JEE Journal of Ecological Engineering

Journal of Ecological Engineering, 2027, 28(1), 510–528
https://doi.org/10.12911/22998993/242648
ISSN 2299–8993, License CC-BY 4.0

Received: 2026.09.25
Accepted: 2026.11.16
Published: 2026.12.01

Toilet paper as a source of microfibers and per- and poly-fluoroalkyl substances and its impact on circular economy implementation at wastewater treatment plants

Krzysztof Fijałkowski[1*], Anna Kwarciak-Kozłowska[1]

[1] Faculty of Infrastructure and Environment, Czestochowa University of Technology, Dąbrowskiego 69, Częstochowa, Poland
* Corresponding author's e-mail: krzysztof.fijalkowski@pcz.pl

ABSTRACT

Toilet paper represents a continuous and globally ubiquitous stream of cellulosic waste introduced into sanitary systems, with annual consumption exceeding 83 million rolls worldwide. Although cellulose – the primary structural component of toilet paper, comprising 60–90% of its dry mass – is theoretically biodegradable, accumulating evidence demonstrates that cellulose microfibers persist in aquatic environments and currently account for 60–80% of all microfibers found in the world's oceans. This review synthesises current knowledge on the environmental fate of toilet paper waste, examining its role as a dual source of cellulose microfibers and per- and poly-fluoroalkyl substances (PFAS) in municipal wastewater systems. We assess the structural and compositional factors governing cellulose biodegradability – including crystallinity, lignin content, and the presence of wet-strength resins such as polyamidoamine-epichlorohydrin (PAE) – which collectively reduce hydrolysis rates by 30–50% underenvironmentally relevant conditions. Cellulose microfibers released from toilet paper (typically 1–3 mm in length, 10–50 µm in diameter) are incompletely removed in conventional primary treatment, with gravitational sedimentation achieving only 30–50% retention. Advanced mechanical separation technologies, including microscreens and rotating belt filters (RBF), increase cellulose recovery to up to 95% and can reduce total energy consumption by at least 40%. However, a critical technological barrier to circular economy implementation is identified: while these systems enable efficient cellulose recovery for valorisation – as biogas feedstock, construction materials, or thermochemical products – they simultaneously concentrate PFAS contaminants in the recovered biosolids. Toilet paper contributes 35–89% of the 6:2 diPAP load entering European wastewater treatment plants. This PFAS co-concentration undermines the safe reuse of recovered materials and necessitates integrated treatment strategies combining separation (adsorption, ion exchange, membrane processes) with destructive technologies (electrochemical oxidation, cold plasma, advanced oxidation processes). These findings highlight a fundamental tension between resource recovery and contaminant control that must be resolved to advance sustainable wastewater management within a circular economy framework.

Keywords: toilet paper, cellulose microfibers, PFAS, wastewater treatment plants, circular economy, biodegradation, mechanical separation

INTRODUCTION

Toilet paper is a specific stream of cellulosic waste continuously introduced into sanitary systems globally, with worldwide consumption estimated at over 83 million rolls annually (X. Wang et al., 2023). Because it is designed for immediate disposal into sewer networks after use, it becomes a major component of the organic suspension entering municipal wastewater treatment plants (Abildgård et al., 2003; Ali & Sreekrishnan, 2001).

It is widely assumed that toilet paper and cellulose fibers of other origins are completely environmentally inert, given that the material's primary structural element is a polymer that is theoretically fully biodegradable (Athey & Erdle, 2022). However, recent evidence challenges this

assumption, showing that under natural conditions, cellulose microfibers do not degrade as easily or as quickly as previously thought (KeChi-Okafor et al., 2023). Consequently, they currently predominate in aquatic environments, accounting for 60% to 80% of all microfibers found in the world's oceans, and their presence has also been recorded in bottom sediments and drinking water sources (Pittura et al., 2022; Volgare et al., 2022).

In this review, we assessed the biodegradability of this material and its role as a source of cellulose microfiber emissions into aquatic environments (Tserendorj et al., 2024). Particular attention was paid to the significance of toilet paper as a transport vector for per- and poly-fluoroalkyl substances (PFAS) into the environment (Shtamm et al., 2015; J. T. Thompson et al., 2023).

Mechanical separation systems, such as microscreens or rotating belt filters (RBF), enable effective cellulose recovery, thereby reducing the organic carbon load on biological reactors (Akyol et al., 2022; Gupta, 2018). However, capturing and recovering material from wastewater also unintentionally concentrates PFAS contaminants in the recovered biosolids (J. T. Thompson et al., 2023). Consequently, these recalcitrant xenobiotics create a barrier to land application or agricultural use of sewage sludge and other materials recovered at wastewater treatment plants (Y.-F. Chen et al., 2024).

This publication aims to synthesize existing research on the environmental fate of waste toilet paper.

TOILET PAPER AS A GENERATOR OF CELLULOSE MICROFIBERS

Physicochemistry of cellulose fibers

Toilet paper is a heterogeneous material in which cellulose is the primary structural polymer, accounting for 60% to over 90% of the product's dry mass (Mushtaq et al., 2023; Nagamine et al., 2022). Cellulose chains - with a degree of polymerization ranging from 300 to 1,700 glucose units - aggregate into microfibrils via numerous hydrogen bonds (Nagamine et al., 2022). This structure comprises highly ordered crystalline regions (with a crystallinity index of 40% to 70%) and disordered amorphous regions; this composition determines the density of fiber linkages and their subsequent susceptibility to disintegration in an aqueous environment (Ahmed, 2020; Bading et al., 2023). In addition to pure cellulose, toilet paper contains hemicelluloses (10% to 30% of the mass), which act as a structural binder, and lignin – an aromatic polymer responsible for rigidity, whose content depends heavily on the manufacturing process (Vikman et al., 2024).

The raw material used during production influences the structure and chemical purity of cellulose fibers, thereby determining their subsequent behavior in sewage systems and surface waters. Premium-grade papers are made from virgin pulp fibers produced through chemical pulping processes (e.g., the sulfate method), which preserve structural integrity and long fiber chains (Vikman et al., 2024). This process enables selective, near-total lignin removal (reducing levels to below 1%), resulting in fibers free of most industrial secondary contaminants (Ali & Sreekrishnan, 2001; Nagamine et al., 2022).

In contrast to virgin pulps, secondary fibers (recycled pulp) derived from wastepaper are physically and chemically degraded (Table 1). Analyses indicate that recycled fibers are significantly shortened and flattened. They also sustain mechanical damage from repeated refining, grinding, and aggressive de-inking processes, leading to their subsequent fragmentation into microfibers (Vikman et al., 2024). Furthermore, secondary fibers contain higher and variable levels of residual lignin (potentially reaching up to 20 percent), originating from admixtures, as well as significant quantities of mineral fillers, binders, and contaminants (Ali & Sreekrishnan, 2001; Vikman et al., 2024). Using recycled paper materials is also associated with unwanted additional transfer of xenobiotics, including significantly elevated concentrations of PFAS from recycled fibers (J. T. Thompson et al., 2023).

Mechanical dispersion and cellulose microfiber release

The first stage in the transformation of toilet paper into a source of pollution is its mechanical dispersion, which begins upon contact with water and leads to a loss of the sheet's structural cohesion (Vikman et al., 2024). Mechanical agitation during flushing and wastewater transport within sewer networks breaks the material into individual fibers and fiber aggregates (Allison et al., 2025; Tserendorj et al., 2024). Hydrodynamic forces and shear stresses within conveyance pipes

Table 1. Comparison of the physicochemical properties of virgin and recycled cellulose fibers used in the production of toilet paper

Property	Virgin fibers	Secondary fibers (recycled/hygiene products)	Source
Average fiber length	0.8–2.34 mm; hardwood: 0.8–1.10 mm; softwood:	0.81–1.11 mm (deinked pulp, DIP); 0.93–1.08 mm (old, corrugated containers, OCC)	(Baral, 2026; Debnath et al., 2021; Zambrano, 2021)
Crystallinity index	87–89%	87%	(Zambrano, 2021)
Residual lignin content	0.7–1.5%	3.65–3.7% (DIP); 0–21% (OCC)	(Baral, 2026; Vikman et al., 2024; Zambrano, 2021)
Hemicellulose content	14.2–19.2%	13.73–18.1% (DIP); 0–21.1% (OCC)	ebnath et al., 2021; Zambrano, 2021)
Effect of mechanical treatment on fibers	Standard surface fibrillation and fiber shortening during refining/grinding	Depends on treatment intensity. Intensive refining of DIP can shorten fibers (from 0.999 to 0.812 mm) and increase the proportion of fines from 6.83% to 14.61%. Mild refining does not cause fiber shortening.	(Baral, 2026; Debnath et al., 2021)
PFAS concentration	0.72–1.2 ng/g d.m. — median total PFAS concentration in tissue paper	3.2 ng/g d.m. — mean concentration of the PFAS precursor 6:2 diPAP in toilet paper; 27–38 ng/g to 121–213 ng/g d.m. — mean PFAS concentrations in cardboard and packaging paper	(Langberg et al., 2023; Mofokeng et al., 2024; J. T. Thompson et al., 2023; X. Wang et al., 2023)
Secondary contaminants and associated substances	Trace residues of bleaching and modifying agents	Possible residues of printing inks, fillers, adhesives, resins, and contaminants carried over from the collected wastepaper stream	(Langberg et al., 2023; Mofokeng et al., 2024; Vikman et al., 2024)

tear apart the weak paper structures (Vikmann). Adding wet-strength agents-specifically polyamide-epichlorohydrin (PAE) resins – inhibits natural disintegration, allowing the paper to maintain structural integrity in the sewer system longer (Allison et al., 2025). Consequently, delayed macroscopic fragmentation-driven by turbulence and flow-results in a prolonged, massive release of secondary cellulose microfibers, exacerbating ecosystem micro-pollution (Allison et al., 2025). These physical interactions generate suspensions of cellulose microfibers. The released toilet paper microfibers exhibit a specific morphology: they typically range from 1 to 3 mm in length and 10 to 50 µm in diameter, resulting in aspect ratios (length-to-diameter) ranging from 20:1 to over 100:1. Studies on the behavior of cellulosic materials in aquatic environments (Allison et al., 2025; Rathinamoorthy et al., 2025) indicate that water flow dynamics and mechanical forces accelerate physical fragmentation and the release of microfibers. The loosening and rupture of the structure increase the specific surface area of the fibers; while this facilitates microbial access to the matrix in subsequent stages, it also increases the amount of suspended matter in the water. A study of a municipal wastewater treatment plant (WWTP) in Budapest conducted by Tserendorj et al. (2024) illustrates the scale of this problem. Microfiber concentrations in the plant's effluent ranged from 1.88–2.84 items/L in February (week 7 of 2023) and rose to 4.25–6.79 items/L in April (week 16 of 2023). Cellulosic fibers predominated, accounting for 53–91% of all microfibers in the effluent, while microfibers accounted for 78.3% (February) and 94.7% (April) of the total particle count, respectively. The median fiber length increased from 650 µm in February to 1250 µm in April, and the median diameter increased from 21 µm to 29 µm. The daily load of microfibers transported into the Danube via treated wastewater ranged from 0.44 to 0.69 billion in February and from 0.94 to 1.53 billion in April, confirming the significance of seasonal variability. Furthermore, a study by Allison et al. (2025), conducted in mesocosms simulating natural river systems, showed that cellulose-based wet wipes persisted in the river environment for five weeks, with partial fragmentation beginning in the third week.

