaerobic, 34 °C, 63 days	Not provided	(Defluorination degree of 14.3%)	(Y. Yu et al., 2022)
U-C5a* (C5HF7O2) (~75 µmol)	WWTP activated sludge, mixed anaerobic defluorinating enrichment culture	Anaerobic, 34 °C, 63 days	Not provided	(Defluorination degree of 14.3%)	(Y. Yu et al., 2022)
TFA (5, 10, 20 mg/L)	WWTP activated sludge	Aerobic, 25 °C, 63 days	Not provided	No degradation occurred with TFA.	(Alexandrino et al., 2018)

na = not available, * See figure below for structures

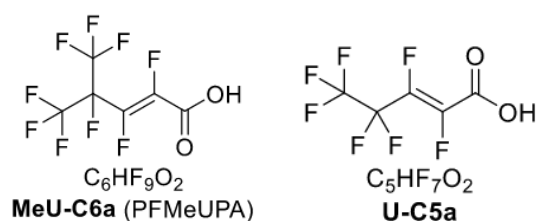


Figure 5-1. Structures of PFMeUPA, PFPeA, U-C5a (C₅HF₇O₂) from (Y. Yu et al., 2022).

PFOA is highly recalcitrant to microbial transformation and defluorination (e.g. reviews by Berhanu et al. 2023, Grgras et al. 2023). Numerous studies investigate PFCA degradation in experimental setups mimicking conditions specifically designed for bioremediation of PFAS or PFCAs, e.g., by employing customized photocatalyst films (Ding et al., 2021), coupled microbial fuel cells (Ji et al., 2023), nano zerovalent iron (Qian et al., 2023) and/or different iron materials such as ferrihydrite (Huang et al., 2022; Huang & Jaffe, 2019; H. Ma et al., 2023) within wastewater treatment systems or microcosms. These studies are largely omitted in the above table as they describe conditions of limited environmental relevance.

Since some of the studies nonetheless demonstrate at least a possible microbial pathway with potential of degrading PFCA in wastewater treatment, a few recent ones are shortly described here. The studies by Huang and Jaffé (2019), Ruiz-Urigüen et al. (2022) and Huang et al. (2022) highlight the ability of the autotroph bacteria *Acidimicrobium* sp. A6 as the best example of a PFOA-defluorinating bacteria. Huang and Jaffé (2019) report >60 % degradation of the recalcitrant PFOA (and PFOS) over 100 days and measured fluoride in WWTP sludge and biosolids augmented with ferrihydrite, facilitating reductive defluorination of PFOA by the Feammox reaction⁹ catalysed by strain A6 (Huang and Jaffé, 2019 and Huang et al., 2022, as cited in (Berhanu et al., 2023)). However, the Feammox reaction requires large amounts of Fe(III) as electron acceptor.

To overcome this limitation, Ruiz-Urigüen et al. (2022) conducted the reaction in a microbial electrolysis cell (MEC), where Fe(III) is replaced with electrodes. In this system, they were able to demonstrate a 77% reduction of PFOA, fluoride release and formation of four short chain PFCAs (PFBA, PFPeA, PFHxA and PFHpA).

Ma et al. (2023) investigated improvement of microbial degradation and performance of the substrate in PFCA (and PFSA) degradation, using different configurations of iron minerals as substrate to remove PFOA (and PFOS) from constructed wetlands (CWs). The addition of iron minerals resulted in 63.9% removal of PFOA (57.2% of PFOS) in the effluent, which was 36.8% (35.0% for PFOS) higher than in the control test. The authors found an enhanced electron transfer in iron mineral-based CWs, increase in cytochrome C reductase and increased Fe(III)/Fe(II) ratio, as well as increasing proportions of microbes with PFAS removal capacity, e.g., *Burkholderiae* sp. and *Pseudomonas* sp.). Based on their findings, the authors conclude that supplementation with iron mineral could enhance PFOA and PFOS removal in CWs.

Chiriach et al. (2023) presents results on PFOA degradation with pure culture strains of *Pseudomonas aeruginosa* and *Pseudomonas putida* during 96 hours, with resulting degradation percentages ranging from 1.84-19.0% depending on applied initial concentration and test system. The study included measured concentrations of shorter chain PFCAs, which are recognized as degradation products. It is noted, however, that the study does not document information on mass balance in the different test systems, fluoride measurements were not conducted and potential sorption issues to test material were not addressed in the study, limiting the reliability of the findings. Also, these

⁹ A microbial anaerobic process ("ferric iron-mediated anaerobic ammonium oxidation") in which ammonium (NH₄⁺) is oxidised using ferric iron (Fe(III)) as the electron acceptor, producing nitrogen species such as dinitrogen (N₂), nitrite (NO₂⁻), or nitrate (NO₃⁻), while Fe(III) is reduced to ferrous iron (Fe(II)).

results were derived in a pure culture study with limited environmental relevance, however, the authors suggest that use of such microorganisms can contribute to the improvement of wastewater treatment processes.

Xiao et al. (2023) observed PFOA removal from synthetic wastewater in subsurface flow treatment wetlands during 7 months, and suggest several removal pathways, of which many could be quantified; substrate adsorption (46.2%), plant uptake (2.0-2.5%), effluent (36.5-40.5%). Removal of PFOA through other pathways accounted for 10.3–11.5%. Since the fraction of PFOA removed by hydrolysis, photodegradation and volatilization is stated to be almost negligible, the authors consider biotransformation may play an important role in PFOA removal in the aqueous and solid phases of the CW systems. The study also highlights that there are several other removal pathways than biodegradation, that are of importance for removal of PFOA in wastewater treatment systems.

Y. Yu et al. (2022) found that unsaturated PFCAs can be biotransformed under anaerobic conditions by reductive defluorination and/or hydrogenation pathways, emphasising that the α,β -unsaturation of unsaturated PFCAs is crucial for biotransformation via reductive defluorination and/or hydrogenation pathways. They do not mention any specific bacteria species or strains involved in the biodegradation, but report that the transformation was carried out by a mixed anaerobic defluorinating enrichment culture.

Liou et al. (2010b) found no evidence of PFOA biotransformation or defluorination. Next to other microbial communities (from industrial site sediment, soils from two fire training areas, and an agriculture soil), they also performed degradation studies with sludge from a municipal WWTP.

Anaerobic incubation with PFOA (concentration of 100 ppm and 100 ppb) as an electron acceptor and acetate, lactate, ethanol, and/or hydrogen gas as electron donors up to 259 days, the authors found no evidence for a reductive defluorination.

The authors also examined the potential co-metabolism of PFOA (at concentration of 100 ppm and 100 ppb) during reductive dechlorination of trichloroethene with 5% inoculum from a sewage treatment plant as an electron acceptor over 65 days. Reductive dechlorination based on co-metabolism of PFOA could not be confirmed.

Additionally, the authors investigated a potential transformation of the PFOA as electron donor (at concentrations 100 ppm and 100 ppb) with five different electron acceptors; aerobic, nitrate reduction, iron reduction, sulfate reduction, and methanogenesis with 5% inoculum from a sewage treatment plant during 110 days of incubation. Significant changes in either PFOA concentration or fluoride concentration were not observed in any of the experiments.

Alexandrino et al. (2018) assessed biodegradation fluoroacetates using microbial samples collected from, amongst others, activated sludge from a municipal WWTP. Biodegradation experiments were conducted under different conditions of substrate supplementation, including use of TFA as sole carbon source or in combination with other fluoroacetates and/or sodium acetate. Biodegradation of TFA (and the other fluoroacetates) was assessed via fluoride ion release. TFA was recalcitrant in all tested conditions.

In summary, available evidence shows that PFCAs, especially PFOA, remain recalcitrant under the aerobic and anaerobic conditions typically encountered in conventional WWTPs and anaerobic digesters. No fluoride release has been reported in such systems, and observed half-lives exceeded the duration of the tests (≥ 110 –259 d). Measurable biotransformation has only been observed when tests are modified with electron donors such as ferrihydrite, electrodes, or other iron-based materials, and when pure/enriched cultures such as *Acidimicrobium sp. A6* or *Pseudomonas spp.* are introduced - set-ups that fall outside conventional WWTP operation. PFCA removal in wastewater

treatment is governed by other removal processes (e.g. sorption, plant uptake) rather than biodegradation.

6. Polyfluoroalkyl acids (PolyFCAs)

Polyfluoroalkyl carboxylic acids (PolyFCAs) consist of a polyfluoroalkyl tail, in which some but not all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, and a hydrophilic -COOH head group. PolyFCAs follow the general molecular formula $C_nF_{(2n-m)}H_mCOOH$, where n represents the number of carbon atoms in the polyfluoroalkyl chain and m the number of hydrogen atoms.

For example, when $n = 8$ and $m = 1$, the formula becomes $C_8F_{15}HCOOH$, corresponding to 7H-Perfluorooctanoic acid, a partially fluorinated analogue of PFOA. Other prominent polyfluorinated carboxylic acids include 6:2 Fluorotelomer carboxylic acid (6:2 FTCA), 5:3 Fluorotelomer carboxylic acid (5:3 FTCA), and 7:3 Fluorotelomer carboxylic acid (7:3 FTCA), which contain hydrogen atoms within the fluorinated carbon chain and are often formed as transformation products of fluorotelomer-based substances.

Summary of review findings:

Polyfluoroalkyl acids (PolyFCAs) are structurally related to PFCAs, with the difference that the carbon chain is not fully fluorinated. Only two studies on degradation of PolyFCAs were found; aside from two laboratory investigations in activated sludge and enrichment cultures, no studies were found on their fate in the atmosphere, natural waters, sediments or soils. The limited data show that certain highly fluorinated propionic, butanoic and pentanoic acids can undergo slow co-metabolic biotransformation in aerobic sludge, while unsaturated C4–C5 analogues appear more labile under anaerobic conditions, yet there is uncertainty about the transformation mechanisms. Transformation pathways appear to involve stepwise defluorination and formation of less-fluorinated acids, but mineralization has not been demonstrated, and the enzymes and organisms responsible were not identified.

Consequently, the environmental behaviour of most PolyFCAs is uncertain, and extrapolation of the few results to real-world wastewater treatment or natural settings is currently not justified.

6.1 Abiotic degradation

6.1.1 Degradation in the atmosphere

No experimental data was found.

6.1.2 Degradation in water

No experimental data was found.

6.2 Biotic degradation

6.2.1 Degradation in water and sediments

No experimental data was found.

6.2.2 Degradation in soil

No experimental data was found.

6.2.3 Degradation in WWTPs

Only two studies were found with information on biotransformation of polyFCAs.

Table 6-1. Biotic degradation of PolyFCAs in WWTP. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
MeU-C4c (C₄H₃F₃O₂)* (10.5 mg/L)	KB1 enrichment culture dominated by <i>Dehalococcoides</i>	Anaerobic (34 °C), 63 days	Not provided	100 % transformation, 10 % (abiotic) defluorination	(Y. Yu et al., 2022)
MeU-C5d (C₅H₂F₆O₂)* (15.6 mg/L)	KB1 enrichment culture dominated by <i>Dehalococcoides</i>	Anaerobic (34 °C), 63 days	Not provided	>90 % transformation, 60–80 % defluorination through abiotic pathways	(Y. Yu et al., 2022)
MeU-C5e (C₅H₅F₃O₂)* (11.6 mg/L)	KB1 enrichment culture dominated by <i>Dehalococcoides</i>	Anaerobic (34 °C), 63 days	Not provided	>90 % biotransformation. No F– release	(Y. Yu et al., 2022)
3,3,3-Trifluoropropionic acid (C₃H₃F₃O₂) (6.4 mg/L)	Activated sludge communities	Aerobic, pH = 7.0, 336 days	Not provided	100 % biotransformation, 84 % defluorination	(Che et al., 2021)
2-Fluoropropionic acid (C₃H₅F₂O₂) (4.6 mg/L)	Activated sludge communities	Aerobic, pH = 7.0, 336 days	Not provided	100 % biotransformation, 21 % defluorination	(Che et al., 2021)
5,5,5-Trifluoropentanoic acid (C₅H₇F₃O₂) (7.8 mg/L)	Activated sludge communities	Aerobic, pH = 7.0, 336 days	Not provided	100 % biotransformation, 71 % defluorination	(Che et al., 2021)
4,5,5-Trifluoropent-4-enoic acid (C₅H₅F₃O₂) (7.7 mg/L)	Activated sludge communities	Aerobic, pH = 7.0, 336 days	Not provided	30 % biotransformation, 37 % defluorination	(Che et al., 2021)

* See Figure below for structures

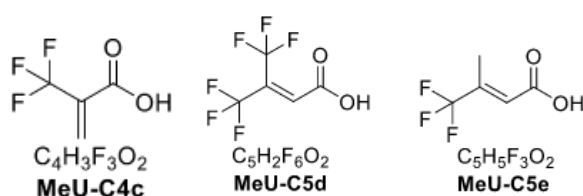


Figure 6-1. Structures of MeU-C4c, MeU-C5d and MeU-C5e (Y. Yu et al., 2022).

Yu et al. (2022) investigated biotransformation and abiotic degradation of the three unsaturated PolyFCAs studied under anaerobic conditions during 63 days incubation with an enrichment culture. Even though transformation of all three compounds, partially even defluorination, could be confirmed, it is not entirely clear, how much can be attributed to biotic and abiotic transformation.

Che et al. (2021) studied structure reactivity relationships in the aerobic defluorination of C3–C5 PolyFCAs by activated sludge communities. Four structures exhibited greater than 20% defluorination, with 3,3,3-trifluoropropionic acid being almost completely defluorinated. They also demonstrated that the defluorination occurred via cometabolism.

Thus, evidence exist that (bio)transformation of PolyFCAs can occur, however, the available studies also demonstrate the specific structures have a fundamental impact in the biodegradation potential, as well as it is unknown to which extent the study results apply to real-world wastewater treatment conditions.

7. Perfluoroalkane sulfonic acids (PFSAs)

Perfluoroalkane sulfonic acids (PFSAs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, and a hydrophilic $-\text{SO}_3\text{H}$ head group. PFSAs follow the general molecular formula $\text{C}_n\text{F}_{(2n+1)}\text{SO}_3\text{H}$, where n represents the number of carbon atoms in the perfluoroalkyl chain.

For example, when $n = 8$, the formula becomes $\text{C}_8\text{F}_{17}\text{SO}_3\text{H}$, corresponding to Perfluorooctanesulfonic acid (PFOS). Its condensed structural formula can be written as $\text{CF}_3-(\text{CF}_2)_7-\text{SO}_3\text{H}$. Other prominent members of the PFSA homologous series include Perfluorobutanesulfonic acid (PFBS) and Perfluorohexanesulfonic acid (PFHxS), which differ in chain length and associated physicochemical properties.

Summary of review findings:

Some individual perfluoroalkane sulfonic acids (PFSAs) are subject to regulation in the EU. Since 2009, perfluorooctane sulfonate (PFOS) and its derivatives have been designated as persistent organic pollutants (POPs) under the international Stockholm Convention.¹⁰ In 2022, perfluorohexane sulfonic acid (PFHxS), its salts, and PFHxS-related compounds were also added to the list of POPs.¹¹

Some perfluoroalkane sulfonic acids, exemplified by PFOS and PFHxS, exhibit greater environmental persistence than their carboxylate analogues: they resist photolysis under natural light, and they show no meaningful degradation in unamended waters, sediments or conventional wastewater treatment. Laboratory studies have revealed that specialised microbes or diatom-nanoparticle systems can cleave C–S and C–F bonds, yielding shorter-chain PFSAs, PFCAs and fluoride, yet these processes require stringent redox control, high pH, iron minerals and/or engineered mediators and have not been demonstrated at field scale, nor do they reflect environmentally relevant conditions. Most studies focus on PFOS, with limited data for other chain lengths, and most studies lack comprehensive mass-balance documentation.

Consequently, based on the found data, PFSAs are currently regarded as terminal environmental contaminants.

7.1 Abiotic degradation

7.1.1 Degradation in the atmosphere

Degradation of PFSAs in the atmosphere is considered negligible as they are strong acids with very low vapor pressures, and due to the strength of the C–F bond, they are resistant to direct photolysis at relevant atmospheric temperatures and wavelengths, and by typical gas-phase oxidants (Rayne & Forest, 2009). There are no experimental gas-phase measurements for atmospheric oxidation of PFSAs as they will partition almost entirely to particles and undergo wet or dry deposition much more rapidly than gas-phase chemistry could occur.

7.1.2 Degradation in water

Compared with PFOA, PFOS is even more resistant to photolytic degradation into short-chain molecules because of differences in their terminal groups (Bentel et al., 2019; Lyu et al., 2015). Computational studies have shown that the carbon–fluorine bond dissociation energies at the α -position $-\text{CF}_2-$ (adjacent to $-\text{COO}-$) are lower than those of the corresponding α -position C–F bonds

¹⁰ List of substances subject to POPs Regulation (ECHA): [Perfluorooctane sulfonic acid \(PFOS\), its salts and PFOS-related compounds C8F17SO2X \(X = OH, Metal salt \(O-M+ \), halide, amide, and other related compounds including polymers\)](#)

¹¹ List of substances subject to POPs Regulation (ECHA): [Perfluorohexane sulfonic acid \(PFHxS\), its salts and PFHxS-related compounds](#)

in PFASs. This difference explains the higher reactivity observed experimentally in PFCAs. Furthermore, in long-chain PFASs (typically with $n \geq 5$), the central CF_2 units within the fluoroalkyl chain exhibit lower bonding energies. As a result, degradation may proceed through an alternative mechanism, in which hydrated electrons (eaq^-) preferentially attack the C–F bonds located in the middle of the PFOS chain during its conversion into shorter-chain PFAS (Bentel et al., 2019).

Taniyasu et al. (2013) suggested that PFOS, generally considered environmentally persistent, can undergo photolysis (see also discussion under PFCAs). However, Z. Wang et al. (2015b) challenged this interpretation, arguing that such photolytic degradation of PFASs is implausible under natural conditions and pointing to missing experimental evidence needed to substantiate the claim. Therefore, the study results are assumed to be not reliable.

As an example for artificial destruction, Lyu et al. (2015) investigated PFOS degradation in an aqueous, catalyst-free system. Under optimized pH (11.8) and temperature conditions (100 °C), rapid PFOS photodegradation was achieved, with a pseudo-first-order rate constant of 0.91 h^{-1} . Hydrated electrons were identified as the dominant photo-generated reductive species driving this process. This study exemplary shows the very harsh conditions and intense UV irradiation (no degradation occurred without UV irradiation) needed for an effective degradation.

7.2 Biotic degradation

PFOS is a well-known sulfonic acid that in general is considered to be recalcitrant and resistant to microbial degradation (Berhanu et al., 2023). Reports of microbial biotransformation of PFOS are usually under very specific conditions with a limited relevance to real environmental conditions.

7.2.1 Degradation in water and sediments

Similar to PFCAs, several studies have been investigated degradation of PFOS by extracellular enzymes, microbial species or in microbial settings related to water/sediments, however, this remains mainly pure culture work with limited relevance to real environmental conditions.

Table 7-1. Biotic degradation of PFASs in water (including pure culture) and sediment. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
PFOS (0.1 and 100 mg/L)	<i>Acidimicrobium sp. A6</i>	Anaerobic, pH 4.5–5, 30°C, 100 days	Not provided	Removal of up to 60%, TPs: HFBA and PFBS	(Huang & Jaffe, 2019)
PFOS (10 mg/l)	Microbial - Synergetic diatom (<i>Chaetoceros muelleri</i>)	25 °C, pH 9, 24h, Synergetic diatom (<i>Chaetoceros muelleri</i>) and iron nanoparticle system (biomicroreactor) ^a	Not provided	89% defluorination, TPs: PFOA, PFHpA, PFPA, PFPrA, TFA	(Albert et al., 2020)
PFOS (2, 60 and 600 ng/l)	Microbial - Marine, Gammaproteobacteria and Roseobacter, Flavobacteria	Aerobic, arctic water and spiked seawater, 6 days	43h	31% PFOS removal, 85% decrease in PFOS when PFOS and PFOA present	(Cerro-Gálvez et al., 2020)
PFOS (0.1 µM)	Laccase (Fungi and laccase mediator systems)	Aerobic, 4 days, 1-hydroxybenzotriazole as mediator	Not provided	No PFOS transformation	(Steffens et al., 2023)
PFOS (1mg/l)	Microbial - <i>Synechocystis sp. PCC 6803</i> (cyanobacterium)	30°C, 4 days (batch experiment), 12-15 days (continuous reactor) ^a	Not provided	71% removal (batch experiment), 88% removal (continuous reactor)	(Marchetto et al., 2021)

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
PFOS (Spiked: 100 ng/L; non-spiked: <0.91-11ng/l)	Plants - <i>Phragmites australis</i>	pH 6-8, 10-20°C, 4x1 week	Not provided	87% PFOS removal (PFOS spiked) 100% PFOS removal (non-spiked)	(Gorito et al., 2018)
PFOS (0.5 mg/L)		ECOHRs, pH 5, 22°C, 162 days, laccase, 1-hydroxybenzotriazole (HBT)	Not provided	Up to 59% transformation, partially fluorinated compounds	(Luo et al., 2018)

^a Limited environmental relevance

7 out of 8 listed experiments in Table 7-1 focus on the long-chain sulfonic acid PFOS. Data for shorter-chain PFCAs (e.g., PFBA, PFHpA) appear only as transformation products, with no dedicated degradation studies. Longer-chain PFCAs are absent altogether, highlighting a major data gap.

Aerobic systems dominate the literature. These include laccase-mediated reactions (Luo et al., 2018; Steffens et al., 2023), phototrophic/microbial consortia (Albert et al., 2020; Marchetto et al., 2021), marine microcosms (Cerro-Gálvez 2020) and planted constructed wetlands (Gorito et al., 2018). Only one clearly anaerobic study was found: *Acidimicrobium* sp. A6 operating at pH 4.5–5 over 100 days (Huang & Jaffe, 2019).

PFOS has been regarded as non-biodegradable, yet several recent laboratory studies demonstrate that under optimized conditions specialised microbial communities can attack the sulfonate, cleave C–S and C–F bonds, and shorten the perfluoroalkyl chain: *Acidimicrobium* sp. A6 (anaerobic, pH 4.5–5) removed up to 60 % of PFOS after 100 days (Huang & Jaffe, 2019).

Synergetic diatom *Chaetoceros muelleri* combined with zero-valent iron reached 89 % defluorination of PFOS in only 24 h under alkaline (pH 9) aerobic conditions (Albert et al., 2020). Arctic seawater microcosms containing Gammaproteobacteria/Roseobacter/Flavobacteria achieved 31 % PFOS loss in 6 days; when PFOS and PFOA were present together, PFOS declined by 85 % ($t_{1/2} \approx 43$ h), while PFOA was unaffected (Cerro-Gálvez et al., 2020). ECOHR (laccase + HBT) transformed ~59 % of PFOS over 162 days (Luo et al., 2018). The cyanobacterium *Synechocystis* sp. PCC 6803 removed 71 % of PFOS in 4 days (batch) and 88 % in 12–15 days (continuous reactor) (Marchetto et al., 2021) and CW microcosms with *Phragmites australis* achieved 87 % PFOS removal in spiked influent and complete (100 %) removal in non-spiked influent over four weekly treatment cycles (Gorito et al., 2018).

Across studies, short-chain fluorinated acids predominate as common transformation products, mainly short-chain PFCAs and PFSA. For example, PFBS (a PFSA) and HFBA (a PFCA) were reported by Huang & Jaffe (2019), while PFOA, PFHpA, PFPA, PFPrA, and TFA (all PFCAs) were identified by Albert (2020). Fluoride and sulfate release were also observed.

Conditions that favour degradation are the presence of suitable electron donors under micro-aerobic/anaerobic conditions (*Acidimicrobium* A6), high pH (~9) with iron nanoparticles, continuous phototrophic culture, and redox-active mediators (laccase + HBT) while conditions that impede degradation are strong sorption to enzymes or biomass causing false positives (Steffens et al., 2023), slow kinetics in mediator-free laccase tests, and the intrinsic recalcitrance of PFOS especially under neutral aerobic conditions.

Consequently, while specialised microbes and microbial settings demonstrate that PFOS is not entirely recalcitrant, the turnover under real environmental conditions is expected to be minimal. Observed degradation relies on strict laboratory conditions; field-scale relevance remains uncertain.

7.2.2 Degradation in soil

Some studies have investigated degradation of PFOS by microbial species related to soil, however, this remains mainly pure culture work with limited relevance to real environmental conditions (Table 7-2).

Table 7-2. Biotic degradation of PFSA in soil. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species origin	or	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
PFOS (0.1 mg/L)	Soil/water microcosm		Aerobic, pH >7, 45 days, lactate, cVOCs, WBC-2	49-101 days	Up to 46% degradation, TPs: PFHxS, PFPeS and PFBS	(Lorah et al., 2024)
PFOS (1000 mg/L)	<i>Pseudomonas plecoglossicida</i> 2.4-D		Aerobic, pH 7, 28 °C, 6 months	Not provided	Complete degradation detected in liquid media, PFHpA and F ⁻ release, 75 % PFOS degradation in soil in 6 months	(S. P. Chetverikov et al., 2017)
PFOS	<i>Ensifer moralensis</i> H16		Aerobic, months	6 Not provided	Up to 88% degradation in soil, TPs: PFHpA, F ⁻	(S. Chetverikov & Hkudaigulov, 2021)
PFOS (250 mg/L)	<i>Pseudomonas mosselii</i> 5(3)		Aerobic, 28 °C, pH 5.2, 7 days	Not provided	100% reduction, TPs: Fluoride, PFHpS, PFHxS, PFHxA (70%)	(S. Chetverikov et al., 2023)
PFOS (1000 mg/L)	<i>Ensifer adhaerens</i> M1		Aerobic, pH 6-8, 26-28 °C, 6 days	Not provided	Complete transformation, TPs: PFHpA, fluoride	(S. P. Chetverikov & Loginov, 2019)
PFOS (1000 mg/L)	<i>Ensifer adhaerens</i> M1		Aerobic, 28 °C, pH 4-5, 140 hours	Not provided	Complete degradation, TPs: 2-perfluorooctane, 3-perfluoroheptanoic acid, fluoride (152 mg/L)	(Danil et al., 2022)

Several recent laboratory studies demonstrate that under optimized conditions specialised microbial communities can attack the sulfonate, cleave C–S and C–F bonds, and shorten the perfluoroalkyl chain: *Ensifer moralensis* H16, isolated from PFOS-contaminated soil, degraded PFOA, yielding PFHpA and four equivalents of fluoride. After treating contaminated soil in a model experiment degradation of PFOS up to 88 % was achieved after 6 months (S. Chetverikov & Hkudaigulov, 2021). Two independent studies with *Ensifer adhaerens* M1 isolated from soil reported almost complete disappearance when using PFOS as a sole carbon source, releasing up to 152 mg/L fluoride from 1000 mg/L PFOS in five days (S. P. Chetverikov & Loginov, 2019; Danil et al., 2022). *Pseudomonas plecoglossicida* 2.4-D achieved complete utilisation of 1000 mg/L PFOS within six days, generating PFHpA and releasing fluoride; soil tests showed 75 % PFOS loss after 6 months (S. P. Chetverikov et al., 2017). *Pseudomonas mosselii* strain 5(3), obtained from herbicide-impacted soil, likewise completely transformed PFOS within five days, but another pathway was assumed (S. Chetverikov et al., 2023).

Beyond pure cultures, bio-augmentation of AFFF-impacted soil with the dehalogenating consortium WBC-2 and chlorinated volatile organic compound (CVOC) co-contaminants showed up to 46±11 % PFOS mass loss in 45 days with specific microcosm treatment; the accompanying enrichment of sulfate-reducing *Desulfosporosinus* and *Sulfurospirillum* as likely initiators of desulfonation (Lorah et al., 2024). Consequently, while specialised soil microbes demonstrate that PFOS are not absolutely recalcitrant, their turnover in most soils is still expected to be minimal, and the environmental significance of the newly discovered routes awaits confirmation.

Although these proofs-of-concept overturn the notion that PFOS is absolutely recalcitrant, they rely on strict laboratory conditions; field-scale relevance remains uncertain.

7.2.3 Degradation in WWTPs

Some studies have been found investigating degradation of PFSA by microbial species related to wastewater treatment, however, many of these studies are based on pure culture work or other experimental conditions with limited relevance to real environmental conditions (Table 7-3). With one exception, all studies are concerned with PFOS, a single study also considering the C4 PFBS.

Table 7-3. Biotic degradation of PFSAs in WWTPs. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
PFOS (50 µg/L)	subsurface flow treatment wetlands microbial consortia in constructed wetland, main bacterial species were Proteobacteria, Firmicutes, Bacteroidetes, Planctomycetes, Chloroflexi, and Actinobacteria	Anaerobic, four kinds of lab-scale, iron-based Constructed wetlands were prepared with magnetite, siderite, sponge iron, and hematite	Not provided	na	(H. Ma et al., 2024)
PFOS (10, 1 and 0.1 mg/L)	WWTP	Aerobic, 25 °C (normal conditions), incubation at 35 °C and 37 °C for bacterial cultures, pH Neutral (~7), 96-hour incubation	Not provided	PFHxA, PFHpA, PFPeA	(Chiriac et al., 2023)
PFOS (100 µg/L)	sewage water, domestic (simulated)	Anaerobic, 35 °C, pH 6, 10 days	Not provided	pentadecafluoroheptane-1-sulfonic acid, 1,1,1,2,2,3,3,4,4,5,5,6,6,7,7-pentadecafluoroheptane, 2,2,3,3,4,4,5,5,6,6,7,7,8,8-pentadecafluorooctanal, 2,2,3,3,4,4,5,5,5-nonafluoropentanal	(De S. Furtado et al., 2023)
PFOS (1.4–1.8 mg/L)	pure culture strain of <i>Pseudomonas aeruginosa</i> HJ4 from WWTP sludge	Aerobic, 30-37 °C, pH 7-9, 2 days	Not provided	Up to 67% PFOS reduction, minor detection of PFBS and PFHxS (4–26 ng/L), no fluoride production	(Kwon et al., 2014)
PFOS (2 mg/L)	Bacterial consortium with <i>Paracoccus</i> , <i>Hyphomicrobium</i> from activated sludge enclosed in capsules	Aerobic, 30 °C, np, 6 weeks	Not provided	PFBS, 3,3,3-trifluoropropionic acid	(Sorn et al., 2023)
13C8 PFOS (Not available)	sludge	Anaerobic, 35 °C, pH 7.6 ± 0.1, 8 weeks	Not provided	C5–C7 PFCAs; PFBS	(Y. Li et al., 2021)
PFOS (~4 mg/L for mixture 1, ~2 mg/L for mixture 2)	sludge	Aerobic and Anaerobic, 2–3.5 months	Not provided	No significant degradation	(Sáez et al., 2008)
PFBS (~4 mg/L for mixture 1, ~2 mg/L for mixture 2)	sludge	Aerobic and Anaerobic, 2–3.5 months	Not provided	No significant degradation	(Sáez et al., 2008)

Based on their previous work showing that added iron minerals can enhance degradation of PFOS (and PFOA) in constructed wetlands (H. Ma et al., 2023), Ma et al. (2024) investigated this topic further to elucidate the microbial extracellular electron transfer mechanisms mediated by iron minerals. The 2024 published experiments showed that addition of iron minerals increased the PFOS and PFOA removal by 7.27% - 16.42% compared to the control group. Removal pathways were 3-6 % substrate adsorption, 1% plant uptake, 26-37% other, 58-71% effluent, where microbial transformation is expected to be in the “other” pathway category (Ma et al., 2024).

The 2023 published experiments showed even higher removal percentages at 57.2% for PFOS (and 63.9% removal of PFOA), which was 35% (36.8% for PFOA) higher than in the control test (H. Ma et al., 2023). The authors found an enhanced electron transfer in iron mineral-based CWs, increase in cytochrome C reductase and increased Fe(III)/Fe(II) ratio, as well as increasing proportions of microbes with PFAS removal capacity, e.g., *Burkholderiae* sp. and *Pseudomonas* sp.). Based on their findings, the authors conclude that supplementation with iron mineral could enhance PFOA and PFOS removal in CWs.

Additional studies involving the addition of different iron minerals in constructed wetlands for wastewater treatment have been found from China (Ji et al., 2023; Kang et al., 2023), but are not discussed here in further detail due to their limited environmental relevance.

Chiriac et al. (2023) presents results on PFOS degradation with pure culture strains of *Pseudomonas aeruginosa* and *Pseudomonas putida* during 96 hours, with resulting degradation percentages ranging from 6.35-46.9% depending on applied initial concentration and test system. It is noted, however, that the study does not document information on mass balance in the different test systems, fluoride measurements were not conducted and potential sorption issues to test material were not addressed in the study, limiting the reliability of the findings. Also, these results were derived pure culture studies with limited environmental relevance. However, the authors suggest that use of such microorganisms can contribute to the improvement of wastewater treatment processes.

De S. Furtado et al. (2023) assessed the degradation of PFOS in anaerobic conditions using simulated domestic sewage as co-substrate. After 10 days, at pH 6 and 35°C, a 24% PFOS degradation was achieved, which the authors interpreted as biodegradation, as they accounted for sorption of PFOS to experimental equipment (bottles) and sludge, and measured degradation products.

Based on the identified degradation products and comparison with other studies, they suggest PFOS degradation process may proceed by three mechanisms:

- i. The first mechanism is the cleavage of the C–S bond, due to its lower bond energy compared to the C–F bond and C–C bonds, followed by the loss of a CF₂ unit and recombination of the perfluoroalkyl chain with the sulfonic group resulting in the degradation product pentadecafluoroheptane-1-sulfonic acid, C₇F₁₅SO₃H. In the next step, the sulfonic group leaves the pentadecafluoroheptane-1-sulfonic acid, followed by the addition of one hydrogen, resulting in the second degradation product of pentadecafluoroheptane, C₇F₁₅H.
- ii. In a second mechanism, PFOS loses the sulphonic group and, after being hydroxylated, generates the alcohol, C₈F₁₇OH. Afterwards, that alcohol is defluorinated generating pentadecafluorooctanal, C₇F₁₅CHO.
- iii. The third mechanism begins with the oxidation of PFOS to the corresponding carboxylic acid PFOA (C₇F₁₅COOH). PFOA is then anticipated to undergo a series of degradation reactions giving rise to a homologous series, whose members differ by –CF₂. Via subsequent hydroxylation and defluorination, the detected degradation product nonafluoropentanal, C₄F₉CHO, was apparently formed. The authors note that this is the first time that this degradation product was reported.

It is noted that De S. Furtado et al. (2023) do not provide information on fluoride release or mass balance, however, due to the experimental setup, it appears likely that PFOS biodegradation actually occurred.

In summary, current evidence indicates that conventional aerobic activated-sludge processes achieve little or no biodegradation of PFOS. Recent studies reporting up to of 67% PFOS biotransformation (Kwon et al. (2014)) rely on pure cultures and/or short-term batch tests lacking mass-balance or fluoride data, limiting their relevance to full-scale WWTPs.

Under anaerobic conditions, some indications of partial PFOS and PFSA defluorination have been reported in specialized systems, including iron-reducing and Feammox-type setups involving microorganisms such as *Acidimicrobium* sp. A6. However, it remains unclear whether PFOS functions as a true terminal electron acceptor supporting microbial growth in these systems. The available evidence may instead suggest that PFOS transformation occurs cometabolically under strongly reducing conditions, with Fe(III) or other electron acceptors serving as the primary drivers of microbial energy conservation.

Overall, the few reported PFSA biodegradation cases occur under special redox and microbial conditions that are not representative of typical environmental conditions in conventional WWTPs or constructed wetlands.

8. Perfluoroalkyl phosphonic acids (PFPAs)

Perfluoroalkyl phosphonic acids (PFPAs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, and a hydrophilic -P(O)(OH)₂ head group. PFPAs follow the general molecular formula $C_nF_{(2n+1)}PO_3H_2$, where n represents the number of carbon atoms in the perfluoroalkyl chain.

For example, when $n = 8$, the formula becomes $C_8F_{17}PO_3H_2$, corresponding to perfluorooctylphosphonic acid (PFOPA). Its condensed structural formula can be written as $CF_3-(CF_2)_7-P(O)(OH)_2$. Other members of the PFPA homologous series include perfluorohexylphosphonic acid (PFHPA) and perfluorobutylphosphonic acid (PFBPA), which differ in chain length and associated physicochemical properties.

Summary of review findings:

Only few studies examining perfluoroalkyl phosphonic acids have been found, all of which report an absence of abiotic or biotic degradation under environmentally relevant conditions. Based on the reviewed studies, PFPAs are non-volatile, hydrolytically stable, and show no measurable transformation in aerobic sludge, where removal of some homologues appears to result solely from sorption to biomass rather than metabolism. In a comparative assessment, (Z. Wang et al., 2016) concluded that, based on the limited information available, PFPAs are likely to exhibit high persistence in the environment.

8.1 Abiotic degradation

8.1.1 Degradation in the atmosphere

No experimental information on atmospheric degradation of PFPAs was found. They are expected to behave similarly to PFCAs and PFSAs (Z. Wang et al., 2016).

8.1.2 Degradation in water

Similarly to PFCAs and PFSAs, PFPAs show high resistance to various agents including heat, oxidants and bases in aqueous media (Z. Wang et al., 2016). PFPAs are not expected to undergo hydrolysis under environmentally relevant conditions (ECHA, 2025).

8.2 Biotic degradation

8.2.1 Degradation in water and sediments

No experimental information was found.

8.2.2 Degradation in soil

No experimental information was found.

8.2.3 Degradation in WWTPs

According to ECHA (2025), biotransformation of PFPAs has not been observed. No transformation was observed in rats exposed to C8 PFPA (Joudan et al., 2017).

Only one study about PFPAs degradation in wastewater treatment was found; (Llorca et al., 2018) studied adsorption/desorption in water and sludge, microbial degradation of PFHxPA, PFOPA, PFDPA in laboratory scale bioreactors under aerobic conditions during a 90 day period according to the OECD 309 aerobic biodegradation test guideline, and they also characterised the microbial communities in the sludge by gene analysis methods. PFHxPA and PFOPA were found to be highly resistant to aerobic biodegradation in sludge. PFDPA was found to be taken up by microorganisms and floc biomass but was not metabolized. The authors observed that flocculation plays a major role for removal of PFDPA from aqueous phase, but species responsible for uptake could not be fully identified. Overall, the experimental findings do not indicate that biodegradation is a relevant degradation pathway for PFPA in aerobic wastewater treatment.

9. Perfluoroalkyl phosphinic acids (PFPIAs)

Perfluoroalkyl phosphinic acids (PFPIAs) consist of two perfluoroalkyl tails, in which all the hydrogen atoms in the carbon backbones are substituted with fluorine atoms, and a hydrophilic -P(O)OH head group. PFPIAs follow the general molecular formula $(C_nF_{(2n+1)})(C_mF_{(2m+1)})PO_2H$, where n and m are typically even numbers between 4 and 12. This is often abbreviated as x:y PFPIA, such as 6:6 or 6:8 PFPIA.

For example, 6:6 PFPIA corresponds to $(C_6F_{13})_2PO_2H$, while 6:8 PFPIA corresponds to $(C_6F_{13})(C_8F_{17})PO_2H$. Prominent members include bis(perfluorohexyl)phosphinic acid (6:6 PFPIA) and perfluorohexyl perfluorooctyl phosphinic acid (6:8 PFPIA).

Summary of review findings:

Current evidence indicates that perfluoroalkyl phosphinic acids are highly persistent (X. Wang et al. 2025); they neither volatilise nor undergo measurable abiotic degradation under normal conditions, and only limited microbial degradation has been observed. In soils, aerobic microbes can slowly cleave the P–C bond, producing the more recalcitrant phosphonic acids, with reported half-lives ranging from roughly four months to over three years depending on chain length and sorption strength; in aquatic systems, strong binding to sediments dominate their fate, and degradation is largely confined to partial *in vivo* cleavage to PFPAs.

With no data found in this review from wastewater treatment and only a few laboratory studies overall, the environmental fate, degradation kinetics and pathways of PFPIAs remain constrained,

and current results support the view that sequestration rather than degradation governs their long-term behaviour.

9.1 Abiotic degradation

9.1.1 Degradation in the atmosphere

No experimental information on atmospheric degradation of PFPIAs was found. They are expected to behave similarly to PFCAs and PFSAs (Z. Wang et al., 2016).

9.1.2 Degradation in water

Hydrolysis of PFPIAs can occur in aqueous media when heated or alkalized, resulting in the cleavage of their P–C bonds and producing PFPAs and perfluoroalkanes (which can later oxidize to PFCAs) (Z. Wang et al., 2016). However, the relevance of abiotic hydrolysis under environmentally relevant conditions remains unclear (Z. Wang et al., 2016).

9.2 Biotic degradation

Recently, X. Wang et al. (2025) analysed the fate of PFPIAs in different environmental compartments which is visually summarized in Figure 9-1. The authors concluded that PFPIAs exhibit strong sorption to soils and sediments, especially longer-chain congeners, leading to environmental persistence and potential long-term contamination. According to X. Wang et al. (2025) they have limited and chain-length dependent microbial biotransformation in soil, rapid sorption to plant roots with slow translocation and metabolism, pronounced bioaccumulation and sex-specific toxicity in aquatic organisms, and partial metabolic cleavage yielding persistent metabolites in mammals, indicating potential entry into food webs and complex toxicokinetic profiles. Individual study results are discussed in the following for the different pathways.

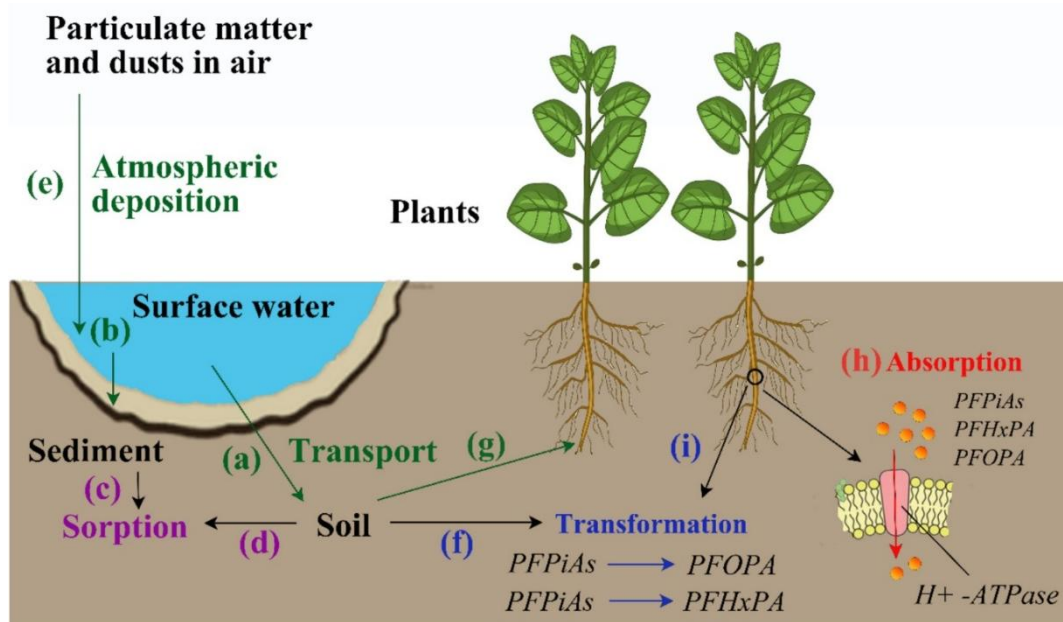


Figure 9-1. Transport and transformation of PFPIAs in the atmosphere, soil sediments, surface water, and plants according to (X. Wang et al., 2025)... Transport and transformation of PFPIAs in the atmosphere, soil sediments, surface water, and plants according to (X. Wang et al., 2025). Transport and transformation of PFPIAs in the atmosphere, soil sediments, surface water, and plants according to (X. Wang et al., 2025).

9.2.1 Degradation in water and sediments

PFPIAs bind tightly to sediments, especially the longer chains (X. Wang et al., 2025). It was found that aquatic organisms such as zebrafish, carp and rainbow trout bioaccumulate PFPIAs could also convert C₆/C₆, C₆/C₈ and C₈/C₈ congeners to the corresponding PFPAs *in vivo* (Lee et al., 2012; M. Liu et al., 2020; Z. Wang et al., 2016).

9.2.2 Degradation in soil

Similar to sediments, PFPIAs demonstrate strong sorption affinity for soils, particularly longer-chain congeners (X. Wang et al., 2025, 2025). Recent studies indicate that aerobic soil microbes can partially transform PFPIAs, although this process is slow and strongly influenced by sorption and chain length. Despite their overall persistence, the formation of corresponding perfluoroalkyl phosphonic acids - PFHxPA from C₆/C₆ and PFOPA from C₈/C₈ - was observed in active soils (Gao et al., 2023). In agricultural loam soils, the degradation of C₆/C₆ and C₈/C₈ PFPIAs exceeded that of sterilized controls by 13.8–23.5% after 224 or 300 days demonstrating a measurable microbial contribution (Gao et al., 2023). C₆/C₆ PFPIA degraded faster (half-lives of 392 and 603 days) than its C₈/C₈ analogue (half-lives of 746 and 1155 days) in two different soils. It is important to note that a decrease in PFPIAs was also observed in sterilized soil, attributed to irreversible sorption due to their strong hydrophobicity.

Hydroponic experiments with wheat demonstrated that both homologues rapidly sorb to root surfaces driven by hydrophobic interactions. A small fraction mediated by H⁺-ATPase enters the tissue; nevertheless, slow phase-I metabolism in plants produced tiny amounts of PFHxPA or PFOPA, along with intermediates such as 1H-PFHx and 1H-PFO, which were translocated to the shoots (J. Zhou et al., 2020).

Collectively, these findings show that soil microbial communities can mediate limited cleavage of the P-C bond in PFPIAs, but the rate is governed by molecular size and bioavailability, leading to the conclusion that sequestration rather than mineralization dominates their long-term fate (X. Wang et al., 2025).

9.2.3 Degradation in WWTPs

PFPIAs were metabolized in male Sprague-Dawley rats to the corresponding PFPAs and 1H-perfluoroalkanes (1H-PFAs), with 70% and 75% biotransformation 2 weeks after a single bolus dose for C₆/C₈ PFPIA and C₈/C₈ PFPIA, respectively (Joudan et al., 2017).

10. Per- and polyfluoroalkylether carboxylic acids (PFECAs)

Per- and polyfluoroalkylether carboxylic acids (PFECAs) consist of a per- or polyfluoroalkyl chain with an ether group -O- and a hydrophilic -COOH head group. A general molecular formula for one type of PFECAs with perfluoroalkyl chains can be for example $C_nF_{(2n+1)}O(CF_2)_mCOOH$, where n and m represent chain lengths. However, different variants are possible, including substances with multiple ether groups.

A prominent example is hexafluoropropylene oxide dimer acid (HFPO-DA, commonly known as GenX), which contains an ether linkage within a branched perfluorinated backbone. Another example is ammonium 4,8-dioxa-3H-perfluorononanoate (ADONA), a short-chain PFECA.

Summary of review findings:

The identified data indicate that investigated per- and polyfluoroalkyl ether carboxylic acids (PFECAs) such as GenX can be as persistent as legacy PFCA: they resist hydrolysis, photolysis and microbial degradation in natural waters, show only trace ($\approx 5\%$) anaerobic biotransformation over many months, and display measurable abiotic degradation only under artificially nitrate-rich, high-irradiance conditions. Soil and field observations suggest some transformation of chloro-substituted PFPECAs to more oxidised analogues. Ether linkages are hypothesized as potential reaction centers under environmental conditions; however, no evidence of ether scission has been identified.

