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Review

Review of water treatment technologies for PFAS from a life cycle perspective, with meta-analysis of financial costs and climate impacts

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ABSTRACT

Per- and polyfluoroalkyl substances (PFAS) contamination of drinking water is now a critical environmental and public health concern. Conventional water treatment is ineffective, prompting investment in solutions like granular activated carbon, ion exchange, membrane filtration, foam-fractionation and electrochemical oxidation. However, selecting appropriate technologies involves trade-offs among performance, resource use and environmental impact criteria. Our analysis aims to offer new insights into the climate impacts per gram of PFAS removed and the annual capital and operational costs per volume water treated. We also highlight critical limitations of environmental assessment of PFAS treatments, particularly regarding toxicity-related impacts, that have not kept pace with developments in life cycle assessment methodology. Our analysis synthesizes data from 17 disparate publications on PFAS treatment technologies. Emissions from innovative treatments vary widely, with climate impacts ranging from 0.1 to 70 190 kg CO_2 eq. per gram of PFAS depending on raw water PFAS concentrations. The economic analysis showed that operational costs span from \$0.03/m^3 to \$28/m^3, while capital expenditures range from \$0.01 to \$0.51/m^3 of water treated and exhibit some economies of scale. This work also underscores the importance of using life cycle assessment and life cycle costing approaches to comprehensively evaluate PFAS removal technologies.

1. Introduction

As more is understood regarding the toxicological impacts of per- and polyfluorinated alkyl substances (PFAS), interest in removing these contaminants from drinking water sources is increasing. The legal limits for PFAS concentration in drinking water have become more restrictive over time. For instance, for 2026, Sweden's Drinking Water Directive sets limit values for PFAS 4[1] and PFAS 21[2] at 4 ng/l and 100 ng/l, respectively (Swedish Food Agency, 2022). Previously, the recommended action level in drinking water was 90 ng/l for PFAS 11[3] (Swedish Food Agency, 2016).

Other countries have different regulations. For instance, Australia's guidelines set limit values for PFOS and PFHxS at 70 ng/l, and for PFOA at 560 ng/l (Australian Government, 2022). The U.S. Environmental Protection Agency (2022) established maximum contaminant levels of 4 ng/l for PFOS and PFOA, and 10 ng/l for PFHxS, PFNA, and HFPO-DA. The Government of Canada (2024) has set a limit of 30 ng/l for the sum of 25 specific PFAS.[4]

PFAS concentrations surpassing recommended thresholds have been identified in drinking water. The substantial health-related costs associated with PFAS exposure, estimated to range from 52 to 84 billion euros annually for the European Economic Area (Goldenman et al., 2019), and 5.52 to 62.6 billion dollars annually for the U.S. (Obsekov et al., 2023) highlight the urgency of effective mitigation strategies. However, conventional treatment methods have proven inadequate in eliminating PFAS from drinking water (Wee and Aris, 2023; Franke et al., 2021; Belkouteb et al., 2020; Appleman et al., 2014). Numerous studies are underway to develop and apply innovative processes for

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[1] Sum of the following four substances: PFOS, PFOA, PFNA and PFHxS. The complete names of the substances can be found in the Supporting Information (SI).

[2] Sum of twenty-one substances: PFBA, PFPA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, PFBS, PFPS, PFHxS, PFHpS, PFOS, PFNS, PFDS, PFUnDS, PFDoDS, PFTrDS, 6:2 FTS.

[3] Sum of eleven substances: PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS, 6:2 FTSA.

[4] Sum of twenty-five substances: PFBA, PFNA, PFPeS, 6:2 FTS, PFMBA, PFPeA, PFDA, PFHxS, 8:2 FTS, NFDHA, PFHxA, PFUnA, PFHpS, HFPO-DA, 9Cl-PF3ONS, PFHpA, PFDoA, PFOS, ADONA, 11Cl-PF3OUdS, PFOA, PFBS, 4:2 FTS, PFMPA, PFEESA.

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treating various waters contaminated with PFAS — processes such as granular activated carbon (GAC) filters, ion exchange (IEX) resins, nanofiltration (NF) and reverse osmosis (RO) membranes, and foam-fractionation (FF). However, these technologies generate PFAS-laden residuals or wastes that require further treatment or disposal. To address this, electrochemical oxidation (EO) is being studied as a potential method for PFAS destruction, in which PFAS compounds are broken down into less harmful substances—ideally carbon dioxide, water, and fluoride ions.