Fate of cellulose microfibers in wastewater treatment plants

Cellulose derived from toilet paper constitutes a significant portion of the suspended matter in raw municipal wastewater, accounting for an estimated 30–50% of suspended solids.

Consequently, the initial separation method significantly influences the subsequent fate of cellulose fibers within the treatment plant. Pathways of toilet paper-derived cellulosic microfibres through municipal wastewater treatment was shown on Figure 1. In conventional primary clarifiers, suspended matter is removed primarily through gravitational sedimentation. However, this mechanism is less effective for cellulose fibers because their density is close to that of water (Zhang et al., 2026). Retention rates in standard primary clarifiers range from about 30% to 50% (Akyol et al., 2022; Gupta, 2018). As a result, a substantial proportion of the fibers remains in the wastewater and proceeds to subsequent treatment stages. To improve initial cellulose fiber separation and enable recovery, more advanced technologies – such as microscreens and rotating belt filters (RBF) are increasingly used (Akyol et al., 2022; Gupta, 2018). Both solutions rely on physical particle retention, but they differ in their operating processes. For microscreens, mesh size is a key parameter that largely determines microfiber retention efficiency (Ruiken et al., 2013).

Li et al. (2020) showed that screens with a 3 mm mesh recover about 46% of cellulose fibers, whereas reducing the mesh size to 0.1-1 mm increase recovery to about 95%. However, using finer mesh sizes makes the screens more prone to clogging. In rotating belt filters, pore sizes range from approximately 10 to 550 μm (Zhang et al., 2026). This technology not only separates cellulose fibers from the main wastewater stream but also concentrates them in the resulting sludge. The use of RBF yields sludge with a cellulose content nearly double (37%) that of sludge from a conventional primary clarifier (18%) (Akyol et al., 2022; Gupta, 2018). Separating cellulose fibers at the preliminary stage also impacts the wastewater treatment plant's energy consumption. According to Ruiken et al. (2013), employing screens as a preliminary treatment step can reduce total energy consumption- including sludge processing- by at least 40%. Cellulose fibers not captured during the mechanical treatment stage proceed with the wastewater to the biological treatment stage. In the activated sludge tank, they can reduce oxygen transfer efficiency from air to wastewater, thereby increasing aeration demand. Ahmed et al. (2020) showed that removing cellulose before aeration using a rotating belt filter can reduce air and energy requirements by about 25%. Consequently, separating fibers before the biological treatment stage can reduce energy consumption. Cellulose biodegradation in activated sludge tanks is relatively slow because most microorganisms cannot directly use insoluble fibers. Breaking down cellulose fibers requires prior enzymatic hydrolysis, which decomposes the fiber structure into simpler compounds accessible to microorganisms (R. Chen et al., 2017). Hydrolysis is considered the rate-limiting step in cellulose biodegradation, and the process may require more time than the typical hydraulic retention time (HRT ≤ 12 h) used in conventional aerobic wastewater treatment systems. Therefore, solids retention time (SRT) appears to be a crucial parameter influencing the extent of fiber degradation. A longer SRT retains microorganisms involved in cellulose hydrolysis and decomposition, promoting biodegradation. Li et al. (2020) showed that the degradation rate of toilet paper fibers increases as SRT increases. At an SRT of 40 days, approximately 83% of the material underwent degradation, and the hydrolysis rate constant increased from 0.025–0.034 d^{-1} (for SRT < 20 days) to 0.116–0.137 d^{-1}. Extending the biomass retention time thus promotes fiber hydrolysis and the growth of microorganisms involved in their breakdown. However, an SRT exceeding 40 days may be difficult to implement in a conventional wastewater treatment plant. Temperature also influences cellulose biodegradation rates. Although natural cellulosic materials are biodegradable, knowledge regarding their degradation rates during biological wastewater treatment- and subsequently in the aquatic environment remains limited.

Studies on the aerobic biodegradation of cellulose, conducted over a three-day contact time and within a temperature range of 13–23°C, showed that temperature strongly influenced the process. A 10°C rise in temperature increased cellulose degradation from 6.7% to 87%. These results confirm that temperature can significantly affect the rate of cellulose fiber transformation. As previously mentioned, not all cellulose fibers entering activated sludge chambers undergo complete biodegradation. Undecomposed fibers can bind to activated sludge flocs and be separated along with them in the secondary clarifier, thereby increasing the volume of excess sludge requiring disposal (Tserendorj et al., 2024).

AQUATIC CELLULOSE BIODEGRADATION

Microbial dynamics in cellulose breakdown

The biological degradation of cellulose found in toilet paper in aquatic environments – and the efficiency of this process – relies on continuous microbial succession and close synergism between specialized taxa of cellulolytic bacteria and fungi (Demain et al., 2005; Flemming & Wingender, 2010). Figure 2 presents a diagram of the process stages.

Aerobic and facultatively anaerobic bacteria play a key role in surface waters and bottom sediments (Table 2). Dominant taxa include bacteria of the genus *Cellulomonas*, which exhibit an exceptional ability to synthesize and secrete extracellular endoglucanases responsible for the early stage of fiber depolymerization. In natural water bodies, as well as in the aerated chambers of biological reactors, *Bacillus* strains are commonly identified; these produce cellulases that remain highly stable under fluctuating environmental conditions. Meanwhile, bacteria of the genus *Cytophaga* employ a mechanism known as contact digestion. These microorganisms adhere closely to the cellulose surface and secrete enzymes directly into the cell-substrate interaction zone, thereby preventing enzyme dispersion and loss (Bhatia et al., 2024).

In anoxic and reducing environments – such as deep bottom sediments and wastewater treatment plant digesters- anaerobic bacteria of the genus *Clostridium* dominate the microbiome structure (R. Chen et al., 2017; Demain et al., 2005). These bacteria (e.g., *Clostridium thermocellum*) produce unique, cell-membrane- anchored multienzyme complexes known as cellulosomes. Composed of scaffoldin proteins that link the catalytic domains of enzymes, cellulosomes ensure optimal cell adhesion to the substrate and highly efficient, loss-free hydrolysis of chains under anaerobic conditions (Demain et al., 2005; L. Yang et al., 2021).

Bacterial succession in aquatic environments is complemented by cellulolytic fungi, which also penetrate deeper layers of the paper matrix (De Vries & Visser, 2001; Sequeira, 2016). Microscopic fungi of the genera *Trichoderma* and *Aspergillus* secrete highly active exoenzymes into the environment, breaking down pure cellulose and hemicelluloses (Bhatia et al., 2024; De Vries & Visser, 2001). Furthermore, fungal hyphae physically penetrate the crevices and micropores of paper fibers, expanding their structure and thereby facilitating the secondary colonization and growth of cellulolytic bacteria (Bhatia et al., 2024; Rathinamoorthy et al., 2025).

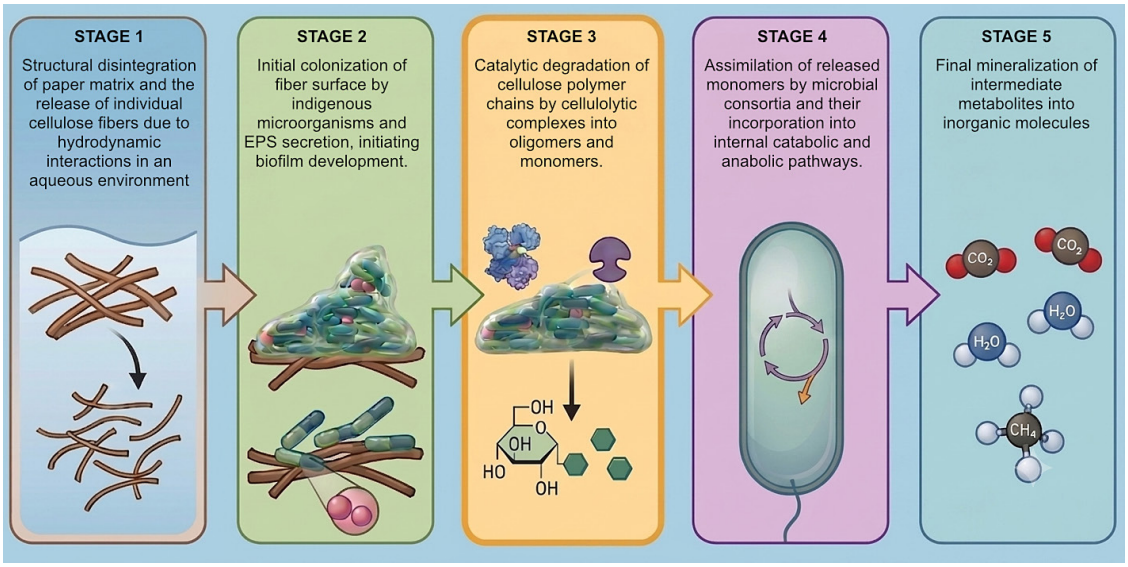

Figure 1. Pathways of toilet paper-derived cellulosic microfibres and PFAS through municipal wastewater treatment and their release into the aquatic environment. Authors' own elaboration based on (Akyol et al., 2022; Allison et al., 2025; Gupta, 2018; Palmieri et al., 2019; Rusten & Ødegaard, 2019; Thompson et al., 2023; Tserendorj et al., 2024; Zhang et al., 2026).
Note: Image generated using ChatGPT/Gemini based on the authors' original prompts and subsequently modified using graphic software.