10.1 Abiotic degradation

10.1.1 Degradation in the atmosphere

No experimental information on atmospheric degradation of PFECAs was found within this review. Based on limited information, perfluoroether chains in PFECAs are expected to have similar resistance to photolysis and reaction with $\cdot\text{OH}$ radicals as PFCAs, under environmentally relevant conditions (Z. Wang et al., 2015a). Gomis et al. (2015) predicted similar physicochemical properties and persistence for PFECAs as compared to their PFCA counterparts. Zhang et al. (2023) found various PFECAs to be resistant to UV photolysis after 8 hours at a UV dose of $\sim 32,000 \text{ mJ/cm}^2$. HFPO-DA, a common PFECA, also known as GenX, has been found to be resistant to degradation by hydroxyl radicals and stable in air with a half-life on the order of years (Pica et al., 2019; USEPA, 2021).

10.1.2 Degradation in water

According to (Z. Wang et al., 2015a), experimental hydrolysis results of different PFECAs (including ADONA, GenX) show essentially no degradation at elevated temperatures, at different pH levels, and when oxidising agents are added, demonstrating that, under environmentally relevant conditions, perfluoroether chains are as resistant to photolysis, reactions with $\cdot\text{OH}$ radicals and hydrolysis as perfluoroalkyl chains are.

In a recent study by (Folkerson & Mabury, 2024) 2H-1:2 polyfluoroalkyl ether carboxylic acid (2H-1:2 PFECA) exhibited no significant loss in Lake Ontario surface water over a 10-day exposure, a finding attributed to the low oxidative capacity of this system and low concentrations of nitrate. However, in nitrate-enriched artificial surface water, 2H-1:2 PFECA showed an observable decrease, with a half-life of 557 ± 25 hours. The perfluorinated ether control (PF5OHxA) did not degrade under either set of conditions, confirming that the presence of polyfluorinated (rather than perfluorinated) carbon in the PFECA structure increases susceptibility to radical-induced abiotic degradation. Dark controls showed negligible losses, pointing to photolysis as the sole or dominant abiotic degradation pathway. This demonstrates that while some polyfluoroether derivatives can undergo abiotic breakdown in radical-rich environments, its environmental persistence is high under more realistic surface water conditions.

10.2 Biotic degradation

10.2.1 Degradation in water and sediments

Recent microcosm tests conducted with a native microbial consortium from an aquifer impacted by PFAS and chlorinated solvents investigated the biodegradation of HFPO-DA in a soil/water system under anaerobic conditions. After 10 months of microbial treatments $5.5 \pm 3.0\%$ of HFPO-DA was biologically removed, with a defluorination yield of $3.0 \pm 0.6\%$ (Tang et al., 2024). The authors proposed a biodegradation pathway that is depicted in Figure 10-1.

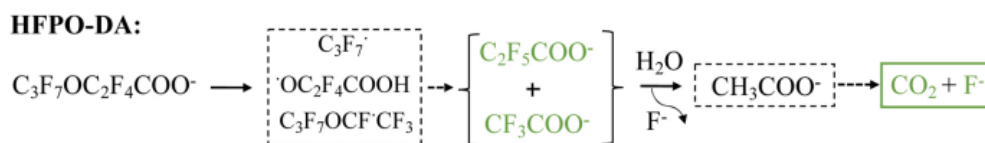


Figure 10-1. Biodegradation pathway of HFPO-DA based on intermediates identified in this study (green) and previous studies (dashed lines) (Tang et al., 2024).

10.2.2 Degradation in soil

One field study was identified that investigated the fate of chloro-perfluoropolyether carboxylates (Cl-PFPECAs) in soil (Evich et al., 2022). Multiple congeners of Cl-PFPECAs were detected in New Jersey soils surrounding a manufacturing facility. Suspected transformation products of these congeners, including H-PFPECAs, epox-PFPECAs, and diOH-PFPECAs, were tentatively identified during the screenings. The degradation pathway was not further investigated in this study; therefore, it is unclear if abiotic, biotic degradation or a mixture of several degradation pathways was observed. Although ether linkages are hypothesized as potential reaction centers under environmental conditions, no evidence of ether scission has been found. Despite exhaustive analysis of high-resolution data from soils containing Cl-PFPECAs, the authors did not find any indication of ether scission.

10.2.3 Degradation in WWTPs

The only study identified that investigated the PFECA GenX (see structure in Figure 10-2) in wastewater treatment, is the study by (Y. Yu et al., 2022) with a 63 days anaerobic test with a defluorinating enrichment culture with two different experimental set-ups. They found that the tertiary sp^3 C–F bond in the branched moiety of GenX was, unlike the studied unsaturated fluorinated carboxylic acids, not microbially cleaved, although the bond dissociation energy of the tertiary sp^3 C–F bond in GenX is rather low. Yu et al. (2022) anticipate that the saturated structure of GenX made it less biodegradable.

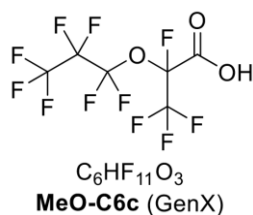


Figure 10-2. Structure of GenX (from Yu et al. 2022)

11. Per- and polyfluoroalkylether sulfonic acids (PFESAs)

Per- and polyfluoroalkylether sulfonic acids (PFESAs) consist of a per- or polyfluoroalkyl chain with an ether group, and a hydrophilic $-\text{SO}_3\text{H}$ head group. A general molecular formula for one type of PFESAs with perfluoroalkyl chains can be, for example, $\text{C}_n\text{F}_{(2n+1)}\text{O}(\text{CF}_2)_m\text{SO}_3\text{H}$, where n and m represent chain lengths. However, different variants are possible, including substances with multiple ether groups, thioethers, and non-fluorinated carbons.

A prominent example is 9-chlorohexadecafluoro-3-oxanonane-1-sulfonic acid (commonly referred to as F-53B), a chlorinated polyfluoroether sulfonic acid.

Summary of review findings:

Per- and polyfluoroalkyl ether sulfonic acids are expected to be similar to persistent perfluoroalkyl acids (Z. Wang et al., 2015a). Current evidence exemplified by the chlorinated surfactant F-53B (6:2 Cl-PFESA) and newer 2H-1:2 PFESA analogues show: they are essentially non-volatile, withstand hydrolysis and natural photolysis, and show no meaningful biodegradation in water, soil or conventional wastewater treatment tests. Only when nitrate-rich artificial waters were irradiated for weeks, a polyfluorinated PFESA lost roughly half its mass, while fully fluorinated ether controls remained intact, indicating that high radical flux is required to degrade the molecule. OECD closed-bottle assays and crop-soil microcosms confirmed that Cl-PFESAs are not readily biodegradable and persist at least several months, with sorption rather than transformation governing their removal.

11.1 Abiotic degradation

11.1.1 Degradation in the atmosphere

No experimental information on atmospheric degradation of PFESAs could be found. Based on limited information, perfluoroether chains in PFESAs are expected to have similar resistance to photolysis and reaction with $\cdot\text{OH}$ radicals as PFASs, under environmentally relevant conditions (Z. Wang et al., 2015a). Gomis et al. (2015) predicted similar physicochemical properties and persistence for PFESAs as compared to their PFSA counterparts. (C. Zhang et al., 2023) found various PFESAs to be resistant to UV photolysis after 8 hours at a UV dose of $\sim 32,000 \text{ mJ/cm}^2$. According to the UK Environment Agency, 6:2 Cl-PFESA, a common PFESA, also known as F-53B (molecular formula $\text{C}_8\text{ClF}_{16}\text{O}_4\text{SK}$), is unlikely to volatilize or be photodegraded in air (UK Environment Agency, 2023).

11.1.2 Degradation in water

Similar to PFESA, 2H-1:2 polyfluoroalkyl ether sulfonic acid (2H-1:2 PFESA) was tested for abiotic degradation under simulated sunlight in both natural Lake Ontario surface water and artificial surface water (ASW) with high nitrate levels by (Folkerson & Mabury, 2024). Minimal degradation of 2H-1:2 PFESA was detected over approximately 10 days in Lake Ontario water, indicating that at environmentally relevant conditions, where reactive radical concentrations such as nitrate are low, PFESA is highly persistent. In contrast, under ASW conditions, designed to maximize hydroxyl radical production, PFESA underwent gradual but measurable phototransformation, with a half-life of about 570 ± 17 hours. Importantly, controls with fully fluorinated PFESA showed negligible abiotic degradation in both conditions. The lack of transformation in dark control samples confirmed photolytic (not microbial) mechanisms as the main pathway for PFESA breakdown. Thus, 2H-1:2 PFESA demonstrates potential for photolytic abiotic degradation under sustained, high-radical conditions, but otherwise resists breakdown in typical surface water environments.

Wang et al. (2013) investigated the biodegradation of F-53B in water and found that the compound degraded only slowly, with less than 24% of theoretical oxygen demand (ThOD) depleted after 21 days and below 44% after 28 days. These results indicate that F-53B does not meet the OECD criteria for ready biodegradability, which require at least 60% removal of ThOD during the test period. Consequently, F-53B is classified as not readily biodegradable, suggesting it is not efficiently broken down in standard wastewater treatment plants. However, the authors noted that the rigor of the OECD 301 test means that failure to meet its criteria does not necessarily preclude some degree of environmental degradation. The findings instead highlight the need for additional studies to fully understand the environmental persistence and eventual breakdown of F-53B and similar PFESA compounds.

11.2 Biotic degradation

11.2.1 Degradation in water and sediments

No experimental data was found.

11.2.2 Degradation in soil

No degradation of F-53B was observed in soil containing maize plants (*Zea mays* L.) over a period of 84 days, under natural conditions designed to simulate normal field growth (including aeration and exposure to light) (Just et al., 2022). Similarly, in another study no significant biotransformation of 6:2 Cl-PFESA and 8:2 Cl-PFESA occurred during 105 days in soil under aerobic conditions (H. Chen et al., 2018). Both substances were also resistant to heat-activated persulfate oxidation. The authors concluded that Cl-PFESAs are equally recalcitrant as PFOS in addition to being more sorptive.

11.2.3 Degradation in WWTPs

Wang et al. (2013) investigated microbial degradation of F-53B in sewage sludge sampled from a sewage treatment facility during a 28-day period, according to OECD Guideline 310D (Closed bottle test, CBT). Thus, the percentage of biodegradation was derived from the biological oxygen demand (BOD) and the theoretical oxygen demand (ThOD).

Biodegradation was not completed during the course of the test for F-53B. Less than 24% of ThOD was depleted after 21 days and <44% after 28 days. F-53B showed slow degradation throughout the test period and did not meet the OECD criteria for ready biodegradation (60% removal of ThOD). F-53B was classified as not readily degradable in the CBT, which implies that it is probably not biodegraded efficiently in wastewater treatment plants.

The degradation of di(polyfluoroether thioether(S)-oate) sulfonate (diFESOS) to its alcohol (FESOH) and carboxylic acid (FESCA) was further investigated by (Joudan & Mabury, 2022).

12. Perfluoroalkyl dicarboxylic acids (PFdiCAs)

Perfluoroalkyl dicarboxylic acids (PFdiCAs) consist of a perfluoroalkyl chain, in which all the hydrogen atoms are substituted with fluorine atoms, and two hydrophilic -COOH head groups. PFdiCAs follow the general molecular formula $(\text{COOH})(\text{CF}_2)_n(\text{COOH})$, where n represents the number of fluorinated carbon atoms.

For example, when $n = 4$, the formula becomes $(\text{CF}_2)_4(\text{COOH})_2$, corresponding to perfluoroadipic acid. Other members of this homologous series include perfluoroglutaric acid and perfluorosebacic acid, which differ in chain length and physicochemical properties.

Summary of review findings:

Despite laboratory experiences indicating that the two electron-withdrawing carboxyl groups appear to accelerate defluorination under aggressive reductive treatments (Bentel et al., 2019), perfluoroalkyl dicarboxylic acids remain almost completely unstudied with respect to natural attenuation. No atmospheric, aquatic, soil or wastewater data were found in this review and the laboratory report demonstrating faster breakdown than PFCAs employed highly non-environmental conditions.

PFdiCAs can be assumed to share persistence of other fully fluorinated acids, but this rests on structural analogy and not on empirical evidence.

13. Perfluoroalkane disulfonic acids (PFdiSAs)

Perfluoroalkane disulfonic acids (PFdiSAs) consist of a perfluoroalkyl chain, in which all the hydrogen atoms are substituted with fluorine atoms, and two hydrophilic -SO₃H head groups. PFdiSAs follow the general molecular formula (SO₃H)(CF₂)_n(SO₃H), where n represents the number of fluorinated carbon atoms.

For example, when $n = 4$, the formula becomes (SO₃H)(CF₂)₄(SO₃H), corresponding to perfluorobutane-1,4-disulfonic acid. Other members of this homologous series include perfluorohexane-1,6-disulfonic acid and perfluorooctane-1,8-disulfonic acid, which differ in chain length and associated physicochemical properties.

Summary of review findings:

Abiotic or biotic degradation studies for perfluoroalkane disulfonic acids have not been found in this review. Their environmental persistence is therefore inferred solely from their structural similarity to the recalcitrant PFsAs. PFdiSAs are expected to be non-volatile, resistant to hydrolysis and natural photolysis, and potentially largely resistant to microbial degradation under environmental conditions.

Consequently, PFdiSAs can be assumed to share persistence of analogous PFsAs or PFcAs, for example, but this rests on structural analogy and not on empirical evidence.

14. Perfluoroalkane sulfinic acids (PFSIAs)

Perfluoroalkane sulfinic acids (PFSIAs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, and a hydrophilic -SO₂H head group. PFSIAs follow the general molecular formula C_nF_(2n+1)SO₂H, where n represents the number of carbon atoms in the perfluoroalkyl chain.

For example, when $n = 8$, the formula becomes C₈F₁₇SO₂H, corresponding to perfluorooctane sulfinic acid. Other members of this homologous series include perfluorohexane sulfinic acid and perfluorobutane sulfinic acid, which differ in chain length and physicochemical properties.

Summary of review findings:

Perfluoroalkane sulfinic acids such as perfluorooctane sulfinate (PFOSI) have scarcely been investigated; apart from an unreliable high-altitude photolysis claim, studies indicating abiotic degradation were not found. One aerobic sludge experiment detected PFOSI as a degradation intermediate from a sulfonamide precursor to PFOS, with a sub-day half-life for PFOSI itself. This observation indicates that sulfinates may be unstable under oxic, biologically active conditions and can oxidise to more recalcitrant sulfonates.

Based on the limited findings for this group, PFSIAs may be viewed as PFAS degradation intermediates potentially transforming to more recalcitrant PFsAs.

14.1 Abiotic degradation

14.1.1 Degradation in the atmosphere

No experimental information was found.

14.1.2 Degradation in water

Taniyasu et al. (2013) reported that following solar irradiation in Mt. Mauna Kea, the PFSIA perfluorooctane sulfinate, PFOSi, was 100% decomposed, and resulted in the formation of 73% PFOA, 6.1% PFHpA, 0.8% PFHxA, 0.3% PFPeA and 7.2% PFOS. However, several inconsistencies in the experimental results were noticed and therefore the study cannot be considered reliable (see chapter on PFCAs).

14.2 Biotic degradation

14.2.1 Degradation in water and sediments

No experimental information was found.

14.2.2 Degradation in soil

No experimental information was found.

14.2.3 Degradation in WWTPs

One study by Rhoads et al. (2008), who investigated aerobic biotransformation of N-ethyl perfluorooctane sulfonamidoethanol (N-EtFOSE) in activated product of EtFOSE. PFOSI itself was found to transform to 78% to PFOS with a degradation half-life of 0.73 days in the 10-day aerobic incubation study, without any detailed description of transformation pathways.

15. Fluorotelomer alcohols (FTOHs)

Fluorotelomer alcohols (FTOHs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, bonded to an ethylene group and a hydrophilic -OH head group. FTOHs follow the general molecular formula $C_nF_{(2n+1)}(CH_2)_mOH$, where n represents the number of fluorinated carbon atoms and m the number of bridging CH_2 units (typically n:2 FTOHs where m = 2).

For example, 8:2 FTOH corresponds to $C_8F_{17}(CH_2)_2OH$, and 6:2 FTOH to $C_6F_{13}(CH_2)_2OH$. Prominent members of this group include 8:2 Fluorotelomer alcohol and 6:2 Fluorotelomer alcohol, which differ in the length of the perfluorinated chain and associated physicochemical properties.

Summary of review findings:

The degradation of n:2 fluorotelomer alcohols (n:2 FTOHs) has been studied in many research articles under different environmental settings and is subject to several review articles (Berhanu et al., 2023; Titaley, 2024; W. Zhang et al., 2021). Based on the available studies, there is a general consensus that, in contrast to other PFAS-groups, FTOHs can be degraded relatively rapidly in the environment. They volatilise into the atmosphere, where reactions with hydroxyl radicals dominate, resulting in formation of fluorotelomer aldehydes, with lifetimes on the order of several days, followed by oxidation to the perfluoroacyl peroxy radicals that eventually yield PFCAs.

In surface waters and especially in aerobic sediments, soils and activated sludges, FTOHs follow analogous stepwise microbial pathways via aldehyde and unsaturated fluorotelomer telomer acids, typically dissipating within days to months. Transformation products that were consistently identified include fluorotelomer carboxylic acids (FTCAs), fluorotelomer unsaturated carboxylic acids (FTUCAs), and PFCAs (W. Zhang et al., 2021).

In general, FTOHs will be mainly transformed to C_{n-2} to C_{n+1} PFCAs as final products. Transformation rates are slow under anaerobic or substrate-rich conditions, and defluorination remains limited, but the overall weight of evidence shows that FTOHs act as mobile, short-lived precursors for PFCAs.

15.1 Abiotic degradation

15.1.1 Degradation in the atmosphere

Numerous studies including smog chamber experiments and relative rate techniques previously examined the atmospheric degradation processes of FTOHs, in which they degrade to form PFCAs. Due to their high volatility, given their high vapour pressures and non-ionic state, FTOHs have been reported to contribute significantly to the presence of PFCAs in remote areas through long-range atmospheric transport (Ellis et al., 2004; Wallington et al., 2006).

Laboratory kinetics show that the dominant removal mechanism for all FTOHs is gas-phase reaction with $\cdot\text{OH}$ radicals. While reactions with Cl atoms are fast, they are insignificant compared to $\cdot\text{OH}$ radical reactions, due to the much higher abundance of $\cdot\text{OH}$ radicals in the troposphere (Ellis et al., 2003). Reactions with ozone, direct photolysis and both wet and dry deposition are also considered negligible compared to reactions with $\cdot\text{OH}$ radicals. (Hurley et al., 2005) and (Ellis et al., 2003) determined lifetimes on the order of years with respect to reaction with ozone and potential for wet and dry deposition, respectively, whereas photolysis of FTOHs is considered unlikely, based on the UV absorption wavelengths of FTOHs being outside the wavelength regions of the lower atmosphere (Waterland et al., 2005). Rather, reaction of even-numbered FTOHs ($n:2$ FTOHs, $\text{CF}_3(\text{CF}_2)_n\text{CH}_2\text{CH}_2\text{OH}$) with $\cdot\text{OH}$ radicals results in atmospheric lifetimes of approximately 10–20 days for $n:2$ FTOHs of varying chain lengths (Ellis et al., 2003; Kelly et al., 2005). By contrast, odd-numbered FTOHs ($n:1$ FTOHs, $\text{CF}_3(\text{CF}_2)_n\text{CH}_2\text{OH}$) react an order of magnitude slower with OH resulting in lifetimes on the order of 4–6 months (L. Chen et al., 2000; Hurley et al., 2004; Papadimitriou et al., 2007). The reactivity of both $n:1$ and $n:2$ FTOHs was found to be independent of chain length (Ellis et al., 2004; Hurley et al., 2004).

Oxidation of FTOHs predominantly produces the corresponding aldehydes (fluorotelomer aldehydes: FTALs, $\text{CF}_3(\text{CF}_2)_n\text{CH}_2\text{CHO}$ for $n:2$ FTOHs and perfluoroaldehydes: PFALs, $\text{CF}_3(\text{CF}_2)_n\text{C}(\text{O})\text{H}$ for $n:1$ FTOHs), which have lifetimes on the order of a few days with respect to reaction with $\cdot\text{OH}$ radicals (Ellis et al., 2004; Hurley et al., 2004; Wallington et al., 2006). The formation of FTALs/PFALs from FTOHs is thought to occur by hydrogen abstraction at the carbon adjacent to the alcohol group, followed by O_2 addition, forming a peroxy radical that rapidly decomposes yielding the corresponding FTALs/PFALs and HO_2 . In low NO_x environments, i.e. remote or non-urban regions, the FTALs/PFALs is the sole primary product resulting from reaction of FTOHs with both Cl atoms and $\cdot\text{OH}$ radicals (Ellis et al., 2003, 2004; Hurley et al., 2004; Kelly et al., 2005). Only in situations with high NO_x , i.e. highly polluted urban regions, will the peroxy radical react with NO or NO_2 , leading to different products that inhibit direct PFCA formation, via a chain shortening mechanism (Ellis et al., 2004; Hurley et al., 2005). In general, the FTALs/PFALs undergo further OH-initiated photo-oxidation to produce perfluoro-acyl peroxy radicals that eventually yield PFCAs. Oxidation of the $n:2$ FTOHs can also yield formation of the $n:2$ fluorotelomer carboxylic acids ($n:2$ FTCAs) (Ellis et al., 2004). Smog chamber experiments and global atmospheric chemistry models conducted by Ellis et al., 2004 and Wallington et al., 2006 observed formation of PFNA, PFOA, and shorter chain PFCAs from the atmospheric oxidation of 8:2 FTOH, with an estimated carbon PFCA yield of 1–10 % and specific PFOA yield of 3–6 %, respectively.

In addition to gas-phase chemistry, heterogeneous photooxidation of FTOHs at metal-rich atmospheric surfaces has been observed. (Styler et al., 2013) conducted experiments with 6:2 FTOH irradiated on TiO_2 , SiO_2 , iron oxides, desert dust and volcanic ash and showed quantitative

production of surface-bound PFCAs, including PFHpA, PFHxA, and PFPeA, implying that mineral aerosols close to source regions may act both as an atmospheric sink for FTOHs and as a direct source of particulate PFCAs that are then available for long-range transport.

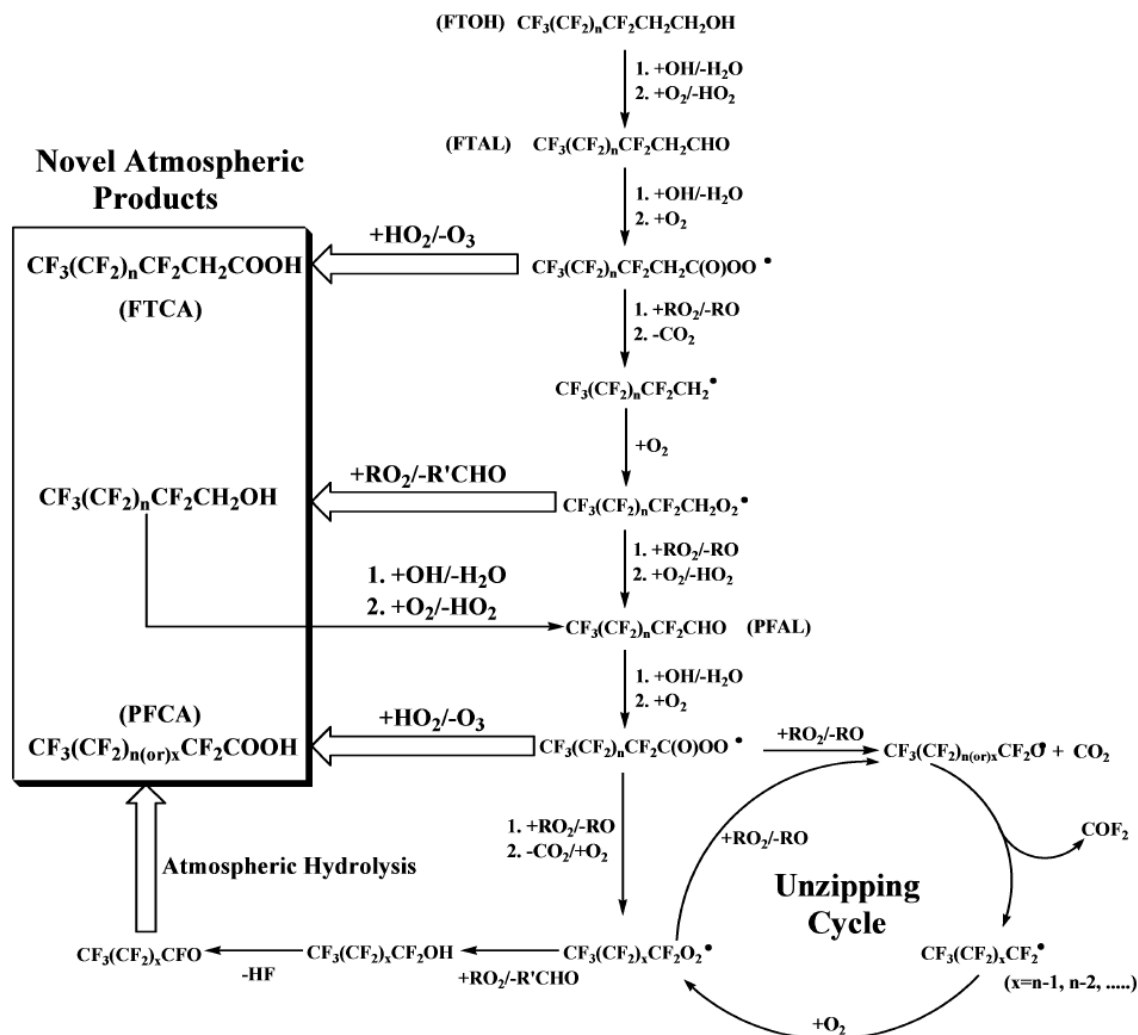


Figure 15-1. $n:2$ and $n:1$ FTOH OH-initiated oxidation pathways leading to formation of PFCAs. R and R' are nonspecific alkyl species in the smog chamber, and x is an integer less than n, whereby $n = 2, 4$, or 6 . Pathway obtained from Ellis et al., 2004.

15.1.2 Degradation in water

Gauthier & Mabury (2005) examined the indirect photolysis of 8:2 FTOH and identified the primary degradation products as 8:2 fluorotelomer aldehyde (8:2 FTAL), 8:2 fluorotelomer acid (8:2 FTCA), and perfluorooctanoate (PFOA). The minor products detected included 8:2 fluorotelomer unsaturated acid (8:2 FTUCA) and perfluorononanoate (PFNA). The authors proposed a photolysis mechanism involving the photodegradation of 8:2 FTCA and 8:2 FTUCA, leading to PFOA as the major end product and PFNA as the minor one. No significant degradation of the parent compound via direct photolysis was observed. The half-life of 8:2 FTOH in lake water samples was estimated to be approximately 93.2 hours.

Further, the photolytic degradation was investigated by (Taniyasu et al., 2013). However, several inconsistencies in the experimental results were noticed and therefore the study is not considered to be reliable (see chapter on PFCAs).

15.2 Biotic degradation

The degradation potential and pathways of FTOHs (usually 8:2 FTOH and 6:2 FTOH) have been studied in detail under different environmental conditions, ranging from sediments, soil, activated sludge and defined microbial communities, selected studies are presented in the following.

Berhanu et al. (2023) suggested a general pathway for aerobic and anaerobic degradation that is illustrated in Figure 15-2, although these pathways may vary slightly depending on the environmental conditions.

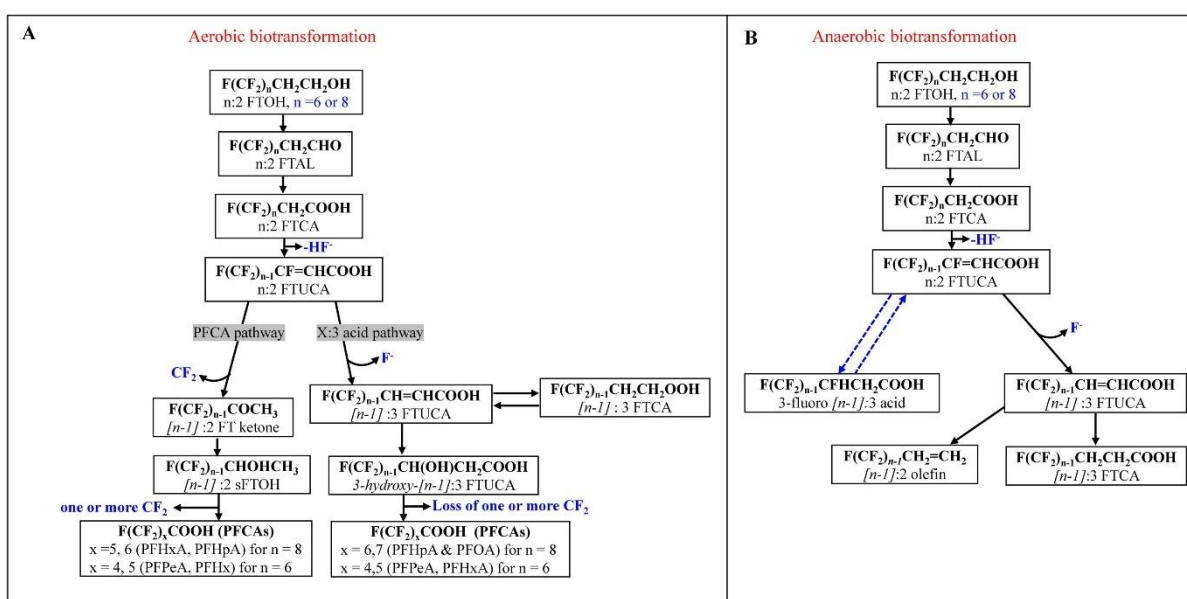


Figure 15-2. General microbial transformation pathways of fluorotelomer alcohols. (A) General aerobic biotransformation pathways for FTOH, (B) anaerobic biotransformation pathways for 6:2 and 8:2 FTOH as presented by (Berhanu et al., 2023).

15.2.1 Degradation in water and sediments

Several studies have investigated biodegradation of FTOHs in water (including pure cell cultures) and sediments using different microbial species (Table 15-1).

Table 15-1. Biotic degradation of FTOHs in water (including pure culture) and sediment. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
8:2 FTOH (0.75 mg/L)	Consortium obtained from sediment and groundwater of a contaminated site	Aerobic, pH 7, 15 days	Not provided	8:2 FTCA, 8:2 FTUCA, PFOA	(Dinglasan et al., 2004)
8:2 FTOH (40 mg/L)	Bacterial culture (<i>Pseudomonas butanovora</i> , <i>Pseudomonas oleovorans</i>)	Aerobic, 30 °C, 28 days	Not provided	8:2 FTUCA, 8:2 FTUCA, 7:2 ketone, 7:2 sFTOH, 7:3 Uacid, 7:3 acid, PFOA, PFHxA	(M. H. Kim et al., 2012)
8:2 FTOH	Landfill leachate and sediment	Aerobic, pH 7.9, 20 °C, 90 days	>365	Transient: FTUCA (1.8%). Terminal: PFOA (2.8%),	(Hamid et al., 2020a)

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
(7.9 mg/L spiked to 232 mg/L)				PFHxA (0.5%), PFHpA (0.5%), 7:3 FTCA (1.5%),	
6:2 FTOH (660 mg/L)	Bacterial culture (Pseudomonas butanovora, Pseudomonas oleovorans)	Aerobic, 30 °C, 2 and 28 days	Not provided	6:2 FTCA, 6:2 FTUCA, 5:2 ketone, 5:2 sFTOH, PFHxA, 5:3 Uacid, 5:3 acid, PFHxA, PFBA	(M. H. Kim et al., 2012)
6:2 FTOH	M. vaccae JOB5, P. fluorescens DSM 8341, P. oleovorans, and P. butanovora	Aerobic, 30 °C, 28 days	Not provided	6:2 FTCA, 6:2 FTUCA, 5:2 Ketone, 5:2 sFTOH, 5:3 acid, 5:3 Uacid, PFHxA, PFPeA, PFBA	(M. H. Kim et al., 2014)
6:2 FTOH (1 mg/L)	River sediment	Aerobic, pH 7, 100 days	1.8	Transient: 6:2 FTCA, 6:2 FTUCA, 5:2 ketone, 5:2 sFTOH. Terminal: PFPeA (10.4%), PFBA (1.5%), PFHxA (8.4%), 5:3 acid (22.4%), 4:3 acid (2.7%)	(L. Zhao et al., 2013)
6:2 FTOH (2 mg/L)	River sediment	Anaerobic, 20 °C, pH 7, 100 days	Not provided	6:2 FTCA (60%), 6:2 FTUCA, 5:3 acid (12%), PFHxA (0.6%)	(S. Zhang et al., 2016)
6:2 FTOH (5 and 15 mg/L)	River sediment	Aerobic, 20-25 °C, 28 days	<3	Transient: 5:2 sFTOH, 6:2 FTUCA, 5:2 ketone, 5:3 Uacid. Terminal: 5:3 acid, PFPeA, PFHxA, PFBA	(S. Zhang et al., 2017)
6:2 FTOH (3 mg/L)	Six different fungal isolates	Aerobic, pH 4.5, 28 days, additional organic nutrients	Not provided	5:3 acid, 6:2 FTUCA, PFHxA, 6:2 FTCA, 5:2 Ketone, 5:2 sFTOH, 5:3 Uacid, PFBA, PFPeA	(Merino et al., 2018)
4:2 FTOH (318 mg/L)	Bacterial culture (Pseudomonas butanovora, Pseudomonas oleovorans)	Aerobic, 30 °C, 2 and 28 days	Not provided	Not determined	(M. H. Kim et al., 2012)

In mixed cultures the degradation of 8:2 FTOH followed oxidation of the 8:2 FTOH to the telomer acid via the transient telomer aldehyde - before β -oxidation was leading to the unsaturated acid and ultimately the terminal product PFOA (Dinglasan et al., 2004). Pure-culture work with *Pseudomonas* strains extended the picture: 4:2, 6:2 and 8:2 FTOHs were defluorinated via two competing routes, one producing x:2 ketones and the parent-chain PFCAs (PFHxA, PFOA), the other yielding x:3 polyfluoro acids and even shorter PFCAs (PFHxA, PFBA); overall defluorination was highest for the 4:2 homologue (M. H. Kim et al., 2012). Additional work investigated the microbial degradation of 8:2 FTOH in landfill leachate and sediment (Hamid et al., 2020a) and brackish water ((Keränen et al., 2013), not shown). In river sediment 6:2 FTOH degraded within days to weeks, (a half-life of <3 d was determined regardless of the initial dose (S. Zhang et al., 2017)), with intermediates such as 6:2 FTCA, 6:2 FTUCA, 5:2 ketone, 5:2 sFTOH and ultimately 5:3 acid, PFPeA, PFHxA and PFBA (S. Zhang et al., 2017; L. Zhao et al., 2013); after 100 d the product distribution was PFPeA (10.4%), PFBA (1.5%), PFHxA (8.4%), 5:3 acid (22.4%), 4:3 acid (2.7%) in one study (L. Zhao et al., 2013). Collectively, these studies indicate that FTOHs biodegrade under aerobic conditions in sediment. Not many studies under anaerobic conditions have been carried out. In one study 6:2 FTCA, 6:2 FTUCA, and 5:3 acid were observed as the major stable products in anaerobic sediment (S. Zhang et al., 2016).

15.2.2 Degradation in soil

FTOHs undergo stepwise biotransformation in soils, ultimately yielding perfluoroalkyl carboxylic acids (PFCAs) (Table 15-2).

Table 15-2. Biotic degradation of FTOHs in soil. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
10:2 FTOH (0.144 nmol/g)	Soil with and without earthworm (<i>Eisenia fetida</i>) and wheat (<i>Triticum aestivum</i> L.)	Aerobic, pH 7.7, 30 days	Not provided	Soil: PFOA (5-10 %), PFNA (4-7 %) and PFDA (60-78%); Earthworm: PFNA, PFDA; Root: PFPeA, PFHxA, PFDA; Shoot: PFUnDA, (PFDA)	(S. Zhao & Zhu, 2017)
8:2 FTOH (100 mg/kg)	Soil microcosm and pure cultures: <i>Pseudomonas</i> OCY4, <i>Pseudomonas</i> OCWA	Aerobic, pH not specified, 7 and 60 days, 30 °C, additional carbon source	Not provided	7:2 sFTOH, 8:2 FTCA, PFOA, minor products (PFPeA, 7:3 FTCA, 7:3 FTUCA)	(J. Liu et al., 2007)
[3-14C] 8-2 FTOH, 8:2 FTOH (1.5 mg/L)	Clay loam soil, sandy loam soil	Aerobic, pH 5.8–7.5, 210 days	Not provided	PFOA (25%), PFHxA, 7:3 FTCA (11%), 7:3 FTUCA, 7:2 sFTOH, 7:2 ketone, 2H-PFOA	(N. Wang et al., 2009)
8:2 FTOH	Sandy loam soil	Aerobic, 101 days	Not provided	8:2 FTCA, 8:2 FTUCA, 7:3 FTUCA, 7:3 FTCA, 7:2 sFTOH, HPFOA, PFOA, PFHpA, PFHxA	(Washington, Jenkins, & Weber, 2015)
8:2 FTOH	Soil (air force base)	Aerobic, 400 days, nitrate reducing conditions	36.5	Transient: 8:2 FTCA, 8:2 FTUA, 7:2 sFTOH, Terminal: C4-C9 PFCAs, PFOA	(Yan et al., 2022)
8:2 FTOH	Soil (air force base)	Aerobic, 400 days, sulfate and iron reducing conditions	Not provided	8:2 FTCA, 8:2 FTUA	(Yan et al., 2022)
8:2 FTOH (0.17 mg/L)	Soil (air force base)	Aerobic, 90 days, bicarbonate buffer	3	Transient: 8:2 FTCA, 8:2 FTUA, 7:2 sFTOH and 7:3 acid, Terminal: PFOA, PFPeA, PFHxA, PFHpA, PFNA, PFDA	(Dong et al., 2023)
8:2 FTOH (215 nmol/L)	Hydroponic plant systems (soybean)	Aerobic, 22-25 °C, 144 hours	Not provided	Soybean: 8:2 FTCA, 8:2 FTUCA, 7:2 sFTOH, 7:3 FTCA (5.57 mol%), PFOA (6.01 mol%), PFHpA, PFHxA	(H. Zhang et al., 2016)
8:2 FTOH (1 mg/kg)	Soil with maize plants (<i>Zea mays</i> L.)	Aerobic, 84 days, pH 5.6	Not provided	PFOA (6.7%), PFHpA (2.6%), PFHxA (1.0%), PFPeA (0.5%), PFBA (0.2%)	(Just et al., 2022)
7:2 sFTOH	Sandy loam soil	Aerobic, 28 days	Not provided	PFOA	(Washington, Jenkins, & Weber, 2015)
6:2 FTOH (1 mg/kg)	Soil with maize plants (<i>Zea mays</i> L.)	Aerobic, 84 days, pH 5.6	Not provided	PFHxA (3.0%), PFPeA (2.6%), PFBA (0.4%)	(Just et al., 2022)
6:2 FTOH (208 nmol/L)	Hydroponic plant systems (soybean)	Aerobic, 144 hours	Not provided	Soybean: 6:2 FTCA, 6:2 FTUCA, 5:3 FTCA, PFHxA, PFPeA, 5:3 FTUCA; Nutrient solution: 6:2 FTCA, 6:2 FTUCA, 5:2 sFTOH, 5:3 FTCA, PFHxA,	(H. Zhang et al., 2020)

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
				PFPeA (after 24 h), PFBA and 4:3 FTCA (after 48 h)	
6:2 FTOH (50 mg/L)	Landfill soil	Aerobic, 32 days	1.5	5:2 sFTOH (12.5 %), 5:3 FTCA (12.5 %), PFHxA (2.0 %), PFPeA (1.6 %), PFBA (1.7 %), TFA (2.3 %)	(M. Sun et al., 2020)
4:2 FTOH (50 mg/L)	Landfill soil	Aerobic, 32 days	Not provided	TFA, (7.8 %), PFPeA (2.6 %), PFBA (17.4 %)	(M. Sun et al., 2020)

Under aerobic conditions, 8:2 FTOH was found to rapidly oxidize to 8:2 FTCA and 8:2 FTUCA; while formation of PFHxA was observed, only trace amounts of unsaturated PFOA (UPFOA) were formed (Washington, Jenkins, & Weber, 2015). The prevailing redox regime strongly modulates these reactions. In strictly nitrate-reducing microcosms, 8:2 FTOH degradation was faster than in sulfate- and iron-reducing conditions, and in addition to 8:2 FTUA, 8:2 FTCA that were found at all redox conditions a broader suite of intermediates including 7:2 sFTOH, and PFOA (Yan et al., 2022). Similarly, in aerobic AFFF-impacted soils with three distinct microbial communities 8:2 FTOH was converted predominantly to 7:2 sFTOH and PFOA during 90 days (Dong et al., 2023). In landfill soil 6:2 FTOH degraded to 5:3 FTCA, PFHxA, PFPeA, PFBA, and ultimately TFA; analogous tests with 4:2 FTOH confirmed successive CF₂ loss down to TFA (M. Sun et al., 2020). In aerobic loam, additional carbon sources inhibited biotransformation of 8:2 FTOH, with inhibition rates increasing as the biodegradability of the co-mediators increased in the order octanol (strongest inhibitor) > ethanol > 1,4-dioxane (weakest inhibitor) (J. Liu et al., 2007).

Plant–soil interactions can stimulate the degradation of FTOHs. In a coupled soil–earthworm–wheat system, 10:2 FTOH was taken up by both biota and transformed various PFCAs; sequestration by organisms appeared to stimulate additional degradation in bulk soil with PFDA as a major degradation product (S. Zhao & Zhu, 2017). Hydroponic soybean studies on 8:2 FTOH metabolism found 7:3 FTCA (roots and stems) and PFOA (leaves) as primary metabolites while up-regulating alcohol dehydrogenase, aldehyde dehydrogenase, and glutathione S-transferase, implicating these enzymes in in-plant metabolism (H. Zhang et al., 2016). Follow-up work with 6:2 FTOH found initial formation of 6:2 FTCA and 6:2 FTUCA, followed by 5:3 FTUCA, 5:3 FTCA, PFHxA, and PFPeA in soybean tissues (H. Zhang et al., 2020). In soil planted long-chain PFCAs largely remained in soil whereas ≤ C7 acids translocated to shoots (Just et al., 2022).

15.2.3 Degradation in WWTPs

About ten studies have investigated biodegradation of FTOHs in WWTP (Table 15-3). Only studies concerned with degradation in “conventional” wastewater treatment have been considered, meaning that studies on wastewater treatment including treatment steps specifically developed for PFAS degradation, have not been included in the review.

Table 15-3. Biotic degradation of FTOHs in WWTPs. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
8:2 FTOH (0.337 mg/L)	Activated sludge	Aerobic, pH 7, 28 days	Not provided	8:2 FTCA (27%), 8:2 FTUCA (6%), PFOA (2.1%), 7:3 acid (2.3%)	(N. Wang, Szostek, Folsom, et al., 2005)

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
14C-8:2 FTOH (1 mg/L)	Activated sludge, mixed bacterial culture	Aerobic, pH 7.21, 90 days	Not provided	7:2 FTOH, 7:3 acid, 8:2 FTAL, 8:2 acid, PFOA: 6%; PFHxA: 0.9%	(N. Wang, Szostek, Buck, et al., 2005)
8:2 FTOH (750 µg 8:2 FTOH per 1 L microcosm, ~50 µg/L aqueous concentration)	Sludge, Ethanol- acclimated mixed culture	Aerobic, pH 7, 16 days	Not provided	8:2 FTAL (aldehyde, transient), 8:2 FTCA (telomer acid) 8:2 FTUCA (dominant metabolite), PFOA (, minor final product)	(Dinglasan-Panlilio, 2008)
8:2 FTOH (~2-4 mg/L)	Sludge	Aerobic and Anaerobic tests; lactate, acetate, pyruvate, ethanol (only for aerobic condition) were provided as extra carbon sources	Not provided	Not conclusively observed; decreases for PFHxA, 6:2 FTOH, 8:2 FTOH could be due to non- biological processes	(Sáez et al., 2008)
[3-14C] 8-2 FTOH (0.182 mg/L)	WWTP digester sludge	Anaerobic, 35 °C, 181 days	181	7:3 Acid; 8:2 FTCA; 8:2 FTUCA	(S. Zhang et al., 2013)
8:2 FTOH (0.5 mg/L)	Activated WWTP sludge	Aerobic, pH 7, 1 day	1	PFBA, PFPeA, PFHxA, PFHpA, transient metabolites (7:2 sFTOH, 8:2 FTUCA)	(X. Yu et al., 2016)
8:2 FTOH (na)	Activated WWTP sludge, mixed culture	Aerobic, 4 days, tests with three FTOH- degrading enrichment cultures, e.g. enriched with n-octane and 1-butanol	Not provided	np	(Lewis, Kim, Wang, et al., 2016)
8:2 FTOH (0.461 mg/L)	WWTP activated sludge (domestic)	Aerobic, 48 h, different autotrophic and heterotrophic conditions, substrates added	Not provided	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA	(X. Yu et al., 2018)
8:2 FTOH (12.1 mg/L)	WWTP activated sludge (industrial)	Aerobic, 20 °C, 192 h, different autotrophic and heterotrophic conditions, substrates added	Not provided	PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA	(X. Yu et al., 2018)
8:2 FTOH (0.147 mg/L)	Activated WWTP sludge	Anaerobic, pH 7.5, 150 days	5	Intermediates: 8:2 FTUA, 8:2 FTUCA, 7:2 sFTOH, 7:3 FTCA. Terminal: PFOA (17%), PFPeA (1.2%), PFBA (1.9%), PFHxA	(F. Li et al., 2018)

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
				(5.4%), PFHpA (8.9%), F ⁻ release	
6:2 FTOH (~2 - 4 mg/L)	Sludge	Aerobic and Anaerobic tests; lactate, acetate, pyruvate, ethanol (only for aerobic condition) were provided as extra carbon sources	Not provided	Not conclusively observed; decreases for PFHxA, 6:2 FTOH could be due to non-biological processes	(Sáez et al., 2008)
6:2 FTOH (1.6 mg/L)	WWTP digester sludge	Anaerobic, 35 °C, 176 days	176	6:2 FTCA; 6:2 FTUCA; 5:3 FTCA	(S. Zhang et al., 2013)
6:2 FTOH (4.125)	Sewage sludge, <i>Pseudomonas</i> <i>fluorescens</i> DSM 8341	Aerobic, pH = 7, 28, 90 and 52 days		PFHxA, PFBA, PFPeA, 5:3 FTCA, 4:3 FTCA	(M. H. Kim et al., 2014)
6:2 FTOH (na)	Activated WWTP sludge, mixed culture	Aerobic, 4 days, tests with three FTOH- degrading enrichment cultures, e.g. enriched with n-octane and 1-butanol	Not provided	Not provided	(Lewis, Kim, Wang, et al., 2016)
6:2 FTOH (12 µL of 4.125 g/L)	Activated WWTP sludge, bioaugmented	Aerobic, 30 days, 30 °C,	Not provided	Not provided	(Lewis, Kim, Wang, et al., 2016)
6:2 FTOH (0.455 mg/L)	WWTP activated sludge (domestic)	48 h, different autotrophic and heterotrophic conditions, substrates added	Not provided	PFBA, PFPeA, PFHxA, PFHpA, PFOA (only traces)	(X. Yu et al., 2018)
6:2 FTOH (100 mg/kg)	Sewage sludge, dominant bacteria were <i>Proteobacteria</i> (56.18%), <i>Firmicutes</i> (15.98%), <i>Chloroflexi</i> (9.72%), and <i>Planctomycetes</i> (5.61%)	Aerobic, 52 days	<1	PFHxA, PFPeA, 5:3 FTCA, 4:3 FTCA	(Qiao et al., 2021)
4:2 FTOH (na)	Activated WWTP sludge, mixed culture	Aerobic, 4 days, tests with three FTOH- degrading enrichment cultures, e.g. enriched with n-octane and 1-butanol	Not provided	Not provided	(Lewis, Kim, Wang, et al., 2016)

Most studies are concerned with aerobic degradation of 8:2 FTOH, while for anaerobic degradation of 8:2 FTOH only three studies have been identified. The database is comparatively rich for 8:2 FTOH and 6:2 FTOH, while only one study has been identified for 4:2 FTOH.