These innovative treatments can significantly decrease PFAS concentrations in drinking water and enhance the management of waste generated during treatment. Given the range of technological options and the broad spectrum of potential tradeoffs—including energy use, material resource demands, environmental impacts and economic costs—technology selection has sustainability consequences. Moreover, there is in principle a concentration limit at which further reduction for PFAS contaminants causes net damage to the environment and thus indirectly to human wellbeing. Therefore, holistic assessments are necessary.

Holistic comparison of environmental impacts and economic costs of treatment options can employ life cycle assessment (LCA) and life cycle costing (LCC). LCA is a technique for assessing the potential environmental impacts associated with a product throughout its life cycle. LCA helps in identifying the main sources of environmental and human health impacts of a process, in improving one area without leading to unintended negative consequences in other areas (burden-shifting), and can provide a foundation for making policy recommendations (Lei et al., 2023). LCC is a method used to evaluate the total cost of a project over its entire life span, and it can provide a comprehensive view of the financial implications of a decision, aiding in determining the most cost-effective option (Rödger et al., 2018). LCC requires consideration of a discount rate to account for the time value of money and determine the net present value of the investment (Hunkeler et al., 2008). The high costs of treating water can impact decision-making on the implementation of novel technologies. Therefore, LCC can offer a comprehensive view of financial impacts over time (Valladares Linares et al., 2016).

The environmental and economic viability of PFAS treatment technologies has been the topic of a few reviews. They shed light on the available technologies for PFAS remediation (Sharma et al., 2024), the current state of research on LCA of PFAS removal (Song et al., 2024), and proposed frameworks for future studies focusing in developing and achieving sustainable PFAS treatment (Tushar et al., 2024). However, the challenge of quantitatively integrating the data presented by different authors using a common metric was not undertaken in these reviews.

The present study aims to fill that gap by extracting and recalculating literature results (here called meta-analysis) of recent case studies, offering new insights into the climate impacts per gram of PFAS removed and the annual capital and operational costs per volume of water treated. Additionally, we aim to provide information about methodological limitations, and information deficiencies in environmental and economic assessments of emerging technologies for PFAS removal. This information can assist analysts, technology developers, and utility managers in evaluating PFAS removal technologies.

We address three key research questions: (1) What is the scope of existing LCA and cost assessments applied to novel PFAS removal technologies? (2) What is the range of potential climate impacts and financial costs associated with PFAS removal in published studies, and what factors drive these outcomes? (3) What shortcomings exist in LCA studies related to innovative PFAS removal technologies? Through this study, we aim to deliver analysis of reported carbon footprints and annual costs associated with water treatment technologies for persistent fluorinated substances, while also identifying current limitations in LCA and cost assessments and highlighting priorities for future research and development.

2. Materials and methods

2.1. Compiling earlier studies

A scoping review (Sutton et al., 2019; Munn et al., 2018; Arksey and O'Malley, 2005) was conducted using Scopus and Web of Science. Documents were retrieved utilizing title, keywords, and abstract as search fields. Relevant publications from any year were sought. The search was conducted in English, from April to October 2024. The final database search string is provided in Table I in Supporting Information (SI), which resulted in 81 articles found.

Based on these results, metadata from Scopus and Web of Science—specifically titles and abstracts—were imported and analyzed using the online tool Rayyan. Rayyan streamlined the initial evaluation by organizing and visually highlighting relevant search terms within the metadata, while its advanced tagging features enabled efficient sorting of papers for further review or exclusion. After removing duplicates, this metadata-based screening identified twenty-six candidate papers (Figure I in SI). A second review then focused on reading each article's methods, results, and conclusions, applying predefined inclusion and exclusion criteria to determine which articles would be included in the final analysis (Table II in SI).

Sixteen papers, culled from the 26 candidate papers, were ultimately selected for detailed reading. Furthermore, two studies were found by other means: (1) the study by McNamara et al. (2018) was identified following a review of Belkouteb et al. (2020), and (2) the study by Li et al. (2022b) was discovered after reading the literature review by Song et al. (2024). However, following the exclusion criteria established, Li et al. (2022b) was not included in the final dataset.