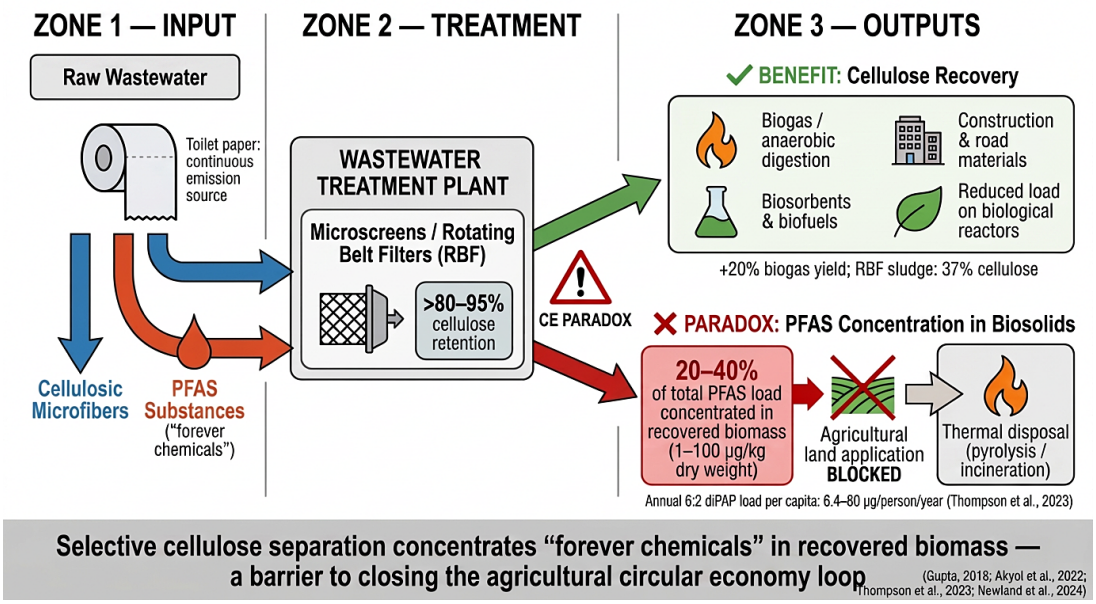

Figure 2. Diagram of the five-stage biodegradation process of toilet paper in an aquatic environment: (1) mechanical dispersion → (2) biofilm formation → (3) enzymatic hydrolysis → (4) intracellular metabolism → (5) final mineralization. Arrows indicate the rate-limiting step (enzymatic hydrolysis). Authors' own elaboration based on (Akyol et al., 2022; Bhatia et al., 2024; Demain et al., 2005; Vikman et al., 2024).
Note: Image generated using ChatGPT/Gemini based on the authors' original prompts and subsequently modified using graphic software.

Fiber properties affecting hydrolysis kinetics

Enzymatic hydrolysis is the key step determining the rate of toilet paper biodegradation in an aquatic environment (Figure 2). Its course and kinetics are directly governed by the physicochemical parameters and structure of the fibers themselves (Ahmed, 2020; Nagamine et al., 2022). The toilet paper manufacturing process affects the proportions of crystalline and amorphous regions in the cellulose, which directly influences the accessibility of beta-1,4-glycosidic bonds to the cellulolytic enzyme complex (Bading et al., 2023).

The crystallinity index is one parameter that determines a fiber's resistance to microbial degradation. In crystalline regions, cellulose molecules form a tightly packed, parallel arrangement stabilized by a strong network of hydrogen bonds; this structure prevents water molecules from penetrating and hinders the binding of the catalytic domains of endo- and exoglucanases. Enzymatic hydrolysis begins in the amorphous regions, which degrade rapidly, leaving behind the more durable crystalline fragments. This phenomenon increases the crystallinity index of the remaining, undegraded material fraction, thereby slowing subsequent biodegradation kinetics over time. Biodegradation studies in aquatic environments confirm that highly crystalline fibers lose only 10% to 20% of their mass within 28 days, whereas fibers rich in amorphous regions lose 40% to 60%t over the same period (Bading et al., 2023; Nagamine et al., 2022).

A second factor influencing enzymatic hydrolysis is the residual lignin content in paper fibers (Vikman et al., 2024). Lignin acts as a physical and chemical barrier surrounding cellulose microfibers, blocking access to extracellular hydrolases (Ganguly & Chakraborty, 2023; Vikman et al., 2024). Research findings (Vikman et al., 2024) indicate a strong negative correlation between lignin presence and the paper's susceptibility to degradation. Including mechanical or unbleached pulp with lignin content exceeding 5 percent reduces the rate of biodegradation in aquatic environments by 30% to 50% compared to fully refined and delignified paper pulps. Furthermore, the degradation of lignin itself in aquatic ecosystems proceeds slowly, requiring oxygen and specific ligninolytic enzymes (laccases, peroxidases) synthesized primarily by white-rot fungi- which means that in the anaerobic conditions

Table 2. Selected microorganisms and enzymatic systems involved in the degradation of cellulose and cellulosic waste – mechanisms of action and experimental and biotechnological significance

Microorganism	Type/Conditions	Enzymatic mechanism	Ecological/biotechnological role	Source
Selected strains of *Bacillus* / *Geobacillus*	Aerobic / facultatively anaerobic, thermophilic	Thermostable cellulases (mainly endoglucanases)	Cellulose degradation at elevated temperatures	(Bhatia et al., 2024)
Clostridium thermocellum	Anaerobic, thermophilic	Cell-surface cellulosomes containing multiple cellulases and scaffold proteins	Efficient hydrolysis of insoluble and crystalline cellulose; conversion of biomass to ethanol	(Demain et al., 2005)
Aspergillus spp.	Aerobic fungi	Endoglucanases, cellobiohydrolases, and β-glucosidases	Hydrolysis of cellulose and other polysaccharides	(De Vries & Visser, 2001)
Cellulomonas spp.	Aerobic / facultatively anaerobic	Synthesis and secretion of extracellular endoglucanases	Early stage of cellulose fiber depolymerization	(Bhatia et al., 2024)
Bacillus spp.	Aerobic / facultatively anaerobic	Secretion of cellulases with high environmental stability	Cellulose degradation in surface waters and biological reactors	
Cytophaga spp.	Aerobic	Contact digestion – direct enzyme secretion at the cell–substrate attachment interface	Efficient degradation with minimal enzyme loss	
Enzymatic system from *Thermobifida fusca*	*In vitro* experiment, aerobic conditions	TfLPMO10A/B system integrated into an artificial cellulosome containing endo- and exocellulases	1.7-fold increase in sugar release compared with free enzymes and 2.6-fold increase compared with cellulases without LPMO	(Arfi et al., 2014)
Enzyme preparation from *Irpex lacteus*	Enzymes obtained from cultures of an aerobic white-rot fungus	Cellulases and hemicellulases, low β-glucosidase activity; ligninolytic enzymes also present	Release of 31.2 [units as reported] of sugars per gram of dry screenings after 48 h; 28% saccharification	(Zarina & Mezule, 2024)
Bjerkandera adusta DSM 23426	Aerobic white-rot fungus	Predominant cellulolytic activity (endo-1,4-β-glucanase); low laccase activity	43% reduction in the total carbohydrate content of screenings after 336 h	(Civzele, 2025)

of bottom sediments, lignin-rich paper fibers can persist for many months (Sequeira, 2016; Vikman et al., 2024).

The role of PAE resins in blocking enzyme access to cellulose

Producing high-quality toilet paper involves synthetic chemical modifiers designed to enhance the product's performance. Wet-strength resins- specifically polyamidoamine-epichlorohydrin (PAE) resins- play the most critical role. These compounds are added to the paper pulp at levels ranging from 0.1% to 1.0% of the dry mass to prevent the product from immediately losing its structural integrity upon wetting. Reactive azetidinium groups within the PAE resin structure react with the carboxyl and hydroxyl groups of cellulose, forming strong covalent bonds (Kwon, 2024). This creates an effective barrier that inhibits subsequent biological degradation processes in aquatic environments. The PAE matrix physically obstructs the free diffusion and access of extracellular endoglucanases, exoglucanases, and accessory proteins to the cellulose surface. Consequently, the normal enzymatic activity of cellulolytic microorganisms is disrupted, reducing the rate of cellulose hydrolysis by 30% to as much as 50% (Ganguly & Chakraborty, 2023; Kwon, 2024).

PAE resins in toilet paper cause a range of negative effects in both engineered systems and natural receiving environments (Ali & Sreekrishnan, 2001; Vikman et al., 2024). Paper containing PAE in sewer systems increases the risk of hydraulic blockages and the formation of fat-fiber conglomerates (so-called "fatbergs") (Allison et al., 2025). Meanwhile, in wastewater treatment plant digesters, the resistance of PAE-crosslinked fibers hinders proper methanogenic fermentation (Angelidaki et al., 2009; Appels et al., 2008). Furthermore, the hydrolysis of PAE resins themselves releases toxic by-products into the aquatic environment, including epichlorohydrin and the genotoxic compound 3-monochloropropane-1,2-diol (3-MCPD). These compounds strongly inhibit cellulolytic microbial biomass (Hanekamp, 2019).

TOILET PAPER AS A SOURCE OF PFAS

PFAS constitute a group of synthetic organofluorine compounds in which strong carbon-fluorine bonds confer exceptional chemical stability, thermal resistance, and hydrophobic and oleophobic properties. Measurements of PFAS in toilet paper samples from various manufacturing facilities have revealed a range of compounds from this group; the most frequently identified substances included polyfluoroalkyl phosphate diesters (such as 6:2 diPAP), as well as – at lower concentrations – perfluoroalkyl acids and their precursors.

PFAS in toilet paper stem from both intentional manufacturing processes and cross-contamination within the raw material supply chain. These compounds are used during paper production as processing aids, coating additives, or sizing agents that provide grease- and water-resistance. Studies indicate that toilet paper produced from secondary (recycled) fibers may have a different profile and PFAS residue content- resulting from raw-material contamination- than products made from virgin pulp (LaFond et al., 2023; J. T. Thompson et al., 2023).

Toilet paper accounts for about 4% of the 6:2 diPAP load entering municipal wastewater treatment plants in North America (and 35%-89% in selected European countries), translating to an annual emission of 88–210 kg of this compound (Table 3). On a per-capita basis, the 6:2 diPAP load discharged via toilet paper ranges from 6.4 to 80 μg per person per year (Y.-F. Chen et al., 2024; K. A. Thompson, 2022).

Studies on Chinese paper products (Wan et al., 2024) provide comparative data on PFAS concentrations in the paper matrix. Total PFAS concentrations in facial tissues ranged from 0.36 to 1.7 ng/g dry weight (median: 0.72 ng/g dry weight), while in toilet paper they ranged from 0.28 to 4.5 ng/g dry weight (median: 1.2 ng/g dry weight). Among next-generation substances, the dominant compounds were chlorinated polyfluorinated ether sulfonates (Cl-PFESAs; median: 0.32 ng/g dry weight; DF >96%), hexafluoropropylene oxide homologs (HFPOs; median: 0.19 ng/g dry weight), and sodium p-perfluorononenyloxybenzenesulfonate (OBS; median: 0.10 ng/g dry weight; DF >96%). These results confirm that traditional analytical methods targeting only classical PFAS may significantly underestimate the actual load of these substances entering treatment systems via toilet wastewater.

Newland et al. (2024) provide broader context on PFAS in the paper industry, noting that 269 pulp and paper mills collectively produce 825 tons of PFAS-coated paper daily. An estimated 45% of tap water in the United States contains at least one type of PFAS. Measured PFOA concentrations in the effluents of selected wastewater treatment plants ranged from 11.7 to 13.5 ng/L, exceeding recommended safety limits. Regarding the bioremediation of pulp and paper wastewater and waste, a mixture of *Klebsiella* sp., *Alcaligenes* sp., and *Cronobacter* sp. strains reduced the biochemical oxygen demand (BOD) by 91%. It lowered the concentration of organic halides by 45.4%. Among physicochemical technologies, Berkey home filters removed 98% of PFAS, while Clearly Filtered and Zero Water systems achieved 100% efficiency, confirming the potential of decentralized water treatment in affected areas.