8:2 FTOH and 6:2 FTOH undergo rapid biodegradation in activated sludge under aerobic conditions with biodegradation efficiencies close to 100% within 1 -16 days (Dinglasan-Panlilio, 2008; Lewis, Kim, Wang, et al., 2016; X. Yu et al., 2018). Under anaerobic conditions, degradation of 8:2 FTOH appears to be somewhat slower, still ranging between 67 and 86 % transformation efficiency within 1-150 days, with half-lives between 0.52 and 5 days (F. Li et al., 2018; X. Yu et al., 2016).

Contrary to the above are the results by Sáez et al. (2008), who observed no significant aerobic or anaerobic biodegradation of 8:2 FTOH by the microbial communities in municipal sewage sludge in a modified OECD 301D closed bottle test. Instead, the observed decreases in FTOH concentrations in their and other studies were attributed to sorption to glassware or septa, or to losses associated with interactions with sludge matrix components and microbial biomass rather than true biodegradation (Sáez et al., 2008).

Degradation products of 8:2 FTOH and 6:2 FTOH comprise n:2 FTAL (transient intermediate), n:2 FTCA, unsaturated telomer acids (n:2 FTUCA), sFTOH (7:2 or 5:2), polyfluorinated acids (5:3 or 7:3 acid). PFCAs form at later stages as final degradation products (Dinglasan-Panlilio, 2008; Lewis et al., 2016; F. Li et al., 2018; N. Wang, Szostek, Buck, et al., 2005; X. Yu et al., 2018, Wang et al. 2012 as cited in ECHA, 2025). Defluorination is found to be minimal under experimental conditions (Dinglasan-Panlilio, 2008; X. Yu et al., 2018).

Availability of co-substrates, such as lactate, glucose, ethanol, yeast extract and/ammonia under diverse experimental conditions can support more efficient generation of PFCAs from FTOHs transformation, especially for short-chain PFCAs generation from 6:2 FTOH (N. Wang, Szostek, Buck, et al., 2005; X. Yu et al., 2018). However, (Lewis, Kim, Wang, et al., 2016) studied different carbon sources' impact on pathway utilization and overall changes in effectiveness. They found that the best overall effectiveness was observed when co-substrate carbon was available at low concentrations and that additional carbon resulted in decreased 6:2 FTOH biotransformation (decrease by 60%). These results indicate that biotransformation of FTOH may be limited under environmental conditions, where diverse carbon sources are available as microbial substrates.

In summary, the available studies document that aerobic and anaerobic pathways exist leading to FTOH biodegradation in WWTP. The extent of biodegradation of FTOH under environmental conditions remains unclear, and findings also imply that FTOHs will be converted mainly to PFCAs during wastewater treatment.

16. Other Fluorotelomers and derivatives

Other fluorotelomers and derivatives consist of a fluorotelomer tail with a terminal perfluoroalkyl segment, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, an ethylene group, and various functional head groups (except OH which are covered under fluorotelomer alcohols) and are based on precursors following the general formula $C_nF_{(2n+1)}(CH_2)_m-R$, where n represents the number of fluorinated carbon atoms, m the number of bridging CH_2 units and R various functional head groups except OH.

Prominent examples include 8:2 Fluorotelomer sulfonate (8:2 FTS), 6:2 Fluorotelomer sulfonamide (6:2 FTSA), and fluorotelomer acrylates used in polymer production.

Summary of review findings:

A broad array of fluorotelomer precursors, including sulfonates (FTSAs), acrylates (FTACs), iodides (FTIs), olefins (FTOs), carboxylic acids (FTCAs), and unsaturated carboxylic acids (FTUCAs), degrade both abiotically and biotically to form PFCAs. The degradation potential and pathway depend on the molecular structure and its functional groups. Volatile species such as FTIs, FTOs, and FTACs degrade in the atmosphere within days to weeks through OH-driven or photolytic pathways that proceed via fluorotelomer or perfluoro-aldehydes and further oxidize to perfluoroacyl fluorides or peroxy radicals that form PFCAs.

Non-volatile fluorotelomer counterparts such as the FTSAs, FTCAs, FTUCAs, and other compounds transform primarily in oxic soils, sediments and activated sludges by initial desulfonation or de-esterification followed by successive oxidation and limited defluorination. Aerobic half-lives can be as short as hours for some compounds like FTUCAs, but many fluorotelomers, like the FTSAs, degrade more slowly or persist in anaerobic zones, to form fluorotelomer intermediates and the resulting PFCAs.

Overall, the literature shows that other fluorotelomers and derivatives will undergo abiotic and biotic degradation to form fluorotelomer intermediates and ultimately yield PFCAs. A wide range of fluorotelomer derivatives exist for which transformation data were not found, in particular information on abiotic degradation in water, degradation under anoxic conditions and in full-scale wastewater treatment, remain scarce.

16.1 Abiotic degradation

16.1.1 Degradation in the atmosphere

Additional fluorotelomers including fluorotelomer olefins (FTOs), fluorotelomer iodides (FTIs), fluorotelomer acrylates (FTAc) and polyfluoroalkylsilanes (i.e. n:2 fluorotelomer silanes) can degrade in the atmosphere to form PFCAs. FTAc and FTIs degrade in the atmosphere to form FTALs as the primary product, similar to FTOHs, whereas FTOs degrade to form the corresponding PFALs. In the presence or absence of NO_x, FTALs degrade to form PFALs, but can also form FTCAs in small yields in the absence of NO_x, both of which will eventually degrade to form PFCAs as the terminal products (Young & Mabury, 2010). Polyfluoroalkylsilanes will also degrade in the atmosphere to form the corresponding PFCAs.

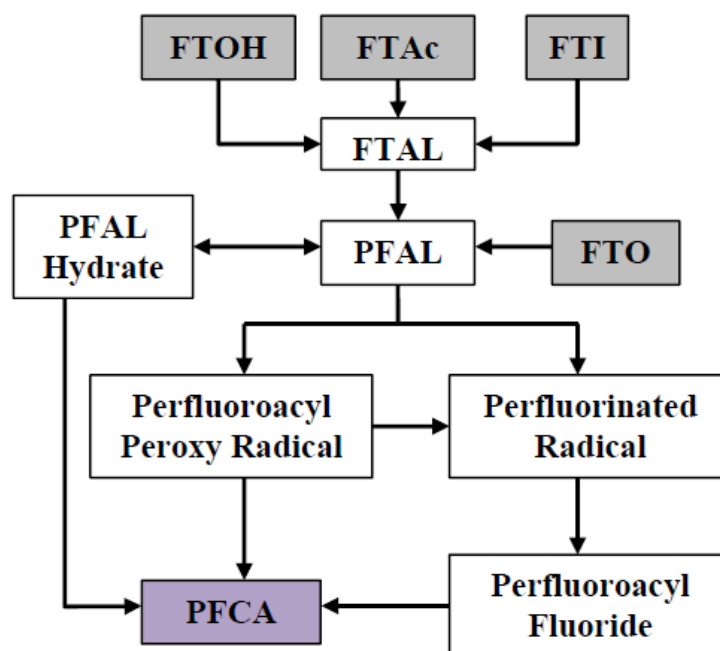


Figure 16-1. Atmospheric fate of fluorotelomer precursors. Reproduced from Young & Mabury, 2010.

To the best of our knowledge, Young et al., 2008 is the only study that has assessed the atmospheric fate of FTIs ($\text{CF}_3(\text{CF}_2)_n\text{CH}_2\text{CH}_2\text{I}$). While the study assessed the reaction rates for the 4:2 FTI ($\text{C}_4\text{F}_9\text{CH}_2\text{CH}_2\text{I}$), rate constants for the longer chain FTIs are expected to be similar. An atmospheric lifetime of 10 days was estimated with respect to reaction with $\cdot\text{OH}$ radicals. However, photolysis was determined to be the dominant atmospheric loss pathway for FTIs as degradation by photolysis was determined to be faster with an estimated lifetime of 1 day up to 1 week, depending on latitude and time of year. In smog chamber experiments conducted with $\cdot\text{Cl}$ radicals and photolysis, the corresponding FTAL was produced as the primary product, followed by the corresponding PFCAs.

Two studies have assessed the atmospheric fate of FTACs ($\text{CF}_3(\text{CF}_2)_n\text{CH}_2\text{CH}_2\text{OC}(\text{O})\text{CH}=\text{CH}_2$), both of which assessed atmospheric degradation of the 4:2 FTAC ($\text{C}_4\text{F}_9\text{CH}_2\text{CH}_2\text{OC}(\text{O})\text{CH}=\text{CH}_2$). Kinetics of reaction for the longer chain FTACs are expected to be similar. An atmospheric lifetime of approximately 1 day for 4:2 FTAC was estimated with respect to reaction with $\cdot\text{OH}$ radicals by both studies (Butt et al., 2009; Y. Sun et al., 2022). Using smog chamber studies Butt et al., 2009 determined reaction rates with Cl atoms and $\cdot\text{OH}$ radicals, which react primarily at the double bond forming a fluorotelomer glyoxylate and formaldehyde. The atmospheric fate of the fluorotelomer glyoxylate is photolysis and reaction with $\cdot\text{OH}$ radicals, leading to formation of the FTAL, which will degrade to form the corresponding PFCAs. Reaction with ozone was not measured but estimated to result in an atmospheric lifetime of 9 days, therefore degradation of FTACs is limited by reaction with $\cdot\text{OH}$ radicals. (Y. Sun et al., 2022) assessed reaction of FTACs with $\cdot\text{OH}$ radicals using density functional theory calculations and came to the same conclusions. The formation of FTALs can occur through a number of pathways initiated by reaction with $\cdot\text{OH}$ radicals or Cl atoms in the presence or absence of NO_x , therefore the yield of FTALs and the corresponding PFCAs will be dependent on local atmospheric conditions.

The atmospheric degradation of FTOs ($\text{CF}_3(\text{CF}_2)_n\text{CH}=\text{CH}_2$) has been assessed in a few studies. Sulbaek Andersen et al., 2005 determined reaction rates with Cl atoms, $\cdot\text{OH}$ radicals, and ozone for various FTO chain lengths, with reaction rates determined to be independent of chain length, except for reaction of $\text{CF}_3\text{CH}=\text{CH}_2$ with ozone which was much faster than for longer chain FTOs. An atmospheric lifetime of approximately 8 days was estimated with respect to reaction with $\cdot\text{OH}$ radicals, contributing to 90% of the FTO loss, and an estimated lifetime of 70 days for reaction with ozone, contributing to 10% of the FTO loss. Nakayama et al., 2007 determined the products of Cl atom and $\cdot\text{OH}$ radical reactions for shorter chain FTOs which react primarily at the double bond. Reaction with Cl atoms results in a small yield of PFALs and various other chlorinated compounds in the absence of NO_x , with additional nitrogen-containing species formed in the presence of NO_x . However, atmospheric degradation of FTOs is dominated by reaction with $\cdot\text{OH}$ radicals which forms PFALs as the primary product in both the presence and absence of NO_x . PFALs then undergo further oxidation to form PFCAs as secondary products. The products of reaction with atmospheric ozone for FTOs have not been measured.

There is minimal information on the atmospheric degradation of polyfluoroalkylsilanes, also referred to as n:2 fluorotelomer silanes. Nielsen, 2014 proposed that the silanes may evaporate and undergo photo-oxidation in the atmosphere due to their appreciable vapour pressures, in which they will photo-oxidize to form the corresponding n:2 FTCAs by oxidation/Si-C cleavage, followed by further degradation into the corresponding PFCAs. (J. Zhu et al., 2019) conducted photodegradation experiments of various n:2 fluorotelomer silanes and determined the formation of FTCAs and FTUCAs as intermediates and the PFCAs of varying chain lengths as terminal products.

16.1.2 Degradation in water

Although no studies on the photolytic degradation of fluorotelomers other than FTOHs were found, the mechanistic consistency suggests that, as with 8:2 FTOH, other fluorotelomers undergo

stepwise phototransformation, with key intermediates such as FTCAs and FTUCAs confirming the route towards PFCAs, including PFOA and PFNA (Gauthier & Mabury, 2005).

Rayne & Forest (2010) estimated the hydrolytic rate constants and half-lives of FTAcS using the SPARC software program. They determined that hydrolytic half-lives for n:2 fluorotelomer acrylate monomers and fluorotelomer methacrylate monomers where n = 2-12, are temperature and solution pH dependent based on assessments over a pH range of 0-14 and at temperatures ranging between 0-100°C. Under environmental conditions (assuming a pH of 8.1 and 15°C), half-lives of 3 and 5 years were estimated for fluorotelomer acrylate and methacrylate monomers, respectively. Rayne & Forest (2010) used the HYDROWIN module of the EPI Suite software program to estimate hydrolytic rate constants and half-lives for FTIs. They determined that hydrolytic half-lives for FTIs are temperature dependent at all pH ranges. Under environmental conditions (assuming a pH of 8.1 and 20°C), a half-life of 126 days was estimated. This estimate was found to decrease from ~8 years at 0°C to ~130 days at 100 °C at a pH of 8.1.

16.2 Biotic degradation

16.2.1 Degradation in water and sediments

Studies that have investigated biodegradation of other fluorotelomers in water (including pure cell cultures) and sediments using different microbial strains are shown in Table 16-1.

Table 16-1. Biotic degradation of other fluorotelomers in water (including pure culture) and sediment. Data was reproduced from(Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
6:2 FTSA	<i>Rhodococcus jostii</i> RHA1	Aerobic, 30 °C, 144 h, S-free medium	Not provided	Transient: 6:2 FTCA, 6:2 FTUSA. Terminal: 6:2 FTUCA, α-OH 5:3 FTCA, PFHpA (minor)	(Yang et al., 2022)
6:2 FTSA	Landfill leachate and sediment	Aerobic, pH = 7.5–7.9, 20 °C, 90 days	Not provided	PFBA (0.6%), PFPeA (5.6%), PFHxA (3.1%), 5:3 FTCA (0.2%), 5:2 sFTOH (0.6%)	(Hamid et al., 2020a)
6:2 FTSA (635-670 mg/L)	Landfill leachate microcosm	Aerobic, pH = 6.5-7.5, 20 °C, 60 days	Not provided	PFBA, PFPeA, PFHxA	(Hamid et al., 2020b)
6:2 FTSA (0.75-0.8 mg/L)	Landfill leachate and sediment	Aerobic, pH = 7.5-7.9, 10 days	Not provided	FHxSA, PFHxS	(Hamid et al., 2020c)
6:2 FTSA (1.3 mg/L)	River sediment	Aerobic, pH = 6.5-6.8, 20 °C, 90 days	Not provided	Nearly complete degradation. TPs - 5:3 acid (16%), PFPeA (21%), PFHxA (20%). Volatile TPs - 5:2 ketone. 5:2 sFTOH, 6:2 FTOH	(S. Zhang et al., 2016)
6:2 FTSA (64.23 mg/L)	<i>Dietzia aurantiaca</i> J3 enrichment	Aerobic, pH = 7, 30 °C, 7 days	Not provided	6:2 FTCA, 6:2 FTUCA, 5:3 FTCA, PFHxA, PFPeA	(Méndez et al., 2022)
6:2 FTSA (25.69 mg/L)	<i>Gordonia</i> sp. strain NB4-1Y	Aerobic, sulfur limiting, 30 °C, 7 days	Not provided	99.9 % degradation. TPs - 6:2 FTCA, 6:2 FTUCA, 5:3 FTCA, 4:3 FTCA, PFHxA, PFPeA, PFBA. Volatile TPs-5:2FT-ketone, 6:2 FTOH, 5:2 sFTOH	(Shaw et al., 2019)
6:2 FTAB (34.22 mg/L)	<i>Gordonia</i> sp. strain NB4-1Y	Aerobic, sulfur limiting, 30 °C, 7 days	Not provided	6:2 FTSA, 6:2 FTCA, 6:2 FTUCA, 5:3 FTCA, 4:3 FTCA, PFHxA, PFPeA, PFBA	(Shaw et al., 2019)
1:3 FTUCA (10.5 mg/L)	KB1 enrichment culture dominated by Dehalococcoides	Anaerobic, 34 °C, 63 days	Not provided	100 % biotransformation, No F- release, TP - 1:3 FTCA	(Y. Yu et al., 2022)

FTMeUPA (C₆H₃F₇O₂) (18 mg/L)	KB1 enrichment culture dominated by <i>Dehalococcoides</i>	Anaerobic (34 °C), 63 days	Not provided	Near complete biotransformation, 4 % defluorination	(Yu et al., 2020, 2022)
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Aerobic studies show that sulfonated fluorotelomers can be readily biodegraded, but its breakdown is governed by the ability of microorganisms to first remove the sulfonate group. Collectively, oxygen supply, sulfur status and co-substrate availability determine how quickly sulfonated fluorotelomers convert.

Mechanistic culture work reinforces the importance of sulfur limitation: In sulfur-rich mediums, alkane monooxygenase and cytochrome P450 were highly expressed, yet defluorination occurred only with 6:2 fluorotelomer alcohol, not 6:2 FTSA. In sulfur-free media, alkanesulfonate monooxygenase facilitated desulfonation, while alkane monooxygenase, haloacid dehalogenase, and cytochrome P450 were linked to defluorination of 6:2 FTSA (Yang et al., 2022). Similarly, under sulfur-limiting conditions *Gordonia sp. NB4-1Y* degraded 99.9 % of 6:2 FTSA and 70 % of its amide analogue 6:2 FTAB within 168 h (Shaw et al., 2019). Major metabolites included 5:2 FT ketone, 5:2 sFTOH, PFPeA, and PFHxA, indicating that both major and minor metabolic routes contribute to fluorotelomer breakdown.

In river-sediment microcosms 6:2 FTSA had a half-life of < 5 days and, within 90 days, yielded a product mixture dominated by 5:3 fluorotelomer carboxylic acid (16 mol %), PFPeA (21 mol %) and PFHxA (20 mol %) (S. Zhang et al., 2016). No transformation occurred under anaerobic conditions within 100 days, indicating that the enzymatic desulfonation step limited 6:2 FTSA biotransformation in anaerobic sediment. (Hamid et al., 2020b) investigated how organic carbon and ammonia concentrations affect 6:2 FTSA biotransformation in leachate sediment, observing higher degradation rates in leachate-added microcosms (~21%) relative to deionized water microcosms (~14%). Furthermore, the presence of ammonia-oxidizing bacteria increased 6:2 FTSA removal rates, likely due to ammonia monooxygenase enzyme activity (Hamid et al., 2020c).

16.2.2 Degradation in soil

The biodegradation of a broad suite of non-alcohol fluorotelomer precursors—ranging from sulfonates and thioethers to acrylates and betaines—have been investigated in soil (Table 16-2), with special emphasis on their stepwise microbial breakdown to delayed PFCA formation.

Table 16-2. Biotic degradation of other fluorotelomers in soil. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
6:2 FTSA (1700 µg/L)	Soil from two different AFFF-impacted sites	Aerobic, 224 days	43.3 and >224	6:2 FTCA, 6:2 FTUA, 5:2 FTOH, 5:3 acid, PFBA, PFPeA, PFHxA, PFHpA, HF	(Yan, Dong, Manz, et al., 2024)
6:2 FTSA (1700 µg/L)	Soil from AFFF-impacted sites	Aerobic, 305 days, dynamic flow conditions	Not provided	really stage TPs: 6:2 FTCA, 6:2 FTUA, Middle stage TPs: 5:2 sFTOH, 5:3 acid, Late-stage TPs: PFBA, PFPeA, PFHxA, PFHpA	(Yan, Dong, Woodcock, et al., 2024)
6:2 FTSA (1.101 nmol/mL)	Hydroponic plant systems (pumpkin)	Aerobic, 22-27 °C, 12 days	Not provided	5:3 FTCA, PFBA, PFPeA, PFHxA, PFHpA, PFPrA, TFA	(S. Zhao et al., 2019)
8:2 FTAc (74.7-111.5 mg/kg)	Soil	Aerobic, pH 5-7, 105 days	≤5	8:2 FTUCA, 8:2 FTCA, 7:3 FTCA, 8:2 FTOH, 7:2 sFTOH, PFHxA (<0.4 mol%), PFHpA, PFOA (up to 8 mol%)	(Royer et al., 2015)

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
8:2 FTCA (0.851 nmol/mL)	Hydroponic plant systems (wheat and pumpkin)	Aerobic, 22-27 °C, 12 days	Not provided	8:2 FTUCA, 7:3 FTCA, PFOA, PFHpA, PFHxA, PFPeA, PFBA, PFPrA, and TFA	(J. Zhao et al., 2023)
6:2 FTAc (1 mg/kg)	Soil with maize plants (<i>Zea mays</i> L.)	Aerobic, pH 5.6, 4 days	Not provided	PFHxA, PFPeA, PFBA	(Just et al., 2022)
8:2 FTMAC (80.7 mg/kg)	Soil	Aerobic, pH 5-7, 105 days	15	8:2 FTUCA, 8:2 FTCA, 7:3 FTCA, 8:2 FTOH, PFHxA (<0.4 mol%), PFHpA, PFOA (9 mol%)	(Royer et al., 2015)
8:2 FTUCA	Sandy loam soil	Aerobic, 28 days	Not provided	7:3 FTUCA, 7:3 FTCA, 7:2 sFTOH, HPFOA, UPFOA, PFOA, PFHpA, PFHxA	(Washington, Jenkins, & Weber, 2015)
6:2 FTAB (10 mg/kg)	Soil impacted by petroleum oil spill-firefighting activities	Aerobic, 60 days, high nutrient	31	Transient: 6:2 FTUA. Terminal: 5:2 ketone	(R. Li et al., 2019)
6:2 FTAB, n:2 FTSAs	Soil after AFFF accident	Aerobic, 2 years, field study	Not provided	Highest levels of PFBA, PFPeA, PFHxA, 6:2 FTSA, 8:2 FTSA, and 6:2 FTAB with an increase of PFCAs after two years	(Mejia-Avendaño et al., 2017)
8:2 Fluorotelomer stearate monoester (250 mg/L)	Soil	Aerobic, pH 5.4, 22 °C, 80-90 days	10.3	8:2 FTUCA, 8:2 FTCA, 7:2 sFTOH, 7:3 acid, PFOA (1.7–4.0%), PFHpA (0.38–0.9%), PFHxA (0.16%)	(Dasu et al., 2012)
n:2 FTAs, n:2 FTSS, n:2 FTABs, n:3 FTBs and n:1:2 FTBs	Soil impacted by petroleum oil spill-firefighting activities	Aerobic, 60 days, different nutrient conditions	Not provided	Concentration decrease: n:2 FTABs, n:3 FTBs and n:1:2 FTBs, 8:2 FTSA, n:2 FTAs; Increase: 6:2 FTSA; lack of detectable increase of PFCAs	(R. Li et al., 2019)
6:2 Fluorotelomer thioether amido sulfonate (6:2 FtTAoS)	Soil (pristine and AFFF-contaminated)	Anaerobic, 30 °C, 300 days, nitrate-reducing conditions	Not provided	thioether propionate product (6:2 FtTP), 6:2 FtSH	(S. Yi et al., 2022)
6:2 FtTAoS	Pristine solids	Anaerobic, 282 days, sulfate-reducing conditions	Not provided	6:2 FtTP (~35 %), 6:2 FtTPIA, 6:2 FtTPIAA, 6:2 FtTPoP, 6:2 FtSiAoS, 6:2 FtSiP	(S. Yi et al., 2018)
FtTAoS (6:2 FtTAoS, 8:2 FtTAoS, 4:2 FtTAoS) (0.03–15.2 mg/L)	Soil microcosms	Aerobic, pH 6.9, 30 °C, 60 days	Not provided	6:2 FTSA (8%), 4:2 FTSA and 8:2 FTSA (<1%), 5:3 FTCA (0.5%), 6:2 FTUCA (0.18%), PFCA (in total 1.5%, mainly PFHxA, PFPeA, and PFBA)	(Harding-Marjanovic et al., 2015)

Degradation rates of 6:2 fluorotelomer sulfonic acid (6:2 FTSA) are much depending on the composition of the soil. Half-lives of 6:2 FTSA in Loring soil was 43.3 days, while >60% of 6:2 FTSA remained after 224 days in Robins soil with cumulative PFCA yields of 13.8 mol % and 1.2 mol % after 224 days, respectively. It was assumed that the reason for the difference in biotransformation rates was attributed to differences in initial sulfate concentration and *Desulfobacterota* appeared to facilitate desulfonation. Flow-through columns confirmed that slower pore-water velocities and intermittent anoxia enhanced desulfonation and shifted the product slate toward C4–C7 PFCAs; attached communities rich in *Rhodococcus* and *Pelotomaculum* drove the reaction (Yan, Dong, Woodcock, et al., 2024). 6:2 FTSA was further metabolized in pumpkin to PFCAs while the concentration in pumpkin roots followed the order: PFHpA > PFBA > PFPrA > TFA > PFHxA ~ PFPeA (S. Zhao et al., 2019).

Microbial mediated degradation of a fluorotelomer acrylate (8:2 FTAc) and metacrylate (8:2 FTMAC) in aerobic soils was investigated over 105 days (Royer et al., 2015). 8:2 FTAc and FTMAC were found to degrade rapidly via 8:2 FTOH to FTCAs and PFCAs with half-lives ≤ 5 d and 15 d, respectively. However, the study had some quality issues (no use of adequate controls and a low mass balance of 50-75%) and therefore the reliability of the study results is considered to be limited. 8:2 FTCA could be bio transformed in wheat and pumpkin to 8:2 FTUCA, 7:3 FTCA and seven terminal different carbon chain length PFCAs (PFOA, PFHpA, PFHXA, PFPeA, PFBA, PFPrA, and TFA) by *Klebsiella sp.* bacteria (J. Zhao et al., 2023). No remaining 6:2 FTCA was detected in soil with maize after 4 days with a low conversion rate of just 1.3% to PFCAs (Just et al., 2022). The degradation of the unsaturated derivative 8:2 FTUCA via β - and α -oxidation was found to generate unsaturated (2uPFCAs) and 2H-substituted PFCAs next to other expected degradation products, indicating that double-bond formation precedes side-chain loss (Washington, Jenkins, & Weber, 2015).

A high number of compounds including the zwitterionic polyfluoroalkyl betaines (n:2 FTABs) were found in soil impacted by petroleum oil spill-firefighting activities even after two years indicating high persistency in this type of soil (R. Li et al., 2019). However, laboratory studies under high nutrient conditions showed that spiked 6:2 FTAB degraded relatively rapid with a half-life of 31 days in the presence of petroleum hydrocarbons. While no PFCAs could be identified suggesting a rather slow process to reach PFCAs from 6:2 FTAB transformation, 6:2 FTUA was observed as an intermediate and increasing amounts of the 5:2 ketone (R. Li et al., 2019). Field evidence from the Lac-Mégantic derailment shows that after two years of weathering the proportion of AFFF-derived FTAB/FTSAs was reduced, while short-chain PFCAs have increased due to biotransformation (Mejia-Avendaño et al., 2017).

The aerobic biodegradation of the 8:2 fluorotelomer stearate monoester was observed to follow the expected pathway via 8:2 FTOH and other transient degradation products to PFCAs. The subsequent degradation products expected from 8:2 FTOH biotransformation were observed including 8:2 FTCA, 8:2 FTUCA, 7:2s FTOH, PFOA, PFHpA, PFHxA, and the 7:3 Acid (Dasu et al., 2012).

In soil microcosms amended with an aqueous film-forming foam (AFFF) solution 4:2, 6:2, and 8:2 fluorotelomer thioether amido sulfonate (FtTAoS) completely degraded in 45 days, however only a limited amount (~ 10 mol%) of degradation products relative to the complete mass balance could be identified including some FtSAs, FtCAs, and PFCAs (Harding-Marjanovic et al., 2015). Under anaerobic sulfate-reducing or nitrate-reducing settings biotransformation of 6:2 FtTAoS is far slower and stops at thioether or sulfinyl derivatives with negligible PFCA release (S. Yi et al., 2018, 2022).

16.2.3 Degradation in WWTPs

Eight studies concerned with biodegradation of other fluorotelomer and FT derivatives in wastewater treatment have been identified (Table 16-3).

Table 16-3. Biotic degradation of other fluorotelomers in WWTPs. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
FTMeUPA* (C6H3F7O2) (~ 75 μ mol)	WWTP activated sludge	Aerobic, 34 °C, 63 days	Not provided	(Nearly complete biotransformation)	(Y. Yu et al., 2022)
FTMePA* (~ 75 μ mol)	WWTP activated sludge	Aerobic, 34 °C, 63 days	Not provided	(No biodegradability)	(Y. Yu et al., 2022)
FTEO (na)	WWTP sludge	Aerobic, 25 °C, 48 days	1	FTEOC	(Frömel & Knepper, 2010)

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
6:2 FTUCA (~75 µmol)	WWTP activated sludge	Aerobic, 34 °C, 63 days	<5	3 fluoro 5:3 acid (82.9%), 5:3 FTCA (17.1%)	(Y. Yu et al., 2022)
1:3 FTUCA (~75 µmol)	WWTP activated sludge	Aerobic, 34 °C, 63 days	Not provided	na	(Y. Yu et al., 2022)
6:2 FTAB, (1.5 mg/L)	WWTP sludge	Aerobic, pH 6.9, 109 days	Not provided	6:2 FTOH, 5:3 FTCA, PFBA, PFPeA, PFHxA	(D'Agostino & Mabury, 2017)
6:2 FTAB 1.2 mg/L	WWTP sludge	Aerobic, 25 °C, 100 days	Not provided	(Slight degradation but no biotransformation products observed)	(Fang et al., 2024)
6:2 FTSA (na)	Wetland slurry	Aerobic, 30, °C, in darkness, pH 7, 142 days	Not provided	5:3 Acid (2.7 mol%), PFPeA (6.1 mol%), PFHxA (2.1 mol%)	(Yin et al., 2019)
6:2 FTSA (na)	Wetland slurry	Anaerobic, 30°C, in darkness, 116 days	Not provided	5:3 Acid (0.7 mol%)	(Yin et al., 2019)
6:2 FTSA 1.2 mg/L	WWTP sludge	Aerobic, 25 °C, 100 days	≈28.8	Seven transformation products: 6:2 FTUSA Ketone-6:2 FTSA OH-6:2 FTSA 5:3 FTU amide 5:3 FT amide-AcA 5:2 FTUCA 4:2 FTUCA	(Fang et al., 2024)
6:2 FTNO 1.2 mg/L	WWTP sludge	Aerobic, 25 °C, 100 days	≈1.2	15 transformation products: Intermediates: 6:2 FTAA (primary intermediate), 6:2 FTUAA, 6:2 FTSeAm, 6:2 FTPrAm, 6:2 FTSAm- PrOH, 6:2 FTSAm-PrA, 6:2 FTSAm- EtOH, 6:2 FTSAm-ForAL, N-Me-6:2 FTSAm-PrA, N-Et-6:2 FTSAm, Final product: 6:2 FTSAm	(Fang et al., 2024)
6:2 FTAA 1.2 mg/L	WWTP sludge	Aerobic, 25 °C, 100 days	≈11.5	6:2 FTUAA, 6:2 FTSeAm, 6:2 FTPrAm, 6:2 FTSAm-PrOH, 6:2 FTSAm-PrA	(Fang et al., 2024)
6:2 FTAA (1.5 mg/L)	WWTP sludge	Aerobic, pH 6.9, 109 days	Not provided	Intermediate product: 6:2 FTOH, 6:2 FTASm; final product: 5:3 FTCA, PFPeA, PFHxA, PFBA	(D'Agostino & Mabury, 2017)
7:3 FTCA (1.4–1.5 mg/L)	WWTP activated sludge	Aerobic, pH ca. 7, 90 days	Not provided („slow“)	only 1.7 mol% conversion to perfluoroheptanoic acid (PFHpA)	(Wang et al., 2012)
7:3 FTCA (~28-33 ng/L in influent)	pilot-scale vertical flow constructed wetlands (VFCW)	32 days	Not provided	na (two PFCAs increased significantly in the effluent this experiment, presumably due to 7:3 FTCA transformation (PFBA and PFNA))	(Lott et al., 2023)
6:3 FTCA (~38-41 ng/L in influent)	pilot-scale vertical flow constructed wetlands (VFCW)	32 days	Not provided	na	(Lott et al., 2023)
5:3 FTCA, (1.4–1.5 mg/L)	WWTP activated sludge	Aerobic, pH ca. 7, 90 days	Not provided	5:3 acid was biotransformed to 4:3 acid (14.2 %), 3:3 acid (0.9 %), PFPeA (5.9 %), PFBA (0.8 %).	(Wang et al., 2012)

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
				Intermediate products: 5:3 Uacid, a-OH 5:3 acid, 5:2 acid, and 5:2 Uacid.	
6:2 FTS (~4 mg/L for mixture 1, ~2 mg/L for mixture 2)	Sludge	Aerobic and anaerobic, pH ca. 7, 2–3.5 months	Not provided	(Not conclusively observed)	(Sáez et al., 2008)
6:2 FTS (na)	Pilot-scale vertical flow constructed wetlands (VFCW)	32 days	Not provided	(na. Concentrations increased in the effluent compared to concentration in influent)	(Lott et al., 2023)
4:2 FTSA (1 mg/L)	WWTP activated sludge	Aerobic, 28 days	Not provided	TFA; possibly other intermediates not fully characterized	(Scheurer et al., 2017)
2:3 FTCA (~75 µmol)	WWTP activated sludge	Aerobic, 34 °C, 63 days	Not provided	na	(Y. Yu et al., 2022)

* See figure below for structure.

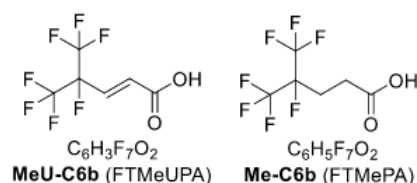


Figure 16-2. Structures of unsaturated (FTMeUPA) and saturated fluorinated carboxylic acids from Y. Yu et al., 2022.

Of the eight studies identified, most studies are concerned with aerobic degradation of fluorotelomers other than FTOH, only two studies investigated for anaerobic degradation.

The fluorotelomers and derivatives 6:2 FTSA (6:2 fluorotelomer sulfonate), 6:2 FTAB (6:2 fluorotelomer sulfonamidoalkyl betaine), 6:2 FTAA (6:2 fluorotelomer acetic acid), 7:3 FTCA (7:3 fluorotelomer saturated carboxylic acid), 6:2 FTS (6:2 fluorotelomer sulfonate) are reasonably well characterised with at least two datapoints available, whereas data is scarce or non-existent for other fluorotelomer saturated and unsaturated carboxylic acids, other fluorotelomer sulfonates, fluorotelomer sulfonamide amine oxides (FTNOs), fluorotelomer ethoxylates (FTEO) or other fluorotelomer derivatives.

Rapid biodegradation in aerobic, experimental systems mimicking different wastewater treatment have been found for FTEO (leading to FTEO carboxylates), 6:2 FTUCA, 6:2 FTNO, 6:2 FTAA with half-lives ranging from 1–11.5 days (Frömel 2010, Yu 2022, Fang et al 2024).

Nearly complete biodegradation within 63 days of incubation time was found for the fluorotelomer unsaturated acid FTMeUPA (Yu et al 2022).

Moderate to slow biotransformation was found for 6:2 FTSA with a half-life of about 28.8 days (Fang et al 2024) on the one hand, <11% transformation within 142 days on the other hand (Yin 2019).

Slow or insignificant biodegradation was observed for 6:2 FTAB (no measurable loss in 100 d; Fang 2024; D'Agostino 2017), the fluorotelomer saturated acid FTMePA (no loss; Yu 2022), 6:2 FTS

(Sáez 2008; Lott et al 2023, Yin 2019); 7:3 FTCA (only 1.7 mol % to PFHpA in 90 d; Wang 2012) and 4:2 FTSA (Scheurer 2017).

Wang et al. 2012 notes the difference in biodegradability potential between 5:3 and 7:3 FTCA, the latter appearing significantly more recalcitrant to biotransformation.

Under anaerobic conditions, the two available studies do not indicate biotransformation of 6:2 FTSA and 6:2 FTS in wastewater treatment systems (Yin 2019, Saez 2008).

Most studies, considering PFCA formation from fluorotelomer and derivatives biodegradation, identified PFCAs, including Wang et al 2012, D'Agostino and Mabury, 2017 and Yin et al 2019, who observed that aerobic biotransformation of FTCA, FTAB and FTSA in WWTP sludge resulted in (among other degradation products) short chain PFCAs. In contrast, Frömel et al. (2010) did not observed PFCA formation attributable to FTEO degradation within the in the experimental timeframe (48 days).

Scheurer et al 2017 suggest biotransformation of several fluorinated pesticides (Fluoxetine, flufenacet, fluopyram) yielding TFA, however, the authors also note that background TFA concentrations may affect measured yields.

In summary, many fluorotelomers other than FTOHs may transform reasonably fast under experimental conditions which appear relevant for environmental conditions as well. However, final degradation products are PFCAs with the potential of accumulating in WWTP effluent and constructed wetland outflows (Lott 2023, Wang et al 2012, D'Agostino and Mabury, 2017 and Yin et al. 2019). Stable species such as 6:2 FTAB or 6:2 FTS may act as long-term PFCA reservoirs. Limited anaerobic degradability implies that sediments, saturated zones and anoxic wetland compartments can function as precursor inventories, thus potentially acting as diffuse sources of short-chain PFCAs in the environment.

17. Perfluoroalkanoyl fluorides (PACFs)

Perfluoroalkanoyl fluorides (PACFs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, and a -COF head group. PACFs follow the general molecular formula $C_nF_{(2n+1)}COF$, where n represents the number of carbon atoms in the perfluoroalkyl chain.

Summary of review findings:

The few data found on perfluoroalkanoyl fluorides were limited to $CF_3C(O)F$ showing that photolytic lifetimes are thousands of years, but atmospheric hydrolytic lifetimes are days to weeks, leading to formation of corresponding PFCAs. No studies were found that have examined longer-chain congeners or assessed abiotic or biotic degradation of in water, sediment, or soil.

17.1 Abiotic degradation

17.1.1 Degradation in the atmosphere

Studies from the pre-2000s have assessed atmospheric photolysis and hydrolysis of $CF_3C(O)F$ in which lifetimes for photolysis are thousands of years, whereas atmospherically relevant hydrolysis occurs on the order of days to weeks (Young & Mabury, 2010). Although no studies were identified that have assessed the atmospheric degradation of longer chain PACFs, it is expected that hydrolysis will be the dominate fate with similar lifetimes. Atmospheric hydrolysis of PACFs results in formation of the corresponding PFCAs (Young & Mabury, 2010).

17.1.2 Degradation in water

No experimental data was found.

17.2 Biotic degradation

No experiment data on biotic degradation of PACFs was found.

18. Perfluoroalkyl iodides (PFAIs)

Perfluoroalkyl iodides (PFAIs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, terminated by an iodine atom. PFAIs follow the general molecular formula $C_nF_{(2n+1)}I$, where n represents the number of carbon atoms in the perfluoroalkyl chain.

Summary of review findings:

A couple studies have proposed that photolysis of PFAIs results in formation of PFCAs through cleavage of the carbon-iodine bond, but no measurement data on reaction rates or transformation pathways was found. Hydrolytic or microbial pathways have not been investigated.

18.1 Abiotic degradation

18.1.1 Degradation in the atmosphere

Nielsen, 2014 and Siegemund et al. (2000) proposed that gas phase photolysis of PFAIs ($CF_3(CF_2)_nI$) will result in formation of PFCAs by reacting preferentially by homolytic cleavage of the C-I bond, resulting in the perfluorinated radical that can then be a source for PFCAs. However, no studies have been identified that measured the reaction rates of PFAI atmospheric degradation.

18.1.2 Degradation in water

No experimental data was found.

18.2 Biotic degradation

No experiment data on biotic degradation of PFAIs was found.

19. Perfluoroalkane sulfonyl fluorides (PASFs)

Perfluoroalkane sulfonyl fluorides (PASFs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, and a $-SO_2F$ head group and follow the general molecular formula $C_nF_{(2n+1)}SO_2F$, where n represents the number of carbon atoms in the perfluoroalkyl chain.

Summary of review findings:

Perfluoroalkane sulfonyl fluorides (PASFs) are synthesized by electrochemical fluorination and are precursors to perfluoroalkane sulfonamides. No laboratory or field data were found that address their abiotic or biotic degradability. One analytical survey demonstrated that PFOSF and

PFHxSF can occur in soils two years after the assumed source, a fluorochemical manufacturing facility, was abandoned, indicating persistence of the PASFs.

In the absence of degradation studies or confirmed transformation products, PASFs degradability in the environment remains uncertain.

19.1 Abiotic degradation

19.1.1 Degradation in the atmosphere

No experimental data was found.

19.1.2 Degradation in water

No experimental data was found.

19.2 Biotic degradation

19.2.1 Degradation in water and sediments

No experimental data was found.

19.2.2 Degradation in soil

A challenge in measuring concentrations of PASFs in environmental samples is a lack of sensitive methods. Bao et al. (2023) discovered a novel chemical derivatization technique for the quantitative analysis of trace PFOSF and PFHxSF in soil collected from an abandoned fluorochemical manufacturing facility, indicating that PFASFs, which ceased production two years ago, persist in the environment and degrade slower than expected.

19.2.3 Degradation in WWTPs

No experimental data was found.

20. Per- and polyfluoroalkylethers

Per- and polyfluoroalkylethers consist of fluorinated alkyl chains, in which all or some of the hydrogen atoms in the alkyl segments are substituted with fluorine atoms, linked by one or more ether oxygen(s). Representatives' molecular formulas for perfluoroalkylethers include $C_nF_{(2n+1)}OC_mF_{(2m+1)}$, where n and m represent perfluorinated chain lengths.

Polyfluoroalkylesters have a similar molecular structure but contain non-fluorinated moieties (e.g., terminal $-CH_3$ groups).

When multiple ether groups are present in a fully fluorinated structure, such substances may formally be referred to as perfluoropolyethers (PFPEs). In this report, a distinction is made between low-molecular-weight PFPE substances, which are discrete molecules with defined molecular weights and structures, and polymeric PFPE materials. The former are considered within the scope of this substance group, whereas polymeric PFPEs are addressed separately under fluoropolymers (see chapter 33).

Summary of review findings:

One of the studies found suggests that perfluoroalkylethers may be resistant to direct photolysis and reaction with $\cdot\text{OH}$ radicals. Another study assessed atmospheric, chlorine-initiated degradation of a methyl perfluoroalkylether ($\text{CF}_3(\text{CF}_2)_n\text{OCH}_3$), with estimated lifetimes on the order of a few years, and formation of carbonyl fluoride and perfluoroalkyl formates that can cleave and form minor yields of PFCAs. No studies were found that document abiotic or biotic degradation of per- or polyfluoroalkylethers in water, sediment, or soil.

20.1 Abiotic degradation

20.1.1 Degradation in the atmosphere

Based on limited information, perfluoroalkylether chains are expected to have similar resistance to photolysis and reaction with $\cdot\text{OH}$ radicals as PFCAs, under environmentally relevant conditions (Z. Wang et al., 2015a). However Nohara et al (2001) assessed the Cl atom-initiated photo-oxidation of methyl perfluoroalkylethers ($\text{CF}_3(\text{CF}_2)_n\text{OCH}_3$) in the absence of NO_x and determined that the perfluoroalkyl group can undergo C-C cleavage to form COF_2 as a product. Photo-oxidation initially occurs via H-abstraction with a Cl atom from the methyl group to produce a perfluoroalkyl radical that is then oxidized to form perfluoroalkyl formates as the sole product for any perfluoroalkyl ether chain length, with estimated lifetimes between 2-4 years with respect to reaction with $\cdot\text{OH}$ radicals. These perfluoroalkyl formates can then undergo C-C cleavage resulting in COF_2 formation (50-75% conversion) or form $\text{CF}_3\text{OOOCF}_3$ from the terminal CF_3 group (20% conversion) or be hydrolyzed to form $\text{CF}_3(\text{CF}_2)_{n-1}\text{COF}$ (12-30% conversion) which will then further hydrolyse to produce the corresponding PFCAs.

20.1.2 Degradation in water

No experimental data was found.

20.2 Biotic degradation

No experimental data was found.

21. Perfluoroalkenes

Perfluoroalkenes consist of fully fluorinated alkene carbon backbones, in which all the hydrogen atoms are substituted with fluorine atoms, and contain at least one C=C double bond.

Summary of review findings:

Perfluoroalkenes can include $\text{CF}_2=\text{CF}_2$ or $\text{R}_f\text{CF}=\text{CF}_2$, $\text{R}_f\text{CF}=\text{CFR}_f'$, and $\text{R}_f\text{C}=\text{CFR}_f'$ where R_f and R_f' are perfluoroalkyl moieties. Based on the found studies, perfluoroalkenes can undergo atmospheric degradation via hydroxyl radicals and chlorine atoms, giving carbonyl fluoride and perfluoroacyl fluorides that can hydrolyse to ultra short-chain PFCAs, such as TFA. Measured tropospheric lifetimes span about a day for tetrafluoroethylene ($\text{CF}_2=\text{CF}_2$) to several weeks for perfluorobutenes (e.g. $(\text{CF}_3)_2\text{C}=\text{CFC}_2\text{F}_5$), whereas bulkier, branched congeners may persist for years. Aside from atmospheric degradation, no studies were found that document abiotic or biotic degradation in water, sediment, or soil.

Consequently, perfluoroalkenes are best viewed as volatile precursors that are efficiently removed from the atmosphere but serve as sources of PFCAs, while their degradation in terrestrial and aquatic compartments remains essentially uncharacterised.

21.1 Abiotic degradation

21.1.1 Degradation in the atmosphere

Perfluoroalkenes containing a $C_nF_{2n+1}CF=$ moiety, such as hexafluoropropene ($CF_3CF=CF_2$) or octafluoro-1-butene ($CF_3CF_2CF=CF_2$) can undergo atmospheric degradation through reactions with $\cdot OH$ radicals and Cl atoms, resulting in formation of perfluoroacyl fluorides ($C_nF_{2n+1}C(O)F$), which subsequently hydrolyse to form PFCAs in the atmosphere, in near quantitative molar yields under typical atmospheric conditions, in both the presence or absence of NO_x (Young & Mabury, 2010). Several studies have examined the rate of reaction of various perfluoroalkenes with a focus on shorter chain short-lived compounds.

Mashino et al. (2000) examined the atmospheric degradation of hexafluoropropene ($CF_3CF=CF_2$) initiated by $\cdot OH$ radicals, Cl atoms, and ozone, using smog chamber techniques. Atmospheric degradation of hexafluoropropene results in formation of COF_2 and $CF_3C(O)F$ in molar yields of 100%, with an atmospheric lifetime of 9 days with respect to reaction with $\cdot OH$ radicals and formation of TFA, HF and CO_2 as final degradation products.

Acerboni et al. (2001) also examined atmospheric degradation of hexafluoropropene ($CF_3CF=CF_2$), as well as perfluorobuta-1,3-diene ($CF_2=CFCF=CF_2$) initiated by reaction with OH radicals, ozone, and nitrate. Hexafluoropropene gave a nearly quantitative yield of $CF_3C(O)F$ and COF_2 as organic products, while perfluorobuta-1,3-diene gave from 130% to 170% of COF_2 . The atmospheric fates of $CF_3C(O)F$ and COF_2 are hydrolysis to form TFA and HF, and HF and CO_2 , respectively. Atmospheric lifetimes were estimated at 6 days for hexafluoropropene and 1.9 days for perfluorobuta-1,3-diene.

Young et al. (2009) examined the atmospheric degradation of perfluorobutenes ($CF_3CF=CFCF_3$ and $CF_3CF_2CF=CF_2$) initiated by $\cdot OH$ radicals and Cl atoms using smog chamber techniques.