The final dataset thus consists of seventeen texts spanning the years 2018–2024. Eight of these focus on the LCA of innovative technologies for treating PFAS-contaminated water and wastewater. Among them, four articles also mention an economic analysis of PFAS treatment: two apply LCC, while the other two evaluate specific costs associated with treatment, such as capital investment and operation and maintenance. Additionally, one article exclusively focuses on LCC to compare the costs of various PFAS treatment technologies over a payback period.

Since only a few articles were found on LCC, eight articles were included in this review where the authors only analyzed the price of water treatment. The main difference between LCC and other cost analyses is that the former encompasses the costs for treatment infrastructure, operation and maintenance over the treatment technology's lifetime, where an annual discount rate is applied. The latter focuses mainly on costs related to the treatment operation phase, annually or over a determined timeframe. To answer the research questions, the selected studies were thus reviewed, and quantitative data was extracted for performing meta-analysis, as described below.

2.2. Meta-analysis of contributions to climate change

We define meta-analysis of LCAs according to Zumsteg et al. (2012), the process of systematically synthesizing results from multiple LCA studies through quantitative methods. This involves pooling data from different LCA studies, which may have varying parameters and methodologies, to achieve a more comprehensive understanding of the environmental impacts associated with a product or system across its life cycle.

In a meta-analysis following the methodology established by Behjat et al. (2022), Hermansson et al. (2019) and Zumsteg et al. (2012), we carefully analyzed environmental impact results from various published LCA papers. The variations in functional units (FUs), system boundaries, environmental impact categories, and specific inventory data across studies made direct comparisons challenging. To address this, each study's details were thoroughly evaluated, ensuring a comprehensive and consistent approach for preliminary evaluations.

Since contributions to climate change (CCC) is the sole

environmental impact category examined across all the LCA studies reviewed, it was selected for the meta-analysis to consolidate data on CO_2 equivalent emissions generated during treatment[5] of PFAS-contaminated waters. In this context, mathematical evaluations were carried out to connect information and data to address queries that cannot be answered by current standalone studies.

Data on inlet concentrations of PFAS were gathered from the LCA articles. When available, outlet concentration data was collected. In some cases, the articles only mentioned the efficiency of the treatment technologies (percentage of PFAS removal), therefore the PFAS concentration in the effluent was calculated. Additionally, information about the volume of water treated, the FU of the LCAs and the total amount of CO_2 equivalent emitted per FU was documented. The study from Boyer et al. (2021) presented climate change impact results normalized to the average annual impact of a U.S. citizen in 2008, so adjustment to the unit of kg CO_2 eq. was required, which was done by multiplying it by a normalization factor of $2.4x10^4$ kg CO_2 eq. given by Ryberg et al. (2014). After that, the total amount of PFAS removed and/or destroyed from the total volume of contaminated waters treated by the technologies was calculated together with the amount of CO_2 eq. emitted per gram of PFAS treated. Of the eight LCA articles included in this review, seven provided sufficient data for this meta-analysis (Figure I in SI). From these, 30 treatment scenarios were extracted and labeled T01 through T30.

The results were thus expressed in kg CO_2 per gram of PFAS treated[6] and categorized according to two distinct phases of the treatment process: installation and operation. This categorization was guided by the systems boundaries defined in the LCAs and the data reported in the relevant studies. Furthermore, when possible, we split each treatment scenario into (1) PFAS removal (for technologies capable of transferring PFAS from the contaminated water to either solid media or a waste stream, such as GAC, IEX and membranes) and (2) PFAS destruction (for technologies that utilize either high temperature alone or a combination of high temperature and pressure to mineralize PFAS compounds, such as incineration and reactivation of GAC, or advanced oxidation processes like electrochemical oxidation).

In some instances, additional assumptions were necessary. For example, the manufacturing of IEX resin and GAC was included under the operation phase, as these sorbents, whether used once or regenerated/reactivated, are integral to the main treatment process and have a shorter lifespan than other inventory items like concrete and steel construction elements. The destruction of PFAS-laden residuals by either incineration or electrochemical oxidation was also included in the operation phase, since it is still part of the treatment process.