Impact of cellulose microfibers and chemical modifiers on aquatic organisms

When aquatic organisms ingest microfibers, they accumulate in the digestive tract, causing epithelial damage, reduced feeding efficiency, and growth inhibition. These effects are observed at concentrations as low as 50 fibers/L in bivalves (Pittura et al., 2022) and at higher exposure concentrations in small crustaceans and fish (Siddiqui et al., 2023).

PFAS compounds have strong potential for bioaccumulation in aquatic organisms, with bioconcentration factors (BCF) ranging from 10^2 to 10^4 in fish (Burkhard, 2021). These substances preferentially accumulate in protein-rich parenchymal organs – such as the liver, kidneys, and blood (D. Liu et al., 2023) – from where they undergo further biomagnification within aquatic food webs. Detection of these substances in toilet paper. Thompson et al. (2022) suggests that this product serves as a direct vector for their release into sewage systems and aquatic ecosystems. Both PFAS and AOX compounds present in cellulose-related wastewater are classified as endocrine-disrupting chemicals (EDCs) (Rosińska, 2022; Shtamm et al., 2015; Tan et al., 2025). They induce, among other effects, pathological suppression or stimulation of vitellogenin synthesis in male fish, thyroid disorders, reduced fertility, and skeletal and embryonic developmental deformities (Shtamm et al., 2015).

Table 3. Types and concentrations of PFAS in toilet paper and hygiene products

PFAS compound	Concentration range (ng/g d.w.)	Region/Study	Source
6:2 diPAP	1.25–9.15	USA, Chile, UK, France, the Netherlands, South Africa, Costa Rica, El Salvador, Uruguay	(K. A. Thompson, 2022)
PFOA	330	China (single toilet paper sample)	(Wan et al., 2024)
PFHxA	0.04–0.36	USA, Chile, UK, France, the Netherlands, South Africa, Costa Rica, El Salvador, Uruguay, China	(K. A. Thompson, 2022)
PFDA (perfluorodecanoic acid)	0.03–0.11	USA, Chile, South Africa, El Salvador	
Cl-PFESAs (chlorinated polyfluorinated ether sulfonates)	Median: 0.32; DF: 100%	China	(Wan et al., 2024)
HFPOs / HFPO-TrA	Median: 0.19; DF: 96%	China	
OBS (sodium *p*-perfluorononenyloxybenzenesulfonate)	Median: 0.10; DF: 96%	China	
Total PFAS — toilet paper	0.28–4.5 (median: 1.2)	China	
Total PFAS — facial tissues	0.36–1.7 (median: 0.72)	China	

Additionally, under environmental stressors (e.g., elevated temperature), cellulose microfibers exacerbate lipid peroxidation in cell membranes and impair the organisms' antioxidant response, making them more susceptible to anthropogenic pressures (Pittura et al., 2022).

CIRCULAR RECOVERY OF CELLULOSE MICROFIBERS AT WWTP

Cellulose fibers present in municipal wastewater can be captured at various stages of the treatment process. Some are separated during mechanical treatment and end up in primary sludge, while fibers that pass to the biological treatment stage may undergo partial biodegradation or become bound to activated sludge flocs, ultimately ending up in excess sludge. If the sludge is further fermented, some undecomposed cellulose may also remain in the digested sludge. This means that the cellulose entering the treatment plant with the wastewater does not undergo complete mineralization but is transferred to the solid phase and subsequent sludge streams. This creates an opportunity for recovery and reuse, aligning with circular economy principles (Glińska et al., 2020). Particularly advantageous is the preliminary separation of cellulose fibers using microsieves or rotating belt filters, which allows production of a concentrated cellulose material. Recovered cellulose fibers can serve as a raw material for further use, including in anaerobic fermentation and methane production processes (R. Chen et al., 2017), as a component of construction materials (Palmieri et al., 2019), and as a feedstock for the production of thermochemical products such as biochar and bio-oil (Khan et al., 2022). Recovered cellulose can also be used to produce adsorbent materials (Honda & Iwahori, 2003). These utilization pathways allow the cellulose in wastewater to be viewed not merely as a component requiring removal, but also as a potential secondary raw material (R. Liu et al., 2022).

The potential for reusing recovered cellulose depends not only on separation efficiency and functional properties but also on the chemical quality of the resulting material. This is particularly significant for fibers derived from municipal wastewater, as they remain in contact with a complex mixture of organic matter, suspended solids, and micropollutants before extraction. Substances relevant to the safety of the recovered material's future use include per- and poly-fluoroalkyl substances (PFAS), which have been detected in products such as toilet paper (J. T. Thompson et al., 2023). PFAS concentrations in influent wastewater range widely, from 0.18 to 66,190 ng/L (Behnami et al., 2024). Interestingly, PFAS concentrations in treated effluent can exceed those in raw wastewater. This phenomenon may stem from the transformation of precursor substances into terminal perfluoroalkyl compounds, including PFOS (Lenka et al., 2022). Clearly, conventional treatment processes do not effectively remove a significant portion of PFAS; consequently, these compounds may be discharged into the environment with treated effluent or transferred to sewage sludge.

However, the presence of PFAS in toilet paper does not mean these compounds remain bound to the fibers during transport through the sewer system, wastewater treatment, and subsequent separation of cellulosic material. The fate of PFAS in a treatment plant depends on the properties of individual compounds and process conditions; a key mechanism is their partitioning between aqueous and solid phases (Arvaniti et al., 2014; Behnami et al., 2024). Arvaniti et al. (2014) demonstrated the sorption of perfluorinated compounds onto primary, secondary, and digested sludge, with the extent of sorption depending on the specific compound's properties and environmental conditions. Meanwhile, Behnami et al. (2024) indicated that PFAS partitioning between the aqueous and sludge phases is influenced by factors such as carbon chain length, sludge protein content, and the presence of cations. Unlike cellulose, which is biodegradable, PFAS are highly persistent and are not effectively degraded during conventional wastewater treatment processes. Consequently, a reduction in the concentration of certain PFAS in the aqueous phase does not necessarily indicate degradation; instead, it may result from sorption onto suspended solids and sludge, followed by their removal alongside the solid phase.

Additionally, during wastewater treatment and sludge processing, PFAS precursors can transform into other compounds within this group, potentially altering the concentrations of specific PFAS at various stages of the process (J. T. Thompson et al., 2023). This issue is particularly significant in the context of the circular economy. On the one hand, recovering cellulose fibers reduces their influx into the biological treatment stage and yields a reusable material. On the other hand, PFAS in the solid streams generated during treatment can affect options for their subsequent management- particularly the use of wastewater treatment plant materials in natural environments or agriculture. Consequently, closing the material loop must not lead to the uncontrolled transfer of persistent pollutants into new products or other environmental compartments. Thus, the mere effective separation of cellulose from wastewater does not guarantee a raw material suitable for reuse; its chemical quality and the presence of persistent contaminants must also be considered. At the same time, direct data conclusively determining the extent to which PFAS remain associated with natural cellulose fibers during separation and further processing are lacking. While existing studies confirm the presence of PFAS in toilet paper and their sorption onto sewage sludge, they do not directly characterize PFAS partitioning during the recovery of natural cellulose fibers from wastewater (Arvaniti et al., 2014; J. T. Thompson et al., 2023). It remains unclear how much PFAS stay bound to the fibers during separation versus partitioning into the sludge alongside other solid-phase constituents. This represents a significant knowledge gap regarding the feasibility of safely recovering and reusing cellulose from wastewater.

The conceptual scheme (Figure 3) illustrates the relationship between cellulose recovery from municipal wastewater and the potential occurrence and management of per- and poly-fluoroalkyl substances (PFAS). Zone 1 (Input) presents municipal wastewater containing cellulose-rich solids associated with toilet paper, together with PFAS and PFAS precursors, including 6:2 diPAP. Zone 2 (Treatment) illustrates rotating belt filtration (RBF) as a separation process for cellulose-rich suspended solids. Reported RBF performance includes 11–74% total suspended solids (TSS) removal, while the recovered cellulosic material has been reported to contain 38 ± 6% pure cellulose after washing. The RBF sludge fraction has also been characterized as containing approximately 37% cellulose. Zone 3 (Outputs) highlights potential resource-recovery pathways, including the use of recovered cellulose in hydraulic lime-based mortars and other potential bio-based applications, as well as the reported reduction in overall sludge production compared with primary clarification.

The red pathway represents the CE barrier associated with PFAS fate and management. PFAS and PFAS precursors have been detected in toilet paper and wastewater sludge; however, the available literature does not establish whether PFAS are co-recovered or concentrated specifically in the cellulose-rich fraction generated by RBF. Therefore, the figure identifies PFAS fate during cellulose recovery as an unresolved research question and a potential barrier to the safe reuse of recovered cellulosic solids, rather than assuming that RBF necessarily concentrates PFAS. The scheme is based on data and findings reported by Gupta (2018), Palmieri et al. (2019), Thompson et al. (2023), and Newland et al. (2024).

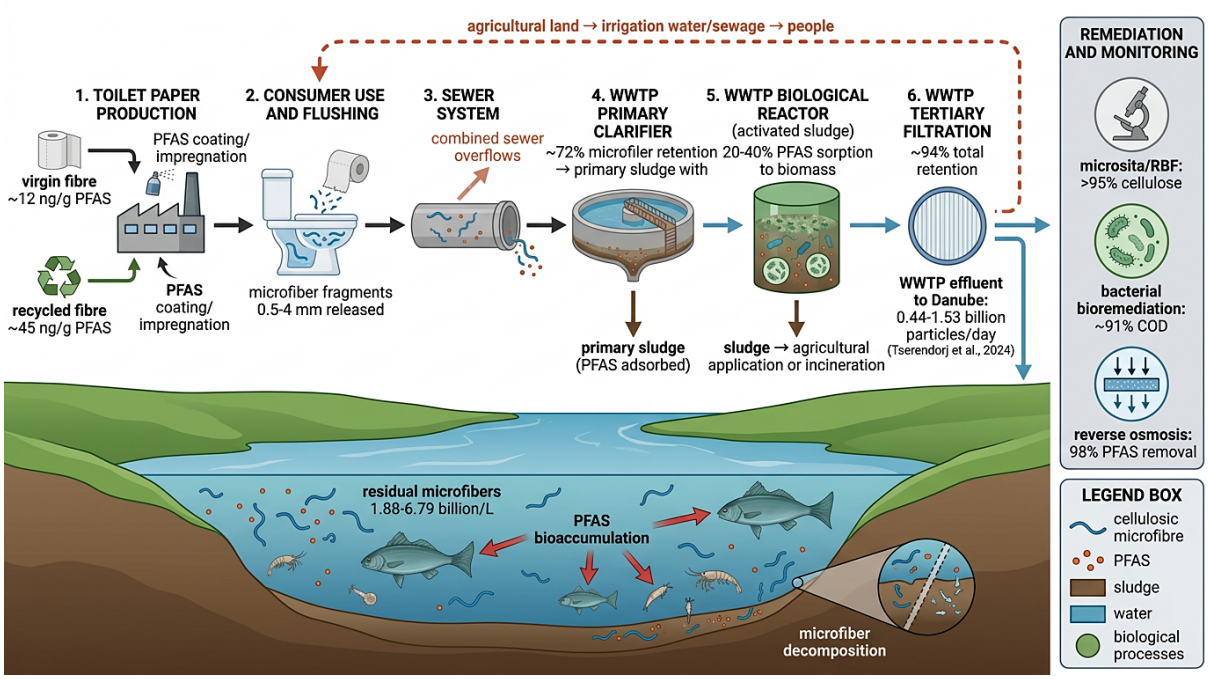

Figure 3. Circular economy barrier in wastewater treatment: cellulose recovery and potential PFAS co-recovery. Legend: 1. PFAS in toilet paper and sources (Newland et al., 2024; J. T. Thompson et al., 2023), 2. RBF performance (Palmieri et al., 2019); 3. Cellulose content and construction applications (Palmieri et al., 2019); 4. Cellulose content in RBF sludge and reduced sludge production (Gupta, 2018); 5. PFAS concentrations in toilet paper and wastewater sludge (J. T. Thompson et al., 2023). **Note:** Image generated using ChatGPT/Gemini based on the authors' original prompts and subsequently modified using graphic software.