Atmospheric lifetimes with respect to reaction with $\cdot OH$ radicals were determined to be 24 days ($CF_3CF=CFCF_3$) and 6 days ($CF_3CF_2CF=CF_2$) respectively. The Cl atom- and $\cdot OH$ radical-initiated oxidation of $CF_3CF=CFCF_3$ in 700 Torr of air gives $CF_3C(O)F$ in molar yields of 196 ± 11 and $218 \pm 20\%$, respectively. Cl atom-initiated oxidation of $CF_3CF_2CF=CF_2$ gives molar yields of $97 \pm 9\%$ $CF_3CF_2C(O)F$ and $97 \pm 9\%$ COF_2 . $\cdot OH$ radical-initiated oxidation of $CF_3CF_2CF=CF_2$ gives molar yields of $110 \pm 15\%$ $CF_3CF_2C(O)F$ and $99 \pm 8\%$ COF_2 . The atmospheric fate of $CF_3CF_2C(O)F$ and $CF_3C(O)F$ is hydrolysis to give $CF_3CF_2C(O)OH$ (PFPrA) and $CF_3C(O)OH$ (TFA), respectively.

Orkin et al. (2011) examined reaction rates with $\cdot OH$ radicals for several perfluoroalkenes including tetrafluoroethylene ($CF_2=CF_2$), hexafluoropropene ($CF_3CF=CF_2$), the perfluorobutene ($CF_3CF=CFCF_3$), and $(CF_3)_2C=CFC_2F_5$. Atmospheric lifetimes of 1.2 days for tetrafluoroethylene, 5.3 days for hexafluoropropene, 21-34 days for the perfluorobutene (depending on isomers), and 182 days for $(CF_3)_2C=CFC_2F_5$ were similar to those observed for the same compounds in other studies. The study concluded that reactions with $\cdot OH$ radicals in the troposphere dictate the atmospheric lifetimes of these perfluoroalkenes. Degradation products were not determined, but based on prior studies, final degradation products would be HF and CO_2 from $CF_2=CF_2$, HF, CO_2 and TFA from $CF_3CF=CF_2$, HF and TFA from $CF_3CF=CFCF_3$ and HF, $CF_3CF_2C(O)OH$ (PFPrA) and TFA from $(CF_3)_2C=CFC_2F_5$.

Qin et al. (2025) examined the rate of reaction of perfluoro(3-isopropyl-4-methyl-2-pentene) ($CF_3CF=C[CF(CF_3)_2]_2$) and perfluoro(2,4-dimethyl-3-ethyl-2-pentene) ($(CF_3)_2C=C(C_2F_5)[CF(CF_3)_2]$) with $\cdot OH$ radicals. Atmospheric lifetimes with respect to reaction with $\cdot OH$ radicals were determined to be 4.7 years for perfluoro(3-isopropyl-4-methyl-2-pentene) and 39 years for perfluoro(2,4-dimethyl-3-ethyl-2-pentene), which are much greater than the lifetimes for the shorter chain perfluoroalkenes examined in other studies. Degradation products were not determined in this study given the long atmospheric lifetimes.

21.1.2 Degradation in water

No experimental data was found.

21.2 Biotic degradation

No experimental data was found.

22. Perfluoroalkanes

Perfluoroalkanes consist of a fully fluorinated alkane chain, in which all the hydrogen atoms are substituted with fluorine atoms and follow the general molecular formula $C_nF_{(2n+2)}$, where n represents the number of carbon atoms.

Summary of review findings:

Perfluoroalkanes such as tetrafluoromethane (CF_4), hexafluoroethane (C_2F_6), and octafluoropropane (C_3F_8) resist degradation by atmospheric oxidants, sunlight, water and microorganisms. Laboratory kinetics combined with atmospheric modelling indicate lifetimes of thousands to tens of thousands of years. Because they are nearly insoluble in water and highly volatile, any environmental release largely partitions to the atmosphere, where it will persist for geological timescales. No studies examining biotic degradation pathways or transformation rates were found.

22.1 Abiotic degradation

22.1.1 Degradation in the atmosphere

Tetrafluoromethane (CF_4), hexafluoroethane (C_2F_6), and octafluoropropane (C_3F_8) have lifetimes on the order of thousands to tens of thousands of years and do not undergo atmospheric degradation (Say et al., 2021).

One study investigated the atmospheric lifetimes of various long-lived halogenated species including CF_4 , C_2F_6 , $c-C_4F_8$, $(CF_3)_2C-C_4F_6$, C_5F_{12} , and C_6F_{14} ((Ravishankara et al., 1993) as cited in (ECHA, 2025)). The researchers evaluated the potential atmospheric loss processes of these gases by measuring their reaction rate coefficients with $O(^1D)$, H, and OH, as well as their absorption cross sections at 121.6 nm in laboratory settings. This data was used in a two-dimensional atmospheric model, indicating that the lifetimes of all studied perfluoroalkane compounds exceeded 2000 years.

22.1.2 Degradation in water

No experimental data was found.

22.2 Biotic degradation

No experimental data was found.

23. Semifluorinated alkanes (SFAs)

Semi-fluorinated alkanes (SFAs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, bonded directly to a hydrocarbon chain and follow the general molecular formula $C_nF_{(2n+1)}C_mH_{(2m+1)}$ ($n, m > 2$),¹² where n represent perfluorocarbon and m represent hydrocarbon segments.

Summary of review findings:

For SFAs ($n, m > 2$), no experimental studies could be found that have tested how the bi-segment architecture affects environmental degradability. For small molecules ($n, m \leq 2$), there is an overlap in the definition of semi-fluorinated alkanes with the group of hydrofluorocarbons (HFCs) that are discussed in the following section. Consequently, the environmental fate of SFAs other than HFCs remain unknown and represent a data gap in current PFAS degradation knowledge.

24. Hydrofluorocarbons/ -ethers / - olefins / -chlorofluorocarbons (HFCs/ HFEs/ HFOs/ HCFCs)

Hydrofluorochlorofluorocarbons (HCFCs), hydrofluorocarbons (HFCs), hydrofluoroethers (HFEs), and hydrofluoroolefins (HFOs) consist of partially fluorinated carbon chains, in which some but not all hydrogen atoms are substituted with fluorine atoms, with inclusion of chlorine atoms for HCFCs, ether linkages in the case of HFEs, and double bonds in the case of HFOs.

Summary of review findings:

HCFCs, HFCs, HFOs, and HFEs have been shown to degrade in the atmosphere, almost exclusively by reaction with hydroxyl radicals and - for the longest-lived members - by stratospheric photolysis with lifetimes ranging from days (HFOs) to several decades (some HFCs). Oxidation of these compounds generally results in formation of intermediates such as trifluoroacetyl fluoride ($CF_3C(O)F$), trifluoroacetaldehyde (CF_3CHO), trifluoromethanol (CF_3OH), and carbonyl fluoride. These intermediates are hydrolysed to yield HF, CO_2 and ultra-short chain PFCAs, most prominently TFA. Reported TFA (or other PFCA) formation from atmospheric degradation results in yields ranging from <10 % to 100 %, depending on molecular structure and atmospheric conditions. Because these substances are highly volatile, abiotic atmospheric oxidation far outweighs any aqueous or biological degradation. No biotic degradation studies could be found. Thus, the studies reviewed here indicate that while many HCFCs, HFCs, HFEs and HFOs do not persist as parent compounds indefinitely, they act as important atmospheric sources of ultra-short chain PFCAs, in particular TFA.

24.1 Abiotic degradation

24.1.1 Degradation in the atmosphere

The atmospheric degradation of HCFCs, HFCs, HFOs, and HFEs have been investigated in various studies, in which the main degradation pathway is $\cdot OH$ radical reaction in the troposphere (Burkholder et al., 2015). These compounds can also be degraded by reaction with Cl atoms and nitrate radicals, but these are not the main degradation pathways. Due to their long atmospheric lifetimes, HCFCs and HFCs can also transport to the stratosphere where they can undergo photolysis

¹² For the purpose of this report, n and m are considered >2 to avoid overlaps with the group of hydrofluorocarbons (HFCs).

or reaction with $O(^1D)$. While the levels of NO_x in the atmosphere can affect the yield and degradation pathways of these compounds, many will generally form intermediate compounds such as trifluoroacetaldehyde (CF_3CHO), trifluoroacetyl fluoride ($CF_3C(O)F$), and/or trifluoromethanol (CF_3OH), depending on the compound. The $CF_3C(O)F$ intermediate is removed almost entirely into water droplets and hydrolyses quickly to form HF and the corresponding short-chain PFCA. The CF_3CHO intermediate undergoes further gas-phase oxidation processes and can either undergo reaction with $\cdot OH$ radicals, hydrolysis or photolysis, with TFA as a potential degradation product of oxidation and hydrolysis. Depending on the compound and reaction pathways, atmospheric degradation typically results in yields ranging from <10% to 100% for TFA (Behringer et al., 2021; Madronich et al., 2023). While not all HCFCs, HFCs, HFOs, and HFEs will oxidize to form PFCAs, the table below reproduced from (Pickard et al., 2020) provides a summary of many of those which can degrade to form short-chain PFCAs, including data on average atmospheric lifetimes and known fractional molar yields of PFCA degradation products as determined by various studies.

Numerous studies have examined atmospheric degradation of hydrochlorofluorocarbons (HCFCs) (Burkholder et al., 2015; Young & Mabury, 2010). Most HCFCs that contain at least one CF_3 group have the potential to degrade in the troposphere to form short-chain PFCAs as degradation products. Some of those HCFCs include HCFC-124, -123, -133a, and -225ca. HCFC-124 (CF_3CH_2Cl) has an estimated tropospheric lifetime of 6-8 years and will react with $\cdot OH$ radicals via a H atom abstraction reaction mechanism in the troposphere to form $CF_3C(O)F$ as the sole product which will subsequently hydrolyse to TFA in ~100% molar yield (Young & Mabury, 2010). HCFC-123 (CF_3CHCl_2) and HCFC-225ca ($CF_3CF_2CHCl_2$) atmospheric degradation is similarly dominated by reaction with $\cdot OH$ radicals in the troposphere, resulting in lifetimes of 1.2-2.1 years (Young & Mabury, 2010). Reaction with Cl atoms has also been examined as well as photolysis through scission of the C-Cl bond and reaction with $O(^1D)$ which can occur in the stratosphere. HCFC-123 degradation results in formation of TFA in >60-100% molar yield whereas degradation of HCFC-225ca results in formation of PFPrA in ~100% molar yield (Young & Mabury, 2010). HCFC-133a (CF_3ClCH_2) similarly degrades in the troposphere via reaction with $\cdot OH$ radicals and by photolysis in the stratosphere and has an estimated atmospheric lifetime of 4.0-4.9 years. Degradation of HCFC-133a results in formation of CF_3CClO via OH reaction and CF_3CHO via photolysis. Hydrolysis of these products can result in formation of TFA with an estimated molar yield of ~50% based on prior studies (McGillen et al., 2015).

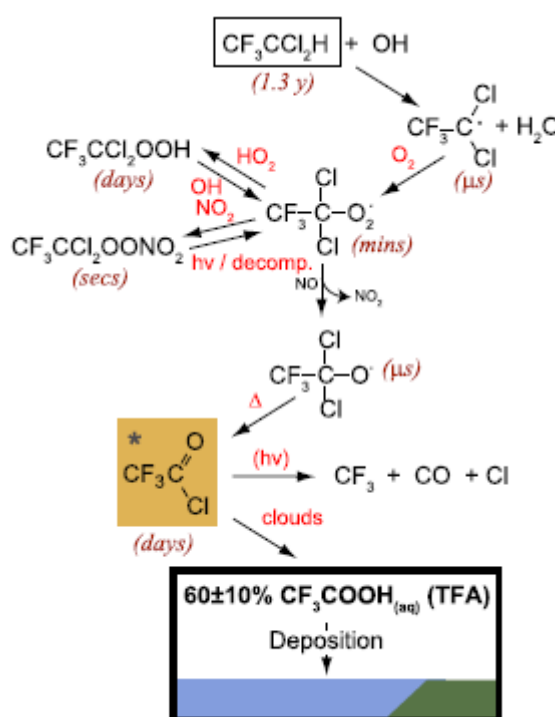


Figure 24-1. Example atmospheric degradation pathways for HCFC-123 (Madronich et al., 2023).

Hydrofluorocarbons (HFCs) are structurally similar to HCFCs but do not contain Cl atoms. Numerous studies have examined atmospheric degradation of HFCs (Burkholder et al., 2015; Young & Mabury, 2010). HFCs that contain at least one CF_3 group and the $-\text{CHF}_2$, $-\text{CH}_2\text{F}$, $-\text{CH}_3$ or $-\text{C}_2\text{H}_5$ moiety at the end of the perfluoroalkyl chain can undergo atmospheric reactions to partially form PFCAs. HFCs with the $-\text{CHF}_2$ moiety including HFC-125 and HFC-329p, which have estimated atmospheric lifetimes ~ 31 -33 years, will react with $\cdot\text{OH}$ radicals in the atmosphere to form a perfluoroalkyl radical. In the presence of NO_x , these compounds will form COF_2 which will then hydrolyse, whereas under low NO_x conditions, oxidation reactions will result in formation of low yields ($<10\%$) of short-chain PFCAs (Young et al., 2009). HFCs with the $-\text{CH}_2\text{F}$ moiety including HFC-134a and HFC-236cb, which have estimated atmospheric lifetimes ~ 13 -14 years, will react with $\cdot\text{OH}$ radicals, followed by reaction with O_2 and NO to yield an alkoxy radical which can either react with O_2 and give $\text{CF}_3\text{C}(\text{O})\text{F}$ or decompose and form $\text{HC}(\text{O})\text{F}$ and CF_3 (Madronich et al., 2023). Reaction with O_2 is favoured at high altitudes and latitudes, which will result in hydrolysis of $\text{CF}_3\text{C}(\text{O})\text{F}$ to form TFA in yields of 7-20% for HFC-134a degradation and $<10\%$ yield of PFPrA for HFC-236cb (Mogelberg et al., 1995; Young & Mabury, 2010). HFCs with the $-\text{CH}_3$ moiety including HFC-143a, which has an atmospheric lifetime of ~ 51 years, will react with $\cdot\text{OH}$ radicals to form CF_3CHO , which is then subject to the three different processes discussed above, some of which may result in low yields ($<10\%$) of TFA (O. J. Nielsen et al., 1994). HFCs with a $-\text{CHF}-\text{R}$ moiety where R is additional C, H, and F groups, including HFC-227ea, HFC-245eb, HFC-236ea, and HFC-43-10-mee, will also undergo atmospheric reactions with $\cdot\text{OH}$ radicals that result in formation of perfluorinated radical that will react with O_2 and NO to yield an alkoxy radical which can either decompose or react with O_2 and give $\text{C}_n\text{F}_{2n+1}\text{C}(\text{O})\text{F}$ that can hydrolyse to form the corresponding PFCAs (Burkholder et al., 2015; Young & Mabury, 2010). HFC-227ea has an estimated lifetime of ~ 39 years and is expected to yield $\sim 100\%$ TFA (Young & Mabury, 2010). Additional HFCs with longer perfluorinated chains may also degrade in the atmosphere to form PFCAs including PFBA and longer chain PFCAs (Z. Wang et al., 2014).

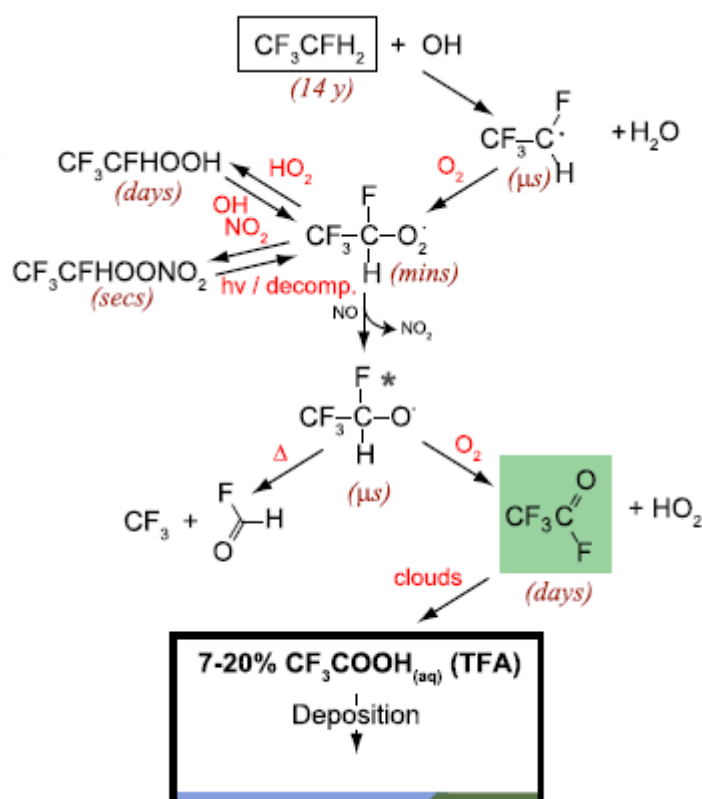


Figure 24-2. Example atmospheric degradation pathways for HFC-134a (Madronich et al., 2023).

Hydrofluoroolefins (HFOs) contain a double bond in their chemical structure which allows them to degrade more rapidly in the atmosphere, reducing their atmospheric lifetimes from years to days. Numerous studies have examined the atmospheric degradation of various HFOs (Hurley et al., 2008, p. 200; Wallington et al., 2015). HFOs rapidly react with $\cdot\text{OH}$ radicals but can also react with Cl atoms and with ozone which is slower. McGillen et al. (2023) studied the ozonolysis of different HFOs and observed that most HFO loss ($\sim 65\%$) due to ozonation happens within the lower 3 km of the atmosphere, and results in formation of TFA, although some TFA could further decompose to CHF_3 through decarboxylation. HFOs that contain the $\text{CF}_3\text{CF}=\text{}$ group largely result in formation of TFA as the primary degradation product. HFO-1234yf has an atmospheric lifetime of ~ 11 days with respect to $\cdot\text{OH}$ radical reaction. Atmospheric degradation is dominated by reaction with $\cdot\text{OH}$ radicals, but HFO-1234yf can also react with Cl atoms, both of which result in formation of the radical that further reacts with O_2 and NO to yield the corresponding $\text{CF}_3\text{C}(\text{O})\text{F}$ which will hydrolyse to TFA in $\sim 100\%$ molar yield (Hurley et al., 2008). Atmospheric degradation of HFO-1225ye(Z) and (E) isomers by reaction with $\cdot\text{OH}$ radicals, Cl atoms, and ozone have been examined and are dominated by $\cdot\text{OH}$ radical reaction with estimated atmospheric lifetimes of 5-9 days (Young & Mabury, 2010). Degradation of these isomers will result in formation of $\text{CF}_3\text{C}(\text{O})\text{F}$ which will hydrolyse to TFA in $\sim 100\%$ molar yield (Tovar et al., 2014). HFOs that contain the $\text{CF}_3\text{CH}=\text{}$ group will primarily form the CF_3CHO as the key intermediate, which has been shown to only yield PFCAs in small amounts relative to other degradation products (Wallington et al., 2015). For example, HFO-1234ze ($\text{CF}_3\text{CH}=\text{CHF}$) predominantly reacts with $\cdot\text{OH}$ radicals to form CF_3CHO within 20 days, which will only result in a TFA yield of $<4\%$, depending on atmospheric conditions (Tewari et al., 2023). Nilsson et al., 2009 examined reaction of HFO-1234ze (E) with $\cdot\text{OH}$ radicals and ozone and determined an atmospheric lifetime of 10 days with respect to reaction with $\cdot\text{OH}$ radicals, which react with the $\text{C}=\text{C}$ double bond to produce CF_3CHO and HCOF in 100% molar yield. Further degradation of those products was not discussed in that study. Additional HFOs have also been estimated to form up to 100% molar yield of TFA depending on their chemical structure and

atmospheric conditions (Behringer et al., 2021). Hydrochlorofluorocarbons (HCFOs) are a class of HFOs that have also been investigated. Sulbaek Andersen et al., 2018 investigated atmospheric degradation of HCFO-1233zd(E) which is initiated by reaction with $\cdot\text{OH}$ radicals and leads to several oxidation products and an estimated atmospheric lifetime of ~ 36 days. CF_3CHO is the key intermediate, resulting in a low TFA yield of 2-30% (Madronich et al., 2023).

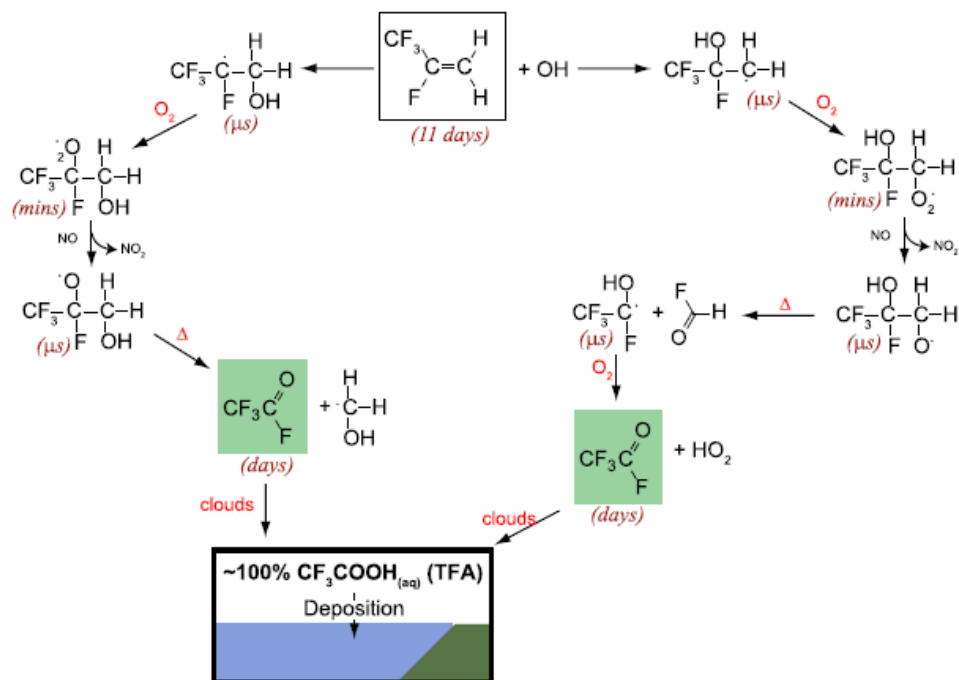


Figure 24-3. Example atmospheric degradation pathways for HFO-1234yf (Madronich et al., 2023).

Hydrofluoroethers (HFEs) are characterized by the inclusion of an ether bond and have atmospheric lifetimes on the order of years (Burkholder et al., 2015; Tsai, 2005). Studies by (Nohara et al., 2001; Tsai, 2005) indicate that HFEs will predominantly undergo $\cdot\text{OH}$ radical initiated hydrogen abstraction, forming intermediates like formate esters ($\text{CF}_3(\text{CF}_2)_n\text{OC(O)H}$), which will cleave to form COF_2 but also hydrolyse to form the corresponding PFCAs. For example, HFE-7100 was shown to undergo $\cdot\text{OH}$ radical oxidation to form the $\text{C}_4\text{F}_9\text{OC(O)H}$ formate ester which degraded to $\text{C}_4\text{F}_9\text{OH}$ which can be readily converted to $\text{C}_3\text{F}_7\text{C(O)F}$ and hydrolyse to form PFBA and potentially shorter-chain PFCAs in low yields (generally $<10\%$) (Nohara et al., 2001; Tsai, 2005).

The table below is adapted from (Pickard et al., 2020) with a summary of HCFCs, HFCs, HFOs, HCFOs, and HFEs that can degrade to short-chain PFCAs, their fractional molar yields from various studies and atmospheric lifetimes from Carpenter & Reimann, 2014, UNEP, 2022 and references therein.

Table 24-1. Chemical formula, atmospheric lifetimes and fractional molar yield of HCFCs, HFCs, HFOs, HCFOs, and HFEs (adapted from (Pickard et al., 2020).

Compound	Chemical Formula	Atmospheric Lifetime (Years)	Fractional Molar Yield			References
			TFA	PFPrA	PFBA	
Halon-2311 (Halothane)	CF_3CHClBr	1.0	~ 1			(Young & Mabury, 2010)
HCFC-123	CF_3CHCl_2	1.3	>0.6			(Madronich et al., 2023; Rayez

Compound	Chemical Formula	Atmospheric Lifetime (Years)	Fractional Molar Yield			References
			TFA	PFPa	PFBA	
						& Lesclaux, 2009)
HCFC-124	CF ₃ CHFCl	5.9	~1			(Rayez & Lesclaux, 2009)
HCFC-133a	CF ₃ ClCH ₂	4.6	0.22-0.55			(Rayez & Lesclaux, 2009) (UNEP, 2022)
HCFC-225ca	CF ₃ CF ₂ CHCl ₂	1.9		~1		(Rayez & Lesclaux, 2009)
HFC-134a	CF ₃ CH ₂ F	14	0.07-0.2			(Madronich et al., 2023; Rayez & Lesclaux, 2009)
HFC-227ea	CF ₃ CHF ₂ CF ₃	38.9	0.94 ± 0.06			(Rayez & Lesclaux, 2009)
HFC-245eb	CF ₃ CHFCH ₂ F	3.2	Unknown			(Burkholder et al., 2015)
HFC-245fa	CF ₃ CH ₂ CHF ₂	7.9	0.01-0.17			(UNEP, 2022)
HFC-236ea	CF ₃ CHFCH ₂ F	11	Unknown			(Burkholder et al., 2015)
HFC-236fa	CF ₃ CH ₂ CF ₃	213	0.2 ± 0.1			(UNEP, 2022)
HFC-43-10mee	CF ₃ CHFCH ₂ CF ₃	17	0.54-0.60	0.54-0.60		(UNEP, 2022)
HFC-125	CF ₃ CHF ₂	31	<0.1			(Young & Mabury, 2010)
HFC-143a	CF ₃ CH ₃	51	0.02-0.30			(Young & Mabury, 2010) (UNEP, 2022)
HFC-329p	CF ₃ CF ₂ CF ₂ CHF ₂	33	<0.1	<0.1	<0.1	(Young & Mabury, 2010)
HFC-365mfc	CF ₃ CH ₂ CF ₂ CH ₃	8.9	0.02-0.30			(UNEP, 2022)
HFC-236cb	CF ₃ CF ₂ CH ₂ F	13		<0.1		(Young & Mabury, 2010)
HFO-1234yf	CF ₃ CF=CH ₂	10.5 days	~1			(Burkholder et al., 2015)
HFO-1234ze (E)	CF ₃ CH=CHF	19 days	0.02-0.30			(UNEP, 2022)
HFO-1225ye (E)	CF ₃ CF=CHF (E)	4.9 days	~1			(Burkholder et al., 2015)
HFO-1225ye (Z)	CF ₃ CF=CHF (Z)	8.5 days	~1			(Burkholder et al., 2015)
HFO-1336mzz (E)	CF ₃ CH=CHCF ₃ (E)	122 days	0.04-0.60			(UNEP, 2022)
HFO-1336mzz (Z)	CF ₃ CH=CHCF ₃ (Z)	27 days	0.04-0.60			(UNEP, 2022)
HCFO-1233zd (E)	CF ₃ CH=CHCl (E)	42 days	0.02-0.30			(UNEP, 2022)
HCFO-1233zd (Z)	CF ₃ CH=CHCl (Z)	13 days	0.02-0.30			(UNEP, 2022)
HFE-236ea2 (Desflurane)	CHF ₂ OCHF ₂ CF ₃	9	0.03-0.20			(UNEP, 2022)
HFE-143a	CF ₃ OCH ₃	4.8	<0.1			(Young & Mabury, 2010)
HFE-245cb2	CF ₃ CF ₂ OCH ₃	5.0	<0.1	<0.1		(Young & Mabury, 2010)
HFE-7100	CF ₃ (CF ₂) ₃ OCH ₃	4.7	<0.1	<0.1	<0.1	(Young & Mabury, 2010)

Compound	Chemical Formula	Atmospheric Lifetime (Years)	Fractional Molar Yield			References
			TFA	PFPaA	PFBA	
HFE-7200	CF ₃ (CF ₂) ₃ OCH ₂ CH ₃	0.80	<0.1	<0.1	<0.1	(Young & Mabury, 2010)
HCFE-235da2 (isoflurane)	CF ₃ CHClOCHF ₂	3.5	0.95 ± 0.03			(Young & Mabury, 2010)
Sevoflurane	(CF ₃) ₂ HCOCH ₂ F	1.4	0.02-0.95			(UNEP, 2022)
FK-5-1-12	CF ₃ CF ₂ C(O)CF(CF ₃) ₂	7 days	~1			(UNEP, 2022)

24.1.2 Degradation in water

No experimental data was found.

24.2 Biotic degradation

No experimental data on biotic degradation of HFCs/ HFEs/ HFOs/ HCFCs could be found. However, within this group typically gaseous compounds are considered and therefore it is expected that biotic degradation does not play a significant role.

25. Perfluoroalkyl and semi-fluorinated ketones

Perfluoroalkyl and semi-fluorinated ketones consist of fluorinated alkyl groups, in which all or some hydrogen atoms are substituted with fluorine atoms, which are bonded to a C=O ketone group. Perfluoroalkyl and semi-fluorinated ketones follow the general molecular formula R_fC(O)R_f', where R_f and R_f' are perfluoroalkyl moieties in the case of perfluoroalkyl ketones or at least one group is a fully or partially fluorinated alkyl moiety in the case of semi-fluorinated ketones.

Summary of review findings:

Three studies have been found that examined the atmospheric degradation of perfluoroalkyl and semi-fluorinated ketones. Photolysis, hydrolysis, and hydration of perfluoroalkyl ketones, such as perfluoro-2-methyl-3-pentanone, shows photolysis to be the dominant degradation pathway, generating carbonyl fluoride and trifluoroacetyl fluoride within days, followed by hydrolysis to form corresponding short-chain PFCAs, such as TFA. In water, the CF-5K fluoroketone is reported to hydrolyse rapidly to TFA, indicating that abiotic processes, rather than microbial activity, determine environmental fate. However, no studies on biotic degradation were found.

25.1 Abiotic degradation

25.1.1 Degradation in the atmosphere

A few studies have examined the atmospheric degradation of perfluoroalkyl ketones, in which photolysis is the predominant reaction pathway, resulting in formation of COF₂ and CF₃C(O)F which can hydrolyse to form TFA, as well as small yields of other ultra short-chain PFCAs, depending on the perfluorinated carbon chain length of the perfluoroalkyl ketone.

Taniguchi et al. (2003) examined the atmospheric degradation of perfluoro-2-methyl-3-pentanone (C₂F₅C(O)CF(CF₃)₂), in which photolysis was the dominant loss mechanism, with an estimated atmospheric lifetime of 7-14 days. The authors did not observe any discernible reactions with ·OH radicals, Cl atoms, or O₃. Photolysis of C₂F₅C(O)CF(CF₃)₂ yielded COF₂ and CF₃C(O)F. CF₃C(O)F was assumed to be formed in 100% molar yield in the absence of NO_x and further undergo hydrolysis to give TFA.

Jackson et al. (2011) also examined the photolysis, hydrolysis, and hydration of perfluoro-2-methyl-3-pentanone. Under normal atmospheric conditions, photolysis dominates hydrolysis as the atmospheric loss pathway, with estimated atmospheric lifetimes of 4-14 days. Under low NO_x conditions, photolysis was observed to give PFPrA in a small yield from the formation of perfluoroalkyl radicals, such as C₂F₅C(O) radicals that can react with excess HO₂ in air. Formation of PFBA was also proposed from this reaction but not detected. However, the predominant reaction pathway from photolysis results in formation of the CF₃C(O)F and the corresponding TFA from hydrolysis.

Ren et al. (2019) also examined the atmospheric degradation of perfluoro-2-methyl-3-pentanone as well as perfluoro-3-methyl-2-butanone (CF₃C(O)CF(CF₃)₂). Photolysis was also found to be the dominant reaction pathway in the troposphere for both compounds. Atmospheric lifetimes with respect to photolysis were estimated at 3-11 days for perfluoro-2-methyl-3-pentanone and ~13 days for perfluoro-3-methyl-2-butanone. Similar to the other studies, photolysis of perfluoro-2-methyl-3-pentanone results in formation of COF₂ and CF₃C(O)F, PFPrA, and TFA. There were no observed photolysis products for perfluoro-3-methyl-2-butanone due to its low photolysis rate and analytical detection limits.

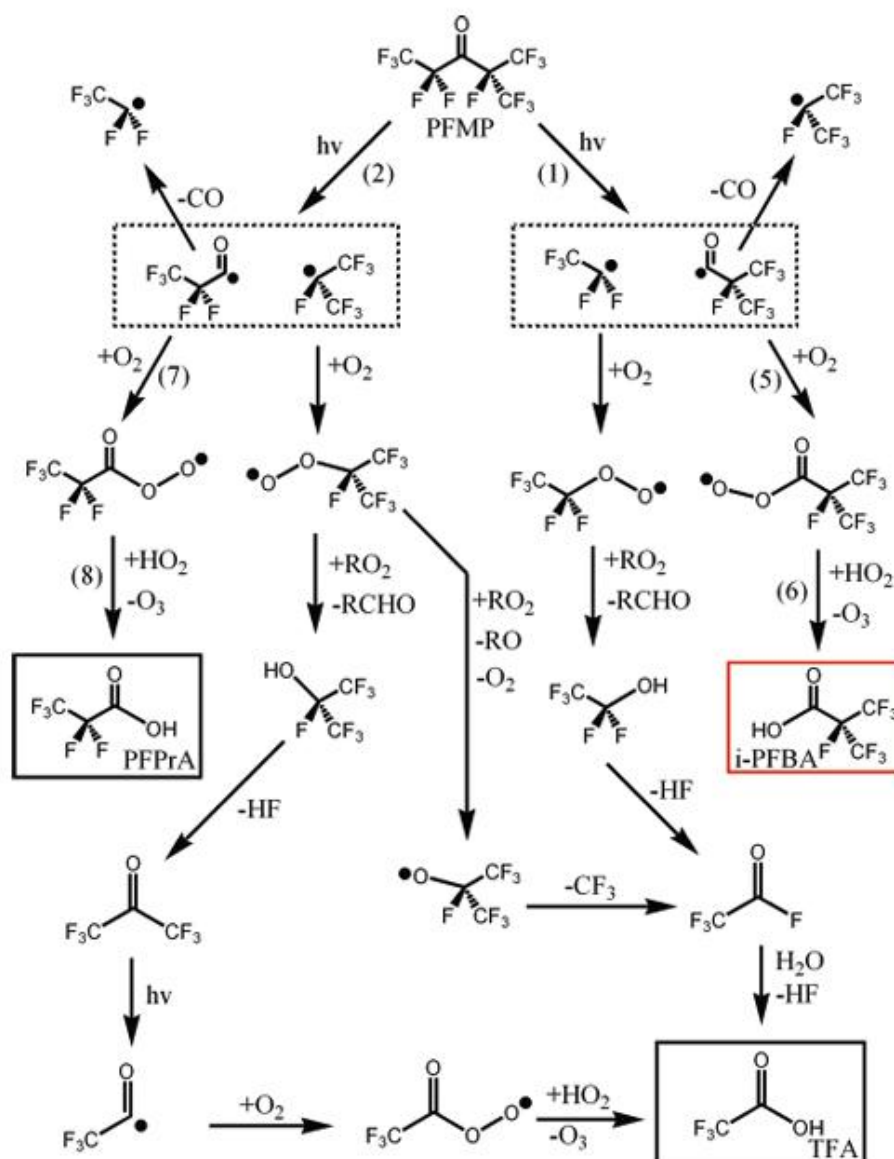


Figure 25-1. Proposed atmospheric degradation pathways from photolysis of perfluoro-2-methyl-3-pentanone (Jackson et al., 2011).

25.1.2 Degradation in water

According to (ECHA, 2025), based on information from the manufacturer 3M, the fluoroketone (CF₃)₂CFC(O)CF₃ (C5-FK) will rapidly hydrolyse upon contact with water, producing TFA as the main hydrolysis product. Consequently, 3M advises against applications of this compound where considerable amounts are intentionally released (ECHA, 2025).

25.2 Biotic degradation

No experiment data on biotic degradation of perfluoroalkyl and semi-fluorinated ketones was found.

26. Per- and polyfluoroalkyl alcohols

Per- and polyfluoroalkyl alcohols consist of fluorinated alkyl tails, in which all or some hydrogen atoms are substituted with fluorine atoms and contain a hydrophilic -OH head group. Perfluoroalkyl alcohols have the general molecular structure $C_nF_{(2n+1)}OH$. Polyfluoroalkyl alcohols include compounds that differ from fluorotelomer alcohols (which have the general molecular structure of $C_nF_{(2n+1)}(CH_2)_2OH$) and instead can have molecular structures that contain substitutions such as ether and sulfide linkages.

Summary of review findings:

Per- and polyfluoroalkyl alcohols may degrade rapidly in the atmosphere. Perfluoroalkyl alcohols, such as trifluoromethanol, are unstable in the atmosphere and decompose spontaneously. Polyfluoroalkyl alcohols, such as FESOH, react within days with photochemical oxidants to yield perfluoroacyl fluorides and ultimately short-chain PFCAs. No data could be found regarding abiotic or biotic degradation in water or soils. One sludge study shows that a long-chain, partially fluorinated alcohol is partially degraded to a mixture of PFCAs and other polyfluorinated acids.

Consequently, per- and polyfluoroalkyl alcohols can be regarded as reactive intermediates whose atmospheric fate derives mainly from their rapid conversion to ultra-short PFCAs. Biodegradation has been documented in a single study with a mixed bacterial sludge culture but information on degradation in other compartments was not identified.

26.1 Abiotic degradation

26.1.1 Degradation in the atmosphere

Perfluoroalkyl alcohols ($C_nF_{2n+1}OH$) are considered unstable in the atmosphere and are typically intermediate degradation products of other PFAS. They are expected to quickly lose HF in the atmosphere to yield the perfluoroacyl fluorides ($C_{n-1}F_{2n-1}C(O)F$), which then readily hydrolyse to form the corresponding PFCAs (Young & Donaldson, 2007). However, only the atmospheric fate for the smallest perfluoroalkyl alcohol, trifluoromethanol (CF_3OH) has been examined. CF_3OH is a chemically unstable gas and decomposes spontaneously to yield COF_2 and HF which is enhanced in the presence of water (Schneider & Wallington, 1994). Young & Donaldson, 2007 showed that CF_3OH can readily complex with water allowing degradation to occur faster than gas-phase decomposition. It is not subject to $\cdot OH$ radical or ozone attack and the most effective loss process is expected to be heterogeneous loss on cloud particles in the troposphere where hydrolysis will occur to yield CO_2 and HF.

Polyfluoroalkyl alcohols that are not fluorotelomer alcohols are included in this category. One study by (Joudan et al., 2021) examined the atmospheric degradation of a novel polyfluoroalkyl alcohol, 2-(1,1,2-trifluoro-2-heptafluoropropoxy-ethylsulfanyl)-ethanol (FESOH, $C_3F_7OCH_2CF_2SCH_2CH_2OH$), from reaction with $\cdot OH$ radicals and Cl atoms. An atmospheric lifetime of ~2.8 days was estimated with respect to reaction with $\cdot OH$ radicals. Atmospheric oxidation of FESOH resulted in formation of two aldehydes and COF_2 , SO_2 , and $C_3F_7OC(O)F$ as secondary products in the gas phase. Formation of aqueous products resulted in additional acidic products, including formation of PFPrA from the hydrolysis of $C_3F_7OC(O)F$. Proposed mechanisms for FESOH oxidation are provided in Figure 26-1Figure .

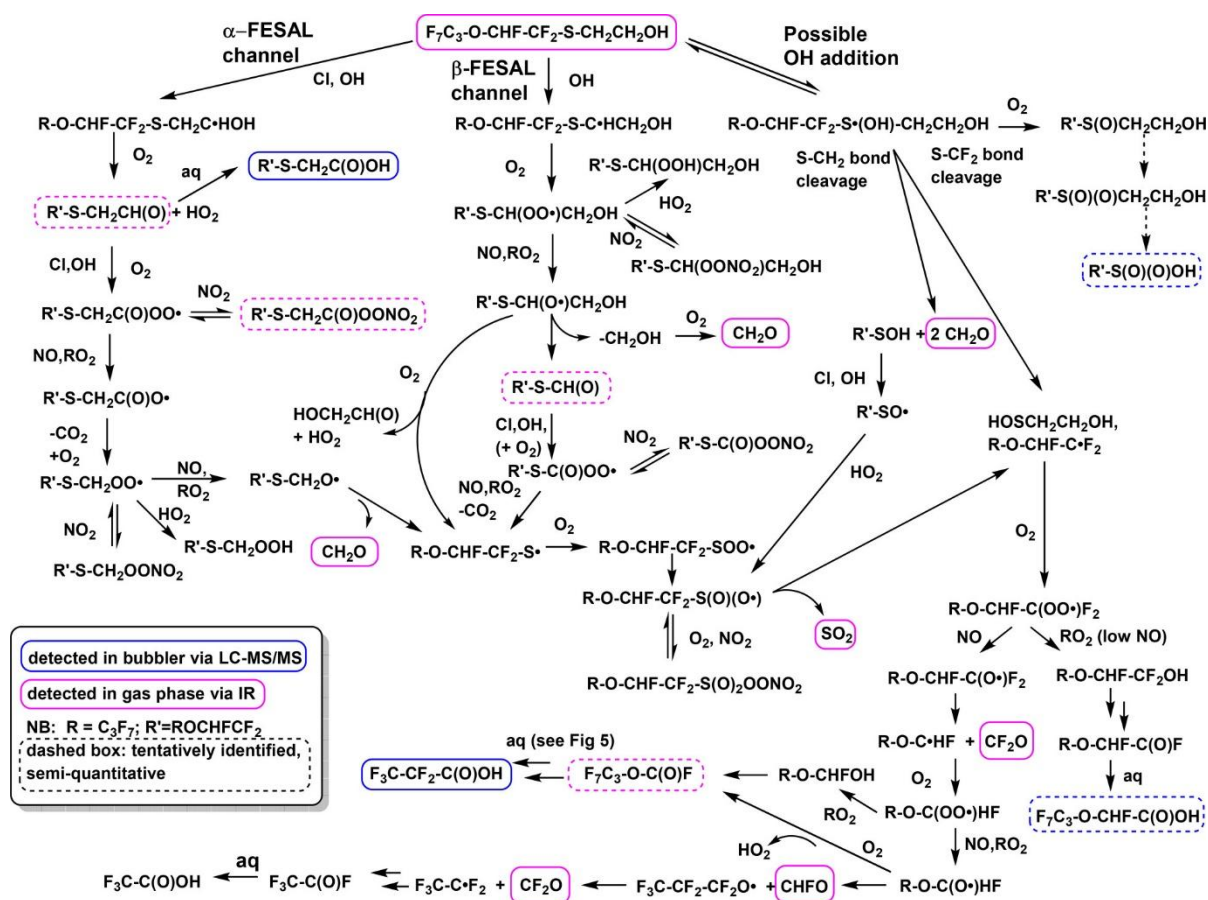


Figure 26-1. Proposed atmospheric degradation mechanism for FESOH with ·OH radicals and Cl atoms (Joudan et al., 2021).

26.1.2 Degradation in water

No experimental data was found.

26.2 Biotic degradation

26.2.1 Degradation in water and sediments

No experimental data was found.

26.2.2 Degradation in soil

No experimental data was found.

26.2.3 Degradation in WWTPs

Arakaki et al. (2010) investigated the biodegradation of 1H,1H,2H,2H,8H,8H-perfluorododecanol, which they named DTFA (degradable telomer fluoroalcohol) in a mixed bacterial culture obtained from activated sludge. The tests were conducted during 70 days at room temperature, with initial concentrations of 100 mg/L under aerobic and closed-bottle conditions.

They found that DTFA partially degraded and noted accumulation of intermediates, suggesting incomplete transformation within 70 days. Half-lives or degradation efficiencies were not mentioned, but PFDoA, 2H,8H,8H-2-PFUDoA, PFBA, PFPeA, PFOA, PFDA, and other short chain mono- and dicarboxylic acids (PFBA, PFBDiA, PFPeDiA, PFPeA, PFHxDiA, PFPrDiA) were identified.

The authors confirm that the mixed bacterial sludge culture is responsible for the observed degradation.

27. Side-chain fluorinated aromatics

Side-chain fluorinated aromatics consist of an aromatic ring with one or more per- or polyfluoroalkyl side chains, in which all or some hydrogen atoms in the side-chain carbon backbone are substituted with fluorine atoms, for example, aromatics with $C_nF_{(2n+1)}$ -side groups, where n represents the number of carbon atoms.

Summary of review findings:

Side-chain fluorinated alcohols can undergo a range of abiotic degradation processes in aqueous environments, such as photolysis and hydrolysis. These mechanisms yield a diverse array of intermediate and final products, notably TFA, fluoride ions, and PFCAs.

Laboratory photolysis of CF_3 -substituted phenols generates fluoride and modest amounts ($\leq \approx 10\%$) of TFA, where degradation is favoured by acidic to neutral pH and the presence of oxidants such as singlet oxygen, as opposed to degradation at alkaline pH.

Biodegradation studies, though scarce, show that specialised microbial consortia or plant–soil systems can strip the CF_3 group and release fluoride, again leaving TFA or other short-chain PFCAs as terminal products. According to (ECHA, 2025), TFA was identified as the predominant degradation metabolite of several side-chain fluorinated aromatics used as pesticide active ingredients. It can be expected that side-chain fluorinated aromatics (C_nF_{2n+1} -Ar) can be degraded and transformed into the corresponding PFCAs ($C_nF_{2n+1}C(O)OH$) including TFA. Overall, existing data indicate that many side-chain fluorinated aromatics are not indefinitely persistent in water or biological matrices, but their partial degradation may result in stable ultra-short PFCAs. No experimental data was found for degradation in the atmosphere.

27.1 Abiotic degradation

27.1.1 Degradation in the atmosphere

No experimental data was found.

27.1.2 Degradation in water

Side-chain fluorinated alcohols can undergo a range of abiotic degradation processes in aqueous environments, such as photolysis and hydrolysis. These mechanisms yield a diverse array of intermediate and final products, notably TFA, fluoride ions, and perfluoroalkyl carboxylic acids (PFCAs).

Studies by (Tisler et al., 2019) demonstrate that **fluoxetine**, a side-chain trifluoromethyl pharmaceutical, degrades in ultrapure water (pH 6, 8, and 10) and surface water (pH 8) under sunlight simulation, forming a complex mixture of products. The metabolite norfluoxetine (NFLX) proved to be a minor transformation product (TP) in photolysis ($\leq 2\%$ of degraded FLX). In addition, 26 TPs were detected, and primarily formed were 3-(methylamino)-1-phenyl-1-propanol (TP 166) and 4-(trifluoromethyl)phenol. TFA formation during direct and indirect photolysis was limited (0.3% and 1.5% respectively after 28 hours). The mineralization of TFA is considered a dead-end, indicating that extended photolysis increases TFA levels as degradation progresses and more intermediates react.

Ellis & Mabury (2000) showed that photolysis of **3-trifluoromethyl-4-nitrophenol (TFM)** in water yielded TFA and fluoride ions, with the half-life and TFA yield varying by pH: 5.1% at pH 9 (22 h

half-life) and 17.8% at pH 7 (91.7 h half-life). Higher pH favours competing defluorination pathways, lowering TFA formation while producing more fluoride ions. The study highlighted that ortho- and para-trifluoromethylphenols follow similar degradation pathways, confirming defluorination as a principal environmental process under abiotic conditions. This type of defluorination degradation pathway (via the deprotonated TFM) has been reported previously for ortho- and para trifluoromethyl phenol (Sakai & Santi, 1973). The authors estimated that photolytic degradation of TFM in natural water systems, such as Lake Ontario, could contribute significantly to ambient TFA concentrations (Ellis and Mabury).