2.3. Meta-analysis of cost assessments

The methodology for the economic analysis follows the LCC structure, specifically dividing costs into capital expenditures (CAPEX) and operational expenditures (OPEX). Decommissioning costs are excluded from this analysis, as they were not assessed in the 13 papers reviewed on costs of PFAS treatment technologies. CAPEX refers to the initial investment cost incurred during the project development and construction. It typically includes expenses for equipment procurement, infrastructure assembly, and construction. OPEX, on the other hand, represents the ongoing expenses throughout the lifespan of the treatment technology, covering operation and maintenance costs. This includes expenditures on electricity, replacement of adsorbents or media, chemicals, transportation, disposal of spent adsorbents, and, in some cases, labor costs associated with operations. From the review sample, 59 distinct treatment scenarios were identified, as detailed in Table V in the SI. Of these, 17 included capital investment costs, while all 59 provided data on operation and maintenance costs (Table VI in SI).

CAPEX and OPEX were annualized according to the project lifetime specified in the studies. Eq. (1) was applied to calculate the annual CAPEX. If not specified in the study, a 20-year time horizon was used for CAPEX payment, based on information gathered from other articles included in this review. Eq. (2) was used to calculate annual CAPEX per cubic meter of PFAS-contaminated water treated. The volume of water treated per year was calculated based on data gathered from the studies. In addition, information about the discount rates used in the LCC studies to calculate the present value of future costs over the treatment technology lifecycle was extracted.

$$Annual\ CAPEX = \frac{Total\ CAPEX}{time\ horizon\ set\ for\ payment} \tag{1}$$

$$Annual \frac{CAPEX}{m3} of\ water\ treated = \frac{Annual\ CAPEX}{Volume\ of\ water\ treated\ per\ year} \tag{2}$$

For OPEX, the majority of studies report operational costs on an annual basis. Eq. (3) was used to calculate the annual OPEX per cubic meter of water treated. In cases where the studies provided the volume of PFAS-contaminated water on a daily or hourly basis, it was assumed that the facility operated continuously, 24 h a day, 365 days a year, unless a different operating schedule was explicitly stated in the study.

$$Annual \frac{OPEX}{m3} of\ water\ treated = \frac{Annual\ OPEX}{Volume\ of\ water\ treated\ per\ year} \tag{3}$$

Some articles expressed the results in currencies other than U.S. dollars (USD), such as euros. In these cases, the values were converted to USD using European Central Bank (2024) exchange rates, either from the year of the original data or, when that information was unavailable, from the year of the article's publication. Additionally, the costs found were adjusted for inflation to the year of 2024 using the consumer price index (CPI) inflation calculator from U.S. Bureau of Labor Statistics (2024).

2.4. Statistical analysis

To provide insights into different variables that influence climate impact and financial aspects, regression analyses were conducted in Microsoft Excel on parameters that could be extracted from the literature. The strength and nature of the correlations were assessed using the coefficient of determination (R^2), while p-values ($p < 0.05$) were used to indicate statistical significance. Specifically, these analyses explored the extent to which climate impact and annual treatment costs for PFAS-contaminated water are driven by inlet concentration, PFAS concentration reduction, outlet concentration, volume of water treated per year, plant scale and the age of the study. In addition to these factors that suited regression analysis, we considered the choice of technology and performed a hotspot analysis to find the parts of the process life cycle that cause the most environmental damage.

3. Results and discussion

3.1. The scope of recent case studies

The LCA studies reviewed were performed to understand and compare the environmental and human health life cycle impacts of the use of different novel technologies to remove PFAS or different configurations of one technology. The studies employ different functional units (FU) as listed in SI Table III. They relate to water volumes treated (sometimes considering a maximum concentration of PFAS in treated water) or are stated as orders of magnitude removal of PFAS from a designated water volume. These FUs are all relatable to the function of

[5] By treatment we mean the removal and/or destruction of PFAS, depending on the technology applied.

[6] By "treated," we refer to the concentration of PFAS that has been removed and/or destroyed, depending on the applied technology, and scaled according to the treatment efficiency.

the water treatment technologies. The water resources under analysis include groundwater (for drinking or remediation purposes), side-streams of water treatment, landfill leachate and fire-extinguishing waters containing aqueous film-forming foams (AFFF).

The studies used different system boundaries, as presented in Fig. 1, but the focus of the majority of studies was on the operation phase of the life cycle of treatment technologies. The majority of the LCAs focused on GAC and IEX resins,[7] featured in 62 % and 75 % of the studies, respectively. Other technologies evaluated included nanofiltration (NF) membranes, reverse osmosis (RO), precipitation agents, electrochemical oxidation (EO), and incineration. These treatment systems were assessed individually or as part of integrated treatment trains (Table III in SI).