COMPARISON OF PFAS REMOVAL AND DEGRADATION TECHNOLOGIES

Table 4 presents a comparative overview of the main groups of technologies used to remove and degrade PFAS compounds. The classification considers the dominant process mechanism (physicochemical separation vs. chemical degradation/ advanced oxidation), the range of effectiveness confirmed by empirical studies, the specific characteristics of the water matrix, the target PFAS homologs, and the key application limitations of each solution. Most experimental data in the literature come from studies conducted on model waters (deionized) and drinking water, whereas comprehensive studies on raw and treated wastewater remain limited. This is due to the complexity of the wastewater matrix, analytical challenges, and the fact that PFAS do not biodegrade in conventional biological wastewater treatment processes (e.g., activated sludge systems). In conventional wastewater treatment plants, these contaminants are only partially removed through adsorption onto sewage sludge. Compounds that are not adsorbed and pass into the effluent directly threaten the aquatic environment.

PFAS removal and degradation technologies are divided into two main functional categories: separation methods (non-destructive) – including adsorption (GAC, PAC), ion exchange (IX), membrane processes (RO, NF, UF), and foam fraction and degradation methods, comprising cold plasma processes, electrochemical oxidation (including on BDD electrodes), photocatalysis, and advanced oxidation processes (AOPs). Separation methods generally perform well for long-chain PFAS compounds in drinking and model waters; however, they do not mineralize contaminants, only transfer them to another phase. This necessitates managing secondary streams (such as concentrates, regenerants, or foam) that are difficult to dispose of. In contrast, degradation methods can permanently break stable C-F bonds and mineralize the fluorocarbon backbone, though their efficiency – specifically the concentration of dissolved organic matter (DOM/ NOM) (Gomez-Ruiz et al., 2017).

Process efficiency is also clearly linked to PFAS alkyl-chain length. All conventional separation methods show a marked decline in effectiveness against short-chain PFAS homologs (C_4-C_7), such as perfluorobutane sulfonic acid (PFBS)

Table 4. Comparison of PFAS removal and degradation technologies

Technology	Mechanism of action	Medium	Removal/ degradation efficiency	Types of PFAS removed	Main limitations	References
Hybrid: foam fractionation + EC/ BDD	PFAS concentration by foam fractionation → electrolysis of the foam concentrate → mineralization on BDD; volume reduction prior to EC	Landfill leachate, wastewater	90–98% total removal at an energy consumption of ~3–8 kWh/m³ (lower than EC alone)	Surface-active PFAS; foam concentration minimizes the volume requiring EC treatment	Requires a sequential treatment system; limited pilot-scale data	(S. Smith, 2023; S. J. Smith et al., 2022)
Hybrid: cyclodextrin monoliths	Macroporous cyclodextrin polymers: inclusion of the hydrophobic PFAS tail within the CD cavity; continuous-flow removal	Model water	>95% for PFOS, PFOA, and C4–C8 PFAS under continuous-flow conditions	Broad spectrum of PFAS, including C4–C6; effective for short-chain PFAS	Laboratory scale only; durability in the presence of DOM unknown; high synthesis cost	(Lin et al., 2025)
Hybrid: redox electrodialysis + electroadsorption	Electroadsorption of PFAS onto redox-active electrodes; elution by potential switching; combined with dialysis for concentration	Model water	>90% for C4–C10 PFAS; effective for ultra-short- and long-chain compounds	Broadest PFAS spectrum, including C4–C6; effective for short-chain PFAS	Laboratory scale only; membrane selectivity in the presence of SO_4^{2-} and Cl^-	(Kim et al., 2024)
Hybrid: UV-regenerable Fe/ AC	UV activates iron-doped activated carbon → *in situ* generation of e^-_{aq}; simultaneous adsorption and degradation; *in situ* regeneration	Drinking water (pilot scale)	80–95% for PFOA and pharmaceutical PFAS; *in situ* regeneration in a 24 h cycle	PFOA, C6–C10 PFAS; pharmaceuticals as co-contaminants	Early pilot stage; limited data for short-chain PFAS; UV lamp costs	(Beak et al., 2026)
Peroxi-electrocoagulation (EC + H_2O_2)	Electrochemical generation of Fe^{2+}/Al^{3+} ions forming flocs; H_2O_2 generates •OH via Fenton-like reactions; flotation of PFAS-containing flocs	Model water, real waters	60–90% for PFOA and PFOS in model water; 30–65% in waters containing DOM	Long-chain PFAS; efficiency decreases for short-chain PFAS and at high DOM levels	DOM strongly inhibits •OH generation; formation of sludge requiring disposal; strong pH dependence	(Hassan et al., 2025; Ryan et al., 2025)
Catalytic ozonation	Ozone reacts with catalyst intermediates; •OH is generated indirectly; surface oxidation of PFAS	Tap water, model water	30–60% for PFOA and PFOS; low efficiency without a catalyst	PFCA (C6–C8); poor performance for PFSA	Weak direct interaction of O_3 with C–F bonds; formation of bromate by-products; incomplete degradation	(Ersan et al., 2024; Franke, Schäfers, et al., 2019)
Photocatalysis (TiO_2, ZnO–Ag, composites)	Photocatalyst activation (hv ≥ Eg); generation of h^+ holes and •OH radicals; PFAS degradation on the catalyst surface	Model water, secondary effluent	40–85% for PFOA after 3–6 h of irradiation; strongly dependent on the matrix	PFOA; limited data for PFOS and short-chain PFAS	Strong attenuation by DOM and turbidity; slow kinetics; incomplete mineralization; TFA by-products	(Xia et al., 2022; H. Yang et al., 2024; Zhu et al., 2026)
UV/sulfite – e^-_{aq} radicals (hydrated-electron AOP)	UV (254 nm) photolysis of sulfite (SO_3^{2-}) generates e^-_{aq}; reductive attack on C–F bonds; gradual defluorination	Model water, IX regenerant	70–98% degradation of PFOA and PFOS; higher efficiency than •OH-based radical AOPs	PFOA, PFOS, PFCA; potential application to ether PFAS	Strong inhibition by O_2, which scavenges e^-_{aq}; NOM and NO_3^- absorb UV; anaerobic conditions required	(Cui & Deng, 2023; Hajalifard et al., 2024; Rao & Liu, 2024)

Electrochemical oxidation (BDD)	Direct oxidation at the BDD electrode (E > 2.8 V vs. SHE); generation of •OH at the electrode surface; indirect oxidation via persulfate	Industrial wastewater, model water	85–99% for PFOA, PFOS, and PFBS; effective for C4–C10	Broad spectrum, including short-chain PFAS; one of the few methods effective for C4–C6	High energy costs (15–50 kWh/m³); high cost of BDD electrodes; formation of ClO_4^- and halogenated by-products	(Gomez-Ruiz et al., 2017; Kim et al., 2024)
Cold atmospheric plasma (CAP/DBD)	Generation of radicals (•OH, •O, O_3, e^-_{aq}), UV radiation, and pressure waves; attack on C–F bonds at the plasma–water interface	Model water (DI)	>90% for PFOA and PFOS in DI water; 8–50% in real matrices	PFOA, PFOS; limited data for ether PFAS	Strong quenching by NOM, Cl^-, and HCO_3^-; high energy demand; incomplete mineralization; formation of shorter-chain PFAS homologues	(Khan et al., 2022; Topolovec et al., 2024)
Foam fractionation	Preferential accumulation of amphiphilic PFAS at the liquid–gas interface; flotation into the foam phase	Landfill leachate, municipal wastewater	60–95% for long-chain PFAS (PFOS, PFOA, PFHxS) in leachate; 40–70% for short-chain PFAS	Surface-active long-chain PFAS; lower efficiency for short-chain PFAS and PFEtherS	Short-chain PFAS exhibit lower surface activity; foam requires further treatment/disposal; matrix sensitivity	(Malovanyy et al., 2025; S. J. Smith et al., 2022; Y. Wang et al., 2023)
Surfactant-enhanced ultrafiltration (*in situ*)	Micellar entrapment of PFAS by surfactant aggregates retained by the UF membrane	Model water	85–96% for PFOS and PFOA at the optimal surfactant concentration	Surface-active PFAS; efficiency decreases for short-chain PFAS	Surfactant dosing required; risk of secondary contamination; pilot-scale performance not demonstrated	(Ren et al., 2026)
Reverse osmosis (RO)	Solute and size exclusion; >99% retention of compounds with MW >100 Da	Drinking water, wastewater	>99% for virtually all PFAS	All PFAS; broadest removal efficiency among membrane-based methods	Highest energy and operating costs; large volumes of retentate; does not destroy PFAS	(Franke, Schäfers, et al., 2019; Rückbeil et al., 2025)
Nanofiltration (NF)	Retention of molecules >200–300 Da; electrostatic (Donnan) exclusion of anionic PFAS	Drinking water, model water	90–99% for PFOA and PFOS; >85% for C6+ PFAS	Long-chain PFAS; lower efficiency for PFAS with MW <300 Da	Membrane fouling; generation of concentrate requiring further treatment; pressure-related costs; short-chain PFAS (<C6) may pass through	(Franke, McCleaf, et al., 2019; Franke, Schäfers, et al., 2019; Nir et al., 2023)
Regenerable ion-exchange resins (RSSCT)	Ion exchange combined with regeneration using NaCl/MeOH or NaOH/MeOH solutions; multiple treatment cycles	Model water, drinking water	80–97% for PFOA and PFOS after multiple cycles	PFCA, PFSA; optimization required for C4–C6 PFAS	Concentrated regenerant requires further treatment (photodegradation, electrolysis); reduced capacity after multiple cycles	(H. Li et al., 2022; B. Liu et al., 2024; Rao & Liu, 2024)
Ion-exchange resins (IX) – single use	Anion exchange: replacement of $PFAS^-$ ions by Cl^-/HCO_3^- on a polymeric matrix containing ammonium groups	Drinking water, model water	85–99% for PFOA and PFOS; 40–65% for ether PFAS (HFPO-DA)	PFCA, PFSA; low selectivity toward ether PFAS	High cost of single-use resin; highly concentrated regenerant requires further treatment/disposal; NO_3^- and SO_4^{2-} reduce capacity	(B. Liu et al., 2024; Rückbeil et al., 2025)