Guo, Tremblay, et al. (2025) investigated the hydrolysis of various **trifluoromethylphenols (TFMPs)** from pH 6 to 10.8, revealing that 2-, 4-, and 2-chloro-4-TFMP undergo significant spontaneous defluorination in water, producing hydroxybenzoic acids and fluoride ions as major products, alongside low but consistent yields of TFA. 3-TFMP, however, remained largely stable, demonstrating the influence of substitution patterns on reactivity. Further work by (Guo, Attar, et al., 2025) explored the photochemical transformation of 4-TFMP, a known fluoxetine metabolite, under UV-B irradiation. TFA yields reached up to 9.2% under acidic conditions, but only 1.3% under alkaline. Fluoxetine's own TFA yield mirrored its 4-TFMP intermediate at acidic/neutral pH but was higher under alkaline conditions, suggesting alternative TFA formation routes. Key reactive oxygen species involved include singlet oxygen and hydroxyl radicals.

These findings collectively show that aqueous degradation of side-chain fluorinated alcohols yields products like TFA alongside transient intermediates, with rates and mechanisms shaped by molecular structure and environmental conditions.

27.2 Biotic degradation

27.2.1 Degradation in water and sediments

Peschka et al. (2008) investigated the biodegradation of **9-[4-(trifluoromethyl)phenoxy] nonane-1-sulfonate** using a standardised fixed-bed bioreactor (FBBR) operating with surface water. After 62 days, the fluoride concentration increased to a value corresponding to 61.8% mineralization of the trifluoromethyl group (without UV decomposition).

In a study by Bygd et al. (2021), 2,2-difluoro-1,3-benzodioxole (DFBD) was shown to be defluorinated by *P.putida* F1 via toluene dioxygenase. These findings may suggest that DFBD and its derivatives will undergo defluorination by dioxygenases found in other known *Pseudomonas* strains and in natural environments. However, several weaknesses in the study were identified, meaning its reliability was considered low.

27.2.2 Degradation in soil

No degradation studies of side-chain fluorinated aromatics were found in this review. However, in the draft updated ECHA background document (Annex B) metabolism studies of flurtamone, saflufenacil, fluazinam, fluometuron, trifloxystrobin, and cyflumetofen in primary crops are discussed based on EFSA assessments (ECHA, 2025). In these studies, TFA was predominantly identified as a metabolite. In cereals/grass (wheat, barley) and oilseeds/pulses (sunflower, peanuts), TFA represented the most abundant metabolite of **flurtamone** (EFSA, 2016 as cited in ECHA, 2025). For **saflufenacil**, TFA was the main metabolite found in maize, soybean, and tomatoes, likely due to soil uptake (EFSA, 2014 as cited in ECHA, 2025). In rotational crop studies with **fluazinam**, TFA was also the only significant compound detected in lettuces, barley grains, and carrots (EFSA et al., 2017 as cited in ECHA, 2025). Similar results were found for **fluometuron**, **trifloxystrobin**, and **cyflumetofen**, where TFA was identified as a significant metabolite (EFSA, Arena, et al., 2017; EFSA et al., 2019; EFSA et al., 2021 as cited in ECHA, 2025).

27.2.3 Degradation in WWTPs

Only few studies investigating the fate of side-chain fluorinated aromatics in wastewater treatment were found.

Scheurer et al. (2017) investigated several CF₃-containing compounds, including five trifluoromethyl substituted aromatics: the pharmaceutical active substances fluoxetine and sitagliptine, and the plant protecting active substances flufenacet, flurtamone and fluopyram, as well as their potential for TFA formation in WWTP.

The substances were inoculated for 28 days with microbial community from WWTP activated sludge according to a modified modified OECD guideline 302B Zahn-Wellens test.

At the end of the test (28 d), the highest removal (67%, after 2 hours) was observed for the fluoxetine, however, the authors anticipated that the removal is not due to biological processes. The same applies for sitagliptine (20 % removal after 2 hours).

All five substances yielded TFA as degradation product, however, the extent of biological transformation was not determined, but most likely are the observed yields due to sorption and other removal processes rather than to biotransformation. The authors also note that background TFA concentrations may affect measured TFA yields.

(Peschka et al., 2008) investigated biodegradation of the fluorosurfactant 9-[4-(trifluoromethyl)phenoxy]nonane-1-sulfonate with unadapted activated sludge from a sewage treatment plant using a standardised method (Zahn-Wellens test, OECD 302B).

They found 22% biodegradation of 9-[4-(trifluoromethyl)phenoxy]nonane-1-sulfonate after 28 days, based on DOC decrease. Additionally, they detected formation of inorganic fluoride and conclude a partial mineralization (28.3%) of the fluorinated methyl group during the test period.

28. Perfluoroalkyl-tert-amines

Perfluoroalkyl-tert-amines consist of three perfluoroalkyl tails, in which all the hydrogen atoms in the carbon backbones are substituted with fluorine atoms, bonded to a tertiary amine nitrogen and follow the general molecular formula (C_nF_(2n+1))₃N, where n represents the number of carbon atoms.

Summary of review findings:

Limited information about environmental degradation of perfluoroalkyl-tert-amines could be found. The only kinetic study on perfluoroalkylamines indicates that even the smallest homologues possess atmospheric lifetimes exceeding three millennia, as neither solar UV nor reactions with excited oxygen atoms provide an efficient sink. No laboratory or field studies could be found that document degradation in water, soils or biological systems. The considerable steric shielding of the tertiary nitrogen together with the fully fluorinated carbon skeleton make biotic cleavage improbable, therefore perfluoroalkyl-tert-amines are expected to behave as persistent, largely volatile chemicals whose ultimate removal occurs only in the upper atmosphere on geological timescales.

28.1 Abiotic degradation

28.1.1 Degradation in the atmosphere

Laboratory experiments were conducted that define the global annually averaged partial and total atmospheric lifetimes of the perfluoroalkylamines N(C₂F₅)₃, N(C₃F₇)₃, and N(C₄F₉)₃ (Bernard et al.,

2020). Reaction with $O(^1D)$ and UV photolysis were evaluated and used in 2-D atmospheric model simulations to determine the global total atmospheric lifetimes of these three compounds which were estimated to be greater than 3,000 years, indicating the stability of these compounds to atmospheric degradation. However, the reliability of this study was assessed to be limited.

28.1.2 Degradation in water

No experimental data was found.

28.2 Biotic degradation

No experimental data was found.

29. Perfluoroalkane sulfonamides (FASAs)

Perfluoroalkane sulfonamides (FASAs) consist of a perfluoroalkyl tail, in which all the hydrogen atoms in the carbon backbone are substituted with fluorine atoms, and a sulfonamide head group. FASAs follow the general molecular formula $C_nF_{(2n+1)}SO_2NR_2$, where n represents the number of carbon atoms and R is H or an alkyl group.

Summary of review findings:

Unlike other PFAS, several studies have shown that FASAs can degrade in the environment, producing fluorinated intermediates that eventually form stable PFAAs (D. Ma & Olivares, 2025). The chemical structure of some subclasses is presented in Figure 29-1. In the atmosphere they exist for several weeks (most atmospheric lifetimes determined in the range of ~20-50 days) before OH-initiated oxidation yields sulfonyl ketones and aldehydes that fragment to ultra-short PFCAs, with little evidence for significant PFSA formation.

In oxic waters, hydroxyl-radical photolysis drives analogous transformation.

In marine sediments, microbial degradation of EtFOSE has been shown with half-lives varying between 44 – 160 days depending on temperature (4 - 25 °C).

In soils and plant-soil systems, FASAs follow a stepwise microbial pathway of N-alkyl oxidation, dealkylation and sulfonamide hydrolysis, ultimately generating PFSA, i.e. PFOS for perfluorinated C8 sulfonamides or PFHxS for C6 analogues, with determined half-lives ranging from ~weeks to years. The findings establish FASAs as long-term, soil-bound sources of perfluoroalkyl sulfonates, with their transformation governed by oxygen supply, pH, sulphur availability, precursor functionality, and biotic activity.

In wastewater treatment systems, oxic and warm (~30 °C) conditions with activated sludge or in constructed wetland systems favour stepwise biotransformation of FASAs with half-lives of about a week or even less. Anoxic or methanogenic conditions impede biotransformation.

Subclass	Chemical structure	Chemical name	Use
<i>The blue 'A' represents alkyl length. For n = 4, 6, 8, A = B (butane), Hx (hexane), O (octane)</i>			
1		where R=H, Perfluoroalkyl sulfonamides (FASA) R=alkyl (methyl, ethyl), N-alkyl FASA (MeFASA, EtFASA)	Major raw material for surfactant and surface protection products
2		where R=H, Perfluoroalkane sulfonamidoethanols (FASE) R=alkyl (methyl, ethyl), N-alkyl FASE (MeFASE, EtFASE)	Major raw material for surfactant and surface protection products
3		Where R=alkyl (methyl, ethyl), N-Alkyl perfluoroalkane sulfonamidoethyl acrylates (FASAC)	Major raw material for surfactant and surface protection products
4		Where R=alkyl (methyl, ethyl), N-Alkyl perfluoroalkane sulfonamidoethyl methacrylates (FASMAC)	Major raw material for surfactant and surface protection products
5		where R=H, Perfluoroalkane sulfonamidoacetic acids (FASAA) Where R=alkyl (methyl, ethyl), N-alkyl FASAA (MeFASAA, EtFASAA)	Intermediate environmental transformation product
6		N-dimethyl ammonio propyl Perfluoroalkane sulfonamide (AmPr-FASA), also referred to as PFASAm	Functioning surfactants in AFFF-formulations
7		N-trimethyl ammonio propyl Perfluoroalkane sulfonamide (TAmPr-FASA), also referred to as PFASAmS	Functioning surfactants in AFFF-formulations
8		Perfluoroalkane sulfonamido amine oxides (PFASNO)	Functioning surfactants in AFFF-formulations
9		Perfluoroalkane sulfonamido betaine (PFASB)	Functioning surfactants in AFFF-formulations

Figure 29-1. Structures of different FASAs from (D. Ma & Olivares, 2025).

29.1 Abiotic degradation

29.1.1 Degradation in the atmosphere

Atmospheric degradation of perfluoroalkane sulfonamides may result in formation of both PFSA and PFCAs. While only a few studies have examined atmospheric degradation of C4-based perfluoroalkane sulfonamides (FASAs) and sulfonamidoethanols (FASEs), reactions are expected to be independent of perfluorinated chain length and result in similar reaction pathways and atmospheric lifetimes for longer chain FASAs and FASEs (Young & Mabury, 2010). To what extent PFCAs and PFSA are formed from degradation of sulfonamides will vary depending on environmental conditions and has not been assessed in detail in available published studies.

Martin et al. (2006) used relative rate methods to measure the gas phase reaction of N-ethyl perfluorobutanesulfonamide (NEtFBSA, C₄F₉SO₂N(H)CH₂CH₃) with ·OH radicals and Cl atoms. Reaction with ·OH radicals is the dominant pathway in the troposphere, resulting in atmospheric lifetimes of 20-50 days for NEtFBSA, as well as for the C8 EtFOSA. (Dreyer et al., 2009) determined similar atmospheric residence times of 27-55 days for NMeFBSA, NMeFOSA, and EtFOSA using measured atmospheric concentrations and the Junge method. Degradation of NEtFBSA results in formation of a ketone (C₄F₉SO₂N(H)COCH₃) and two aldehydes (C₄F₉SO₂N(H)(CH₂)_{n-1}CHO) as primary products. Further degradation results in formation of ultra-short chain PFCAs (PFBA, PFPrA, and TFA) through cleavage of the sulfonyl group forming the C₄F₉ radical which may under sequential CF₂-loss to form the shorter chain-length PFCAs.

D'éon et al. (2006) used relative rate methods to measure the gas phase reaction of N-methyl perfluorobutane sulfonamidoethanol (NMeFBSE, C₄F₉SO₂N(CH₃)CH₂CH₂OH) with ·OH radicals. The

atmospheric lifetime with respect to $\cdot\text{OH}$ radical oxidation is ~ 2 days. (Dreyer et al., 2009) determined much longer atmospheric lifetimes of 42-50 days for NMeFBSE and NMeFOSE using the Junge method. Degradation of NMeFBSE results in formation of the aldehyde ($\text{C}_4\text{F}_9\text{SO}_2\text{N}(\text{CH}_3)\text{CH}_2\text{CHO}$) as a primary product, as well as formation of N-methyl perfluorobutane sulfonamide (NMeFBSA, $\text{C}_4\text{F}_9\text{SO}_2\text{NH}(\text{CH}_3)$), and the ultra-short chain PFCAs (PFBA, PFPrA, and TFA) from $\cdot\text{OH}$ radical attack at the C-S bond. PFBS may also be formed as a product by addition of the $\cdot\text{OH}$ radical to the sulfone double bond resulting in a sulfonyl radical that can cleave the S-N bond to form PFBS as a product. The ratio of PFBS to PFCAs formed from NMeFBSE oxidation is $\sim 1:10$, suggesting higher probability of C-S bond breakage.

Sun et al. (2013) used density functional theory calculations to investigate the atmospheric reaction mechanisms and rate constants for NMeFBSE, NMeFBSA, and EtFOSA with $\cdot\text{OH}$ radicals. The authors determined similar atmospheric lifetimes of 20-40 days for these compounds with respect to $\cdot\text{OH}$ radical reaction. While the addition of $\cdot\text{OH}$ radical to the sulfone double bond is a possible pathway, the authors postulated that the potential barrier is too high for this to occur under normal atmospheric conditions, and therefore formation of PFSAs is unlikely. Rather, $\cdot\text{OH}$ radical-initiated reaction will primarily proceed via direct H-abstraction mechanism, with no effect on chain-length, followed by further reaction with O_2 or NO to produce the corresponding PFCAs.

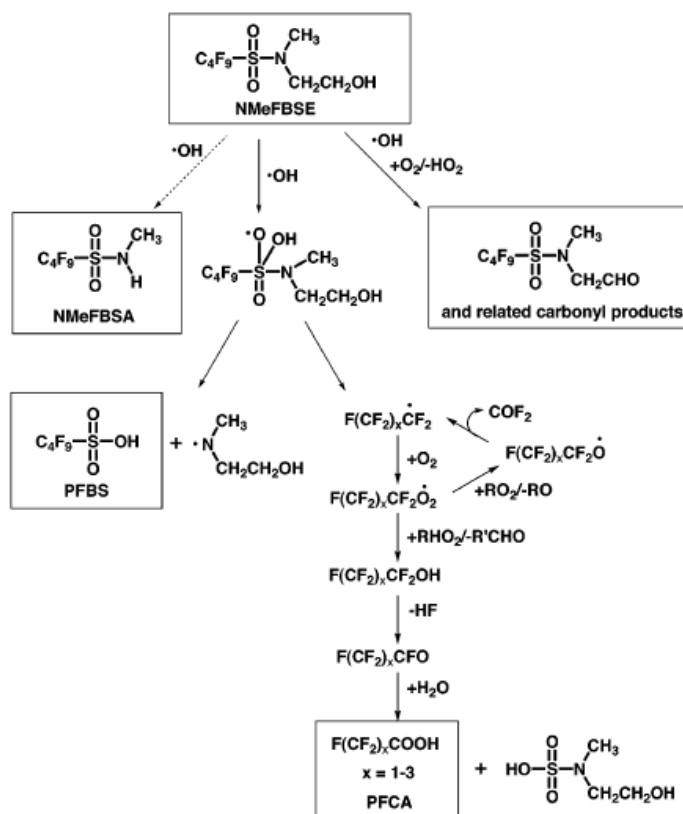


Figure 29-2. Proposed mechanisms of $\cdot\text{OH}$ radical-initiated oxidation of NMeFBSE (D'Eon et al., 2006).

29.1.2 Degradation in water

Plumlee et al. (2009) studied transformation of FASAs under aquatic environmental conditions by irradiation in aqueous hydrogen peroxide solutions using artificial sunlight. They observed indirect photolysis mediated by hydroxyl radical for N-ethyl perfluorooctane sulfonamidoethanol (**EtFOSE**), N-ethyl perfluorooctane sulfonamidoacetate (**EtFOSAA**), N-ethyl perfluorooctane sulfonamide (**EtFOSA**), and perfluorooctane sulfonamide acetate (**FOSAA**). An upper limit for the bimolecular reaction rate constant for reaction of $\cdot\text{OH}$ and EtFOSAA was determined to be $(1.7 \pm 0.7) \times 10^9 \text{ M}^{-1}$.

1 s^{-1} . A proposed reaction pathway for degradation of EtFOSE to the other perfluoroalkane sulfonamides and PFOA includes oxidation and N-dealkylation steps. FOSA and PFOA were the final degradation products of $\cdot\text{OH}$ oxidation.

29.2 Biotic degradation

Several studies have investigated the biotic degradation of FASAs.

29.2.1 Degradation in water and sediments

Benskin et al. (2013) investigated the biodegradation of **EtFOSE** in marine sediments during a 120-day incubation at 4 °C and 25 °C (Table). At both temperatures, EtFOSE was transformed into a several degradation products that included N-ethyl perfluorooctane sulfonamidoacetate (EtFOSAA), perfluorooctane sulfonamidoacetate (FOSAA), N-ethyl-perfluorooctane sulfonamide (EtFOSA), perfluorooctane sulfonamide (FOSA) and PFOS. Transformation proceeded much faster at 25 °C, where the apparent half-life ($t_{1/2}$) was 44 ± 3.4 days, compared with a markedly slower $t_{1/2}$ of 160 ± 17 days at 4 °C. These findings demonstrate that while EtFOSE is susceptible to microbial degradation in marine sediments, the rate of transformation and therefore the persistence of precursor and intermediate compounds, is strongly temperature-dependent.

Table 29-1. Biotic degradation of FASAs in water and sediment. Data was reproduced from (Berhanu et al., 2023).

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
EtFOSE (0.0625 mg/L)	Marine sediments	Aerobic, 25 °C, 120 days	44	EtFOSA aldehyde, EtFOSAA, EtFOSA, FOSE, FOSA aldehyde, FOSA, PFOSi, PFOS	(Benskin et al., 2013)
EtFOSE (0.0625 mg/L)	Marine sediments	Aerobic, 4 °C, 120 days	160	EtFOSA aldehyde, EtFOSAA, EtFOSA, FOSE, FOSA aldehyde, FOSA, PFOSi, PFOS	(Benskin et al., 2013)

29.2.2 Degradation in soil

FASAs show different fates in aerobic soils, with transformation rates and product spectra strongly influenced by molecular structure and soil properties (Table 29-2).

Table 29-2. Biotic degradation of FASAs in soil. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
FOSA (0.5 mg/kg)	Soil plant systems (carrot and lettuce)	Aerobic, 108 days, 20-25 °C	np	PFOS	(Bizkarguenaga et al., 2016)
FOSA (1.07 nmol/g dry weight in the sand)	Earthworm (CYP450, GST enzymes)	Aerobic, 20 days, 22 °C	63 (in sand)	PFOS	(S. Zhao et al., 2020)
FOSA (0.926 mg/L)	Hydroponic plant systems (soybean and pumpkin)	Aerobic, 12 days, 22-25 °C	np	PFOS, PFHxS, PFBS	(S. Zhao, Liang, et al., 2018)
FOSA (1.9 nmol/ml)	Hydroponic plant system (wheat)	Aerobic, 12 days, 22-25 °C, pH 7	np	PFOS, PFHxS, PFBS	(S. Zhao, Zhou, et al., 2018)
FOSA (0.14 nmol/g)	Soil with wheat or earthworms	Aerobic, 30 days, 22-25 °C, pH 7	np	PFOS, PFHxS, PFBS	(S. Zhao, Zhou, et al., 2018)

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (days)	Degradation products	Reference
FOSA (0.14 nmol/g)	Soil with earthworms	Aerobic, 30 days, 22-25 °C, pH 7	np	PFOS	(S. Zhao, Zhou, et al., 2018)
EtFOSA (1.64 mg/L)	Soil	Aerobic, 22 °C, 182 days, pH 7.4	13.9	FOSA, FOSAA, EtFOSAA, PFOS (4%)	(Mejia Avendaño & Liu, 2015)
EtFOSA (0.78 nmol/g)	Quartz sand	Aerobic, 10 days, 22 °C, in the dark	26.7	FOSA (5.6 %), PFOS (7.4 %)	(S. Zhao, Wang, et al., 2018)
EtFOSA (3.8 mg/kg)	Soil plant systems (acidic sandy loam soil and carrots)	Aerobic, 81 days, 18-25 °C, pH 5.7	33.6-35.8	FOSAA, FOSA, PFOS (34%)	(Zabaleta et al., 2018)
EtFOSE (3.71 mg/L)	Soil	Aerobic, 22 °C, 182 days, pH 7.4	5.23	FOSA, FOSAA, EtFOSAA, PFOS	(Mejia Avendaño & Liu, 2015)
EtFOSE (3.7 mg/kg)	Soil (agricultural loam)	Aerobic, 180 days, 22 °C, pH 7.8	31	EtFOSAA, FOSAA, FOSA, PFOS (5.5%)	(L. Zhang et al., 2017)
EtFOSE (3.7 mg/kg)	Soil (acidic forest silt loam)	Aerobic, 210 days, 22 °C, pH 5.5	26	EtFOSAA, FOSAA, FOSA, PFOS (1.0%)	(L. Zhang et al., 2017)
EtFOSE (0.172 nmol/g soil)	Soil with earthworms	Aerobic, 30 days (exposure) and 28 days (depuration), 22 °C	np	EtFOSAA, FOSAA, FOSA, PFOS	(S. Zhao et al., 2016)
EtFOSE	Soil	Aerobic, 105 days, 22 °C, pH 7.4	9.6	n-FOSAA (49%), n-FOSA (0.47%), n-FOSAA (3.6 %), n-FOSA (12%), and n-PFOS (7.9%)	(J. Liu et al., 2019)
Br-EtFOSE	Soil	Aerobic, 105 days, 22 °C, pH 7.4	8.7	br-EtFOSAA, br-PFOS, br-FOSAA, br-FOSA	(J. Liu et al., 2019)
AmPr-FHxSA	Soil (AFFF-impacted)	Aerobic, 70 days, 30 °C	np	FHxSA, PFHxS	(Cook et al., 2022)
AmPr-FOSA (PFOSAm) (2.0 mg/kg)	Soil	Aerobic, 90 days, 22 °C, pH 7.2	47.5	FOSAA, FOSA, PFOS	(M. Liu et al., 2021)
CMeAmPr-FOSA (PFOSB) (1.8 mg/kg)	Soil	Aerobic, 150 days, 22 °C, pH 7.2	675	FOSA, FOSAA, PFOS	(M. Liu et al., 2021)
TAmPr-FOSA (PFOSAmS)	Soil	Aerobic, 6 months, 22°C, pH 5	np	PFOSAm (10%), PFOS (0.3%), FOSA (<0.005%)	(Mejia-Avendaño et al., 2016)
OAmPr-FOSA (PFOSNO) (2 mg/kg)	Sandy loam and loam	Aerobic, 90 days	15	PFOS (2%)	(H. Chen et al., 2020)
AmPr-FASA-PrA, TAmPr-FASA-AA, (CEt-AmPr-FAAd), CEt-AmPr-FASA-PrA, AmPr-FASA-PrA	Soil (AFFF impacted)	Aerobic, 308 days, sodium azide	np	PFCA (C4-C9), PFSA (C2-C10), FASA (C3-C6), M-AmPr-FASA, OAmPr-FASA, N-MeFASAA, N-EtFASAA, OAmPr-FAAd	(Dong et al., 2024)

Sulfonamides such as EtFOSA and EtFOSE typically degrade through stepwise N-alkyl oxidation, dealkylation, and sulfonamide hydrolysis, with half-lives ranging from days to weeks. Mejia Avendaño & Liu, 2015 investigated the aerobic biodegradation of EtFOSE and EtFOSA in soil and found rapid degradation of EtFOSA, with a half-life of 13.9 days, resulting in three major metabolites

(EtFOSAA, FOSA, and FOSAA) and continuous PFOS production, ultimately reaching a yield of 4.0 mol% after 182 days. The sulfonamidoacetic acid metabolites in the pathway from EtFOSE to PFOS were found to be the most persistent ($t_{1/2} \geq 3$ months) in two different soils (an acidic forest soil and a high pH loam), potentially limiting short-term PFOS generation in many soils due to the absence of a direct pathway from EtFOSE, yielding up to 5.49 mol% within 180 days in high pH loam soil (L. Zhang et al., 2017).

A similar rate of biotransformation was observed for branched and linear isomers of EtFOSE, with estimated half-lives of 8.7 and 9.6 days, respectively (J. Liu et al., 2019). However, substitution on the nitrogen had a significant influence. FASAs containing quaternary ammonium or betaine groups (PFOSAmS, PFOSB) showed high stability in soils, with DT_{50} ranging from weeks to years, while those with tertiary amine or amine oxide groups degraded comparably faster (M. Liu et al., 2021). Accordingly, the biotransformation of perfluorooctane sulfonamide quaternary ammonium salt (PFOSAmS) was very slow, with only small amounts of PFOS (0.3%) observed after 6 months (Mejia-Avendaño et al., 2016). In contrast, amine-oxide analogues (PFOANO and PFOSNO) degraded rapidly with half-lives of 3–15 days in live soil microcosms (H. Chen et al., 2020), with PFOANO degrading to PFOA at a yield of 15–21 mol%, and PFOSNO producing PFOS at a yield of 2 mol% after 90 days.

Plant-soil systems or earthworms can modulate both rates and product spectra. Zabaleta et al., 2018 investigated the fate of EtFOSA in a soil–carrot (*Daucus carota* ssp *sativus*) microcosm, finding degradation into FOSAA, FOSA, and PFOS (up to 34%) after 81 days. EtFOSAA, FOSAA, FOSA, and PFOS metabolites were detected during the degradation of EtFOSE in soil with earthworms (S. Zhao et al., 2016). Earthworms and plant–soil systems further facilitated FASA transformation; for example, carrots drove complete FOSA to PFOS conversion, whereas lettuce did not (Bizkarguenaga et al., 2016). Transformation of FOSA to PFOS in earthworms was found to be mainly mediated by cytochrome P450 (CYP450) and glutathione S-transferase (GST) activities rather than gut microbes (S. Zhao et al., 2020). The generation of other shorter-chain perfluoroalkane sulfonates (PFASs) like PFHxS and PFBS was observed in wheat (*Triticum aestivum* L.), but not in soil and earthworms (S. Zhao, Zhou, et al., 2018). In another study, FOSA was readily taken up by soybean (*Glycine max* L. Merrill) and pumpkin roots (*Cucurbita maxima* L.), then translocated to shoots and metabolized to PFOS, PFHxS, and PFBS (S. Zhao, Liang, et al., 2018).

Recent work at AFFF-impacted sites shows that multiple classes of ECF-based precursors (e.g., C4–C8 AmPr-FASA) are transformable, confirming their role as ongoing sources of PFAAs (Dong et al., 2024). This study demonstrated the biotransformation of AmPr-FASA-PrA/TAmPr-FASA-AA (and the amine derivative CEt-AmPr-FAAd), while CEt-AmPr-FASA-PrA was found to be persistent under the experimental conditions tested.

These findings collectively establish FASAs as long-term, soil-bound sources of perfluoroalkyl sulfonates, with their transformation governed by oxygen supply, pH, sulfur availability, precursor functionality, and biotic activity.

29.2.3 Degradation in WWTPs

Seven primary articles concerning the degradation of FASAs in wastewater treatment have been reviewed (Table 29-3).

Table 29-3. Biotic degradation of FASAs in WWTPs. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
EtFOSE (0.1 mg/L)	WWTP activated sludge	Aerobic, 30 °C, 10 days	0.71	Mainly EtFOSAA (ca. 50%), FOSA, FOSAA, PFOSI, PFOS	(Rhoads et al., 2008)
EtFOSE (1.01 mg/L)	Municipal digester sludge	Anaerobic, methanogenic, 20-26 °C, 108 days	1860	EtFOSAA (1.7%), PFOSI (0.3%)	(Lange, 2018)
EtFOSE (0.047 µg/mL)	WWTP activated sludge	Aerobic, 28 ± 2 °C, pH 7.4, 35 days	≤2 (low dose)	EtFOSAA (34.9%), M556 (48.9%), FOSA (5%), PFOSI (3.5%), PFOS (7%), EtFOSA (0.1%)	(Lange, 2000, 2018)
EtFOSE (2.38 µg/mL)	WWTP activated sludge	Aerobic, 28 ± 2 °C, pH 7.4, 35 days	2-3 (high dose)	EtFOSAA and M556 (together accounting for 85%), minor degradation products: FOSA, PFOSI, PFOS, EtFOSA	(Lange, 2000)
MeFOSAA (~18-37 ng/L)	pilot-scale vertical flow constructed wetlands (VFCW)	32 days	Not reported	PFOS	(Lott et al., 2023)
EtFOSAA (~12-28 ng/L)	pilot-scale vertical flow constructed wetlands (VFCW)	32 days	Not reported	PFOS	(Lott et al., 2023)
EtFOSAA (0.1 mg/L)	WWTP activated sludge	Aerobic, 30 °C, 10 days	7.5	EtFOSA (1%), FOSAA (5%), FOSA (18%), PFOS (10%)	(Rhoads et al., 2008)
EtFOSA (1 µg/L)	slurry wetland	from Anaerobic, 30 °C, 191 days	Not reported	Biotransformation was not observed	(Yin et al., 2018)
EtFOSA (1 µg/L)	slurry wetland	from Anoxic - nitrate reducing conditions, 30 °C, pH 7, 154 days	Not reported	Biotransformation was not observed	(Yin et al., 2018)
EtFOSA (1 µg/L)	slurry wetland	from Aerobic, 30 °C, 91 days	0.51	FOSAA, FOSA, PFOS	(Yin et al., 2018)
EtFOSA (0.1 mg/L)	WWTP activated sludge	Aerobic, 30 °C, 10 days	0.75	FOSA (55%), FOSAA (2%), PFOSI (2%), PFOS (41%)	(Rhoads et al., 2008)
FOSAA (0.1 mg/L)	WWTP activated sludge	Aerobic, 30 °C, 10 days	1.89+/- 0.79	FOSA (38%), PFOSI (2%), PFOS (21%)	(Rhoads et al., 2008)
FOSA (2000 mg/L)	activated sludge, Acinetobacter sp. Strain M	Aerobic, 30 °C, pH 7, 35 hours	Not reported	PFOS	(Hao et al., 2019)
FOSA (~4 mg/L for mixture 1, ~2 mg/L for mixture 2)	Sludge	Aerobic and anaerobic tests, 2-3.5 months	Not reported	Not conclusively observed	(Sáez et al., 2008)
FOSA (0.1 mg/L)	WWTP activated sludge	Aerobic, 30 °C, 10 days	9.2	PFOSI (2%), PFOS (38%)	(Rhoads et al., 2008)

Compound (initial concentration)	Microbial species origin	or	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
PFHxSAmS (14-95 nM)	slurry		Aerobic, 29 °C, pH 6.0-7.2, 45 days	na ¹	FHxSA, PFHxS	(Ruyle et al., 2023)
PFHxSAmS (14-95 nM)	slurry		Aerobic, 19 °C, pH 6.0-7.2, 60 days	na ¹	FHxSA, PFHxS	(Ruyle et al., 2023)
PFHxSAm (14-95 nM)	slurry		Aerobic, 29 °C, pH 6.0-7.2, 45 days	na ¹	na	(Ruyle et al., 2023)
PFHxSAm (14-95 nM)	slurry		Aerobic, 19 °C, pH 6.0-7.2, 60 days	na ¹	na	(Ruyle et al., 2023)
FHxSA (14-95 nM)	slurry		Aerobic, 29 °C, pH 6.0-7.2, 45 days	na ¹	na	(Ruyle et al., 2023)
FHxSA (14-95 nM)	slurry		Aerobic, 19 °C, pH 6.0-7.2, 60 days	na ¹	na	(Ruyle et al., 2023)
MeFBSE (0.976 mg/L)	Municipal digester sludge		Anaerobic, methanogenic, 20-26 °C, 108 days	35.8	MeFBSAA (57%), PFBSI (40%)	(Lange, 2018)

¹ Ruyle et al., 2023 provides half-lives based on sorption/biosorption, not for degradation

Across the seven studies identified for FASAs in wastewater systems, more tests have been conducted under aerobic conditions than under anaerobic conditions. Microbial species are usually not reported, but tests employed activated sludge, slurries or simulated constructed wetlands for wastewater treatment. Substance-specific coverage is heavily skewed toward the C8 homologues of the FASAs, such as EtFOSE, EtFOSA and their degradation products.

In the aerobic degradation studies, significant degradation of FASAs has been found with half-lives ranging from <1 – 9.2 days, indicating ready biodegradability of EtFOSE (Lange, 2018). Intermediate degradation products are (Et- or Me-)FOSAA, (Et- or Me-)FOSA, FOSA and PFOSI, the latter finally degrades to PFOS (Lange, 2018; Rhoads et al., 2008; Yin et al., 2018). For C6 analogues, PFHxS is the principal sink (Ruyle et al., 2023). Nitrification steps and S–N bond cleavage are identified as rate-limiting (Yin et al., 2018).

Mass balance between the parent material and measured products was achieved by Lange, 2018, implying that all significant degradation products of EtFOSE were identified.

Only three anaerobic or methanogenic tests were found (Lange, 2018; Sáez et al., 2008; Yin et al., 2018) and only Lange, 2018 reports a half-life of 35.8 days and degradation products MeFBSAA and PFBSI from a test with MeFBSE during 108 days degradation in an anaerobic digester. Yin et al., 2018 did not observe any biodegradation under anaerobic or anoxic conditions. This indicates a limited degradation potential of FASAs in anaerobic WWTP, which aligns with the findings of the review by Berhanu et al., 2023b.

Overall, oxic, warm (≈ 30 °C) activated sludge or wetland systems favour stepwise biotransformation of FASAs, while anoxic or methanogenic conditions impede it. Since

biotransformation of FASAs to PFASs can occur WWTPs and engineered wetlands, these wastewater treatment systems can act as ongoing sources of regulated PFAS to receiving waters.

30. Perfluoroalkyl Phosphate Esters (PAPs)

Perfluoroalkyl phosphate esters (PAPs) consist of fluorotelomer alcohol-derived tails, giving terminal perfluoroalkyl segments attached to short carbon chains, bonded to a phosphate ester core. PAPs follow structures like $(C_nF_{(2n+1)}C_mH_{(2m)}O)_xPO(OH)_{(3-x)}$ ($x=1-3$), for example 6:2-diPAP ($n=6, m=2, x=2$).

Summary of review findings:

PAPs are considered as a group of FTOH-based surfactants (Berhanu et al., 2023). The few studies found indicate notable resistance to hydrolysis with experimental and computed half-lives ranging from years to decades in water. No data was found on their atmospheric chemistry.

PAPs are readily degraded by microbes in oxic media: in activated sludge, 6:2 and 8:2 di- and mono-PAPs hydrolyse within days to the corresponding fluorotelomer alcohols, which are then oxidised through FTCA and FTUCA intermediates to short-chain PFCAs such as PFHxA, PFPeA, PFHpA and PFOA.

In aerobic soils, findings indicate that 6:2 diPAP degrades within weeks, whereas its 8:2 homologue persists for months-years and yields PFOA and related acids; plant roots and earthworms further enhance these transformations.

In sediments, behaviour diverges: DiSAmPAP has been found to be recalcitrant in marine settings but decays to PFOS over months in freshwater. According to (ECHA, 2025), it can be expected that n:2 monoPAPs and n:2 diPAPs will be degraded and transformed into C_x-PFCAs (with x= n-2, n-1, n, n+1) following a similar pathway as n:2 FTOH after initial hydrolysis.

The two identified studies on PAPs' fate in wastewater treatment provide evidence for the biotransformation of PAPs into PFCAs during wastewater treatment.

30.1 Abiotic degradation

30.1.1 Degradation in the atmosphere

No experimental data was found.

30.1.2 Degradation in water

Hydrolytic stability of both 8:2 monoPAPs and 8:2 diPAPs was investigated under aggressive conditions of pH 9 and 50°C. D'Eon & Mabury (2007) observed <0.1% degradation over a 2-week period for both 8:2 PAPs congeners, corresponding to a minimum lifetime of 26 years with respect to hydrolysis.

Rayne & Forest (2010) could not generate similar half-life estimates using the SPARC software and instead estimated half-lives on the order of seconds to days depending on the rate constants used. They suggested that either the current version of SPARC at the time could not model hydrolysis for PAPs or that estimates provided by D'Eon & Mabury are overestimated by up to several orders of magnitude. Estimated hydrolytic half-lives for SAmPAP and DiSAmPAP using the SPARC software were determined to be on the order of thousands of years at pH 7 and a temperature of 25°C. Under the conditions used by D'Eon & Mabury of pH 9 and 50°C, half-lives estimates were 2.7 and 19 years for SAmPAP and DiSAmPAP, respectively. No experimental hydrolysis measurements were found for comparison to benchmark these preliminary estimates.

30.2 Biotic degradation

30.2.1 Degradation in water and sediments

Different degrees of biodegradation were observed for DiSAMPAP in marine and freshwater sediments, displaying significant resistance to degradation in marine environments while being more readily biotransformed in freshwater settings (W. Zhang et al., 2021). A 120-day study by (Benskin et al., 2013) examined the biotransformation of DiSAMPAP in urban marine sediments at temperatures of 4°C and 25°C. Their findings revealed that DiSAMPAP was highly recalcitrant to microbial degradation, with calculated half-lives >380 days at 25°C and >3400 days at 4°C. It is hypothesized that the large molecular size and high hydrophobicity of DiSAMPAP reduce its bioavailability, thereby limiting its biotransformation by bacteria in sediments (W. Zhang et al., 2021).

In contrast, Zhang et al. (2018) explored the biotransformation of DiSAMPAP in freshwater sediments from Taihu Lake, China. Their study demonstrated that DiSAMPAP biodegraded into PFOS, with intermediate products such as PFOSA, EtFOSAA, EtFOSA, and EtFOSE observed during the process. The authors calculated a half-life of 88 days for DiSAMPAP in freshwater sediments. No degradation was observed for the triester TriSAMPAP under the same study conditions, with an estimated degradation half-life >281 days at 25 °C.

Further research by (Gaillard et al., 2017) investigated the biotransformation of DiSAMPAP in fish through dietary exposure. DiSAMPAP was biotransformed, and metabolites such as EtFOSAA, FOSAA, FOSA, and PFOS were detected in the fish. The proposed metabolic pathway for the biotransformation of DiSAMPAP is detailed in Figure .

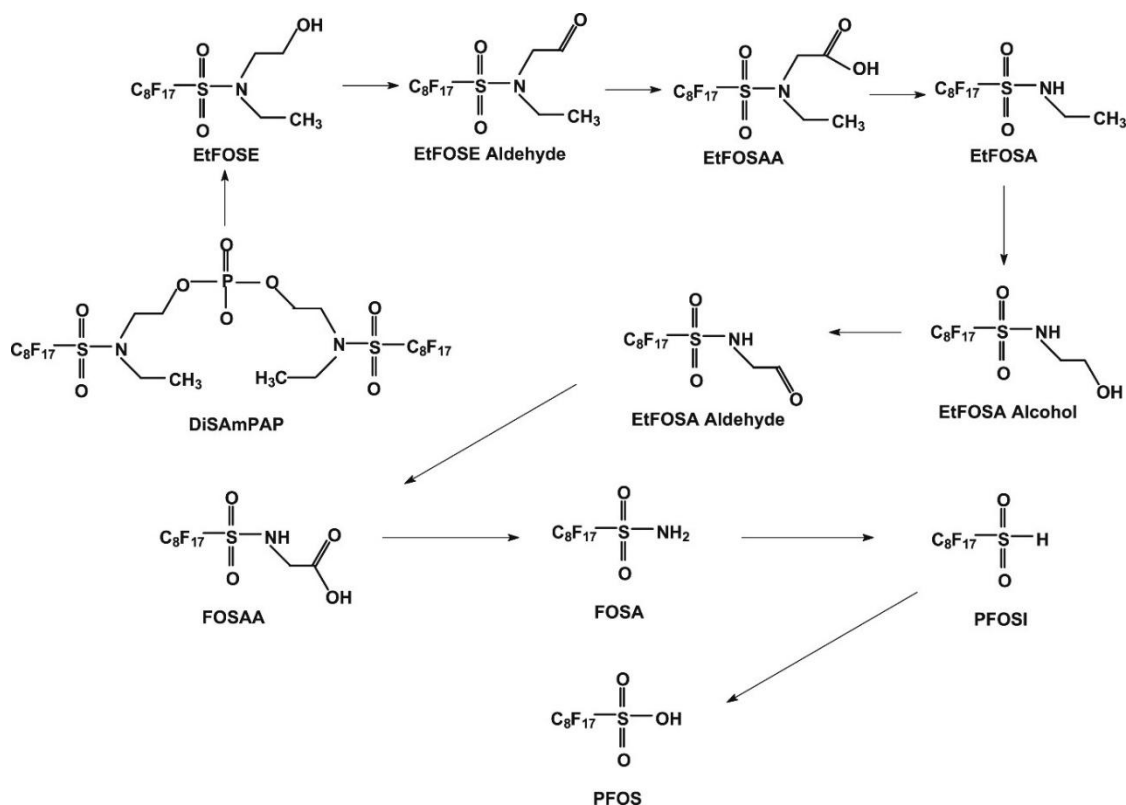


Figure 30-1. Metabolic pathways for the biotransformation of SAMPAP proposed by (W. Zhang et al., 2021).

30.2.2 Degradation in soil

Aerobic soils can hydrolyse and oxidise polyfluoroalkyl phosphate esters, but the rate and product spectrum depend on chain length, substitution pattern and soil biota (Figure 30-1).

Table 30-1. Biotic degradation of PAPs in soil. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
6:2 monoPAP (1 mg/kg soil)	Soil with maize plants (<i>Zea mays</i> L.)	Aerobic, 84 days, pH 5.6	Not provided	PFHxA, PFPeA, PFBA	(Just et al., 2022)
6:2 diPAP (4220 mg/kg soil)	Soil microcosm	Aerobic, 112 days	12 d	5:3 acid (9.3 %), PFPeA (6.4 %), PFHxA (6 %)	(C. Liu & Liu, 2016)
6:2 diPAP (1 mg/kg soil)	Soil with maize plants (<i>Zea mays</i> L.)	Aerobic, 118 days, pH 5.6	33 d	PFHpA, PFHxA, PFPeA, PFBA	(Just et al., 2022)
6:2 diPAP (0.38 mg/kg soil)	Soil and earthworms (<i>M. guillelmi</i>)	Aerobic, 21 days, 20 °C	Not provided	Soil: monoPAP (0.27 %), FPePA (0.27 %), PFHxA (7.76 %), Earthworms: 6:2 FTOH-sulfate, PFHxA, FPePA, PFPeA, FHEA, PFHpA, monoPAP	(Y. Zhu et al., 2021)
6:2 diPAP	Greenhouse microcosm	Aerobic, 165 days	2 months	6:2 FTCA, 6:2 FTUCA, 5:3 FTCA, 5:3 FTUCA, PFBA, PFPeA, PFHxA, minor PFHpA	(Lee et al., 2013)
8:2 monoPAP (1 mg/kg soil)	Soil with maize plants (<i>Zea mays</i> L.)	Aerobic, 84 days, pH 5.6	Not provided	PFOA, PFHpA, PFHxA, PFPeA, PFBA	(Just et al., 2022)
8:2 diPAP (1 mg/kg soil)	Soil with maize plants (<i>Zea mays</i> L.)	Aerobic, 118 days, pH 5.6	78 d	PFOA, PFHpA, PFHxA, PFPeA, PFBA	(Just et al., 2022)
8:2 diPAP (3337 mg/kg soil)	Soil microcosm	Aerobic, 112 days	>1000 d	PFOA (2.1 %)	(C. Liu & Liu, 2016)
8:2 diPAP (0.5 mg/kg soil)	Soil plant systems (carrot and lettuce)	Aerobic, 108 days, 20-25 °C	Not provided	PFOA (major), 8:2 FTCA, 8:2 FTUCA, PFBA, PFPeA, PFHxA, PFHpA, and PFNA	(Bizkarguenaga et al., 2016)

One mechanistic study suggested that hydrolysis first liberates the corresponding fluorotelomer alcohols, which are then funneled through FTCA/FTUCA intermediates to PFCAs (Lee et al., 2013). Shorter-chain 6:2 diPAP were found to degrade comparatively quickly in aerobic soil ($DT_{50} \approx 12$ d), whereas 8:2 diPAP persists for many months ($DT_{50} > 1000$ d) (C. Liu & Liu, 2016). In this experiment, the primary product of 8:2 diPAP was PFOA (2.1%), the major and terminal degradation products of 6:2 diPAP were 5:3 fluorotelomer carboxylic acid (5:3 acid, 9.3%), PFPeA (6.4%), PFHxA (6.0%) and PFBA (0.73%). In pot trials without vegetation, also PFOA dominated the 8:2 diPAP product slate (up to 62 % of recovered mass), while crops shifted the profile: carrots generated a broader mix of PFNA, PFHpA, PFHxA, PFPeA and PFBA in the cultivation media, whereas lettuce yielded almost exclusively PFOA (Bizkarguenaga et al., 2016). Maize experiments confirmed strong soil retention of di- and mono-PAPs yet still recorded mean PFCA conversion rates of 5–24 %, with α -oxidation signatures most evident for 8:2 diPAP (Just et al., 2022). Lifetimes of 33 and 78 days were calculated for 6:2 diPAP and 8:2 diPAP in this study, respectively. Soil fauna can further accelerate breakdown: earthworms shortened the 6:2 diPAP half-life and produced persistent PFCAs alongside the phase-II metabolite 6:2 FTOH-sulfate, highlighting in-organism

conjugation pathways (Y. Zhu et al., 2021). Collectively these studies show that PAPs can act as sources of short- to mid-chain PFCAs in agricultural soils.

30.2.3 Degradation in WWTPs

Only two studies were found that deal with the biotransformation of PAPs during wastewater treatment (Table 30-2).

Table 30-2. Biotic degradation of PAPs in WWTPs.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
4:2, 6:2, 8:2, and 10:2 monoPAPs; 6:2 diPAP (1 mg/L)	WWTP activated sludge	Aerobic, 92 days	Not provided	FTOHs (mainly 6:2 FTOH) and its degradation PFCA metabolites (PFHxA, PFPeA, PFHpA, 6:2 FTUCA, 5:3 FTCA) and transient 6:2 FTCA	(Lee et al., 2010)
6:2 PAPs (contained 11.5% equivalent of 6:2 FTOH if completely hydrolyzed) (not clearly provided)	WWTP activated sludge	Aerobic, 30 °C, 30 days	Not provided	6:2 FTOH, 6:2 FTCA, 6:2 FTUCA, 5:2 ketone, 5:2 sFTOH, PFHxA, 5:3 acid, 5:3 Uacid, PFHpA (0.04 mol%), PFPeA (0.47 mol%), 5:2 sFTOH (most abundant product on day 30, ca. 16 mol%)	(Lewis, Kim, Liu, et al., 2016)

Lee et al. (2010) investigated the biodegradation of PAPs within a WWTP-simulated system will contribute to PFCA release into the environment with WWTP effluent.

PAPs undergo microbial hydrolysis to produce FTOHs, which further oxidize to PFCAs. The majority of the oxidation products observed indicate biodegradation via β -oxidation-like mechanism. Additionally, the study detected odd-chain PFCAs, suggesting that other pathways

may also occur. The authors conclude that microbially mediated biodegradation of PAPs to PFCAs can occur in aerobic WWTP sludge.

M. Lewis et al. (2016) studied biodegradation of PAPs both with pure, known FTOH-degrading *Pseudomonas* strains, and with activated sludge. They found that activated sludge was more efficient at transforming 6:2 PAPs, with a yield of 22 mol% on a gram-of-biomass basis of the initially applied 6:2 PAPs. and produced more transformation products including PFHpA and PFPeA and 5:2 sFTOH as the most abundant degradation product on day 30. Based on the degradation results with both activated sludge and the *Pseudomonas* tests, the authors conclude that PAPs could be transformed to PFCAs both in wastewater treatment plants and in the environment at large.

Thus, both available studies provide evidence for the biotransformation of PAPs into PFCAs during wastewater treatment.

In addition, *in vivo* studies in Sprague-Dawley rats were found (D'Eon & Mabury, 2007, 2011).

31. Other non-polymeric PFAS

Other non-polymeric PFAS comprise structurally diverse fluorinated substances that meet the PFAS definition but are not covered by the specific substance groups defined in this report. These compounds may contain a variety of functional groups and structural features, including partially fluorinated carbon chains, branched backbones, cyclic structures, or mixed functional moieties.

Summary of review findings:

The heterogeneous set of “other” non-polymeric PFAS comprises intermediates such as perfluoro-aldehydes, polyfluoro-amides, fluorinated acrylates and methacrylates as well as a few specific fluorinated gases, for which literature data were found. In the atmosphere these compounds are generally short-lived -hours to weeks- undergoing OH- or photolysis-driven fragmentation that funnels a substantial fraction of fluorine into ultra-short PFCAs, most notably TFA, PFPrA and PFBA.

In aquatic and soil systems, knowledge is sparse and compound-specific: some polyfluoro-amides and trifluoromethyl acrylates are slowly degraded to PFCAs over weeks to months, while small sulfonates can even experience complete microbial defluorination under specialised conditions.

Aerobic wastewater sludges rapidly transform fluorinated ether/thio-ether alcohols to intermediate ether-PFCAs that currently appear stable.

Given the diverse chemistries, predicting environmental degradability, degradation pathways and degradation products for this diverse category remains uncertain.

31.1 Abiotic degradation

31.1.1 Degradation in the atmosphere

Perfluoroalkyl aldehydes ($C_nF_{2n+1}CHO$) are generated during the atmospheric oxidation of n:2 fluorotelomer alcohols and can undergo several mechanisms by which PFCAs can be generated. Trifluoroacetaldehyde (CF_3CHO), the smallest perfluoroalkyl aldehyde, is an intermediate degradation product of numerous PFAS precursors. Hurley et al. (2006) examined the Cl-atom-initiated oxidation of $C_nF_{2n+1}CHO$ ($n = 1, 2, 3, 4$) in the presence of NO_x and focused on the fate of the $C_nF_{2n+1}CO$ radical. Under natural atmospheric conditions these radicals can decompose to give C_nF_{2n+1} and CO and can also react with O_2 to form $C_nF_{2n+1}C(O)O_2$ radicals. The $C_nF_{2n+1}C(O)O_2$ radical can proceed via three pathways, ~50% of which results in formation of PFCAs (Behringer et al., 2021). Photolysis of CF_3COH has been investigated in a couple of studies, in which it has either been observed to form trifluoromethane (HFC-23, CF_3H) as the primary product, or produce the CF_3 radical that forms COF_2 as the dominant product, under atmospheric conditions (Hansen et al., 2021; Sulbaek Andersen & Nielsen, 2022).

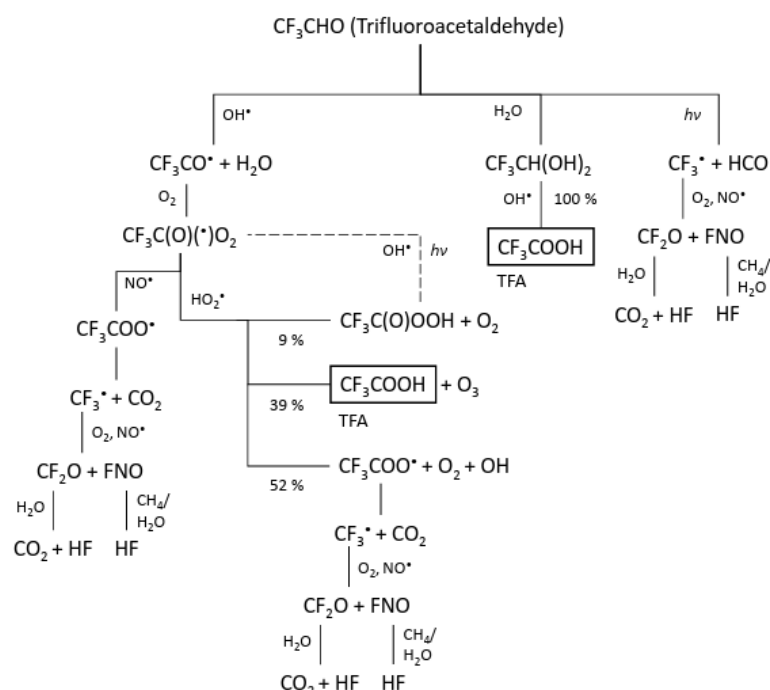


Figure 31-1. Atmospheric degradation pathways for trifluoroacetaldehyde (CF_3CHO) from (Behringer et al., 2021).

Polyfluoroalkyl amides ($\text{C}_n\text{F}_{2n+1}\text{C}(\text{O})\text{N}(\text{H})\text{R}$), an unintentional byproduct of per- and polyfluorinated sulfonamide synthesis by electrochemical fluorination, are semi-volatile compounds that may be degraded in the atmosphere via $\cdot\text{OH}$ radical-initiated oxidation. (Jackson et al., 2013) examined the atmospheric oxidation of the C4-based N-ethylperfluorobutyramide (EtFBA, $\text{C}_3\text{F}_7\text{C}(\text{O})\text{N}(\text{H})\text{CH}_2\text{CH}_3$) by reaction with Cl atoms and $\cdot\text{OH}$ radicals. EtFBA was not expected to react significantly with ozone or undergo photolysis in the troposphere. Reaction of EtFBA with OH is the dominant loss pathway in the troposphere with an estimated atmospheric lifetime of ~ 4.4 days. Reaction of EtFBA with either Cl atoms or $\cdot\text{OH}$ radicals result in formation of three potential compounds, the perfluorinated amine and two carbonyl products, all of which can undergo further reactions in the atmosphere to form the corresponding PFCAs. The authors observed formation of small yields ($<1\%$) of TFA and PFPrA, but a greater yield of (16%) of PFBA through an unconfirmed mechanism, suggesting that the historic C8-based polyfluoroalkyl amides could have been an atmospheric source for PFOA.

Other **fluorinated acrylates** and **methacrylates** can undergo atmospheric degradation to form PFCAs. (García et al., 2020) examined the OH-initiated oxidation for a series of fluorinated acrylates and methacrylates including 2,2,2-trifluoroethylmethacrylate, 1,1,1,3,3,3-hexafluoroisopropylacrylate, 1,1,1,3,3,3-hexafluoroisopropylmethacrylate, and 2,2,2-trifluoroethylacrylate. Reaction with $\cdot\text{OH}$ radicals yield estimated atmospheric lifetimes between 5-11 hours for these compounds, which is much faster those estimated for reaction with Cl atoms. Products of oxidation were not discussed in this study but expected to yield esters, fluorinated pyruvates, and glyoxylates that can undergo further subsequent reactions with $\cdot\text{OH}$ radicals to form PFCAs.

In addition to the HCFCs, HFCs, HFOs, and HFEs, there are other types of fluorinated gases that may degrade in the atmosphere to form PFCAs. **Heptafluoroisobutyronitrile** (C4-FN, $(\text{CF}_3)_2\text{CFCN}$) is a fluoronitrile gas with an atmospheric lifetime of ~ 22 years. (Sulbaek Andersen et al., 2017) examined the atmospheric degradation of C4-FN by reaction with Cl atoms, $\cdot\text{OH}$ radicals, and ozone. Reaction with $\cdot\text{OH}$ radicals is the main atmospheric sink, proceeding via addition of OH to the nitrile group, resulting in formation of NO, COF_2 , and $\text{CF}_3\text{C}(\text{O})\text{F}$ as the sole degradation products. $\text{CF}_3\text{C}(\text{O})\text{F}$

is formed in a molar yield of 100% and will undergo hydrolysis in the atmosphere to form TFA. **2-bromo-3,3,3-trifluoropropene (BTP, $\text{CH}_3=\text{CBrCF}_3$)** is an industrial raw material and used as a substitute for halons. (Patten et al., 2011) examined the atmospheric degradation of BTP with $\cdot\text{OH}$ radicals and estimated an atmospheric lifetime of 4–7 days in the troposphere. The authors did not measure the yields of oxidation products in this study but discussed the potential oxidation pathways in which OH-radical initiated oxidation may result in either formation of $\text{Br} + \text{CF}_3\text{C}(\text{O})\text{CH}_2\text{OH}$ or $\text{CH}_2\text{O} + \text{CF}_3\text{CBrO}$, depending on which C is attacked by OH. The authors speculated that $\text{CF}_3\text{C}(\text{O})\text{CH}_2\text{OH}$ is likely to be soluble and removed by wet deposition, whereas CF_3CBrO may quickly undergo hydrolysis to yield HBr and TFA or photolysis, in which it forms the $\text{CF}_3\text{C}(\text{O})\text{O}_2$ radical which can also yield TFA via one of the degradation pathways.

31.1.2 Degradation in water

No experimental data was found.

31.2 Biotic degradation

31.2.1 Degradation in water and sediments

Defluorination of partially fluorinated small-molecule sulfonates under aerobic, sulfur-limiting conditions by *Pseudomonas* sp. D2 strain in a defined mineral medium was investigated by Key et al., 1998. Full defluorination of **difluoromethane sulfonate (DFMS)** was found under tested conditions. Growth and defluorination were further observed for **2,2,2-trifluoroethane sulfonate (TES)** containing hydrogen, but not for **trifluoromethane sulfonate (TFMS)**. However, the environmental relevance of these laboratory studies is considered to be limited. *Pseudomonas* sp. D2 strain completely defluorinated DFMS under aerobic, sulfur-limiting conditions in a defined mineral medium. Growth and defluorination were further observed for 2,2,2-trifluoroethane sulfonate (TES) containing hydrogen and not for TFMS (Key et al., 1998).

31.2.2 Degradation in soil

Four studies were identified concerned with degradation in soil for other non-polymeric PFAS than mentioned in the previous chapters. Three studies are concerned with perfluoroalkyl amide derivatives (PFOAAm, PFOAB, PFOANO, PFOAAmS), which are zwitterionic substances used in aqueous film-forming foams (AFFFs). Moreover, one study concerned degradation in soil of a polyfluorinated acrylic acid use in polymer synthesis was found (Table 31-1. Biotic degradation of other non-polymeric PFAS in soil.).

Table 31-1. Biotic degradation of other non-polymeric PFAS in soil.

Compound (Initial concentration)	Microbia l species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
PFOAAm (2 mg/kg)	Soil	Aerobic, 180 days, 22 °C, pH 7.2	14	Not provided	(M. Liu et al., 2021)
PFOAB (1.8 mg/kg)	Soil	Aerobic, 150 days, 22 °C, pH 7.2	266-630	PFOA (5.8-32.6%)	(M. Liu et al., 2021)
OAmPr-FOAA (PFOANO) (0.85 mg/kg)	Sandy loam and loam	Aerobic, 90 days	3-7	PFOA (15–21 %)	(H. Chen et al., 2020)

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
TAmPr-FOAA (PFOAms)	Soil	Aerobic, 6 months, 22°C, pH 5	127-142	PFOA (30 %), PFOAAm, perfluorooctanamide propionate	(Mejia-Avendaño et al., 2016)
TFMAA	Landfill soil	Aerobic, 32 days, room temperature		PFBA (1.3 %), TFA (6.3 %)	(M. Sun et al., 2020)

Liu et al. (2021) investigated and compared the stability of nitrogen-containing polyfluoroalkyl substances, i.e. sulfonamide (FASAs) and amide derivatives, in aerobic soils. They found that perfluoroalkyl sulfonamide-based compounds (compare section on FASAs) showed higher microbial stability than the corresponding perfluoroalkyl amide-based ones such as PFOAAm and PFOAB.

While degradation of PFOAAm was not observed, its potential intermediate products PFOAB and PFOANO yielded PFOA as a stable degradation product with half-lives of days to years (3 – 630 days) depending on soil type and experimental conditions ((M. Liu et al., 2021), (H. Chen et al., 2020)).

In a 32-day study of aerobic microbial degradation in landfill soil, **(trifluoromethyl)acrylic acid (TFMAA)**, used in polymer synthesis, was investigated as a representative fluorinated intermediate. The study revealed low biotransformation efficiency compared to other fluorochemicals, with PFBA and TFA detected in the medium at 1.3 mol% and 6.3 mol%, respectively, by the end of the experiment. Notably, TFA production increased rapidly from 1.9 mol% on day 28 to 6.3 mol% on day 32, suggesting complex intermediate processes and indicating that longer culture times might enhance TFA yield from TFMAA biodegradation (M. Sun et al., 2020).

31.2.3 Degradation in WWTPs

Studies about biotransformation of other non-polymeric PFAS are shown in Table 31-2.

Table 31-2. Biotic degradation of other non-polymeric PFAS in WWTPs.

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
Methylpolyfluoroether thioether alcohol (MeFESOH) Propylpolyfluoroether thioether alcohol (FESOH) Methylpolyfluorodiether alcohol (MeFdiEOH) Propylpolyfluorodiether alcohol (ProFdiEOH) Propylpolyfluoroether alcohol (ProFEOH) (each at 1.2 µM)	WWTP sludge	Aerobic, 27 days	Not provided	FESCA (polyfluoroalkyl ether thioether carboxylic acid), MeFESCA, (methyl polyfluoro ether thioether carboxylic acid), MeFdiECA (methyl polyfluoroalkyl diether carboxylic acid), ProFdiECA (Propyl polyfluoroalkyldiether carboxylic acid), PFECA (polyfluoroalkyl ether carboxylic acid), ProFECA (propyl polyfluoroalkyl ether carboxylic acid) final: 2H-1:2 PFECA (from MeFESOH) and 2H-3:2 PFECA (from FESOH)	(Folkerson & Mabury, 2024)
2H-1:2 PFESA and 2H-1:2 PFECA (1.2 µM)	WWTP sludge	Aerobic, 74 days	Not provided	no further transformation	(Folkerson & Mabury, 2024)
1-(2,2,3,3,4,4,4-heptafluorobutoxy)-propan-2-ol (HFBPrOH) (10 mg/L)	WWTP effluent water	Aerobic, 20-25 °C, 35 days (77 days)	Not provided	Final product: PFBA. Several acidic biotransformation intermediates, as well as	(Frömel & Knepper, 2015)

Compound (initial concentration)	Microbial species or origin	Conditions	$t_{1/2}$ (days)	Degradation products	Reference
				heptafluorobutan-1-ol, were identified.	
3-(trifluoromethoxy)-propan-1-ol (TFMP_{ROH}) (10 mg/L)	WWTP effluent water	Aerobic, 20-25 °C, 77 days	Not provided	TFMC was also detected during biotransformation of TFMP _{ROH} ; however, the major transformation product was 3-trifluoromethoxypropanoic acid (TFMP _{PrA} , ca. 85%), which did not show any further degradation within a 47-day period.	(Frömel & Knepper, 2015)
6-(trifluoromethoxy)-hexan-1-ol (TFMH_{xOH}) (10 mg/L)	WWTP effluent water	Aerobic, 20-25 °C, 35 days (77 days)	Not provided	6-trifluoromethoxyhexanoic acid (TFMH _{xA}) as well as trifluoromethyl carbonate (TFMC) were identified as transient transformation products, both of which were entirely degraded. This difference in mineralization yield between TFMP _{ROH} and TFMH _{xOH} cannot be entirely explained because no further transformation products were detected in the course of this experiment.	(Frömel & Knepper, 2015)
10-(trifluoromethoxy)decane-1-sulfonate (400 mg/L)	sludge, non-adapted	Aerobic, room temperature, neutral pH, 28 days	Not provided	(not identified in the Zahn-Wellens test, degradation is quantified by DOC decrease)	(Peschka et al., 2008)

In a study by Frömel & Knepper (2015) three fluorinated alcohols were assessed for aerobic biotransformation with effluent water from a municipal wastewater treatment plant as inoculum. The three substances were 3-(trifluoromethoxy)-propan-1-ol (TFMP_{ROH}), 6-(trifluoromethoxy)-hexan-1-ol (TFMH_{xOH}) and 1-(2,2,3,3,4,4,4-heptafluorobutoxy)-propan-2-ol (HFBP_{ROH}). The major transformation product of TFMP_{ROH} was 3-trifluoromethoxypropanoic acid (TFMP_{PrA}), which did not show any further degradation within a 47 days period. Moreover, TFMP_{ROH} yielded approximately 15% fluoride. In contrast, TFMH_{xOH} reached nearly quantitative release of fluoride (~100%) after 37 days. HFBP_{ROH} yielded PFBA as a stable biotransformation product. Reasons for the varying degradation results for the three fluorinated alcohols could not be entirely explained in the study.

Folkerson & Mabury (2024) utilized microbial microcosms from WWTP sludge to investigate the impact of different functional groups such as heteroatom linkages, different chain lengths and hydrofluoroethers (HFEs) of five novel fluorosurfactants; methylpolyfluoroether thioether alcohol (MeFESOH), propylpolyfluoroether thioether alcohol (FESOH), methylpolyfluorodiether alcohol (MeFdiEOH), propylpolyfluorodiether alcohol (ProFdiEOH) and propylpolyfluoroether alcohol (ProFEOH). Additionally, they also studied biotransformation of two degradation products (2H-3:2 polyfluoroalkylether carboxylic acid [PFECA]) and 2H-1:2PFECA) in a follow-up study.

All HFE alcohol congeners displayed relatively rapid transformation, observable losses ranging from 50% to 75% over the 27 days with the exception of ProFEOH. The HFE alcohols congeners with a thioether (FESOH and MeFESOH) transformed faster than the ether congeners and also produced second-generation HFE acid products (PFECA and 2H-1:2 PFECA). The follow-up experiment with the degradation products 2H-1:2 PFESA and 2H-1:2 PFECA displayed no further transformation over 74 days.

Peschka et al. (2008) studied biodegradation of 10-(trifluoromethoxy)decane-1-sulfonate with the Zahn-Wellens test (OECD Guideline) over a period of 28 days with non-adapted sewage sludge, as

well as the formation of metabolites to elucidate the mechanism of biotransformation. They found that 10-(trifluoromethoxy)dec-ane-1-sulfonate was inherently biodegradable (>75%) in within 28 days based on DOC (dissolved organic carbon) decrease. Information about degradation products were not available from the Zahn-Wellens test.

32. Fluoropolymers

Fluoropolymers (FPs) consist of repeating fully or partially fluorinated carbon backbones, with high molecular weights. Examples include polytetrafluoroethylene (PTFE), polyvinylidene fluoride (PVDF), fluorinated ethylene propylene (FEP), ethylene tetrafluoroethylene (ETFE), and perfluoroalkoxy polymer (PFA).

Summary of review findings:

Fluoropolymers such as PTFE, PVDF, FEP, ETFE and PFA are considered resistant to degradation under normal environmental conditions. Their non-volatile, high-molecular-weight chains resist photolysis, hydrolysis, oxidation and microbial degradation. Laboratory studies on PTFE following OECD protocols have shown no measurable breakdown or fluoride release in water, soils, sediments or sewage sludge, whether under aerobic or anaerobic conditions. Only under extreme thermal conditions can the fluoropolymer backbone be degraded, leading to small perfluorinated fragments. Consequently, fluoropolymers are considered stable and inert under aerobic and anaerobic conditions in relevant environmental compartments.

32.1 Abiotic degradation

32.1.1 Degradation in the atmosphere

No studies have been found in the review that examined atmospheric degradation of fluoropolymers due to their high stability and the fact that they are not volatile. Photoinduced reactions with fluoropolymers are not expected to occur (Henry et al., 2018). PTFE for example has been shown to be resistant to photolysis (Henry & Timmer, 2025).

Fluoropolymers are thermodynamically stable under normal environmental conditions. Ellis et al. (2001) examined the thermolysis of fluoropolymers at temperatures up to 500°C, which can lead to the formation of PFCAs, in particular TFA, however, this requires temperatures that far exceed those observed in the atmosphere (fluoropolymers shown to be thermally stable at 260°C for PTFE and PFA, 200°C for FEP, and 150°C for ETFE (Henry et al., 2018)).

32.1.2 Degradation in water

Fluoropolymers are hydrolytically stable and are not subject to hydrolysis catalysed degradation (Henry et al., 2018).

The study of Joudan et al. (2024) has been assessed, however, the results were not considered relevant for extrapolation to real-world environmental conditions.

32.2 Biotic degradation

Fluoropolymers are considered to be biologically inert and do not undergo degradation by microorganisms under aerobic or anaerobic conditions (Henry et al., 2018). PTFE for example has been shown to be resistant to aerobic and anaerobic conditions in soil, water and sediment, and activated sewage sludge (Henry & Timmer, 2025).

33. Perfluoropolyethers (PFPEs)

Perfluoropolyethers (PFPEs) are polymers consisting of perfluoro-alkyl groups joined by ether linkages and feature, for example, repeating units like $-(C_mF_{2m}O)_n-$ or variants.

Summary of review findings:

Current evidence demonstrates that perfluoropolyethers (PFPEs) are stable to hydrolysis, photolysis, oxidation, and aerobic and anaerobic biodegradation (Javed et al., 2025). Examples of commercially available PFPEs include PFPE-D ($F-(CF_2CF_2CF_2O)_xC_nF_{2n+1}$), PFPE-K ($CF_3CF_2CF_2O-[CF(CF_3)CF_2O]_xC_nF_{2n+1}$), PFPE-Y ($CF_3-[(OCF(CF_3)CF_2)_x(OCF_2)_x]OC_nF_{2n+1}$) and PFPE-Z ($CF_3O-[CF_2CF_2O][CF_2O]CF_3$). These PFPEs are all thermally stable at common environmental temperature ranges. However, studies on environmental degradation are limited.¹³

According to OECD (2024), perfluoroether-based non-polymeric PFASs such as HFPO-DA ("GenX") and ADONA (see Chapter 10) also exhibit high environmental persistence. While these low-molecular-weight perfluoroether carboxylic acids differ structurally and functionally from polymeric PFPEs, their persistence has been attributed in part to the chemical stability of the perfluoroether ($-C-O-C-$) linkage in a fully fluorinated carbon framework. In this context, the resistance of the ether bond to abiotic and biotic cleavage supports the expectation that PFPEs may likewise exhibit high environmental stability, although their polymeric nature and physicochemical properties differ substantially from those of PFECAs.

33.1 Abiotic degradation

33.1.1 Degradation in the atmosphere

Young et al. (2006) investigated the atmospheric behaviour of a commercial mixture containing perfluoropolymethylisopropyl ethers (PFPMIEs). They examined a fraction that boiled at 70°C, primarily composed of $CF_3OCF(CF_3)(CF_2O)_2CF_3$, with smaller quantities of $CF_3OCF(CF_3)(CF_2O)_3CF_3$ as well as longer-chain analogues. This fraction had an average molecular weight of 410 Daltons. The authors observed low reactivity in the troposphere and noted that the PFPE's lifespan is constrained by transport to the mesosphere, where efficient photolysis by Lyman-alpha radiation occurs. They estimated the lower limit for the total atmospheric lifetime of these PFPEs (i.e. PFPMIEs) to be around 800 years, with degradation via C-C or C-O bond cleavage in $CF_3OCF(CF_3)-$ or $CF_3OCF_2OCF_2-$ structures resulting in the formation of 2 or 4 molecules of COF_2 , respectively (Young et al., 2006 as cited in OECD (2024)).

33.1.2 Degradation in water

Javed et al. (2025) concludes that four commercially relevant PFPEs (Table 33-1) exhibit stability under hydrolytic, photolytic, and oxidative conditions. According to the authors, these PFPEs do not decompose into species with a molecular weight of less than 1000 when exposed to such conditions. This conclusion is based on the data provided below.

Tsuda et al. (2023) tested hydrolysis and indirect photolysis of a linear PFPE polymer sold under the commercial trade name DEMNUM (CAS 151772-59-7), a carboxylic acid derivative of PFPE-D.

¹³ While there is limited information on environmental degradation available, a number of studies have investigated use-related degradation of PFPEs, particularly when used as lubricants in different industrial applications. For example, Zhao et al. (2000) reported no obvious effect of ambient temperature up to 70 °C on the decomposition of the PFPE using several experimental approaches. Other investigations summarised in (OECD, 2024) indicate that notable scission or volatilisation of PFPE chains generally requires temperatures above ~100 °C. Because such conditions are not encountered in typical environmental pathways, these "use-related" degradation studies are not covered in detail here; readers are referred to (OECD, 2024) for further information.

Hydrolysis experiments using OECD Test No. 111 showed that DEMNUM remained essentially intact at all three tested pH (pH 4, 7, 9). After five days at 50 °C, fewer than a dozen PFPE carboxylic acids were detectable, all below the method's 0.1 µg L⁻¹ limit of quantification; this corresponds to roughly two parts per million relative to the initial polymer mass and is far below the 10 % loss that would mandate extended testing. On this basis the hydrolytic half-life was estimated to exceed one year.

Indirect photolysis following US EPA guidelines (OPPTS 835.5270) was likewise ineffective. When the polymer was irradiated in synthetic humic water, trace quantities of about a dozen LMWS appeared, peaking near 150 ppm - only about 0.01 % of the polymer's mass - after fourteen days before declining slightly, a pattern that suggests the preferential transformation of minor impurities rather than backbone scission. The derived first-order rate constant for this process is on the order of 10⁻⁶ day⁻¹, confirming the extraordinary resistance of PFPE chains to cleavage by water and sunlight under environmentally relevant conditions.

According to Javed et al. (2025), industry data suggests that PFPE-Y and PFPE-Z oils, similar to other PFPEs, exhibit excellent hydrolytic, chemical, oxidative, and thermal stability (data could not be verified). Additionally, PFPE-K oils are reported to be stable against photolytic degradation based on tests conducted by Chemours, as cited by Javed et al. (2025)¹⁴. The photolytic stability of PFPE-K oils was assessed by placing a 0.5 mL sample into a medium-walled 5 mm outer diameter borosilicate NMR tube (Wilmad-Labglass 504-PP) under ambient air. A baseline 19F NMR spectrum was acquired before irradiation. The sample was then irradiated for 31 hours using a Xe arc lamp. Subsequently, a post-irradiation spectrum was obtained and compared to the spectrum of an unirradiated sample.

Table 33-1. Four commercial relevant PFPEs according to Javed et al. (2025).

Acronym	Name	CAS	Linear/Branched
PFPE-D	Oxetane, 2,2,3,3- tetrafluoro-, homopolymer, fluorinated	113114-19-5	Linear
PFPE-Z	Ethene, tetrafluoro-, oxidized, polymd	69991-61-3	Linear
PFPE-K	Poly[oxy(trifluoro (trifluoromethyl)-1,2-ethanediyl)], α-(1,1,2,2,2-pentafluoroethyl)-ω-[tetrafluoro (trifluoromethyl) ethoxy]-	60164-51-4	Branched
PFPE-Y	1-Propene, 1,1,2,3,3,3-hexafluoro-, oxidized, polymd.	69991-67-9	Branched

33.2 Biotic degradation

Similar to their stability under abiotic conditions, Javed et al. (2025) concludes that the four commercially relevant PFPEs (Table 33-1) are stable against biodegradation under both aerobic and anaerobic conditions. This conclusion is primarily based on the biodegradation tests conducted by Tsuda et al. (2023), as described below.

33.2.1 Degradation in water and sediments

No experimental data was found.

¹⁴ Primary data could not be verified

33.2.2 Degradation in soil

No studies have been identified regarding the degradation of PFPEs in soil. However, Evich et al. (2022) did not find any convincing evidence of scission at ether linkages in the perfluoroether chain of a structurally related chloro-perfluoropolyether carboxylates (Cl-PFECAs) in contaminated soil samples, despite extensive investigatory efforts (see chapter 10.2.2 for more details).

33.2.3 Degradation in WWTPs

The biodegradability of DEMNUM, the carboxylic acid derivatives of PFPE-D, was determined using Zahn-Wellens/EMPA test per OECD Method 302B, and US EPA OPPTS 835.3200 (Tsuda et al., 2023). The test demonstrated that typical aerobic sludge communities do not appreciably degrade the testing substance. DEMNUM introduced at 0.1 g L^{-1} produced no measurable low molecular weight substances (LMWS) in unconcentrated samples, and only after four-fold concentration could fewer than a dozen degradation products be discerned. The maximum total LMWS amounted to approximately eighty parts per million - roughly 0.03 % of the polymer mass on the twenty-seventh day, and the corresponding first-order rate constant was roughly 10^{-6} day^{-1} . The size distribution of the acids detected in sludge shifted toward slightly larger molecules than those seen in the photolysis test, yet their concentrations remained minute. These findings indicate that the DEMNUM is essentially resistant to microbial degradation under the testing conditions.

34. Perfluoroalkylether side-chain fluorinated polymers

Perfluoroalkylether side-chain fluorinated polymers consist of perfluoroalkylether pendants (e.g., $-\text{O}(\text{CF}_2\text{CF}_2\text{O})_m\text{R}_f$), attached to a fluorinated or non-fluorinated polymer backbone. In these materials, the fluorinated moieties are incorporated as side chains rather than forming the main polymer backbone.

Summary of review findings:

No experimental information was found on the environmental degradation of perfluoroalkylether side-chain fluorinated polymers. By analogy with other side-chain fluoropolymers, the perfluoroalkylether side-chain may have the potential to be cleaved from the polymer backbone under environmental conditions, liberating the perfluoroalkylether moiety. Information on perfluoroalkylether side-chain fluorinated polymers and their potential for degradation represents a data gap.

35. Side-chain fluorinated polymers

Side-chain fluorinated polymers (SCFPs) consist of a non-fluorinated polymer backbone of variable composition and per- and polyfluorinated side chains. The fluorinated moieties are typically incorporated as pendant groups, often via spacer units such as $-(\text{CH}_2)_m-$, and may include perfluoroalkyl or fluorotelomer-based structures.

Summary of review findings:

Side-chain fluorinated polymers (SCFPs) include fluorinated acrylate and methacrylate polymers, fluorinated urethane polymers, and fluorinated oxetane polymers. The per- and polyfluorinated side chains are often labile and have the potential to be cleaved from the polymer backbone under environmental conditions, liberating the PFAS.

Abiotic and biotic degradation of acrylate and urethane SCFPs have been examined by several studies. Although side-chain fluorinated polymers are generally regarded as less reactive than their small-molecule analogues, they are not completely inert. Laboratory hydrolysis studies show that polymers with ester linkage can degrade under neutral conditions over decades to form the n:2 fluorotelomer alcohols and resulting PFCAs.

In soils, this process can be accelerated by microbial activity, root exudates or biosolid amendments, yet even under favourable conditions the inferred half-lives for complete side-chain removal range from several years to centuries, indicating that SCFPs can serve as long-term, low-level sources of PFCAs. No data were found for atmospheric or biotic degradation in aquatic or sedimentary compartments.

35.1 Abiotic degradation

35.1.1 Degradation in the atmosphere

No studies have been found in this review which examined atmospheric degradation of SCFPs, however numerous studies have examined the atmospheric degradation for the various classes of per- and polyfluorinated side chains that would be connected to the polymer backbone (e.g. fluorotelomer acrylates and fluorinated acrylates/methacrylates) as discussed in other sections.

35.1.2 Degradation in water

Washington & Jenkins (2015) conducted a study on the abiotic hydrolysis of a commercial acrylate fluorotelomer-based polymer (AFTP) across eight different pHs ranging from 5-12 over a 77-day timeframe. The AFTP contained ~50% 8:2 and ~30% 10:2 mass of fluorotelomer components. Reaction rates and half-lives for the AFTP were determined as a function of the 8:2 and 10:2 FTOH degradation products. The reaction rate was determined to be a function of $[\text{OH}^-]$ with the reaction mechanism being base-catalysed nucleophilic acyl substitution at high pH (>9) and mediated by water at intermediate pH (5–8). The authors found that the hydrolysis of the AFTP proceeds with half-life estimates ranging from 55 to 89 years under circum-neutral conditions. Specifically, the half-life of 8:2 FTOH in the tested AFTP at 25°C is ~55 years at low pH levels where water hydrolysis dictates reaction rates and significantly decreases to ~0.7 years at pH 12. Modelling data for 10:2 FTOH suggests a half-life of up to 89 years at low pH, but the data for 8:2 FTOH is considered more reliable due to its higher concentration in the original AFTP. While not examined in this study, the n:2 FTOHs are expected to undergo further hydrolysis to form the corresponding PFCAs.

Washington, Jenkins, Rankin, et al. (2015) conducted a 376-day study on the degradability of a commercial AFTP in water in conjunction with soil experiments, to assess the stability of the AFTP in water. The AFTP contained ~50% 8:2 and ~30% 10:2 mass of fluorotelomer components. The AFTP in the water control (neutral pH) degraded at a similar rate as the AFTP in soil (estimated half-lives of 54-63 years), resulting in high concentrations of n:2 FTOHs, and lower concentrations of the corresponding PFCAs, suggesting abiotic hydrolytic scission at the ester linkage can occur to form the FTOHs. Follow-up experiments conducted at pH 3 and 10 showed that the AFTP degraded roughly 10-fold faster in water at pH 10 with a modelled half-life of ~5.5 years.

35.2 Biotic degradation

35.2.1 Degradation in water and sediments

No experimental data was found.

35.2.2 Degradation in soil

Results from studies that have assessed biotic degradation of acrylate and urethane fluorotelomer-based polymers are summarized in Table 35-1. Based on available data, these SCFPs are shown to degrade in a similar manner as other small molecule precursors into the corresponding PFCAs, but with much longer half-lives.

Table 35-1. Biotic degradation of side-chain fluorinated polymers in soil. Data was reproduced from (Berhanu et al., 2023) and supplemented with selected studies identified within the project.

Compound (initial concentration)	Microbial species or origin	Conditions	t _{1/2} (years)	Degradation products	Reference
Acrylate fluorotelomer-based polymer (AFTP) (800 mg/kg soil)	Sandy loam and loam soil microcosms	Aerobic, 20 °C, 730 days	1200-1700	AFTP → 8:2 FTOH → 8:2 FTCA, 8:2 FTUCA, 7:2 sFTOH, 7:3 FTCA, C8-C11 PFCAs	((Russell et al., 2008)
Acrylate fluorotelomer-based polymer (AFTP) (10 mg)	Sandy loam soil microcosm	Aerobic, 25 °C, 497 and 546 days	870-1400	AFTP → 8:2 FTAc, 8:2 FTOH → 8:2 FTCA, 8:2 FTUCA, 7:3 FTUCA, 7:3 FTCA, PFOA, 2HPFOA, other C6-C11 PFCAs	(Washington et al., 2009)
Acrylate fluorotelomer-based polymer (AFTP) (50 mg)	Sandy loam soil & alfalfa plant-soil microcosms	Aerobic, 21-25 °C, 167 days	8-111	AFTP → 8:2 FTAc, 8:2 FTOH → 8:2 FTCA, 8:2 FTUCA, 7:3 FTUCA, 7:3 FTCA, C6-C10 PFCAs	(Rankin et al., 2014)
Acrylate fluorotelomer-based polymer (AFTP)	Sandy loam and loam soil microcosms, endogenous microbes	Aerobic, 376 days	33-112	AFTP → n:2 FTAc, n:2 FTOH → n:2 FTCA, n:2 FTUCA, n:3 FTUCA, n:3 FTCA, C6-C16 PFCAs (n:2=8, 10, 12, 14; n:3=5, 7, 9)	(Washington, Jenkins, Rankin, et al., 2015)
Urethane fluorotelomer-based polymer (UFTP) (200 mg)	Sandy loam and loam soil microcosms	Aerobic, 20 °C, 730 days	28-241	UFTP → 8:2 FTOH → 8:2 FTCA, 8:2 FTUCA, 7:2 sFTOH, 7:3 FTCA, C8-C11 PFCAs	(Russell et al., 2010)
Urethane fluorotelomer-based polymer (UFTP) (100 ug/g soil)	Silty clay loam and forest soil microcosms	Aerobic, 22 °C, 117 and 180 days	0.25-1.85	UFTP → 8:2 FTOH → 7:2 sFTOH, C6-C8 PFCAs	(Dasu & Lee, 2016)

Russell et al., (2008) conducted a two-year study on the degradability of AFTP in four aerobic soils. The AFTP contained mostly 8:2 and 10:2 fluorotelomer components. The AFTP showed a slight extent of potential biodegradation of the n:2 FTOH side chains under the experimental conditions of the study for all soil types. Terminal biotransformation products of n:2 FTOH were measured including C8-C11 PFCAs and 7:3 FTCA. A biodegradation half-life of 1200-1700 years for all soils combined was calculated for the AFTP based on the rate of formation of PFOA.

Washington et al. (2009) conducted a 500–550-day study on the degradability of an AFTP in a soil microcosm. The AFTP contained mostly 8:2 and 10:2 fluorotelomer components. The AFTP showed potential biodegradation of the n:2 FTAc and n:2 FTOH side chains under the experimental conditions of the soil microcosm. Terminal biotransformation products of n:2 FTOH were measured including the C6-C11 PFCAs and 7:3 FTCA. The authors also observed PFCA loss between the two treatment periods suggesting possible polymer catalysed degradation. An AFTP biodegradation half-life of 870-1400 years was derived and modelling suggested that more finely grained polymers in soils might have half-lives of about 10-17 years assuming degradation is surface-mediated.

Rankin et al. (2014) conducted a 5.5-month study on the degradability of an 8:2 AFTP in a soil-plant microcosm in the absence and presence of wastewater treatment plant biosolids. The AFTP is a homopolymer of 8:2 FTAc. The AFTP showed potential biodegradation of the 8:2 FTAc side chains resulting in terminal degradation products including the C6-C10 PFCAs, and 7:3 FTCA, with PFOA

as the predominant product in the soil only and plant-soil microcosm. An AFTP biodegradation half-life range of 8-111 years was inferred from the 8:2 FTOH equivalent of the AFTP across all microcosm experiments in this study, but the biodegradation rate was faster in the presence of plant than just in soil.

Washington, Jenkins, Rankin, et al. (2015) conducted a 376-day study on the degradability of two commercial AFTPs in four saturated soil microcosms. The AFTP contained ~50% 8:2 and ~30% 10:2 mass of fluorotelomer components. Modelling of seven sampling rounds across microcosm treatments for one AFTP yielded half-life estimates of 65-112 years and 33-112 years for the other AFTP. The AFTPs showed potential biodegradation of the n:2 fluorotelomer side chains resulting in numerous n:2 (n=6, 8, 10, 12, 14, 16) fluorotelomer components including ethers, acetates, acrylates and the FTOHs and resultant degradation products including the n:3 FTCAs (n=7, 9, 11) and the C6-C16 PFCAs, of which PFOA was the dominant terminal product.

Russell et al. (2010) conducted a two-year study on the degradability of a urethane fluorotelomer-based polymer (UFTP) in four aerobic soils. The UFTP contained ~30% 6:2 and 30% 8:2 FTOH and smaller percentages of longer chain FTOHs as raw material. The UFTP showed potential biodegradation of the n:2 FTOH side chains under the experimental conditions of the study for all soil types. Terminal biotransformation products of 8:2 FTOH were measured including C8-C11 PFCAs and 7:3 FTCA. A biodegradation half-life of 102 years (range: 28-241) was calculated for the UFTP based on regression analysis of the rate of PFOA formation in the four aerobic soils.

Dasu & Lee (2016) conducted a 6-month study on the degradability of two UFTPs: toluene-2,4-di (8:2 fluorotelomer urethane) (FTU) and hexamethylene-1,6 di (8:2 fluorotelomer urethane) (HMU) in a forest soil, with only FTU degradability assessed in an agricultural silty clay loam soil. Negligible degradation of the FTU occurred in the agricultural soil over the study duration, but in the forest soil, both FTU and HMU degraded by up to 24 and 27%, respectively, with C6-C8 PFCAs observed as transformation products, dominated by PFOA. A biodegradation half-life of 3-5 months was estimated for FTU and 16-22 months for HMU in the forest soil experiment.

35.2.3 Degradation in WWTPs

Rankin et al. (2014) conducted a 5.5-month study on the degradability of an 8:2 AFTP in a soil-plant microcosm in the absence and presence of wastewater treatment plant biosolids. The AFTP showed potential biodegradation of the 8:2 FTAc side chains resulting in terminal degradation products including the C6-C10 PFCAs, and 7:3 FTCA, with highest concentrations for PFOA in the biosolid-amended microcosm. The authors observed enhanced biodegradation in the microbially enriched soils with biosolids relative to the soil only and soil-plant microcosms. The AFTP biodegradation range across all studies was determined to be 8-111 years, but the biodegradation rate was found to be fastest for the soil-plant microcosm in the presence of biosolids.