Adsorption on PAC (powdered activated carbon)	Same mechanism as GAC; higher specific surface area provides faster adsorption kinetics	Drinking water, wastewater	65–95% for C8+ PFAS; single-use application	PFOS, PFOA, long-chain PFCA and PFSA	Requires separation after application; higher cost than GAC; no *in situ* regeneration	(Kebede et al., 2025; Rückbeil et al., 2025)
Adsorption on GAC (granular activated carbon)	Hydrophobic surface interactions; electrostatic adsorption at non-ionized active sites	Model water, drinking water	70–99% for C8+ (PFOS, PFOA); 20–45% for C4–C6	Long-chain PFAS (C8+); limited efficiency for PFBS and PFPeA	Competition with NOM/DOM; regeneration requires high-temperature treatment; poor efficiency for short-chain PFAS	

Note: GAC – granular activated carbon; PAC – powdered activated carbon; NOM/DOM – natural/dissolved organic matter; BDD – boron-doped diamond electrode; DI – deionized water; PFCA – perfluoroalkyl carboxylic acids; PFSA – perfluoroalkyl sulfonic acids; PFOS – perfluorooctane sulfonate; PFOA – perfluorooctanoic acid; PFBS – perfluorobutane sulfonate; PFHxS – perfluorohexane sulfonate; HFPO-DA – hexafluoropropylene oxide dimer acid; TFA – trifluoroacetic acid; IX – ion exchange; EC – electrocoagulation; AOP – advanced oxidation process; eaq- – hydrated electron; CAP – cold atmospheric plasma; DBD – dielectric barrier discharge.

or perfluoropentanoic acid (PFPeA). Among degradation techniques, electrochemical oxidation using boron-doped diamond (BDD) electrodes stands out for its exceptional efficiency, with degradation rates exceeding 90% recorded for both long- and short-chain PFAS in real wastewater [27]. Hybrid systems – combining initial separation-based concentration with subsequent destruction – show particular application potential, as they enable broad coverage of the contaminant spectrum while significantly reducing specific energy consumption (Beak et al., 2026; Kim et al., 2024; Lin et al., 2025).

An analysis of the data in Table 4 reveals a distinct lack of research on PFAS behavior and removal in wastewater. Because PFAS molecules resist biodegradation in conventional biological processes (e.g., activated sludge systems), the only mechanism for their partial removal from the wastewater stream is sorption onto sludge flocs. The fraction that does not sorb and passes into the treatment plant effluent requires advanced physicochemical processes. A major operational challenge with separation techniques is generating low-volume but highly concentrated secondary waste streams, such as ion-exchange resin regenerant eluates, membrane system retentates, or concentrated foam from flotation processes. These streams do not solve the problem; they merely transfer the pollutant load to another phase, requiring costly final disposal stages – typically thermal methods or electrochemical degradation (Rao & Liu, 2024; S. J. Smith et al., 2022, 2022).

CONCLUSIONS

The key conclusions from the presented review are as follows:

- The biodegradability of toilet paper depends on its origin (virgin vs. recycled fibers) and the degree of cellulose crystallinity; crystalline regions hinder water penetration and the access of cellulolytic enzymes. Residual lignin and synthetic chemical modifiers – such as PAE resins used to enhance wet strength – create a robust physical and covalent barrier that reduces the rate of cellulose hydrolysis by 30% to 50%.
- The implementation of mechanical separation systems (such as microsieves or rotary belt filters) enables the efficient recovery of cellulose from wastewater, thereby relieving the load on biological reactors and reducing the energy consumption associated with aeration by approximately 25%. The recovered cellulose material can be utilized for energy generation, in construction applications, or in anaerobic digestion processes.
- Toilet paper serves as a significant vector for the transport of persistent chemical pollutants, including PFAS substances (such as 6:2 di-PAP) and their precursors. These compounds are non-biodegradable and have a high potential for bioaccumulation in aquatic organisms, accumulating in visceral organs and causing endocrine and toxic effects.
- The implementation of circular economy principles faces a technological barrier. Recovered

cellulose remains in contact with a complex wastewater matrix and may coexist with undecomposed xenobiotics and PFAS. Because the literature does not conclusively determine the extent of PFAS recovery or concentration in the cellulose fraction, agricultural or environmental applications of recovered sludge risk uncontrolled spread of persistent pollutants.

Acknowledgements

This research was funded by the statutory subvention of Czestochowa University of Technology, Faculty of Infrastructure and Environment.

REFERENCES

1. Abildgård, A., Mikkelsen, S., & Lauridsen, F. (2003). Survey of chemical substances in paper handkerchiefs and toilet paper. *Copenhagen: Danish Environmental Protection Agency*, *34*.
2. Ahmed, A. S. (2020). *Biodegradability Of Cellulose And Its Impact On Oxygen Transfer Efficiency And Biological Nutrients Removal*.
3. Akyol, Ç., Eusebi, A. L., Cipolletta, G., Bruni, C., Foglia, A., Giosuè, C., Frison, N., Tittarelli, F., Canestrari, F., & Fatone, F. (2022). Cellulosic materials recovery from municipal wastewater: From treatment plants to the market. In *Clean Energy and Resource Recovery* (pp. 125–136). Elsevier. https://doi.org/10.1016/B978-0-323-90178-9.00013-5
4. Ali, M., & Sreekrishnan, T. (2001). Aquatic toxicity from pulp and paper mill effluents: A review. *Advances in Environmental Research*, *5*(2), 175–196.
5. Allison, T., Ward, B. D., Harbottle, M., & Durance, I. (2025). Cellulose-based wet wipes undergo limited degradation in river environments. *Environmental Pollution*, *384*, 126971. https://doi.org/10.1016/j.envpol.2025.126971
6. Angelidaki, I., Alves, M., Bolzonella, D., Borzacconi, L., Campos, J. L., Guwy, A. J., Kalyuzhnyi, S., Jenicek, P., & Van Lier, J. B. (2009). Defining the biomethane potential (BMP) of solid organic wastes and energy crops: A proposed protocol for batch assays. *Water Science and Technology*, *59*(5), 927–934. https://doi.org/10.2166/wst.2009.040
7. Appels, L., Baeyens, J., Degrève, J., & Dewil, R. (2008). Principles and potential of the anaerobic digestion of waste-activated sludge. *Progress in Energy and Combustion Science*, *34*(6), 755–781. https://doi.org/10.1016/j.pecs.2008.06.002
8. Arfi, Y., Shamshoum, M., Rogachev, I., Peleg, Y., & Bayer, E. A. (2014). Integration of bacterial lytic polysaccharide monooxygenases into designer cellulosomes promotes enhanced cellulose degradation. *Proceedings of the National Academy of Sciences*, *111*(25), 9109–9114. https://doi.org/10.1073/pnas.1404148111
9. Arvaniti, O. S., Andersen, H. R., Thomaidis, N. S., & Stasinakis, A. S. (2014). Sorption of Perfluorinated Compounds onto different types of sewage sludge and assessment of its importance during wastewater treatment. *Chemosphere*, *111*, 405–411. https://doi.org/10.1016/j.chemosphere.2014.03.087
10. Athey, S. N., & Erdle, L. M. (2022). Are We Underestimating Anthropogenic Microfiber Pollution? A Critical Review of Occurrence, Methods, and Reporting. *Environmental Toxicology and Chemistry*, *41*(4), 822–837. https://doi.org/10.1002/etc.5173
11. Bading, M., Olsson, O., & Kümmerer, K. (2023). *Analysis of Environmental Biodegradability of Cellulose-Based Pharmaceutical Excipients in Aqueous Media*. SSRN. https://doi.org/10.2139/ssrn.4639208
12. Baral, A. (2026). Sustainable hygiene product development from recycled pulp: Surfactant-driven enhancement. *ENGINEERING Biomass*, *1*(1), 83. https://doi.org/10.2738/engb.2026.0005
13. Beak, S., Kim, E., Jung, T., Lee, J., & Bae, J. (2026). Pilot-scale performance of a UV-regenerable Iron-based activated carbon system for pharmaceutical and PFAS removal. *Journal of Water Process Engineering*, *82*, 109468. https://doi.org/10.1016/j.jwpe.2026.109468
14. Behnami, A., Zoroufchi Benis, K., Pourakbar, M., Yeganeh, M., Esrafili, A., & Gholami, M. (2024). Biosolids, an important route for transporting poly- and perfluoroalkyl substances from wastewater treatment plants into the environment: A systematic review. *Science of The Total Environment*, *925*, 171559. https://doi.org/10.1016/j.scitotenv.2024.171559
15. Bhatia, T., Bose, D., Sharma, D., & Patel, D. (2024). A Review on Cellulose Degrading Microbes and its Applications. *Industrial Biotechnology*, *20*(1), 26–39. https://doi.org/10.1089/ind.2023.0025
16. Burkhard, L. P. (2021). Evaluation of published bioconcentration factor (BCF) and bioaccumulation factor (BAF) data for per- and poly-fluoroalkyl substances across aquatic species. *Environmental Toxicology and Chemistry*, *40*(6), 1530–1543. https://doi.org/10.1002/etc.5010
17. Chen, R., Nie, Y., Kato, H., Wu, J., Utashiro, T., Lu, J., Yue, S., Jiang, H., Zhang, L., & Li, Y.-Y. (2017). Methanogenic degradation of toilet-paper cellulose upon sewage treatment in an anaerobic membrane bioreactor at room temperature. *Bioresource Technology*, *228*, 69–76. https://doi.org/10.1016/j.biortech.2016.12.089