36. References

- Acerboni, G., Beukes, J. A., Jensen, N. R., Hjorth, J., Myhre, G., Nielsen, C. J., & Sundet, J. K. (2001). Atmospheric degradation and global warming potentials of three perfluoroalkenes. *Atmospheric Environment*, 35(24), 4113–4123. [https://doi.org/10.1016/S1352-2310\(01\)00209-6](https://doi.org/10.1016/S1352-2310(01)00209-6)
- Albert, K., Hsieh, P.-Y., Chen, T.-H., Hou, C.-H., & Hsu, H.-Y. (2020). Diatom-assisted bioreactor targeting the complete removal of perfluorinated compounds. *Journal of Hazardous Materials*, 384. Scopus. <https://doi.org/10.1016/j.jhazmat.2019.121491>
- Alexandrino, D. A. M., Ribeiro, I., Pinto, L. M., Cambra, R., Oliveira, R. S., Pereira, F., & Carvalho, M. F. (2018). Biodegradation of mono-, di- and trifluoroacetate by microbial cultures with different origins. *New Biotechnology*, 43, 23–29. Scopus. <https://doi.org/10.1016/j.nbt.2017.08.005>

- Arakaki, A., Ishii, Y., Tokuhisa, T., Murata, S., Sato, K., Sonoi, T., Tatsu, H., & Matsunaga, T. (2010). Microbial biodegradation of a novel fluorotelomer alcohol, 1H,1H,2H,2H,8H,8H-perfluorododecanol, yields short fluorinated acids. *Applied Microbiology and Biotechnology*, 88(5), 1193–1203. Scopus. <https://doi.org/10.1007/s00253-010-2815-9>
- Bao, M., Feng, H., Zheng, Y., Luo, H., Sun, C., & Pan, Y. (2023). Determination of Perfluorooctane Sulfonyl Fluoride and Perfluorohexane Sulfonyl Fluoride in Soil by Chemical Derivatization and Liquid Chromatography–Tandem Mass Spectrometry. *Environmental Science & Technology*, 57(10), 4180–4186. <https://doi.org/10.1021/acs.est.2c06958>
- Behringer, Dr. Felix Heydel, Barbara Gschrey, Steffi Osterheld, Winfried Schwarz, Kristina Warncke, Finnian Freeling, Dr. Karsten Nödler, Dr. Stephan Henne, Dr. Stefan Reimann, Markus Blepp, Wolfram Jörß, Ran Liu, Dr. Sylvie Ludig, Ina Rüdenauer, & Dr. Stefan Gartiser. (2021). *Persistent degradation products of halogenated refrigerants and blowing agents in the environment: Type, environmental concentrations, and fate with particular regard to new halogenated substitutes with low global warming potential* (Nos Texte , 73/2021). Umweltbundesamt.
- Benskin, J. P., Ikonomou, M. G., Gobas, F. A. P. C., Begley, T. H., Woudneh, M. B., & Cosgrove, J. R. (2013). Biodegradation of *N*-Ethyl Perfluorooctane Sulfonamido Ethanol (EtFOSE) and EtFOSE-Based Phosphate Diester (SAMPAP Diester) in Marine Sediments. *Environmental Science & Technology*, 47(3), 1381–1389. <https://doi.org/10.1021/es304336r>
- Bentel, M. J., Yu, Y., Xu, L., Li, Z., Wong, B. M., Men, Y., & Liu, J. (2019). Defluorination of Per- and Polyfluoroalkyl Substances (PFASs) with Hydrated Electrons: Structural Dependence and Implications to PFAS Remediation and Management. *Environmental Science & Technology*, 53(7), 3718–3728. <https://doi.org/10.1021/acs.est.8b06648>
- Berhanu, A., Mutanda, I., Taolin, J., Qaria, M. A., Yang, B., & Zhu, D. (2023). A Review of Microbial Degradation of Per- and Polyfluoroalkyl Substances (PFAS): Biotransformation Routes and Enzymes. *Sci. Total Environ.*, 859, 160010.
- Bernard, F., Papanastasiou, D. K., Portmann, R. W., Papadimitriou, V. C., & Burkholder, J. B. (2020). Atmospheric lifetimes and global warming potentials of atmospherically persistent N(C F) , x = 2–4, perfluoroamines. *Chemical Physics Letters*, 744, 137089. <https://doi.org/10.1016/j.cplett.2020.137089>
- Bizkarguenaga, E., Zabaleta, I., Prieto, A., Fernández, L. A., & Zuloaga, O. (2016). Uptake of 8:2 perfluoroalkyl phosphate diester and its degradation products by carrot and lettuce from compost-amended soil. *Chemosphere*, 152, 309–317. <https://doi.org/10.1016/j.chemosphere.2016.02.130>
- Burkholder, J. B., Cox, R. A., & Ravishankara, A. R. (2015). Atmospheric Degradation of Ozone Depleting Substances, Their Substitutes, and Related Species. *Chemical Reviews*, 115(10), 3704–3759. <https://doi.org/10.1021/cr5006759>
- Butt, C. M., Young, C. J., Mabury, S. A., Hurley, M. D., & Wallington, T. J. (2009). Atmospheric Chemistry of 4:2 Fluorotelomer Acrylate [C₄ F₉ CH₂ CH₂ OC(O)CH=CH₂]: Kinetics, Mechanisms, and Products of Chlorine-Atom- and OH-Radical-Initiated Oxidation. *The Journal of Physical Chemistry A*, 113(13), 3155–3161. <https://doi.org/10.1021/jp810358k>
- Bygd, M. D., Aukema, K. G., Richman, J. E., & Wackett, L. P. (2021). Unexpected Mechanism of Biodegradation and Defluorination of 2,2-Difluoro-1,3-Benzodioxole by *Pseudomonas putida* F1. *mBio*, 12(6), e03001-21. <https://doi.org/10.1128/mBio.03001-21>
- Carpenter, L. J., & Reimann, S. (2014). Ozone-Depleting Substances (ODSs) and Other Gases of Interest to the Montreal Protocol. In *Chapter 1 in Scientific Assessment of Ozone Depletion: 2014, Global Ozone Research and Monitoring Project – Report No. 55, World Meteorological Organization, Geneva, Switzerland*.
- Cerro-Gálvez, E., Roscales, J. L., Jiménez, B., Sala, M. M., Dachs, J., & Vila-Costa, M. (2020). Microbial responses to perfluoroalkyl substances and perfluorooctanesulfonate (PFOS) desulfurization in the Antarctic marine environment. *Water Research*, 171. Scopus. <https://doi.org/10.1016/j.watres.2019.115434>
- Che, S., Jin, B., Liu, Z., Yu, Y., Liu, J., & Men, Y. (2021). Structure-specific aerobic defluorination of short-chain fluorinated carboxylic acids by activated sludge communities. *Environmental*

- Science and Technology Letters*, 8(8), 668–674. Scopus. <https://doi.org/10.1021/acs.estlett.1c00511>
- Chen, H., Choi, Y. J., & Lee, L. S. (2018). Sorption, Aerobic Biodegradation, and Oxidation Potential of PFOS Alternatives Chlorinated Polyfluoroalkyl Ether Sulfonic Acids. *Environmental Science & Technology*, 52(17), 9827–9834. <https://doi.org/10.1021/acs.est.8b02913>
- Chen, H., Liu, M., Munoz, G., Duy, S. V., Sauvé, S., Yao, Y., Sun, H., & Liu, J. (2020). Fast Generation of Perfluoroalkyl Acids from Polyfluoroalkyl Amine Oxides in Aerobic Soils. *Environmental Science & Technology Letters*, 7(10), 714–720. <https://doi.org/10.1021/acs.estlett.0c00543>
- Chen, L., Fukuda, K., Takenaka, N., Bandow, H., & Maeda, Y. (2000). Kinetics of the gas-phase reaction of CF₃CF₂CH₂OH with OH radicals and its atmospheric lifetime. *International Journal of Chemical Kinetics*, 32(2), 73–78. [https://doi.org/10.1002/\(SICI\)1097-4601\(2000\)32:2%253C73::AID-KIN1%253E3.0.CO;2-V](https://doi.org/10.1002/(SICI)1097-4601(2000)32:2%253C73::AID-KIN1%253E3.0.CO;2-V)
- Chetverikov, S., & Hkudaigulov, G. (2021). NEW ENSIFER MORALENSIS H16 STRAIN FOR PERFLUOROOCTANE SULPHONATE DESTRUCTION. *Research Report*.
- Chetverikov, S., Hkudaygulov, G., Sharipov, D., Starikov, S., & Chetverikova, D. (2023). Biodegradation Potential of C7-C10 Perfluorocarboxylic Acids and Data from the Genome of a New Strain of *Pseudomonas mosselii* 5(3). *Toxics*, 11(12), 1001. <https://doi.org/10.3390/toxics11121001>
- Chetverikov, S. P., & Loginov, O. N. (2019). A New Ensifer adhaerens Strain M1 is Capable of Transformation of Perfluorocarboxylic Acids. *Microbiology*, 88(1), 115–117. <https://doi.org/10.1134/S0026261718060085>
- Chetverikov, S. P., & Sharipov, D. A. (2022). Biodegradation of Perfluorooctanoic Acid by *Pseudomonas plecoglossicida* Strain DD4. *Baghdad Science Journal*, 19(6(Suppl.)), 1502. <https://doi.org/10.21123/bsj.2022.6825>
- Chetverikov, S. P., Sharipov, D. A., Korshunova, T. Yu., & Loginov, O. N. (2017). Degradation of perfluorooctanyl sulfonate by strain *Pseudomonas plecoglossicida* 2.4-D. *Applied Biochemistry and Microbiology*, 53(5), 533–538. <https://doi.org/10.1134/S0003683817050027>
- Chiriac, F. L., Stoica, C., Iftode, C., Pirvu, F., Petre, V. A., Paun, I., Pascu, L. F., Vasile, G. G., & Nita-Lazar, M. (2023). Bacterial Biodegradation of Perfluorooctanoic Acid (PFOA) and Perfluorosulfonic Acid (PFOS) Using Pure *Pseudomonas* Strains. *Sustainability*, 15(18), 14000. <https://doi.org/10.3390/su151814000>
- Colosi, L. M., Pinto, R. A., Huang, Q., & Weber Jr., W. J. (2009). Peroxidase-mediated degradation of perfluorooctanoic acid. *Environmental Toxicology and Chemistry*, 28(2), 264–271. Scopus. <https://doi.org/10.1897/08-282.1>
- Cook, E. K., Olivares, C. I., Antell, E. H., Yi, S., Nickerson, A., Choi, Y. J., Higgins, C. P., Sedlak, D. L., & Alvarez-Cohen, L. (2022). Biological and Chemical Transformation of the Six-Carbon Polyfluoroalkyl Substance N-Dimethyl Ammonio Propyl Perfluorohexane Sulfonamide (AmPr-FHxSA). *Environ. Sci. Technol.*, 56(22), 15478.
- D'Agostino, L. A., & Mabury, S. A. (2017). Aerobic biodegradation of 2 fluorotelomer sulfonamide-based aqueous film-forming foam components produces perfluoroalkyl carboxylates. *Environmental Toxicology and Chemistry*, 36(8), 2012–2021. Scopus. <https://doi.org/10.1002/etc.3750>
- Danil, S., Sergey, S., & Sergey, C. (2022). Biodefluorination of Perfluorooctanesulphonate by Ensifer adhaerens M1. *HAYATI Journal of Biosciences*, 30(2), 313–320. <https://doi.org/10.4308/hjb.30.2.313-320>
- Dasu, K., & Lee, L. S. (2016). Aerobic biodegradation of toluene-2,4-di(8:2 fluorotelomer urethane) and hexamethylene-1,6-di(8:2 fluorotelomer urethane) monomers in soils. *Chemosphere*, 144, 2482–2488. <https://doi.org/10.1016/j.chemosphere.2015.11.021>
- Dasu, K., Liu, J., & Lee, L. S. (2012). Aerobic Soil Biodegradation of 8:2 Fluorotelomer Stearate Monoester. *Environmental Science & Technology*, 46(7), 3831–3836. <https://doi.org/10.1021/es203978g>

- De S. Furtado, R. X., Sabatini, C. A., Sakamoto, I. K., Zaiat, M., & Azevedo, E. B. (2023). Biodegradation mechanism of perfluorooctane sulfonic acid (PFOS) in domestic sewage: Specific methanogenic activity, molecular biology, and ecotoxicological aspects. *Journal of Water Process Engineering*, 51, 103453. <https://doi.org/10.1016/j.jwpe.2022.103453>
- D'Eon, J. C., Hurley, M. D., Wallington, T. J., & Mabury, S. A. (2006). Atmospheric Chemistry of *N*-methyl Perfluorobutane Sulfonamidoethanol, C₄F₉SO₂N(CH₃)CH₂CH₂OH: Kinetics and Mechanism of Reaction with OH. *Environmental Science & Technology*, 40(6), 1862–1868. <https://doi.org/10.1021/es0520767>
- D'Eon, J. C., & Mabury, S. A. (2007). Production of perfluorinated carboxylic acids (PFCAs) from the biotransformation of polyfluoroalkyl phosphate surfactants (PAPS): Exploring routes of human contamination. *Environmental Science and Technology*, 41(13), 4799–4805. Scopus. <https://doi.org/10.1021/es070126x>
- D'Eon, J. C., & Mabury, S. A. (2011). Exploring Indirect Sources of Human Exposure to Perfluoroalkyl Carboxylates (PFCAs): Evaluating Uptake, Elimination, and Biotransformation of Polyfluoroalkyl Phosphate Esters (PAPs) in the Rat. *Environmental Health Perspectives*, 119(3), 344–350. <https://doi.org/10.1289/ehp.1002409>
- Ding, R., Wu, Y., Yang, F., Xiao, X., Li, Y., Tian, X., & Zhao, F. (2021). Degradation of low-concentration perfluorooctanoic acid via a microbial-based synergistic method: Assessment of the feasibility and functional microorganisms. *Journal of Hazardous Materials*, 416. Scopus. <https://doi.org/10.1016/j.jhazmat.2021.125857>
- Dinglasan, M. J. A., Ye, Y., Edwards, E. A., & Mabury, S. A. (2004). Fluorotelomer Alcohol Biodegradation Yields Poly- and Perfluorinated Acids. *Environmental Science & Technology*, 38(10), 2857–2864. <https://doi.org/10.1021/es0350177>
- Dinglasan-Panlilio, M. J. (2008). *Biodegradation of fluorinated telomer compounds as a source of perfluorinated acid formation in the environment*. <http://hdl.handle.net/1807/122808>
- Dong, S., Yan, P.-F., Liu, C., Manz, K. E., Mezzari, M. P., Abriola, L. M., Pennell, K. D., & Cápiro, N. L. (2023). Assessing aerobic biotransformation of 8:2 fluorotelomer alcohol in aqueous film-forming foam (AFFF)-impacted soils: Pathways and microbial community dynamics. *Journal of Hazardous Materials*, 446, 130629. <https://doi.org/10.1016/j.jhazmat.2022.130629>
- Dong, S., Yan, P.-F., Manz, K. E., Abriola, L. M., Pennell, K. D., & Cápiro, N. L. (2024). Fate and Transformation of 15 Classes of Per- and Polyfluoroalkyl Substances in Aqueous Film-Forming Foam (AFFF)-Amended Soil Microcosms. *Environmental Science & Technology*, 58(51), 22777–22789. <https://doi.org/10.1021/acs.est.4c08665>
- Dreyer, A., Weinberg, I., Temme, C., & Ebinghaus, R. (2009). Polyfluorinated compounds in the atmosphere of the Atlantic and Southern Oceans: Evidence for a global distribution. *Environmental Science and Technology*, 43(17), 6507–6514. Scopus. <https://doi.org/10.1021/es9010465>
- ECHA. (2025). *Draft Annex B to the Background Document to the Opinion on the Annex XV dossier proposing restrictions on Per- and polyfluoroalkyl substances (PFASs)*.
- EFSA. (2014). Reasoned opinion on the setting of MRLs for saflufenacil in various crops, considering the risk related to the metabolite trifluoroacetic acid (TFA). *EFSA Journal*, 12(2). <https://doi.org/10.2903/j.efsa.2014.3585>
- EFSA, Abdourahime, H., Anastassiadou, M., Brancato, A., Brocca, D., Carrasco Cabrera, L., De Lentdecker, C., Ferreira, L., Greco, L., Jarrah, S., Kardassi, D., Leuschner, R., Lostia, A., Lythgo, C., Medina, P., Miron, I., Molnar, T., Nave, S., Pedersen, R., ... Villamar-Bouza, L. (2019). Review of the existing maximum residue levels for fluometuron according to Article 12 of Regulation (EC) No 396/2005. *EFSA Journal*, 17(1). <https://doi.org/10.2903/j.efsa.2019.5560>
- EFSA, Anastassiadou, M., Bernasconi, G., Brancato, A., Carrasco Cabrera, L., Ferreira, L., Greco, L., Jarrah, S., Kazocina, A., Leuschner, R., Magrans, J. O., Miron, I., Nave, S., Pedersen, R., Reich, H., Rojas, A., Sacchi, A., Santos, M., Scarlato, A. P., ... Verani, A. (2021). Modification of the existing maximum residue levels for cyflumetofen in various crops. *EFSA Journal*, 19(2). <https://doi.org/10.2903/j.efsa.2021.6373>

- EFSA, Arena, M., Auteri, D., Barmaz, S., Bellisai, G., Brancato, A., Brocca, D., Bura, L., Byers, H., Chiusolo, A., Court Marques, D., Crivellente, F., De Lentdecker, C., De Maglie, M., Egsmose, M., Erdos, Z., Fait, G., Ferreira, L., Goumenou, M., ... Villamar-Bouza, L. (2017). Peer review of the pesticide risk assessment of the active substance trifloxystrobin. *EFSA Journal*, 15(10). <https://doi.org/10.2903/j.efsa.2017.4989>
- EFSA, Brancato, A., Brocca, D., De Lentdecker, C., Erdos, Z., Ferreira, L., Greco, L., Jarrah, S., Kardassi, D., Leuschner, R., Lythgo, C., Medina, P., Miron, I., Molnar, T., Nougadere, A., Pedersen, R., Reich, H., Sacchi, A., Santos, M., ... Villamar-Bouza, L. (2017). Modification of the existing maximum residue levels for fluazinam in onions, shallots and garlic. *EFSA Journal*, 15(7). <https://doi.org/10.2903/j.efsa.2017.4904>
- Ellis, D. A., Hanson, M. L., Sibley, P. K., Shahid, T., Fineberg, N. A., Solomon, K. R., Muir, D. C. G., & Mabury, S. A. (2001). The fate and persistence of trifluoroacetic and chloroacetic acids in pond waters. *Chemosphere*, 42(3), 309–318. [https://doi.org/10.1016/S0045-6535\(00\)00066-7](https://doi.org/10.1016/S0045-6535(00)00066-7)
- Ellis, D. A., & Mabury, S. A. (2000). The Aqueous Photolysis of TFM and Related Trifluoromethylphenols. An Alternate Source of Trifluoroacetic Acid in the Environment. *Environmental Science & Technology*, 34(4), 632–637. <https://doi.org/10.1021/es990422c>
- Ellis, D. A., Martin, J. W., De Silva, A. O., Mabury, S. A., Hurley, M. D., Sulbaek Andersen, M. P., & Wallington, T. J. (2004). Degradation of fluorotelomer alcohols: A likely atmospheric source of perfluorinated carboxylic acids. *Environmental Science and Technology*, 38(12), 3316–3321. Scopus. <https://doi.org/10.1021/es049860w>
- Ellis, D. A., Martin, J. W., Mabury, S. A., Hurley, M. D., Sulbaek Andersen, M. P., & Wallington, T. J. (2003). Atmospheric Lifetime of Fluorotelomer Alcohols. *Environmental Science & Technology*, 37(17), 3816–3820. <https://doi.org/10.1021/es034136j>
- Evich, M. G., Davis, M., Weber, E. J., Tebes-Stevens, C., Acrey, B., Henderson, W. M., Goodrow, S., Bergman, E., & Washington, J. W. (2022). Environmental Fate of Cl-PFPEACs: Predicting the Formation of PFAS Transformation Products in New Jersey Soils. *Environmental Science & Technology*, 56(12), 7779–7788. <https://doi.org/10.1021/acs.est.1c06126>
- Fang, B., Zhang, Y., Chen, H., Qiao, B., Yu, H., Zhao, M., Gao, M., Li, X., Yao, Y., Zhu, L., & Sun, H. (2024). Stability and Biotransformation of 6:2 Fluorotelomer Sulfonic Acid, Sulfonamide Amine Oxide, and Sulfonamide Alkylbetaine in Aerobic Sludge. *Environ. Sci. Technol.*, 58, 2446.
- Folkerson, A. P., & Mabury, S. A. (2024). A Comparative Biodegradation Study to Assess the Ultimate Fate of Novel Highly Functionalized Hydrofluoroether Alcohols in Wastewater Treatment Plant Microcosms and Surface Waters. *Environmental Toxicology and Chemistry*, 43(11), 2297–2305. <https://doi.org/10.1002/etc.5815>
- Frömel, T., & Knepper, T. P. (2010). Biodegradation of fluorinated alkyl substances. *Reviews of Environmental Contamination and Toxicology*, 208, 161–177. Scopus. https://doi.org/10.1007/978-1-4419-6880-7_3
- Frömel, T., & Knepper, T. P. (2015). Aerobic biotransformation studies of two trifluoromethoxy-substituted aliphatic alcohols and a novel fluorinated C3-based building block. *Journal of Fluorine Chemistry*, 177, 80–89. <https://doi.org/10.1016/j.jfluchem.2015.06.015>
- Gaillard, J., Veyrand, B., Thomas, M., Dauchy, X., Boiteux, V., Marchand, P., Le Bizec, B., Banas, D., & Feidt, C. (2017). Tissue Uptake, Distribution, and Elimination of Perfluoroalkyl Substances in Juvenile Perch through Perfluorooctane Sulfonamidoethanol Based Phosphate Diester Dietary Exposure. *Environmental Science & Technology*, 51(13), 7658–7666. <https://doi.org/10.1021/acs.est.6b05598>
- Gao, Z., Yi, S., Xue, M., Zhu, K., Yang, R., Wang, T., Sun, H., & Zhu, L. (2023). Microbial biotransformation mechanisms of PFPIAs in soil unveiled by metagenomic analysis. *Journal of Hazardous Materials*, 448, 130896. <https://doi.org/10.1016/j.jhazmat.2023.130896>
- García, P. L., Rivela, C. B., Gibilisco, R. G., Salgado, S., Wiesen, P., Teruel, M. A., & Blanco, M. B. (2020). Degradation of a series of fluorinated acrylates and methacrylates initiated by OH radicals at different temperatures. *RSC Advances*, 10(8), 4264–4273. <https://doi.org/10.1039/C9RA08034A>

- Gauthier, S. A., & Mabury, S. A. (2005). Aqueous photolysis of 8:2 fluorotelomer alcohol. *Environmental Toxicology and Chemistry*, 24(8), 1837–1846. <https://doi.org/10.1897/04-591R.1>
- Gomis, M. I., Wang, Z., Scheringer, M., & Cousins, I. T. (2015). A modeling assessment of the physicochemical properties and environmental fate of emerging and novel per- and polyfluoroalkyl substances. *Science of The Total Environment*, 505, 981–991. <https://doi.org/10.1016/j.scitotenv.2014.10.062>
- Gorito, A. M., Ribeiro, A. R., Gomes, C. R., Almeida, C. M. R., & Silva, A. M. T. (2018). Constructed wetland microcosms for the removal of organic micropollutants from freshwater aquaculture effluents. *Science of the Total Environment*, 644, 1171–1180. Scopus. <https://doi.org/10.1016/j.scitotenv.2018.06.371>
- Guo, Z., Attar, A. A., Qiqige, Q., Lundgren, R. J., & Joudan, S. (2025). Photochemical Formation of Trifluoroacetic Acid: Mechanistic Insights into a Fluoxetine-Related Aryl-CF₃ Compound. *Environmental Science & Technology*, 59(2), 1367–1377. <https://doi.org/10.1021/acs.est.4c10777>
- Guo, Z., Tremblay, G. W., Chen, J., & Joudan, S. (2025). Spontaneous aqueous defluorination of trifluoromethylphenols: Substituent effects and revisiting the mechanism. *Environmental Science: Processes & Impacts*, 27(7), 1852–1863. <https://doi.org/10.1039/D4EM00739E>
- Hamid, H., Li, L. Y., & Grace, J. R. (2020a). Aerobic biotransformation of fluorotelomer compounds in landfill leachate-sediment. *Science of The Total Environment*, 713, 136547. <https://doi.org/10.1016/j.scitotenv.2020.136547>
- Hamid, H., Li, L. Y., & Grace, J. R. (2020b). Effect of substrate concentrations on aerobic biotransformation of 6:2 fluorotelomer sulfonate (6:2 FTS) in landfill leachate. *Chemosphere*, 261. Scopus. <https://doi.org/10.1016/j.chemosphere.2020.128108>
- Hamid, H., Li, L. Y., & Grace, J. R. (2020c). Formation of perfluorocarboxylic acids from 6:2 fluorotelomer sulfonate (6:2 FTS) in landfill leachate: Role of microbial communities. *Environmental Pollution*, 259. Scopus. <https://doi.org/10.1016/j.envpol.2019.113835>
- Hansen, C., Campbell, J., & Kable, S. (2021). *Photodissociation of CF₃CHO provides a new source of CHF₃ (HFC-23) in the atmosphere: Implications for new refrigerants*. In Review. <https://doi.org/10.21203/rs.3.rs-199769/v1>
- Hao, J., Wang, P., Kang, Y., He, H., Luo, H., Kim, S., Niu, L., Jiang, H., & Ma, K. (2019). Degradation of Perfluorooctane Sulfonamide by Acinetobacter Sp. M and Its Extracellular Enzymes. *Chemistry - An Asian Journal*, 14(16), 2780–2784. Scopus. <https://doi.org/10.1002/asia.201900638>
- Harding-Marjanovic, K. C., Houtz, E. F., Yi, S., Field, J. A., Sedlak, D. L., & Alvarez-Cohen, L. (2015). Aerobic Biotransformation of Fluorotelomer Thioether Amido Sulfonate (Lodyne) in AFFF-Amended Microcosms. *Environmental Science & Technology*, 49(13), 7666–7674. <https://doi.org/10.1021/acs.est.5b01219>
- Harris, J. D., Coon, C. M., Doherty, M. E., McHugh, E. A., Warner, M. C., Walters, C. L., Orahood, O. M., Loesch, A. E., Hatfield, D. C., Sitko, J. C., Almand, E. A., & Steel, J. J. (2022). Engineering and characterization of dehalogenase enzymes from Delftia acidovorans in bioremediation of perfluorinated compounds. *Synthetic and Systems Biotechnology*, 7(2), 671–676. <https://doi.org/10.1016/j.synbio.2022.02.005>
- Henry, B. J., Carlin, J. P., Hammerschmidt, J. A., Buck, R. C., Buxton, L. W., Fiedler, H., Seed, J., & Hernandez, O. (2018). A critical review of the application of polymer of low concern and regulatory criteria to fluoropolymers. *Integrated Environmental Assessment and Management*, 14(3), 316–334. Scopus. <https://doi.org/10.1002/ieam.4035>
- Henry, B. J., & Timmer, N. (2025). Environmental fate and behavior studies of a polymeric PFAS, polytetrafluoroethylene (PTFE) – results and application to risk assessment. *Chemosphere*, 385, 144569. <https://doi.org/10.1016/j.chemosphere.2025.144569>
- Huang, S., & Jaffe, P. R. (2019). Defluorination of Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonate (PFOS) by Acidimicrobium sp. Strain A6. *Environmental Science and Technology*, 53(19), 11410–11419. Scopus. <https://doi.org/10.1021/acs.est.9b04047>

- Huang, S., Sima, M., Long, Y., Messenger, C., & Jaffé, P. R. (2022). Anaerobic degradation of perfluorooctanoic acid (PFOA) in biosolids by *Acidimicrobium* sp. Strain A6. *Journal of Hazardous Materials*, 424. Scopus. <https://doi.org/10.1016/j.jhazmat.2021.127699>
- Hurley, M. D., Ball, J. C., Wallington, T. J., Sulbaek Andersen, M. P., Nielsen, O. J., Ellis, D. A., Martin, J. W., & Mabury, S. A. (2006). Atmospheric Chemistry of n -C_xF_{2x+1}CHO ($x = 1, 2, 3, 4$): Fate of n -C_xF_{2x+1}C(O) Radicals. *The Journal of Physical Chemistry A*, 110(45), 12443–12447. <https://doi.org/10.1021/jp064029m>
- Hurley, M. D., Misner, J. A., Ball, J. C., Wallington, T. J., Ellis, D. A., Martin, J. W., Mabury, S. A., & Sulbaek Andersen, M. P. (2005). Atmospheric Chemistry of CF₃CH₂CH₂OH: Kinetics, Mechanisms and Products of Cl Atom and OH Radical Initiated Oxidation in the Presence and Absence of NO_x. *The Journal of Physical Chemistry A*, 109(43), 9816–9826. <https://doi.org/10.1021/jp0535902>
- Hurley, M. D., Sulbaek Andersen, M. P., Wallington, T. J., Ellis, D. A., Martin, J. W., & Mabury, S. A. (2004). Atmospheric Chemistry of Perfluorinated Carboxylic Acids: Reaction with OH Radicals and Atmospheric Lifetimes. *The Journal of Physical Chemistry A*, 108(4), 615–620. <https://doi.org/10.1021/jp036343b>
- Hurley, M. D., Wallington, T. J., Javadi, M. S., & Nielsen, O. J. (2008). Atmospheric chemistry of CF₃CFCH₂: Products and mechanisms of Cl atom and OH radical initiated oxidation. *Chemical Physics Letters*, 450(4–6), 263–267. <https://doi.org/10.1016/j.cplett.2007.11.051>
- Jackson, D. A., Wallington, T. J., & Mabury, S. A. (2013). Atmospheric oxidation of polyfluorinated amides: Historical source of perfluorinated carboxylic acids to the environment. *Environmental Science and Technology*, 47(9), 4317–4324. Scopus. <https://doi.org/10.1021/es400617v>
- Jackson, D. A., Young, C. J., Hurley, M. D., Wallington, T. J., & Mabury, S. A. (2011). Atmospheric Degradation of Perfluoro-2-methyl-3-pentanone: Photolysis, Hydrolysis and Hydration. *Environmental Science & Technology*, 45(19), 8030–8036. <https://doi.org/10.1021/es104362g>
- Javed, H., Ansell, G. K., Adamsky, F., Malvasi, M., Mulvaney, P. K., Guarda, P. A., Moffett, R. H., Genna, S., Johnston, S., & Liang, X. (2025). A critical review of the application of polymer of low concern and regulatory criteria to perfluoropolyethers. *Journal of Fluorine Chemistry*, 285–286, 110459. <https://doi.org/10.1016/j.jfluchem.2025.110459>
- Ji, B., Zhao, Y., Yang, Y., Li, Q., Man, Y., Dai, Y., Fu, J., Wei, T., Tai, Y., & Zhang, X. (2023). Curbing per- and polyfluoroalkyl substances (PFASs): First investigation in a constructed wetland-microbial fuel cell system. *Water Research*, 230. Scopus. <https://doi.org/10.1016/j.watres.2022.119530>
- Joudan, S., Gauthier, J., Mabury, S. A., & Young, C. J. (2024). Aqueous Leaching of Ultrashort-Chain PFAS from (Fluoro)polymers: Targeted and Nontargeted Analysis. *Environmental Science & Technology Letters*, 11(3), 237–242. <https://doi.org/10.1021/acs.estlett.3c00797>
- Joudan, S., & Mabury, S. A. (2022). Aerobic biotransformation of a novel highly functionalized polyfluoroether-based surfactant using activated sludge from a wastewater treatment plant. *Environmental Science: Processes & Impacts*, 24(1), 62–71. <https://doi.org/10.1039/D1EM00358E>
- Joudan, S., Orlando, J. J., Tyndall, G. S., Furlani, T. C., Young, C. J., & Mabury, S. A. (2021). Atmospheric Fate of a New Polyfluoroalkyl Building Block, C₃F₇OCHF₂SCH₂CH₂OH. *Environmental Science & Technology*, 56(10), 6027–6035. <https://doi.org/10.1021/acs.est.0c07584>
- Joudan, S., Yeung, L. W. Y., & Mabury, S. A. (2017). Biological Cleavage of the C–P Bond in Perfluoroalkyl Phosphinic Acids in Male Sprague-Dawley Rats and the Formation of Persistent and Reactive Metabolites. *Environmental Health Perspectives*, 125(11), 117001. <https://doi.org/10.1289/EHP1841>
- Just, H., Göckener, B., Lämmer, R., Wiedemann-Krantz, L., Stahl, T., Breuer, J., Gassmann, M., Weidemann, E., Bücking, M., & Kowalczyk, J. (2022). Degradation and Plant Transfer Rates

- of Seven Fluorotelomer Precursors to Perfluoroalkyl Acids and F-53B in a Soil-Plant System with Maize (*Zea mays* L.). *Journal of Agricultural and Food Chemistry*, 70(29), 8920–8930. Scopus. <https://doi.org/10.1021/acs.jafc.1c06838>
- Kang, Y., Guo, Z., Ma, H., Wu, H., & Zhang, J. (2023). Enhanced Removal of Perfluorooctanoic Acid (PFOA) and Perfluorooctane Sulfonate (PFOS) in Constructed Wetlands: Iron Cycling and Microbial Mechanisms. *ACS ES and T Water*, 3(2), 287–297. Scopus. <https://doi.org/10.1021/acsestwater.2c00250>
- Kelly, T., Bossoutrot, V., Magneron, I., Wirtz, K., Treacy, J., Mellouki, A., Sidebottom, H., & Le Bras, G. (2005). A Kinetic and Mechanistic Study of the Reactions of OH Radicals and Cl Atoms with 3,3,3-Trifluoropropanol under Atmospheric Conditions. *The Journal of Physical Chemistry A*, 109(2), 347–355. <https://doi.org/10.1021/jp0467402>
- Keränen, J., Ahkola, H., Knuutinen, J., Herve, S., Reinikainen, M., & Koistinen, J. (2013). Formation of PFOA from 8:2 FTOH in closed-bottle experiments with brackish water. *Environmental Science and Pollution Research*, 20(11), 8001–8012. <https://doi.org/10.1007/s11356-013-1975-4>
- Key, B. D., Howell, R. D., & Criddle, C. S. (1998). Defluorination of Organofluorine Sulfur Compounds by *Pseudomonas* Sp. Strain D2. *Environmental Science & Technology*, 32(15), 2283–2287. <https://doi.org/10.1021/es9800129>
- Kim, B. R., Suidan, M. T., Wallington, T. J., & Du, X. (2000). Biodegradability of trifluoroacetic acid. *Environmental Engineering Science*, 17(6), 337–342. Scopus. <https://doi.org/10.1089/ees.2000.17.337>
- Kim, M. H., Wang, N., & Chu, K. H. (2014). 6:2 Fluorotelomer alcohol (6:2 FTOH) biodegradation by multiple microbial species under different physiological conditions. *Applied Microbiology and Biotechnology*, 98(4), 1831–1840. <https://doi.org/10.1007/s00253-013-5131-3>
- Kim, M. H., Wang, N., McDonald, T., & Chu, K. (2012). Biodefluorination and biotransformation of fluorotelomer alcohols by two alkane-degrading *Pseudomonas* strains. *Biotechnology and Bioengineering*, 109(12), 3041–3048. <https://doi.org/10.1002/bit.24561>
- Kwon, B. G., Lim, H.-J., Na, S.-H., Choi, B.-I., Shin, D.-S., & Chung, S.-Y. (2014). Biodegradation of perfluorooctanesulfonate (PFOS) as an emerging contaminant. *Chemosphere*, 109, 221–225. <https://doi.org/10.1016/j.chemosphere.2014.01.072>
- Lange, C. C. (2000). *The Aerobic Biodegradation of N-EtFOSE Alcohol by the Microbial Activity Present in Municipal Wastewater Treatment Sludge*.
- Lange, C. C. (2018). Anaerobic biotransformation of N-methyl perfluorobutanesulfonamido ethanol and N-ethyl perfluorooctanesulfonamido ethanol. *Environmental Toxicology and Chemistry*, 37(3), 768–779. Scopus. <https://doi.org/10.1002/etc.4014>
- Lee, H., De Silva, A. O., & Mabury, S. A. (2012). Dietary Bioaccumulation of Perfluorophosphonates and Perfluorophosphinates in Juvenile Rainbow Trout: Evidence of Metabolism of Perfluorophosphinates. *Environmental Science & Technology*, 46(6), 3489–3497. <https://doi.org/10.1021/es204533m>
- Lee, H., Deon, J., & Mabury, S. A. (2010). Biodegradation of polyfluoroalkyl phosphates as a source of perfluorinated acids to the environment. *Environmental Science and Technology*, 44(9), 3305–3310. Scopus. <https://doi.org/10.1021/es9028183>
- Lee, H., Tevlin, A. G., Mabury, S. A., & Mabury, S. A. (2013). Fate of Polyfluoroalkyl Phosphate Diesters and Their Metabolites in Biosolids-Applied Soil: Biodegradation and Plant Uptake in Greenhouse and Field Experiments. *Environmental Science & Technology*, 48(1), 340–349. <https://doi.org/10.1021/es403949z>
- Lewis, M., Kim, M. H., Liu, E. J., Wang, N., & Chu, K.-H. (2016). Biotransformation of 6:2 Polyfluoroalkyl Phosphates (6:2 PAPs): Effects of Degradative Bacteria and Co-Substrates. *J. Hazard. Mater.*, 320, 479.
- Lewis, M., Kim, M.-H., Wang, N., & Chu, K.-H. (2016). Engineering artificial communities for enhanced FTOH degradation. *Science of The Total Environment*, 572, 935–942. <https://doi.org/10.1016/j.scitotenv.2016.07.223>

- Li, F., Su, Q., Zhou, Z., Liao, X., Zou, J., Yuan, B., & Sun, W. (2018). Anaerobic biodegradation of 8:2 fluorotelomer alcohol in anaerobic activated sludge: Metabolic products and pathways. *Chemosphere*, 200, 124–132. <https://doi.org/10.1016/j.chemosphere.2018.02.065>
- Li, R., Munoz, G., Liu, Y., Sauvé, S., Ghoshal, S., & Liu, J. (2019). Transformation of novel polyfluoroalkyl substances (PFASs) as co-contaminants during biopile remediation of petroleum hydrocarbons. *Journal of Hazardous Materials*, 362, 140–147. <https://doi.org/10.1016/j.jhazmat.2018.09.021>
- Li, Y., Bräunig, J., Angelica, G. C., Thai, P. K., Mueller, J. F., & Yuan, Z. (2021). Formation and partitioning behaviour of perfluoroalkyl acids (PFAAs) in waste activated sludge during anaerobic digestion. *Water Research*, 189, 116583. <https://doi.org/10.1016/j.watres.2020.116583>
- Liou, J. S.-C., Szostek, B., DeRito, C. M., & Madsen, E. L. (2010). Investigating the biodegradability of perfluorooctanoic acid. *Chemosphere*, 80(2), 176–183. Scopus. <https://doi.org/10.1016/j.chemosphere.2010.03.009>
- Liu, C., & Liu, J. (2016). Aerobic biotransformation of polyfluoroalkyl phosphate esters (PAPs) in soil. *Environmental Pollution*, 212, 230–237. <https://doi.org/10.1016/j.envpol.2016.01.069>
- Liu, F., Guan, X., & Xiao, F. (2022). Photodegradation of per- and polyfluoroalkyl substances in water: A review of fundamentals and applications. *Journal of Hazardous Materials*, 439, 129580. <https://doi.org/10.1016/j.jhazmat.2022.129580>
- Liu, J., Lee, L. S., Nies, L. F., Nakatsu, C. H., & Turco, R. F. (2007). Biotransformation of 8:2 Fluorotelomer Alcohol in Soil and by Soil Bacteria Isolates. *Environmental Science & Technology*, 41(23), 8024–8030. <https://doi.org/10.1021/es0708722>
- Liu, J., Zhong, G., Li, W., & Mejia Avendaño, S. (2019). Isomer-Specific Biotransformation of Perfluoroalkyl Sulfonamide Compounds in Aerobic Soil. *Sci. Total Environ.*, 651, 766.
- Liu, M., Dong, F., Yi, S., Zhu, Y., Zhou, J., Sun, B., Shan, G., Feng, J., & Zhu, L. (2020). Probing Mechanisms for the Tissue-Specific Distribution and Biotransformation of Perfluoroalkyl Phosphonic Acids in Common Carp (*Cyprinus carpio*). *Environmental Science & Technology*, 54(8), 4932–4941. <https://doi.org/10.1021/acs.est.0c00359>
- Liu, M., Munoz, G., Vo Duy, S., Sauvé, S., & Liu, J. (2021). Stability of nitrogen-containing polyfluoroalkyl substances in aerobic soils. *Environ. Sci. Technol.*, 55(8), 4698.
- Llorca, M., Farré, M., Sánchez-Melsió, A., Villagrasa, M., Knepper, T. P., & Barceló, D. (2018). Perfluoroalkyl phosphonic acids adsorption behaviour and removal by wastewater organisms. *Science of The Total Environment*, 636, 273–281. <https://doi.org/10.1016/j.scitotenv.2018.04.271>
- Lorah, M. M., He, K., Blaney, L., Akob, D. M., Harris, C., Tokranov, A., Hopkins, Z., & Shedd, B. P. (2024). Anaerobic biodegradation of perfluorooctane sulfonate (PFOS) and microbial community composition in soil amended with a dechlorinating culture and chlorinated solvents. *Science of The Total Environment*, 932, 172996. <https://doi.org/10.1016/j.scitotenv.2024.172996>
- Lott, D. J., Robey, N. M., Fonseca, R., Bowden, J. A., & Townsend, T. G. (2023). Behavior of Per- and polyfluoroalkyl substances (PFAS) in Pilot-Scale vertical flow constructed wetlands treating landfill leachate. *Waste Management*, 161, 187–192. Scopus. <https://doi.org/10.1016/j.wasman.2023.03.001>
- Luo, Q., Liang, S., & Huang, Q. (2018). Laccase induced degradation of perfluorooctanoic acid in a soil slurry. *Journal of Hazardous Materials*, 359, 241–247. <https://doi.org/10.1016/j.jhazmat.2018.07.048>
- Luo, Q., Lu, J., Zhang, H., Wang, Z., Feng, M., Chiang, S.-Y. D., Woodward, D., & Huang, Q. (2015). Laccase-Catalyzed Degradation of Perfluorooctanoic Acid. *Environmental Science & Technology Letters*, 2(7), 198–203. <https://doi.org/10.1021/acs.estlett.5b00119>
- Luo, Q., Wang, Z., Feng, M., Chiang, D., Woodward, D., Liang, S., Lu, J., & Huang, Q. (2017). Factors controlling the rate of perfluorooctanoic acid degradation in laccase-mediator systems: The impact of metal ions. *Environmental Pollution*, 224, 649–657. <https://doi.org/10.1016/j.envpol.2017.02.050>

- Lyu, X.-J., Li, W.-W., Lam, P. K. S., & Yu, H.-Q. (2015). Insights into perfluorooctane sulfonate photodegradation in a catalyst-free aqueous solution. *Scientific Reports*, 5(1), 9353. <https://doi.org/10.1038/srep09353>
- Ma, D., & Olivares, C. I. (2025). Perfluoroalkane Sulfonamides and Derivatives, a Different Class of PFAS: Sorption and Microbial Biotransformation Insights. *Environmental Science & Technology*, 59(22), 10734–10749. <https://doi.org/10.1021/acs.est.5c00906>
- Ma, H., Kang, Y., Li, M., Dong, J., Wang, Y., Xiao, J., & Guo, Z. (2023). Enhancement of perfluorooctanoic acid and perfluorooctane sulphonic acid removal in constructed wetland using iron mineral: Performance and mechanisms. *Journal of Hazardous Materials*, 447, 130819. <https://doi.org/10.1016/j.jhazmat.2023.130819>
- Ma, H., Zhang, L., Wang, Y., Kong, G., Yu, X., Guo, Z., Kang, Y., Kuang, S., & Zhang, J. (2024). Strengthening effect of different iron minerals for perfluorooctanoic acid and perfluorooctane sulphonic acid removal in constructed wetlands: Mechanisms of electron transfer and microbial effect. *Chemical Engineering Journal*, 486. Scopus. <https://doi.org/10.1016/j.cej.2024.150199>
- Madronich, S., Sulzberger, B., Longstreth, J. D., Schikowski, T., Andersen, M. P. S., Solomon, K. R., & Wilson, S. R. (2023). Changes in tropospheric air quality related to the protection of stratospheric ozone in a changing climate. *Photochemical & Photobiological Sciences*, 22(5), 1129–1176. <https://doi.org/10.1007/s43630-023-00369-6>
- Mahendra, S., Rome, L. H., Kickhoefer, V. A., & Wang, M. (2016). Bioaugmentation with Vaults: Novel In Situ Remediation Strategy for Transformation of Perfluoroalkyl Compounds. *SERDP Project ER-2422*. Scopus. <https://www.scopus.com/inward/record.uri?eid=2-s2.0-85207223690&partnerID=40&md5=979019e468294cbc05c1bfd3b12a0976>
- Marchetto, F., Roverso, M., Righetti, D., Bogialli, S., Filippini, F., Bergantino, E., & Sforza, E. (2021). Bioremediation of Per- and Poly-Fluoroalkyl Substances (PFAS) by *Synechocystis* sp. PCC 6803: A Chassis for a Synthetic Biology Approach. *Life*, 11(12). Scopus. <https://doi.org/10.3390/life11121300>
- Martin, J. W., Ellis, D. A., Mabury, S. A., Hurley, M. D., & Wallington, T. J. (2006). Atmospheric chemistry of perfluoroalkanesulfonamides: Kinetic and product studies of the OH radical and Cl atom initiated oxidation of N-ethyl perfluorobutanesulfonamide. *Environmental Science and Technology*, 40(3), 864–872. Scopus. <https://doi.org/10.1021/es051362f>
- Mashino, M., Ninomiya, Y., Kawasaki, M., Wallington, T. J., & Hurley, M. D. (2000). Atmospheric Chemistry of CF₃CF=CF₂: Kinetics and Mechanism of Its Reactions with OH Radicals, Cl Atoms, and Ozone. *The Journal of Physical Chemistry A*, 104(31), 7255–7260. <https://doi.org/10.1021/jp000498r>
- McGillen, M. R., Bernard, F., Fleming, E. L., & Burkholder, J. B. (2015). HCFC-133A (CF₃CH₂Cl): OH rate coefficient, UV and infrared absorption spectra, and atmospheric implications. *Geophysical Research Letters*, 42(14), 6098–6105. <https://doi.org/10.1002/2015GL064939>
- McGillen, M. R., Fried, Z. T. P., Khan, M. A. H., Kuwata, K. T., Martin, C. M., O'Doherty, S., Pecere, F., Shallcross, D. E., Stanley, K. M., & Zhang, K. (2023). Ozonolysis can produce long-lived greenhouse gases from commercial refrigerants. *Proceedings of the National Academy of Sciences*, 120(51), e2312714120. <https://doi.org/10.1073/pnas.2312714120>
- Mejia Avendaño, S., & Liu, J. (2015). Production of PFOS from Aerobic Soil Biotransformation of Two Perfluoroalkyl Sulfonamide Derivatives. *Chemosphere*, 119, 1084.
- Mejia-Avendaño, S., Munoz, G., Vo Duy, S., Desrosiers, M., Benoit, P., Sauvé, S., & Liu, J. (2017). Novel Fluoroalkylated Surfactants in Soils Following Firefighting Foam Deployment During the Lac-Mégantic Railway Accident. *Environmental Science & Technology*, 51(15), 8313–8323. <https://doi.org/10.1021/acs.est.7b02028>
- Mejia-Avendaño, S., Vo Duy, S., Sauvé, S., & Liu, J. (2016). Generation of Perfluoroalkyl Acids from Aerobic Biotransformation of Quaternary Ammonium Polyfluoroalkyl Surfactants. *Environ. Sci. Technol.*, 50(18), 9923.