18. Chen, Y.-F., Liu, T., Hu, L.-X., Chen, C.-E., Yang, B., & Ying, G.-G. (2024). Unveiling per- and polyfluoroalkyl substance contamination in Chinese paper products and assessing their exposure risk. *Environment International*, *185*, 108540. https://doi.org/10.1016/j.envint.2024.108540
19. Civzele, A. (2025). Biodegradation of screenings from sewage treatment by white rot fungi. *Fungal Biology and Biotechnology*, *12*(1). https://doi.org/10.1186/s40694-025-00198-5
20. Cui, J., & Deng, Y. (2023). Hydrated Electron Degradation of PFOA Laden on Ion-Exchange Resins in the Presence of Natural Organic Matter. *ACS ES&T Engineering*, *3*(1), 86–93. https://doi.org/10.1021/acsestengg.2c00253
21. De Vries, R. P., & Visser, J. (2001). *Aspergillus* enzymes involved in degradation of plant cell wall polysaccharides. *Microbiology and Molecular Biology Reviews*, *65*(4), 497–522. https://doi.org/10.1128/MMBR.65.4.497-522.2001
22. Debnath, M., Salem, K. S., Naithani, V., Musten, E., Hubbe, M. A., & Pal, L. (2021). Soft mechanical treatments of recycled fibers using a high-shear homogenizer for tissue and hygiene products. *Cellulose*, *28*(12), 7981–7994. https://doi.org/10.1007/s10570-021-04024-0
23. Demain, A. L., Newcomb, M., & Wu, J. H. D. (2005). Cellulase, clostridia, and ethanol. *Microbiology and Molecular Biology Reviews*, *69*(1), 124–154. https://doi.org/10.1128/MMBR.69.1.124-154.2005
24. Ersan, M. S., Wang, B., Wong, M. S., & Westerhoff, P. (2024). Advanced oxidation processes may transform unknown PFAS in groundwater into known products. *Chemosphere*, *349*, 140865. https://doi.org/10.1016/j.chemosphere.2023.140865
25. Flemming, H.-C., & Wingender, J. (2010). The biofilm matrix. *Nature Reviews Microbiology*, *8*(9), 623–633. https://doi.org/10.1038/nrmicro2415
26. Franke, V., McCleaf, P., Lindegren, K., & Ahrens, L. (2019). Efficient removal of per- and polyfluoroalkyl substances (PFASs) in drinking water treatment: Nanofiltration combined with active carbon or anion exchange. *Environmental Science: Water Research & Technology*, *5*(11), 1836–1843. https://doi.org/10.1039/C9EW00286C
27. Franke, V., Schäfers, M. D., Joos Lindberg, J., & Ahrens, L. (2019). Removal of per- and polyfluoroalkyl substances (PFASs) from tap water using heterogeneously catalyzed ozonation. *Environmental Science: Water Research & Technology*, *5*(11), 1887–1896. https://doi.org/10.1039/C9EW00339H
28. Ganguly, R. K., & Chakraborty, S. K. (2023). Lignocellulosic biodegradation of solid organic paper mill waste: Succession of enzymes and microbial consortium towards waste valorisation. In *Recent Trends in Solid Waste Management* (pp. 137–154). Elsevier. https://doi.org/10.1016/B978-0-443-15206-1.00013-X
29. Glińska, K., Stüber, F., Fabregat, A., Giralt, J., Font, J., Mateo-Sanz, J. M., Torrens, E., & Bengoa, C. (2020). Moving municipal WWTP towards circular economy: Cellulose recovery from primary sludge with ionic liquid. *Resources, Conservation and Recycling*, *154*, 104626. https://doi.org/10.1016/j.resconrec.2019.104626
30. Gomez-Ruiz, B., Gómez-Lavín, S., Diban, N., Boiteux, V., Colin, A., Dauchy, X., & Urtiaga, A. (2017). Efficient electrochemical degradation of poly- and perfluoroalkyl substances (PFASs) from the effluents of an industrial wastewater treatment plant. *Chemical Engineering Journal*, *322*, 196–204. https://doi.org/10.1016/j.cej.2017.04.040
31. Gupta, M. (2018). *Microsieving as a primary treatment for biological nitrogen removal from municipal wastewater*. https://uwo.scholaris.ca/items/a5bddff4-8c92-4fc6-9bf2-f150b398afda
32. Hajalifard, Z., Kazemi, S., Eftekhari, S., Rezaei, S., Mousazadeh, M., & Usman, M. (2024). Per- and polyfluoroalkyl substances degradation using hydroxyl- and sulphate- radical-based advanced oxidation from water matrices: Which one is the best approach? *International Journal of Environmental Analytical Chemistry*, *104*(20), 9128–9152. https://doi.org/10.1080/03067319.2023.2225412
33. Hanekamp, J. C. (2019). *Risk assessments of 1,3-Dichloro-2-propanol (1,3-DCP; CAS # 96-23-1) and (RS)-3-monochloro-1,2-propanediol (3-MCPD; CAS # 96-24-2)*.
34. Hassan, R. K., Sallam, N. A., Alrefaey, K. A., Shekib, M. A., Su, Y., Salatein, Nahla. M., & Fahim, I. S. (2025). Enhancing PFAS removal: Feasibility of an integrated electrocoagulation and adsorptive membrane approach. *Discover Chemical Engineering*, *5*(1), 23. https://doi.org/10.1007/s43938-025-00089-6
35. Honda, S., & Iwahori, K. (2003). Production of New Adsorbent by Cellulose Recovered from Primary Sludge. *Japanese Journal of Water Treatment Biology*, *39*(2), 67–74. https://doi.org/10.2521/jswtb.39.67
36. Kebede, M. M., Masud, M. A. A., Ortbal, S., Shin, W. S., Mekonnen, M. M., Clement, T. P., & Terry, L. G. (2025). Adsorptive removal of PFAS from aqueous solutions using GAC, PAC and ball-milled colloidal activated carbon: Characterizing Efficiency, Kinetics, and Mechanisms. *ACS ES&T Water*, *5*(11), 6554–6566. https://doi.org/10.1021/acsestwater.5c00641
37. KeChi-Okafor, C., Khan, F. R., Al-Naimi, U., Béguerie, V., Bowen, L., Gallidabino, M. D., Scott-Harden, S., & Sheridan, K. J. (2023). Prevalence and characterisation of microfibres along the Kenyan

and Tanzanian coast. *Frontiers in Ecology and Evolution*, *11*, 1020919. https://doi.org/10.3389/fevo.2023.1020919

38. Khan, M. J., Jovicic, V., Zbogar-Rasic, A., Poser, A., Freichels, K., & Delgado, A. (2022). Effectiveness of non-thermal plasma induced degradation of per- and poly-fluoroalkyl substances from water. *Water*, *14*(9), 1408. https://doi.org/10.3390/w14091408
39. Kim, N., Elbert, J., Shchukina, E., & Su, X. (2024). Integrating redox-electrodialysis and electrosorption for the removal of ultra-short- to long-chain PFAS. *Nature Communications*, *15*(1), 8321. https://doi.org/10.1038/s41467-024-52630-w
40. Kwon, S. (2024). Effect of paper-making additives on biodegradation of lignocellulosic fibers. *BioResources*, *19*(4), 8028–8043. https://doi.org/10.15376/biores.19.4.8028-8043
41. LaFond, J. A., Hatzinger, P. B., Guelfo, J. L., Millerick, K., & Jackson, W. A. (2023). Bacterial transformation of per- and poly-fluoroalkyl substances: A review for the field of bioremediation. *Environmental Science: Advances*, *2*(8), 1019–1041. https://doi.org/10.1039/D3VA00031A
42. Langberg, H. A., Arp, H. P. H., Castro, G., Asimakopoulos, A. G., & Knutsen, H. (2023). *Recycling of Paper and Cardboard, and Pfas?* SSRN. https://doi.org/10.2139/ssrn.4580333
43. Lenka, S. P., Kah, M., & Padhye, L. P. (2022). Occurrence and fate of poly- and perfluoroalkyl substances (PFAS) in urban waters of New Zealand. *Journal of Hazardous Materials*, *428*, 128257. https://doi.org/10.1016/j.jhazmat.2022.128257
44. Li, H., Chang, F., Li, Z., & Cui, F. (2022). The role of extracellular polymeric substances in the toxicity response of anaerobic granule sludge to different metal oxide nanoparticles. *International Journal of Environmental Research and Public Health*, *19*(9), 5371. https://doi.org/10.3390/ijerph19095371
45. Li, S., Wu, Z., Wu, Z., & Liu, G. (2020). Enhancing fiber recovery from wastewater may require toilet paper redesign. *Journal of Cleaner Production*, *261*, 121138. https://doi.org/10.1016/j.jclepro.2020.121138
46. Lin, Z.-W., Kim, S., Dyakiv, Y., Gedangoni, E. F. P., Packman, A. I., Helbling, D. E., Getzinger, G. J., & Dichtel, W. R. (2025). A macroporous cyclodextrin monolith for continuous removal of per- and poly-fluoroalkyl substances from water. *Journal of the American Chemical Society*, *147*(37), 33761–33769. https://doi.org/10.1021/jacs.5c09865
47. Liu, B., Liu, Y.-L., & Sun, M. (2024). Remove legacy perfluoroalkyl acids and emerging per- and polyfluoroalkyl ether acids by single-use and regenerable anion exchange resins: Rapid small-scale column tests and model fits. *Water Research*, *257*, 121661. https://doi.org/10.1016/j.watres.2024.121661
48. Liu, D., Yan, S., Wang, P., Chen, Q., Liu, Y., Cui, J., Liang, Y., Ren, S., & Gao, Y. (2023). Perfluorooctanoic acid (PFOA) exposure in relation to the kidneys: A review of current available literature. *Frontiers in Physiology*, *14*, 1103141. https://doi.org/10.3389/fphys.2023.1103141
49. Liu, R., Li, Y., Zhang, M., Hao, X., & Liu, J. (2022). Review on the fate and recovery of cellulose in wastewater treatment. *Resources, Conservation and Recycling*, *184*, 106354. https://doi.org/10.1016/j.resconrec.2022.106354
50. Malovanyy, A., Forsén, E., & Lihammar, R. (2025). Removal of per- and polyfluoroalkyl substances (PFAS) from municipal wastewater by foam fractionation. *Water Research*, *268*, 122660. https://doi.org/10.1016/j.watres.2024.122660
51. Mofokeng, N. N., Madikizela, L. M., Tiggelman, I., Sanganyado, E., & Chimuka, L. (2024). Determination of per- and polyfluoroalkyl compounds in paper recycling grades using ultra-high-performance liquid chromatography–high-resolution mass spectrometry. *Environmental Science and Pollution Research*, *31*(20), 30126–30136. https://doi.org/10.1007/s11356-024-33250-9
52. Mushtaq, S., Jamil, F., Hussain, M., Inayat, A., Akhter, P., Majeed, K., Khurram, M. S., Aslam, M., Lee, J., & Park, Y.-K. (2023). Unearthing the potential sustainability of cellulose and exploring its source, fate, and recovery. *Environmental Engineering Research*, *28*(6), 230054–0. https://doi.org/10.4491/eer.2023.054
53. Nagamine, R., Kobayashi, K., Kusumi, R., & Wada, M. (2022). Cellulose fiber biodegradation in natural waters: River water, brackish water, and seawater. *Cellulose*, *29*(5), 2917–2926. https://doi.org/10.1007/s10570-021-04349-w
54. Newland, A., Halamek, J., & Ramkumar, S. (2024). Remediation of per- and poly-fluoroalkyl substances in nonwoven production and paper sectors: A brief review. *AATCC Journal of Research*, *11*(3), 229–238. https://doi.org/10.1177/24723444241237301
55. Nir, O., Chaudhary, M., Sela-Adler, M., & Ronen, A. (2023). *Efficient PFOA removal from drinking water by a dual-functional mixed-matrix-composite nanofiltration membrane*. In Review. https://doi.org/10.21203/rs.3.rs-2815368/v1
56. Palmieri, S., Cipolletta, G., Pastore, C., Giosuè, C., Akyol, Ç., Eusebi, A. L., Frison, N., Tittarelli, F., & Fatone, F. (2019). Pilot scale cellulose recovery from sewage sludge and reuse in building and construction material. *Waste Management*, *100*, 208–218. https://doi.org/10.1016/j.wasman.2019.09.015
57. Pittura, L., Nardi, A., Cocca, M., De Falco, F., d'Errico, G., Mazzoli, C., Mongera, F., Benedetti, M., Gorbi, S., Avella, M., & Regoli, F. (2022). Cellular disturbance and thermal stress response in mussels exposed to synthetic and natural microfibers.

Frontiers in Marine Science, *9*, 981365. https://doi.org/10.3389/fmars.2022.981365

58. Rao, D., & Liu, J. (2024). Photochemical PFAS degradation in ion exchange resin regeneration brine: Effects of water matrix components and technical solutions. ChemRxiv, 18 March. https://doi.org/10.26434/chemrxiv-2024-h4bsz
59. Rathinamoorthy, R., Raja Balasaraswathi, S., & Abhinaya, V. (2025). Critical review of cellulosic microfiber pollution: Assessing abundance, sources, and treatment effects on environmental degradation. *Cellulose*, *32*(16), 9209–9230. https://doi.org/10.1007/s10570-025-06793-4
60. Ren, Z., Feng, Z., Liang, H., Zhu, Z., Yang, Y., & Lu, X. (2026). *In situ* surfactant-assisted assembly for efficient removal of PFAS by low-pressure ultrafiltration membrane process. *Environmental Science & Technology*, *60*(2), 2207–2218. https://doi.org/10.1021/acs.est.5c13478
61. Rosińska, A. (2022). *Emerging pollutants wyzwaniem dla gospodarki wodno-ściekowej: Monografia*. Wydawnictwo Politechniki Częstochowskiej.
62. Rückbeil, F. E., Sperlich, A., Kuckelkorn, J., von Wichert, A., Dietrich, C., Hartmann, A., Behnisch, P. A., & Besselink, H. (2025). *Deliverable D4.1 – Performance and assessment of drinking water treatment trains for removal of PFAS and industrial chemicals* (Version This deliverable is subject to final acceptance by the European Comission). Zenodo. https://doi.org/10.5281/ZENODO.14923803
63. Ruiken, C. J., Breuer, G., Klaversma, E., Santiago, T., & Van Loosdrecht, M. C. M. (2013). Sieving wastewater – cellulose recovery, economic and energy evaluation. *Water Research*, *47*(1), 43–48. https://doi.org/10.1016/j.watres.2012.08.023
64. Rusten, B., & Ødegaard, H. (2019). Primary treatment: Particle separation by rotating belt sieves. In G. Mannina, G. Ekama, H. Ødegaard, & G. Olsson, *Advances in Wastewater Treatment*. IWA Publishing. https://doi.org/10.2166/9781780409719_0001
65. Ryan, D. R., Baldus, C. K., Kteeba, S. M., Samuel, M., Dong, Q., Wang, Y., Guo, L., Mayer, B. K., & McNamara, P. (2025). Peroxi-Electrocoagulation for PFAS Mitigation: The Impact of Water Quality and Dissolved Organic Matter on Removal Pathways. *ACS ES&T Engineering*, *5*(5), 1202–1214. https://doi.org/10.1021/acsestengg.4c00854
66. Sequeira. (2016).
67. Shtamm, E. V., Skurlatov, Yu. I., Shvydkii, V. O., Baikova, I. S., & Vichutinskaya, E. V. (2015). Nature of toxic effects caused by sewage from pulp and paper industry on aquatic ecosystems. *Russian Journal of Physical Chemistry B*, *9*(3), 421–428. https://doi.org/10.1134/S1990793115030197
68. Siddiqui, S., Hutton, S. J., Dickens, J. M., Pedersen, E. I., Harper, S. L., & Brander, S. M. (2023). Natural and synthetic microfibers alter growth and behavior in early life stages of estuarine organisms. *Frontiers in Marine Science*, *9*, 991650. https://doi.org/10.3389/fmars.2022.991650
69. Smith, S. (2023). *Innovative treatment technologies for PFAS-contaminated water—Utilizing foam partitioning and exploring electrochemical oxidation: 2023:53*. Swedish University of Agricultural Sciences. https://doi.org/10.54612/a.6p8i5lbs8l
70. Smith, S. J., Wiberg, K., McCleaf, P., & Ahrens, L. (2022). Pilot-scale continuous foam fractionation for the removal of per- and poly-fluoroalkyl substances (PFAS) from landfill leachate. *ACS ES&T Water*, *2*(5), 841–851. https://doi.org/10.1021/acsestwater.2c00032
71. Tan, J., Gao, Y., Xia, Y., Sun, P., & Qin, W. (2025). Investigating the impact of dioxins, furans, and coplanar polychlorinated biphenyls on mortality, inflammatory states, and chronic diseases: An integrative epidemiological analysis. *Ecotoxicology and Environmental Safety*, *289*, 117463. https://doi.org/10.1016/j.ecoenv.2024.117463
72. Thompson, J. T., Chen, B., Bowden, J. A., & Townsend, T. G. (2023). Per- and poly-fluoroalkyl substances in toilet paper and the impact on wastewater systems. *Environmental Science & Technology Letters*, *10*(3), 234–239. https://doi.org/10.1021/acs.estlett.3c00094
73. Thompson, K. A. (2022). Poly- and per-fluoroalkyl substances in municipal wastewater treatment plants in the United States. *ACS ES&T Water*, *2*(5), 690–700. https://doi.org/10.1021/acsestwater.1c00377
74. Topolovec, B., Jovanovic, O., Puac, N., Skoro, N., Lumbaque, E. C., & Petrovic, M. (2024). Plasma water treatment for PFAS: Study of degradation of perfluorinated substances and their byproducts by using cold atmospheric pressure plasma jet. *Journal of Environmental Chemical Engineering*, *12*(3), 112979. https://doi.org/10.1016/j.jece.2024.112979
75. Tserendorj, D., Illés, Á., Károly, Á., Stadler-Szalai, R., Sandil, S., Mireisz, T., Dobosy, P., Pomázi, F., Baranya, S., Adányi, M., & Záray, G. (2024). Microfiber emission from a municipal wastewater treatment plant in Hungary. *Scientific Reports*, *14*(1), 12041. https://doi.org/10.1038/s41598-024-62817-2
76. Vikman, M., Mikkelson, A., & Rautkoski, H. (2024). The impact of lignin content on the biodegradation of virgin paper pulps in soil and marine environment. *BioResources*, *19*(2), 2452–2465. https://doi.org/10.15376/biores.19.2.2452-2465
77. Volgare, M., Avolio, R., Castaldo, R., Errico, M. E., El Khiar, H., Gentile, G., Sinjur, A., Susnik, D., Znidarsic, A., & Cocca, M. (2022). Microfiber contamination in potable water: Detection and mitigation using a filtering device. *Microplastics*, *1*(3), 322–333. https://doi.org/10.3390/microplastics1030024
78. Wan, N., Liu, Y., Zhang, X., Liu, Z., Wang, Q., Liu,

S., Zhang, M., & Mai, B. (2024). Legacy and emerging per- and polyfluoroalkyl substances in tissue and toilet paper from China. *Journal of Hazardous Materials Letters*, *5*, 100109. https://doi.org/10.1016/j.hazl.2024.100109

79. Wang, X., Liu, G., Sun, W., Cao, Z., Liu, H., Xiong, Y., Li, B., Sun, X., Li, Y., Xu, R., Huang, D., & Gao, P. (2023). Removal of toilet paper fibers from residential wastewater: A life cycle assessment. *Environmental Science and Pollution Research*, *30*(35), 84254–84266. https://doi.org/10.1007/s11356-023-28291-5
80. Wang, Y., Ji, Y., Tishchenko, V., & Huang, Q. (2023). Removing per- and polyfluoroalkyl substances (PFAS) in water by foam fractionation. *Chemosphere*, *311*, 137004. https://doi.org/10.1016/j.chemosphere.2022.137004
81. Xia, C., Lim, X., Yang, H., Goodson, B. M., & Liu, J. (2022). Degradation of per- and polyfluoroalkyl substances (PFAS) in wastewater effluents by photocatalysis for water reuse. *Journal of Water Process Engineering*, *46*, 102556. https://doi.org/10.1016/j.jwpe.2021.102556
82. Yang, H., Joe, H. J., Park, S.-J., Kim, S. K., & Lee, C.-G. (2024). Enhanced removal of perfluorooctanoic acid (PFOA) from water using ZnO-Ag-ZnAl2O4 composites: Performance and mechanistic insights. *Journal of Water Process Engineering*, *61*, 105288. https://doi.org/10.1016/j.jwpe.2024.105288
83. Yang, L., Ma, X., Luan, Z., & Yan, J. (2021). The spatial-temporal evolution of heavy metal accumulation in the offshore sediments along the Shandong Peninsula over the last 100 years: Anthropogenic and natural impacts. *Environmental Pollution*, *289*, 117894. https://doi.org/10.1016/j.envpol.2021.117894
84. Zambrano, M. C. (2021). Impact of dyes and finishes on the aquatic biodegradability of cotton textile fibers and microfibers released on laundering clothes. *Marine Pollution Bulletin*, *165*, 112030. https://doi.org/10.1016/j.marpolbul.2021.112030
85. Zarina, R., & Mezule, L. (2024). Enzymatic hydrolysis of waste streams originating from wastewater treatment plants using enzyme cocktail. *Biotechnology for Biofuels and Bioproducts*, *17*(1). https://doi.org/10.1186/s13068-024-02553-x
86. Zhang, Z., Zhang, J., Dai, Y., Wang, L., Wu, Z., & Wang, Q. (2026). Microsieving-Based Advanced Primary Treatment: A Promising Technology for Carbon Redistribution and Recovery for Wastewater Treatment. *Processes*, *14*(9), 1412. https://doi.org/10.3390/pr14091412
87. Zhu, Y., Leary, R. N., Xu, T., He, K., Blaney, L., Hao, X., & Zhao, D. (2026). Photocatalytic degradation of PFAS under field water matrix conditions using an adsorptive photocatalyst. *Water Research X*, *30*, 100485. https://doi.org/10.1016/j.wroa.2026.100485