- Méndez, V., Holland, S., Bhardwaj, S., McDonald, J., Khan, S., O'Carroll, D., Pickford, R., Richards, S., O'Farrell, C., Coleman, N., Lee, M., & Manefield, M. J. (2022). Aerobic biotransformation of 6:2 fluorotelomer sulfonate by *Dietzia aurantiaca* J3 under sulfur-limiting conditions. *Science of The Total Environment*, 829, 154587. <https://doi.org/10.1016/j.scitotenv.2022.154587>
- Merino, N., Wang, M., Ambrocio, R., Mak, K., O'Connor, E., Gao, A., Hawley, E. L., Deeb, R. A., Tseng, L. Y., & Mahendra, S. (2018). Fungal biotransformation of 6:2 fluorotelomer alcohol. *Remediation Journal*, 28(2), 59–70. <https://doi.org/10.1002/rem.21550>
- Mogelberg, T. E., Feilberg, A., Giessing, A. M. B., Sehested, J., Bilde, M., Wallington, T. J., & Nielsen, O. J. (1995). Atmospheric Chemistry of HFC-236cb: Spectrokinetic Investigation of the CF₃CF₂CFHO₂ Radical, Its Reaction with NO and NO₂, and the Fate of the CF₃CF₂CFHO Radical. *The Journal of Physical Chemistry*, 99(48), 17386–17393. <https://doi.org/10.1021/j100048a013>
- Nakayama, T., Takahashi, K., Matsumi, Y., Toft, A., Sulbaek Andersen, M. P., Nielsen, O. J., Waterland, R. L., Buck, R. C., Hurley, M. D., & Wallington, T. J. (2007). Atmospheric Chemistry of CF₃CHCH₂ and C₄F₉CHCH₂: Products of the Gas-Phase Reactions with Cl Atoms and OH Radicals. *The Journal of Physical Chemistry A*, 111(5), 909–915. <https://doi.org/10.1021/jp066736l>
- Nielsen, C. J. (2014). *Potential PFOA Precursors – Literature study and theoretical assessment of abiotic degradation pathways leading to PFOA* (M 231/2014; p. 46). Norwegian Environment Agency (Miljødirektoratet). <https://www.miljodirektoratet.no/globalassets/publikasjoner/M231/M231.pdf>
- Nielsen, O. J., Gamborg, E., Sehested, J., Wallington, T. J., & Hurley, M. D. (1994). Atmospheric Chemistry of HFC-143a: Spectrokinetic Investigation of the CF₃CH₂O₂.bul. Radical, Its Reactions with NO and NO₂, and the Fate of CF₃CH₂O. *The Journal of Physical Chemistry*, 98(38), 9518–9525. <https://doi.org/10.1021/j100089a026>
- Nilsson, E. J. K., Nielsen, O. J., Johnson, M. S., Hurley, M. D., & Wallington, T. J. (2009). Atmospheric chemistry of cis-CF₃CHCHF: Kinetics of reactions with OH radicals and O₃ and products of OH radical initiated oxidation. *Chemical Physics Letters*, 473(4–6), 233–237. <https://doi.org/10.1016/j.cplett.2009.03.076>
- Nohara, K., Toma, M., Kutsuna, S., Takeuchi, K., & Ibusuki, T. (2001). Cl Atom-Initiated Oxidation of Three Homologous Methyl Perfluoroalkyl Ethers. *Environmental Science & Technology*, 35(1), 114–120. <https://doi.org/10.1021/es000895f>
- OECD. (2021). *Reconciling Terminology of the Universe of Per- and Polyfluoroalkyl Substances: Recommendations and Practical Guidance*. OECD. <https://doi.org/10.1787/e458e796-en>
- OECD. (2024). *Synthesis report on understanding Perfluoropolyethers (PFPEs) and their life cycle*. OECD. <https://doi.org/10.1787/99ee2d3e-en>
- Orkin, V. L., Poskrebyshev, G. A., & Kurylo, M. J. (2011). Rate Constants for the Reactions between OH and Perfluorinated Alkenes. *The Journal of Physical Chemistry A*, 115(24), 6568–6574. <https://doi.org/10.1021/jp201663j>
- Papadimitriou, V. C., Papanastasiou, D. K., Stefanopoulos, V. G., Zaras, A. M., Lazarou, Y. G., & Papagiannakopoulos, P. (2007). Kinetic Study of the Reactions of Cl Atoms with CF₃CH₂CH₂OH, CF₃CF₂CH₂OH, CHF₂CF₂CH₂OH, and CF₃CHFCF₂CH₂OH. *The Journal of Physical Chemistry A*, 111(45), 11608–11617. <https://doi.org/10.1021/jp074728e>
- Park, J., Huang, S., Koel, B. E., & Jaffé, P. R. (2023). Enhanced Feammox Activity and Perfluorooctanoic Acid (PFOA) Degradation by Acidimicrobium Sp. Strain A6 Using PAA-Coated Ferrihydrite as an Electron Acceptor. *J. Hazard. Mater.*, 459, 132039.
- Patten, K. O., Khamaganov, V. G., Orkin, V. L., Baughcum, S. L., & Wuebbles, D. J. (2011). OH reaction rate constant, IR absorption spectrum, ozone depletion potentials and global warming potentials of 2-bromo-3,3,3-trifluoropropene: BTP k(OH), IR ABSORPTION, ODP, AND GWP. *Journal of Geophysical Research: Atmospheres*, 116(D24), n/a-n/a. <https://doi.org/10.1029/2011JD016518>

- Peschka, M., Frömel, T., Fichtner, N., Hierse, W., Kleineidam, M., Montenegro, E., & Knepper, T. P. (2008). Mechanistic studies in biodegradation of the new synthesized fluorosurfactant 9-[4-(trifluoromethyl)phenoxy]nonane-1-sulfonate. *Journal of Chromatography A*, 1187(1–2), 79–86. Scopus. <https://doi.org/10.1016/j.chroma.2008.02.007>
- Pica, N. E., Funkhouser, J., Yin, Y., Zhang, Z., Ceres, D. M., Tong, T., & Blotevogel, J. (2019). Electrochemical Oxidation of Hexafluoropropylene Oxide Dimer Acid (GenX): Mechanistic Insights and Efficient Treatment Train with Nanofiltration. *Environmental Science & Technology*, 53(21), 12602–12609. <https://doi.org/10.1021/acs.est.9b03171>
- Pickard, H. M., Criscitiello, A. S., Persaud, D., Spencer, C., Muir, D. C. G., Lehnher, I., Sharp, M. J., De Silva, A. O., & Young, C. J. (2020). Ice Core Record of Persistent Short-Chain Fluorinated Alkyl Acids: Evidence of the Impact From Global Environmental Regulations. *Geophysical Research Letters*, 47(10), e2020GL087535. <https://doi.org/10.1029/2020GL087535>
- Plumlee, M. H., McNeill, K., & Reinhard, M. (2009). Indirect Photolysis of Perfluorochemicals: Hydroxyl Radical-Initiated Oxidation of *N*-Ethyl Perfluorooctane Sulfonamido Acetate (*N*-EtFOSAA) and Other Perfluoroalkanesulfonamides. *Environmental Science & Technology*, 43(10), 3662–3668. <https://doi.org/10.1021/es803411w>
- Prevedouros, K., Cousins, I. T., Buck, R. C., & Korzeniowski, S. H. (2006). Sources, Fate and Transport of Perfluorocarboxylates. *Environmental Science & Technology*, 40(1), 32–44. <https://doi.org/10.1021/es0512475>
- Qian, X., Huang, J., Yan, C., Xiao, J., Li, X., Wang, L., & Wei, Z. (2023). Comparison of eco-improvement on constructed wetlands with nano zero valent iron introduction under different levels of PFOA stress: Perspectives on plant, microbe, and PFOA removal. *Science of the Total Environment*, 891. Scopus. <https://doi.org/10.1016/j.scitotenv.2023.164052>
- Qiao, W., Miao, J., Jiang, H., & Yang, Q. (2021). Degradation and effect of 6:2 fluorotelomer alcohol in aerobic composting of sludge. *Biodegradation*, 32(1), 99–112. <https://doi.org/10.1007/s10532-020-09924-9>
- Qin, S., Wu, J., Zheng, R., Du, X., & Zhang, Q. (2025). Global warming potentials of two C9-perfluoroalkenes. *Chemical Physics Letters*, 868, 142025. <https://doi.org/10.1016/j.cplett.2025.142025>
- Qingguo, J. H. (2013). *Remediation of Perfluoroalkyl Contaminated Aquifers Using an In-situ Two-layer Barrier: Laboratory Batch and Column Study*. Scopus. <https://www.scopus.com/inward/record.uri?eid=2-s2.0-85031737617&partnerID=40&md5=41ce237f189a1317e7b6f8757b2044f2>
- Qu, Y., Zhang, C., Li, F., Chen, J., & Zhou, Q. (2010). Photo-reductive defluorination of perfluorooctanoic acid in water. *Water Research*, 44(9), 2939–2947. <https://doi.org/10.1016/j.watres.2010.02.019>
- Rankin, K., Lee, H., Tseng, P. J., & Mabury, S. A. (2014). Investigating the biodegradability of a fluorotelomer-based acrylate polymer in a soil-plant microcosm by indirect and direct analysis. *Environmental Science and Technology*, 48(21), 12783–12790. Scopus. <https://doi.org/10.1021/es502986w>
- Ravishankara, A. R., Solomon, S., Turnipseed, A. A., & Warren, R. F. (1993). Atmospheric Lifetimes of Long-Lived Halogenated Species. *Science*, 259(5092), 194–199. <https://doi.org/10.1126/science.259.5092.194>
- Rayez, J., & Lesclaux, R. (2009). Mechanisms of Atmospheric Oxidation of the Alkanes. By Jack G. Calvert et al.. *Angewandte Chemie International Edition*, 48(28), 5075–5075. <https://doi.org/10.1002/anie.200901202>
- Rayne, S., & Forest, K. (2009). Perfluoroalkyl sulfonic and carboxylic acids: A critical review of physicochemical properties, levels and patterns in waters and wastewaters, and treatment methods. *Journal of Environmental Science and Health, Part A*, 44(12), 1145–1199. <https://doi.org/10.1080/10934520903139811>
- Rayne, S., & Forest, K. (2010). Modeling the hydrolysis of perfluorinated compounds containing carboxylic and phosphoric acid ester functions and sulfonamide groups. *Journal of*

- Environmental Science and Health, Part A*, 45(4), 432–446.
<https://doi.org/10.1080/10934520903538731>
- Ren, Y., Bernard, F., Daële, V., & Mellouki, A. (2019). Atmospheric Fate and Impact of Perfluorinated Butanone and Pentanone. *Environmental Science & Technology*, 53(15), 8862–8871.
<https://doi.org/10.1021/acs.est.9b02974>
- Rhoads, K. R., Janssen, E. M.-L., Luthy, R. G., & Criddle, C. S. (2008). Aerobic Biotransformation and Fate of N-Ethyl Perfluorooctane Sulfonamidoethanol (N-EtFOSE) in Activated Sludge. *Environ. Sci. Technol.*, 42(8), 2873.
- Rich, S. L., Hafner, J., Salz, M., Qanbarzadeh, M., Geng, F., Yan, L., Liu, J., Helbling, D. E., Higgins, C. P., & Fenner, K. (2025). FAIR and Effective Communication of Data on Chemical Contaminant Biotransformation in the Environment. *Environmental Science & Technology Letters*, 12(11), 1462–1470. <https://doi.org/10.1021/acs.estlett.5c00753>
- Royer, L. A., Lee, L. S., Russell, M. H., Nies, L. F., & Turco, R. F. (2015). Microbial transformation of 8:2 fluorotelomer acrylate and methacrylate in aerobic soils. *Chemosphere*, 129, 54–61.
<https://doi.org/10.1016/j.chemosphere.2014.09.077>
- Ruiz-Urigüen, M., Shuai, W., Huang, S., & Jaffé, P. R. (2022). Biodegradation of PFOA in microbial electrolysis cells by Acidimicrobiaceae sp. Strain A6. *Chemosphere*, 292. Scopus.
<https://doi.org/10.1016/j.chemosphere.2021.133506>
- Russell, M. H., Berti, W. R., Szostek, B., & Buck, R. C. (2008). Investigation of the biodegradation potential of a fluoroacrylate polymer product in aerobic soils. *Environmental Science and Technology*, 42(3), 800–807. Scopus. <https://doi.org/10.1021/es0710499>
- Russell, M. H., Berti, W. R., Szostek, B., Wang, N., & Buck, R. C. (2010). Evaluation of PFO formation from the biodegradation of a fluorotelomer-based urethane polymer product in aerobic soils. *Polymer Degradation and Stability*, 95(1), 79–85. Scopus.
<https://doi.org/10.1016/j.polymdegradstab.2009.10.004>
- Ruyle, B. J., Schultes, L., Akob, D. M., Harris, C. R., Lorah, M. M., Vojta, S., Becanova, J., McCann, S., Pickard, H. M., Pearson, A., Lohmann, R., Vecitis, C. D., & Sunderland, E. M. (2023). Nitrifying Microorganisms Linked to Biotransformation of Perfluoroalkyl Sulfonamido Precursors from Legacy Aqueous Film-Forming Foams. *Environmental Science & Technology*, 57(14), 5592–5602. <https://doi.org/10.1021/acs.est.2c07178>
- Sáez, M., De Voogt, P., & Parsons, J. R. (2008). Persistence of perfluoroalkylated substances in closed bottle tests with municipal sewage sludge. *Environmental Science and Pollution Research*, 15(6), 472–477. <https://doi.org/10.1007/s11356-008-0020-5>
- Sakai, T. T., & Santi, D. V. (1973). Hydrolysis of hydroxybenzotrifluorides and fluorinated uracil derivatives. General mechanism for carbon-fluorine bond labilization. *Journal of Medicinal Chemistry*, 16(10), 1079–1084. <https://doi.org/10.1021/jm00268a003>
- Say, D., Manning, A. J., Western, L. M., Young, D., Wisher, A., Rigby, M., Reimann, S., Vollmer, M. K., Maione, M., Arduini, J., Krummel, P. B., Mühle, J., Harth, C. M., Evans, B., Weiss, R. F., Prinn, R. G., & O'Doherty, S. (2021). Global trends and European emissions of tetrafluoromethane (CF₄), hexafluoroethane (C₂F₆) and octafluoropropane (C₃F₈). *Atmospheric Chemistry and Physics*, 21(3), 2149–2164. <https://doi.org/10.5194/acp-21-2149-2021>
- Scheurer, M., Nödler, K., Freeling, F., Janda, J., Happel, O., Riegel, M., Müller, U., Storck, F. R., Fleig, M., Lange, F. T., Brunsch, A., & Brauch, H.-J. (2017). Small, mobile, persistent: Trifluoroacetate in the water cycle – Overlooked sources, pathways, and consequences for drinking water supply. *Water Research*, 126, 460–471.
<https://doi.org/10.1016/j.watres.2017.09.045>
- Schneider, W. F., & Wallington, T. J. (1994). Thermochemistry of COF₂ and Related Compounds. *The Journal of Physical Chemistry*, 98(31), 7448–7451.
<https://doi.org/10.1021/j100082a009>
- Shaw, D. M. J., Munoz, G., Bottos, E. M., Duy, S. V., Sauv  , S., Liu, J., & Van Hamme, J. D. (2019). Degradation and defluorination of 6:2 fluorotelomer sulfonamidoalkyl betaine and 6:2 fluorotelomer sulfonate by *Gordonia* sp. Strain NB4-1Y under sulfur-limiting conditions.

- Science of The Total Environment*, 647, 690–698. <https://doi.org/10.1016/j.scitotenv.2018.08.012>
- Siegemund, G., Schwertfeger, W., Feiring, A., Smart, B., Behr, F., Vogel, H., & McKusick, B. (2000). Fluorine Compounds, Organic. In Wiley-VCH (Ed.), *Ullmann's Encyclopedia of Industrial Chemistry* (1st edn). Wiley. https://doi.org/10.1002/14356007.a11_349
- Smith, S. J., Lauria, M., Higgins, C. P., Pennell, K. D., Blotevogel, J., & Arp, H. P. H. (2024). The Need to Include a Fluorine Mass Balance in the Development of Effective Technologies for PFAS Destruction. *Environmental Science & Technology*, 58(6), 2587–2590. <https://doi.org/10.1021/acs.est.3c10617>
- Sorn, S., Hara-Yamamura, H., Vet, S., Xiao, M., Hoek, E. M. V., & Honda, R. (2023). Biological treatment of perfluorooctanesulfonic acid (PFOS) using microbial capsules of a polysulfone membrane. *Chemosphere*, 329, 138585. <https://doi.org/10.1016/j.chemosphere.2023.138585>
- Steffens, S. D., Antell, E. H., Cook, E. K., Rao, G., Britt, R. D., Sedlak, D. L., & Alvarez-Cohen, L. (2023). An Artifact of Perfluoroalkyl Acid (PFAA) Removal Attributed to Sorption Processes in a Laccase Mediator System. *Environmental Science and Technology Letters*, 10(4), 337–342. Scopus. <https://doi.org/10.1021/acs.estlett.3c00173>
- Styler, S. A., Myers, A. L., & Donaldson, D. J. (2013). Heterogeneous Photooxidation of Fluorotelomer Alcohols: A New Source of Aerosol-Phase Perfluorinated Carboxylic Acids. *Environmental Science & Technology*, 47(12), 6358–6367. <https://doi.org/10.1021/es4011509>
- Sulbaek Andersen, M. P., Kyte, M., Andersen, S. T., Nielsen, C. J., & Nielsen, O. J. (2017). Atmospheric Chemistry of (CF₃)₂CF–C≡N: A Replacement Compound for the Most Potent Industrial Greenhouse Gas, SF₆. *Environmental Science & Technology*, 51(3), 1321–1329. <https://doi.org/10.1021/acs.est.6b03758>
- Sulbaek Andersen, M. P., & Nielsen, O. J. (2022). Tropospheric photolysis of CF₃CHO. *Atmospheric Environment*, 272, 118935. <https://doi.org/10.1016/j.atmosenv.2021.118935>
- Sulbaek Andersen, M. P., Nielsen, O. J., Toft, A., Nakayama, T., Matsumi, Y., Waterland, R. L., Buck, R. C., Hurley, M. D., & Wallington, T. J. (2005). Atmospheric chemistry of C F₂+1CH CH₂ (x= 1, 2, 4, 6, and 8): Kinetics of gas-phase reactions with Cl atoms, OH radicals, and O₃. *Journal of Photochemistry and Photobiology A: Chemistry*, 176(1–3), 124–128. <https://doi.org/10.1016/j.jphotochem.2005.06.015>
- Sulbaek Andersen, M. P., Schmidt, J. A., Volkova, A., & Wuebbles, D. J. (2018). A three-dimensional model of the atmospheric chemistry of E and Z-CF₃CH=CHCl (HCFO-1233(zd) (E/Z)). *Atmospheric Environment*, 179, 250–259. <https://doi.org/10.1016/j.atmosenv.2018.02.018>
- Sun, M., Cui, J., Guo, J., Zhai, Z., Zuo, P., & Zhang, J. (2020). Fluorochemicals biodegradation as a potential source of trifluoroacetic acid (TFA) to the environment. *Chemosphere*, 254, 126894. <https://doi.org/10.1016/j.chemosphere.2020.126894>
- Sun, X., Ding, L., Zhang, Q., & Wang, W. (2013). Atmospheric oxidation mechanism of polyfluorinated sulfonamides—A quantum chemical and kinetic study. *Canadian Journal of Chemistry*, 91(6), 472–478. <https://doi.org/10.1139/cjc-2012-0486>
- Sun, Y., Liu, L., Li, M., Xu, F., & Yu, W. (2022). Theoretical evidence for the formation of perfluorocarboxylic acids from atmospheric oxidation degradation of fluorotelomer acrylates. *Environmental Science and Pollution Research*, 29(36), 55092–55104. <https://doi.org/10.1007/s11356-022-19788-6>
- Taatjes, C. A., Khan, M. A. H., Eskola, A. J., Percival, C. J., Osborn, D. L., Wallington, T. J., & Shallcross, D. E. (2019). Reaction of Perfluorooctanoic Acid with Criegee Intermediates and Implications for the Atmospheric Fate of Perfluorocarboxylic Acids. *Environmental Science & Technology*, 53(3), 1245–1251. <https://doi.org/10.1021/acs.est.8b05073>
- Tang, Z., Vogel, T. M., Wang, Q., Wei, C., Ali, M., & Song, X. (2024). Microbial defluorination of TFA, PFOA, and HFPO-DA by a native microbial consortium under anoxic conditions. *Journal of Hazardous Materials*, 465, 133217. <https://doi.org/10.1016/j.jhazmat.2023.133217>

- Taniguchi, N., Wallington, T. J., Hurley, M. D., Guschin, A. G., Molina, L. T., & Molina, M. J. (2003). Atmospheric Chemistry of $C_2F_5C(O)CF(CF_3)_2$: Photolysis and Reaction with Cl Atoms, OH Radicals, and Ozone. *The Journal of Physical Chemistry A*, 107(15), 2674–2679. <https://doi.org/10.1021/jp0220332>
- Taniyasu, S., Yamashita, N., Yamazaki, E., Petrick, G., & Kannan, K. (2013). The environmental photolysis of perfluorooctanesulfonate, perfluorooctanoate, and related fluorochemicals. *Chemosphere*, 90(5), 1686–1692. <https://doi.org/10.1016/j.chemosphere.2012.09.065>
- Tewari, S. G., Bell, J. P., Budgen, N., Platz, S., Gibbs, M., Newham, P., & Kimko, H. (2023). Pressurized metered-dose inhalers using next-generation propellant HFO-1234ze(E) deposit negligible amounts of trifluoroacetic acid in the environment. *Frontiers in Environmental Science*, 11, 1297920. <https://doi.org/10.3389/fenvs.2023.1297920>
- Tisler, S., Zindler, F., Freeling, F., Nödler, K., Toelgyesi, L., Braunbeck, T., & Zwiener, C. (2019). Transformation Products of Fluoxetine Formed by Photodegradation in Water and Biodegradation in Zebrafish Embryos (*Danio rerio*). *Environmental Science & Technology*, 53(13), 7400–7409. <https://doi.org/10.1021/acs.est.9b00789>
- Titaley, I. A. (2024). Chemical transformation, exposure assessment, and policy implications of fluorotelomer alcohol partitioning from consumer products to the indoor and outdoor environment—From production to end-of-life. *Environmental Science: Advances*, 3(10), 1364–1384. <https://doi.org/10.1039/D4VA00019F>
- Tovar, C. M., Blanco, M. B., Barnes, I., Wiesen, P., & Teruel, M. A. (2014). Gas-phase reactivity study of a series of hydrofluoroolefins (HFOs) toward OH radicals and Cl atoms at atmospheric pressure and 298 K. *Atmospheric Environment*, 88, 107–114. <https://doi.org/10.1016/j.atmosenv.2014.01.060>
- Trang, B., Li, Y., Xue, X.-S., Ateia, M., Houk, K. N., & Dichtel, W. R. (2022). Low-temperature mineralization of perfluorocarboxylic acids. *Science*, 377(6608), 839–845. <https://doi.org/10.1126/science.abm8868>
- Tsai, W. (2005). Environmental risk assessment of hydrofluoroethers (HFEs). *Journal of Hazardous Materials*, 119(1–3), 69–78. <https://doi.org/10.1016/j.jhazmat.2004.12.018>
- Tsuda, N., Honda, Y., Schaefer, E., Lian, P., Muneer, A., Blake, T. J., & Hammad, L. A. (2023). The environmental degradability of DEMNUM, a typical PFPE polymer. *Chemosphere*, 337, 139331. <https://doi.org/10.1016/j.chemosphere.2023.139331>
- UK Environment Agency. (2023). *Environmental risk evaluation report: Potassium 9-chlorohexadecafluoro-3-oxanonane-1-sulfonate [F-53B] (CAS no. 73606-19-6)*.
- UNEP. (2022). *Environmental Effects of Stratospheric Ozone Depletion, UV Radiation, and Interactions with Climate Change—2022 Assessment Report*. <https://ozone.unep.org/system/files/documents/EEAP-2022-Assessment-Report-May2023.pdf>
- USEPA. (2021). *Human Health Toxicity Values for Hexafluoropropylene Oxide Dimer Acid and Its Ammonium Salt, Also Known as "GenX Chemicals"*. https://www.epa.gov/system/files/documents/2021-10/genx-chemicals-toxicity-assessment_tech-edited_oct-21-508.pdf
- Vaalgamaa, S., Vähätalo, A. V., Perkola, N., & Huhtala, S. (2011). Photochemical reactivity of perfluorooctanoic acid (PFOA) in conditions representing surface water. *Science of The Total Environment*, 409(16), 3043–3048. <https://doi.org/10.1016/j.scitotenv.2011.04.036>
- Wallington, T. J., Hurley, M. D., Xia, J., Wuebbles, D. J., Sillman, S., Ito, A., Penner, J. E., Ellis, D. A., Martin, J., Mabury, S. A., Nielsen, O. J., & Sulbaek Andersen, M. P. (2006). Formation of $C_7F_{15}COOH$ (PFOA) and Other Perfluorocarboxylic Acids during the Atmospheric Oxidation of 8:2 Fluorotelomer Alcohol. *Environmental Science & Technology*, 40(3), 924–930. <https://doi.org/10.1021/es051858x>
- Wallington, T. J., Sulbaek Andersen, M. P., & Nielsen, O. J. (2015). Atmospheric chemistry of short-chain haloolefins: Photochemical ozone creation potentials (POCPs), global warming potentials (GWPs), and ozone depletion potentials (ODPs). *Chemosphere*, 129, 135–141. <https://doi.org/10.1016/j.chemosphere.2014.06.092>

- Wang, N., Szostek, B., Buck, R. C., Folsom, P. W., Sulecki, L. M., Capka, V., Berti, W. R., & Gannon, J. T. (2005). Fluorotelomer Alcohol Biodegradation Direct Evidence that Perfluorinated Carbon Chains Breakdown. *Environmental Science & Technology*, 39(19), 7516–7528. <https://doi.org/10.1021/es0506760>
- Wang, N., Szostek, B., Buck, R. C., Folsom, P. W., Sulecki, L. M., & Gannon, J. T. (2009). 8-2 Fluorotelomer alcohol aerobic soil biodegradation: Pathways, metabolites, and metabolite yields. *Chemosphere*, 75(8), 1089–1096. <https://doi.org/10.1016/j.chemosphere.2009.01.033>
- Wang, N., Szostek, B., Folsom, P. W., Sulecki, L. M., Capka, V., Buck, R. C., Berti, W. R., & Gannon, J. T. (2005). Aerobic Biotransformation of ¹⁴C-Labeled 8-2 Telomer B Alcohol by Activated Sludge from a Domestic Sewage Treatment Plant. *Environmental Science & Technology*, 39(2), 531–538. <https://doi.org/10.1021/es049466y>
- Wang, S., Huang, J., Yang, Y., Hui, Y., Ge, Y., Larssen, T., Yu, G., Deng, S., Wang, B., & Harman, C. (2013). First Report of a Chinese PFOS Alternative Overlooked for 30 Years: Its Toxicity, Persistence, and Presence in the Environment. *Environmental Science & Technology*, 47(18), 10163–10170. <https://doi.org/10.1021/es401525n>
- Wang, X., Lv, Q., Xu, Y. J., Fu, R., Xie, Y., Dai, C., Zhou, N., Wei, Y., Ji, X., Mao, B., Jia, S., & Liu, Z. (2025). Environmental behaviors, biotransformation, and toxicological risks of PFPIAs: A comprehensive review of an emerging PFAS. *Science of The Total Environment*, 996, 180119. <https://doi.org/10.1016/j.scitotenv.2025.180119>
- Wang, Z., Cousins, I. T., Berger, U., Hungerbühler, K., & Scheringer, M. (2016). Comparative assessment of the environmental hazards of and exposure to perfluoroalkyl phosphonic and phosphinic acids (PFPAAs and PFPIAs): Current knowledge, gaps, challenges and research needs. *Environment International*, 89–90, 235–247. <https://doi.org/10.1016/j.envint.2016.01.023>
- Wang, Z., Cousins, I. T., & Scheringer, M. (2015b). Comment on “The environmental photolysis of perfluorooctanesulfonate, perfluorooctanoate, and related fluorochemicals”. *Chemosphere*, 122, 301–303. <https://doi.org/10.1016/j.chemosphere.2014.03.066>
- Wang, Z., Cousins, I. T., Scheringer, M., Buck, R. C., & Hungerbühler, K. (2014). Global emission inventories for C4-C14 perfluoroalkyl carboxylic acid (PFCA) homologues from 1951 to 2030, Part I: Production and emissions from quantifiable sources. *Environment International*, 70, 62–75. Scopus. <https://doi.org/10.1016/j.envint.2014.04.013>
- Wang, Z., Cousins, I. T., Scheringer, M., & Hungerbuehler, K. (2015a). Hazard assessment of fluorinated alternatives to long-chain perfluoroalkyl acids (PFAAs) and their precursors: Status quo, ongoing challenges and possible solutions. *Environment International*, 75, 172–179. Scopus. <https://doi.org/10.1016/j.envint.2014.11.013>
- Washington, J. W., Ellington, J. J., Jenkins, T. M., Evans, J. J., Yoo, H., & Hafner, S. C. (2009). Degradability of an acrylate-linked, fluorotelomer polymer in soil. *Environmental Science and Technology*, 43(17), 6617–6623. Scopus. <https://doi.org/10.1021/es9002668>
- Washington, J. W., & Jenkins, T. M. (2015). Abiotic Hydrolysis of Fluorotelomer-Based Polymers as a Source of Perfluorocarboxylates at the Global Scale. *Environmental Science & Technology*, 49(24), 14129–14135. <https://doi.org/10.1021/acs.est.5b03686>
- Washington, J. W., Jenkins, T. M., Rankin, K., & Naile, J. E. (2015). Decades-Scale Degradation of Commercial, Side-Chain, Fluorotelomer-Based Polymers in Soils and Water. *Environmental Science & Technology*, 49(2), 915–923. <https://doi.org/10.1021/es504347u>
- Washington, J. W., Jenkins, T. M., & Weber, E. J. (2015). Identification of Unsaturated and 2H Polyfluorocarboxylate Homologous Series and Their Detection in Environmental Samples and as Polymer Degradation Products. *Environmental Science & Technology*, 49(22), 13256–13263. <https://doi.org/10.1021/acs.est.5b03379>
- Waterland, R. L., Hurley, M. D., Misner, J. A., Wallington, T. J., Melo, S. M. L., Strong, K., Dumoulin, R., Castera, L., Stock, N. L., & Mabury, S. A. (2005). Gas phase UV and IR absorption spectra of CF₃CH₂CH₂OH and F(CF₂CF₂)_xCH₂CH₂OH (x=2, 3, 4). *Journal of Fluorine Chemistry*, 126(9–10), 1288–1296. <https://doi.org/10.1016/j.jfluchem.2005.06.010>

- Xiao, J., Huang, J., Wang, Y., & Qian, X. (2023). The fate and behavior of perfluorooctanoic acid (PFOA) in constructed wetlands: Insights into potential removal and transformation pathway. *Science of the Total Environment*, 861. Scopus. <https://doi.org/10.1016/j.scitotenv.2022.160309>
- Yan, P.-F., Dong, S., Manz, K. E., Liu, C., Woodcock, M. J., Mezzari, M. P., Abriola, L. M., Pennell, K. D., & Cápiro, N. L. (2022). Biotransformation of 8:2 Fluorotelomer Alcohol in Soil from Aqueous Film-Forming Foams (AFFFs)-Impacted Sites under Nitrate-, Sulfate-, and Iron-Reducing Conditions. *Environmental Science & Technology*, 56(19), 13728–13739. <https://doi.org/10.1021/acs.est.2c03669>
- Yan, P.-F., Dong, S., Manz, K. E., Woodcock, M. J., Liu, C., Mezzari, M. P., Abriola, L. M., Pennell, K. D., & Cápiro, N. L. (2024). Aerobic biotransformation of 6:2 fluorotelomer sulfonate in soils from two aqueous film-forming foam (AFFF)-impacted sites. *Water Research*, 249, 120941. <https://doi.org/10.1016/j.watres.2023.120941>
- Yan, P.-F., Dong, S., Woodcock, M. J., Manz, K. E., Garza-Rubalcava, U., Abriola, L. M., Pennell, K. D., & Cápiro, N. L. (2024). Biotransformation of 6:2 fluorotelomer sulfonate and microbial community dynamics in water-saturated one-dimensional flow-through columns. *Water Research*, 252, 121146. <https://doi.org/10.1016/j.watres.2024.121146>
- Yang, S.-H., Shi, Y., Strynar, M., & Chu, K.-H. (2022). Desulfonation and Defluorination of 6:2 Fluorotelomer Sulfonic Acid (6:2 FTSA) by *Rhodococcus Jostii* RHA1: Carbon and Sulfur Sources, Enzymes, and Pathways. *J. Hazard. Mater.*, 423, 127052.
- Ye, D., Wang, Z., Qian, X., Ouyang, K., Wu, D., Tang, F., Hrynsphan, D., Savitskaya, T., & Chen, J. (2025). Biodegradation of per- and polyfluoroalkyl substances: Microbes, enzymes and their interactions. *Reviews in Environmental Science and Bio/Technology*, 24(1), 43–62. <https://doi.org/10.1007/s11157-025-09721-x>
- Yi, L. B., Chai, L. Y., Xie, Y., Peng, Q. J., & Peng, Q. Z. (2016). Isolation, identification, and degradation performance of a PFOA-degrading strain. *Genetics and Molecular Research*, 15(2). <https://doi.org/10.4238/gmr.15028043>
- Yi, S., Harding-Marjanovic, K. C., Houtz, E. F., Antell, E., Olivares, C., Nichiporuk, R. V., Iavarone, A. T., Zhuang, W.-Q., Field, J. A., Sedlak, D. L., & Alvarez-Cohen, L. (2022). Biotransformation of 6:2 Fluorotelomer Thioether Amido Sulfonate in Aqueous Film-Forming Foams under Nitrate-Reducing Conditions. *Environmental Science & Technology*, 56(15), 10646–10655. <https://doi.org/10.1021/acs.est.2c00063>
- Yi, S., Harding-Marjanovic, K. C., Houtz, E. F., Gao, Y., Lawrence, J. E., Nichiporuk, R. V., Iavarone, A. T., Zhuang, W.-Q., Hansen, M., Field, J. A., Sedlak, D. L., & Alvarez-Cohen, L. (2018). Biotransformation of AFFF Component 6:2 Fluorotelomer Thioether Amido Sulfonate Generates 6:2 Fluorotelomer Thioether Carboxylate under Sulfate-Reducing Conditions. *Environmental Science & Technology Letters*, 5(5), 283–288. <https://doi.org/10.1021/acs.estlett.8b00148>
- Yin, T., Te, S. H., Reinhard, M., Yang, Y., Chen, H., He, Y., & Gin, K. Y.-H. (2018). Biotransformation of Sulfluramid (N-ethyl perfluorooctane sulfonamide) and dynamics of associated rhizospheric microbial community in microcosms of wetland plants. *Chemosphere*, 211, 379–389. <https://doi.org/10.1016/j.chemosphere.2018.07.157>
- Yin, T., Tran, N. H., Huiting, C., He, Y., & Gin, K. Y.-H. (2019). Biotransformation of polyfluoroalkyl substances by microbial consortia from constructed wetlands under aerobic and anoxic conditions. *Chemosphere*, 233, 101–109. Scopus. <https://doi.org/10.1016/j.chemosphere.2019.05.227>
- Young, C. J., & Donaldson, D. J. (2007). Overtone-Induced Degradation of Perfluorinated Alcohols in the Atmosphere. *The Journal of Physical Chemistry A*, 111(51), 13466–13471. <https://doi.org/10.1021/jp075607h>
- Young, C. J., Hurley, M. D., Wallington, T. J., & Mabury, S. A. (2006). Atmospheric Lifetime and Global Warming Potential of a Perfluoropolyether. *Environmental Science & Technology*, 40(7), 2242–2246. <https://doi.org/10.1021/es052077z>
- Young, C. J., Hurley, M. D., Wallington, T. J., & Mabury, S. A. (2008). Atmospheric Chemistry of 4:2 Fluorotelomer Iodide (n -C₄ F₉ CH₂ CH₂ I): Kinetics and Products of Photolysis and

- Reaction with OH Radicals and Cl Atoms. *The Journal of Physical Chemistry A*, 112(51), 13542–13548. <https://doi.org/10.1021/jp807322x>
- Young, C. J., Hurley, M. D., Wallington, T. J., & Mabury, S. A. (2009). Atmospheric chemistry of perfluorobutenes (CF₃CF₂CF₃ and CF₃CF₂CF₂): Kinetics and mechanisms of reactions with OH radicals and chlorine atoms, IR spectra, global warming potentials, and oxidation to perfluorocarboxylic acids. *Atmospheric Environment*, 43(24), 3717–3724. <https://doi.org/10.1016/j.atmosenv.2009.04.025>
- Young, C. J., & Mabury, S. A. (2010). Atmospheric Perfluorinated Acid Precursors: Chemistry, Occurrence, and Impacts. In P. De Voogt (Ed.), *Reviews of Environmental Contamination and Toxicology Volume 208* (Vol. 208, pp. 1–109). Springer New York. https://doi.org/10.1007/978-1-4419-6880-7_1
- Yu, X., Nishimura, F., & Hidaka, T. (2018). Effects of microbial activity on perfluorinated carboxylic acids (PFCAs) generation during aerobic biotransformation of fluorotelomer alcohols in activated sludge. *Science of The Total Environment*, 610–611, 776–785. <https://doi.org/10.1016/j.scitotenv.2017.08.075>
- Yu, X., Takabe, Y., Yamamoto, K., Matsumura, C., & Nishimura, F. (2016). Biodegradation Property of 8:2 Fluorotelomer Alcohol (8:2 FTOH) under Aerobic/Anoxic/Anaerobic Conditions. *Journal of Water and Environment Technology*, 14(3), 177–190. <https://doi.org/10.2965/jwet.15-056>
- Yu, Y., Che, S., Ren, C., Jin, B., Tian, Z., Liu, J., & Men, Y. (2022). Microbial Defluorination of Unsaturated Per- and Polyfluorinated Carboxylic Acids under Anaerobic and Aerobic Conditions: A Structure Specificity Study. *Environ. Sci. Technol.*, 56(8), 4894.
- Yu, Y., Zhang, K., Li, Z., Ren, C., Chen, J., Lin, Y.-H., Liu, J., & Men, Y. (2020). Microbial Cleavage of C–F Bonds in Two C₆ Per- and Polyfluorinated Compounds via Reductive Defluorination. *Environ. Sci. Technol.*, 54(22), 14393.
- Zabaleta, I., Bizkarguenaga, E., Nunoo, D. B. O., Schultes, L., Leonel, J., Prieto, A., Zuloaga, O., & Benskin, J. P. (2018). Biodegradation and Uptake of the Pesticide Sulfluramid in a Soil-Carrot Mesocosm. *Environmental Science and Technology*, 52(5), 2603–2611. Scopus. <https://doi.org/10.1021/acs.est.7b03876>
- Zhang, C., Tang, T., & Knappe, D. R. U. (2023). Oxidation of Per- and Polyfluoroalkyl Ether Acids and Other Per- and Polyfluoroalkyl Substances by Sulfate and Hydroxyl Radicals: Kinetic Insights from Experiments and Models. *Environmental Science & Technology*, 57(47), 18970–18980. <https://doi.org/10.1021/acs.est.3c00947>
- Zhang, H., Wen, B., Hu, X., Wu, Y., Pan, Y., Huang, H., Liu, L., & Zhang, S. (2016). Uptake, Translocation, and Metabolism of 8:2 Fluorotelomer Alcohol in Soybean (*Glycine max* L. Merrill). *Environmental Science & Technology*, 50(24), 13309–13317. <https://doi.org/10.1021/acs.est.6b03734>
- Zhang, H., Wen, B., Huang, H., Wang, S., Cai, Z., & Zhang, S. (2020). Biotransformation of 6:2 fluorotelomer alcohol by the whole soybean (*Glycine max* L. Merrill) seedlings. *Environmental Pollution*, 257, 113513. <https://doi.org/10.1016/j.envpol.2019.113513>
- Zhang, L., Lee, L. S., Niu, J., & Liu, J. (2017). Kinetic analysis of aerobic biotransformation pathways of a perfluorooctane sulfonate (PFOS) precursor in distinctly different soils. *Environmental Pollution*, 229, 159–167. <https://doi.org/10.1016/j.envpol.2017.05.074>
- Zhang, S., Lu, X., Wang, N., & Buck, R. C. (2016). Biotransformation potential of 6:2 fluorotelomer sulfonate (6:2 FTSA) in aerobic and anaerobic sediment. *Chemosphere*, 154, 224–230. Scopus. <https://doi.org/10.1016/j.chemosphere.2016.03.062>
- Zhang, S., Merino, N., Wang, N., Ruan, T., & Lu, X. (2017). Impact of 6:2 fluorotelomer alcohol aerobic biotransformation on a sediment microbial community. *Science of the Total Environment*, 575, 1361–1368. Scopus. <https://doi.org/10.1016/j.scitotenv.2016.09.214>
- Zhang, S., Peng, H., Mu, D., Zhao, H., & Hu, J. (2018). Simultaneous determination of (N-ethyl perfluorooctanesulfonamido ethanol)-based phosphate diester and triester and their biotransformation to perfluorooctanesulfonate in freshwater sediments. *Environmental Pollution*, 234, 821–829. <https://doi.org/10.1016/j.envpol.2017.12.021>

- Zhang, S., Szostek, B., McCausland, P. K., Wolstenholme, B. W., Lu, X., Wang, N., & Buck, R. C. (2013). 6:2 and 8:2 Fluorotelomer Alcohol Anaerobic Biotransformation in Digester Sludge from a WWTP under Methanogenic Conditions. *Environmental Science & Technology*, 47(9), 4227–4235. <https://doi.org/10.1021/es4000824>
- Zhang, W., Pang, S., Lin, Z., Mishra, S., Bhatt, P., & Chen, S. (2021). Biotransformation of perfluoroalkyl acid precursors from various environmental systems: Advances and perspectives. *Environmental Pollution*, 272, 115908. <https://doi.org/10.1016/j.envpol.2020.115908>
- Zhao, J., Yang, L., Yang, X., Zhao, X., Li, M., Zhao, S., Zhu, L., & Zhan, J. (2023). Degradation of 8:2 fluorotelomer carboxylic acid (8:2 FTCA) by plants and their co-existing microorganisms. *Journal of Hazardous Materials*, 451, 131129. <https://doi.org/10.1016/j.jhazmat.2023.131129>
- Zhao, L., Folsom, P. W., Wolstenholme, B. W., Sun, H., Wang, N., & Buck, R. C. (2013). 6:2 Fluorotelomer alcohol biotransformation in an aerobic river sediment system. *Chemosphere*, 90(2), 203–209. <https://doi.org/10.1016/j.chemosphere.2012.06.035>
- Zhao, S., Liang, T., Zhou, T., Li, D., Wang, B., Zhan, J., & Liu, L. (2018). Biotransformation and responses of antioxidant enzymes in hydroponically cultured soybean and pumpkin exposed to perfluorooctane sulfonamide (FOSA). *Ecotoxicology and Environmental Safety*, 161, 669–675. Scopus. <https://doi.org/10.1016/j.ecoenv.2018.06.048>
- Zhao, S., Liang, T., Zhu, L., Yang, L., Liu, T., Fu, J., Wang, B., Zhan, J., & Liu, L. (2019). Fate of 6:2 fluorotelomer sulfonic acid in pumpkin (*Cucurbita maxima* L.) based on hydroponic culture: Uptake, translocation and biotransformation. *Environmental Pollution*, 252, 804–812. Scopus. <https://doi.org/10.1016/j.envpol.2019.06.020>
- Zhao, S., Ma, X., Fang, S., & Zhu, L. (2016). Behaviors of N-ethyl perfluorooctane sulfonamide ethanol (N-EtFOSE) in a soil-earthworm system: Transformation and bioaccumulation. *Science of The Total Environment*, 554–555, 186–191. <https://doi.org/10.1016/j.scitotenv.2016.02.180>
- Zhao, S., Wang, B., Zhong, Z., Liu, T., Liang, T., & Zhan, J. (2020). Contributions of Enzymes and Gut Microbes to Biotransformation of Perfluorooctane Sulfonamide in Earthworms (*Eisenia fetida*). *Chemosphere*, 238, 124619.
- Zhao, S., Wang, B., Zhu, L., Liang, T., Chen, M., Yang, L., Lv, J., & Liu, L. (2018). Uptake, elimination and biotransformation of N-ethyl perfluorooctane sulfonamide (N-EtFOSA) by the earthworms (*Eisenia fetida*) after in vivo and in vitro exposure. *Environmental Pollution*, 241, 19–25. <https://doi.org/10.1016/j.envpol.2018.05.046>
- Zhao, S., Zhou, T., Wang, B., Zhu, L., Chen, M., Li, D., & Yang, L. (2018). Different biotransformation behaviors of perfluorooctane sulfonamide in wheat (*Triticum aestivum* L.) from earthworms (*Eisenia fetida*). *Journal of Hazardous Materials*, 346, 191–198. <https://doi.org/10.1016/j.jhazmat.2017.12.018>
- Zhao, S., & Zhu, L. (2017). Uptake and metabolism of 10:2 fluorotelomer alcohol in soil-earthworm (*Eisenia fetida*) and soil-wheat (*Triticum aestivum* L.) systems. *Environmental Pollution*, 220, 124–131. <https://doi.org/10.1016/j.envpol.2016.09.030>
- Zhao, X., Bhushan, B., & Kajdas, C. (2000). Lubrication studies of head-disk interfaces in a controlled environment Part 2: Degradation mechanisms of perfluoropolyether lubricants. *Proceedings of the Institution of Mechanical Engineers, Part J: Journal of Engineering Tribology*, 214(6), 547–559. <https://doi.org/10.1243/1350650001543412>
- Zhou, J., Yang, Z., Liu, Q., Liu, Y., Liu, M., Wang, T., & Zhu, L. (2020). Insights into Uptake, Translocation, and Transformation Mechanisms of Perfluorophosphinates and Perfluorophosphonates in Wheat (*Triticum aestivum* L.). *Environmental Science & Technology*, 54(1), 276–285. <https://doi.org/10.1021/acs.est.9b05656>
- Zhou, L., Li, W., Zhang, J., & Mao, H. (2023). Removal of perfluorooctanoic acid (PFOA) in the liquid culture of *Phanerochaete chrysosporium*. *Chemosphere*, 345. Scopus. <https://doi.org/10.1016/j.chemosphere.2023.140427>

- Zhu, H., Xia, Y., Zhang, Y., Kang, Y., Ding, Y., Chen, R., & Feng, H. (2024). Distribution characteristics and transformation mechanism of per- and polyfluoroalkyl substances in drinking water sources: A review. *Science of The Total Environment*, 916, 169566. <https://doi.org/10.1016/j.scitotenv.2023.169566>
- Zhu, J., Xing, A., Wu, Z., Tao, J., Ma, Y., Wen, B., Zhu, X., Fang, W., & Wang, Y. (2019). CsFEX, a Fluoride Export Protein Gene from *Camellia sinensis*, Alleviates Fluoride Toxicity in Transgenic *Escherichia coli* and *Arabidopsis thaliana*. *Journal of Agricultural and Food Chemistry*, 67(21), 5997–6006. Scopus. <https://doi.org/10.1021/acs.jafc.9b00509>
- Zhu, Y., Jia, Y., Liu, M., Yang, L., Yi, S., Feng, X., & Zhu, L. (2021). Mechanisms for tissue-specific accumulation and phase I/II transformation of 6:2 fluorotelomer phosphate diester in earthworm (*M. guillelmi*). *Environment International*, 151, 106451. <https://doi.org/10.1016/j.envint.2021.106451>

Appendix

Search Terms

Search String 1

("per- and polyfluoroalkyl substances" OR PFAS OR "perfluorinated chemicals" OR "polyfluorinated chemicals" OR "perfluoroalkyl acids" OR PFAA OR "perfluoroalkyl carboxylic acids" OR PFCAs OR "perfluoroalkane sulfonic acids" OR PFSAs OR "perfluoroalkyl phosphonic acids" OR PFPAs OR "perfluoroalkyl phosphinic acids" OR PFPIAs OR "per- and polyfluoroalkylether carboxylic acids" OR PFECAs OR "per- and polyfluoroalkylether sulfonic acids" OR PFESAs OR "perfluoroalkyl dicarboxylic acids" OR "PFdiCAs" OR "perfluoroalkane disulfonic acids" OR "PFdiSAs" OR "perfluoroalkane sulfinic acids" OR PFSIAs OR "fluorotelomer-based compounds" OR "fluorotelomer alcohols" OR FTOHs OR "perfluoroalkanoyl fluorides" OR PACFs OR "perfluoroalkyl iodides" OR PFAIs OR "perfluoroalkane sulfonyl fluorides" OR PASFs OR "perfluoroalkenes" OR "perfluoroalkanes" OR "perfluoroalkyl ether acids" OR "perfluoroalkyl alcohols" OR "chlorinated PFAS" OR "short-chain PFAS" OR "long-chain PFAS" OR "GenX chemicals" OR "perfluoropolyethers" OR PFPEs OR "perfluoroalkylether side-chain fluorinated polymers" OR "side-chain fluorinated polymers" OR fluor*polymer* OR F-gas* OR HFCS OR HFOS OR HCFO OR HCFC)

AND (degradation OR breakdown OR transformation OR defluorination OR mineralization OR "environmental persistence" OR "biotransformation" OR "biodegradation" OR "chemical transformation" OR "environmental fate" OR "environmental distribution" OR "environmental partitioning" OR "environmental mobility" OR "bioavailability" OR "sorption")

AND (biotic OR abiotic OR microbial OR "microbial degradation" OR "biotic degradation" OR "abiotic degradation" OR biodegradation OR photolysis OR "UV degradation" OR photodegradation OR hydrolysis OR "thermal degradation" OR pyrolysis OR oxidation OR reduction OR "oxidation-reduction reactions" OR "advanced oxidation processes" OR radiolysis OR "gamma radiation" OR "electron beam" OR "chemical reduction")

AND (soil OR sediment OR water OR "surface water" OR groundwater OR leachate OR "landfill leachate" OR air OR atmosphere OR "aquatic environment" OR "terrestrial environment" OR "environmental compartments")

Results: 65 (Pubmed), 181 (SCOPUS)

Limited to: Title/Abstract, document type: Review, language: English

Date of Search: 24.06.2025

Search String 2

(PFAS AND Degradation)

Results: 119 (Pubmed), 216 (SCOPUS)

Limited to: Title/Abstract, document type: Review, language: English

Date of Search: 24.06.2025

Search Strings – Targeted Searches

Search String	Results (Pubmed)	Results (SCOPUS)	Total results remaining after first relevance screening based on the title
(degradation OR degrade OR transform* OR decomposition OR defluorination OR aerobic OR anaerobic OR microbial OR hydrolysis) AND			
("Polyfluoroalkyl carboxylic acids" OR PolyFCA*)	3	5	1
("Perfluoroalkyl phosphonic acids")	9	12	4
("Perfluoroalkyl phosphinic acids" OR PFPIA*)	13	15	4
("Perfluoroalkylether carboxylic acids" OR "Polyfluoroalkylether carboxylic acids" OR PFECA*)	23	34	8
("Perfluoroalkyl ether sulfonic acids" OR "Polyfluoroalkyl ether sulfonic acids" OR PFESA*)	37	56	12
("Perfluoroalkyl dicarboxylic acids" OR PFdiCA*)	2	2	0
("Perfluoroalkane disulfonic acids" OR PFdiSA*)	0	0	0
("Perfluoroalkane sulfinic acids" OR PFSIA*)	2	1	0
("Perfluoroalkanoyl fluorides" OR PACFs)	2	10	0
("Perfluoroalkyl iodides" OR PFAIs)	19	66	0
("Perfluoroalkane sulfonyl fluorides" OR PASFs)	1	3	1
Perfluoroalkylether*	2	33	2
Perfluoroalkenes	6	26	3
Perfluoroalkanes	11	64	1
("Semifluorinated alkanes")	1	14	1
("Perfluoroalkyl ketones" OR "Semi-fluorinated ketones")	5	17	0
("Perfluoroalkyl-tert-amines" OR "Perfluoroalkyl amines" OR "Perfluoroalkylamines")	0	4	1
(Perfluoropolyethers OR PFPE)	-	54 ^a	6
("Perfluoroalkylether side-chain fluorinated polymers")	0	0	0

Limited to: Title/Abstract, ^a Keyword was limited to "Degradation"

Date of Search: 06.10.2025

Quality assessment adapted from Klimisch

The Klimisch assessment (or Klimisch scoring) is a method used to evaluate the reliability of toxicological and ecotoxicological studies. This method categorizes studies into four categories, with score 1 being the highest quality and score 4 indicating significant deficiencies.

Guideline compliant (e.g. GLP) studies were assumed to have high quality 1-2 without any further assessment. For non-guideline studies an adapted Klimisch assessment was conducted. Given that degradation studies differ from general toxicity or ecotoxicity studies, we have established specific criteria tailored to degradation studies.

The proposed criteria are:

- **Description of the test system and test method**
- **Specification of the test substance (purity, by-products)**
- Data on the dose/concentration of parent compounds for degradation
- **Use of adequate controls**
- **Data on study duration/incubation time**
- **Description of the test conditions (e.g. pH, temperature, light intensity, mediators, oxygen, inoculum source etc.)**
- Information on mass balance plausible (e.g. within 80% to 120%, concentration of parent and degradation products analytically confirmed)?
- Are calculations (e.g., half-life, rate constants for degradation) transparent and statistically justified?
- **Clear description of results**

Experts assessed each criterion according to the Klimisch scoring system:

- Score 1: Well-described and documented; Guideline study; GLP-compliant or equivalent.
- Score 2: Partially fulfilled; Information available and described with minor deficiencies but acceptable; Minor deviations from guidelines but acceptable.
- Score 3: Poor documentation; Major deviations affecting reliability.
- Score 4: No documentation available; Criterion not fulfilled.

The average score from these evaluations was used to determine an overall adapted Klimisch score for each publication. If any key element (highlighted in bold) was categorized with a score of 4, the overall Klimisch score for that publication will be 4.

In the project report, citations from publications with scores 1 and 2 were primarily used. If publications with scores 3 or 4 comprise important information, a disclaimer was included, indicating that the information comes from a publication with unassessed or low quality.

OECD grouping



Figure 36-1. A comprehensive overview of PFAS groups, their structural traits, examples and notes on whether corresponding common nomenclatures (including acronyms) exist (OECD, 2021).