From the 13 cost-focused studies we reviewed, 3 applied LCC. The LCC approach was used to assess and compare the total cost of PFAS treatment technologies across their full life cycle, using net present value - NPV (Ellis et al., 2023; Quinnan et al., 2023; Feng et al., 2021). The studies considered capital investment, operational and maintenance expenses, and disposal costs, all discounted over a specific time frame. 6 studies focused exclusively on operating costs (Jiang et al., 2024; Malovanyy et al., 2023; Franke et al., 2021; Murray et al., 2021; Belkouteb et al., 2020; Emery et al., 2019), while 4 also included capital expenditures for acquiring innovative PFAS treatment technologies (Ling et al., 2023; Kanchanapiya and Tantisattayakul, 2022; Moeini et al., 2022; McNamara et al., 2018). To address uncertainty, these studies conducted sensitivity analyses for examining variables such as energy prices or media replacement rates and their impact on overall costs. From these 13 studies, only 4 combined cost assessments with LCA to provide a more comprehensive evaluation of both economic and environmental impacts (Ellis et al., 2023; Moeini et al., 2022; Feng et al., 2021; Emery et al., 2019).

For assessing technologies' climate impact, all LCA studies used global warming potential characterization factors (CFs) based on a 100-year period. In general, the life cycle impact assessment (LCIA) methods are based on internationally accepted models and factors, produced by the Intergovernmental Panel on Climate Change - IPCC (Hauschild et al., 2013), hence, providing similar results and using the same units (kg CO_2 eq.).

Table 1 presents a brief technical description of the 30 PFAS treatment scenarios extracted from the LCA studies and Fig. 1 shows the general system boundaries of the LCA studies. The technologies were applied to flows that differed in terms of PFAS species and concentration as well as physicochemical characteristics. In addition, the treatment technologies had different efficiencies for PFAS removal and/or destruction and different lifespan.

3.2. Contributions to climate change per gram of PFAS treated

For the treatment of a gram of PFAS, the median climate impact was 88 kg. In 53 % of cases, the innovative treatment trains emit 10 to 1 000 kg of CO_2 equivalent per gram of PFAS treated (Table VII in SI). For PFAS removal (Fig. 2), activated carbon shows relatively low emissions, usually ranging from 1 to 80 kg of CO_2 eq./g PFAS removed, with exception of two scenarios, T13 and T19, which present 443 and 13 321 kg of CO_2 eq./g PFAS removed respectively. These two present the lowest PFAS inlet concentrations among the activated carbon treatment scenarios and removal efficiencies of 86 % and 99 %, correspondingly.

The emissions caused by removing a gram of PFAS with IEX as the first step vary from 0.4 to 462 kg of CO_2 eq. Membrane filtration presented emissions between 82 and 122 kg of CO_2 eq./g PFAS removed. The three highest climate impacts (ranging from 57 900 to 70 190 kg CO_2 eq./g PFAS removed) concern small-scale, point-of-use (POU) systems that combine GAC, IEX and reverse osmosis (RO) membranes in different configurations to treat contaminated groundwater with the lowest inlet concentration of PFAS among the studies—0.02 µg/L for the sum of PFOS and PFOA. The destruction of PFAS by electrochemical oxidation presented emissions between 65 and 1 896 kg of CO_2 eq./g PFAS destroyed. Incinerating spent media or PFAS-laden residuals generates climate impact ranging from 0.02 to 142 kg CO_2 eq./g PFAS destroyed (Fig. 3).

The astonishing range of emission estimates underscores the importance of selecting appropriate treatment methods to minimize the environmental impacts. The higher numbers also beg the question: are the environmental costs of eliminating the risks associated with ingesting PFAS-contaminated water justified? Alternatively, do the significant CO_2 emissions linked to certain treatment technologies provide sufficient grounds to advocate for a comprehensive ban on PFAS production and use in industry? These questions should also be considered in relation to the financial cost of treatment and the toxicity of PFAS—the subjects of the next sections of this paper.

Heterogeneity is to be expected in a meta-analysis, given that the studies were conducted by different teams on different waters with PFAS contamination (Abolli et al., 2023). Our examination of the underlying factors points to several influential parameters. The most apparent of these is the concentration of PFAS in raw water. This is also reflected in regression analysis on the mass of CO_2 eq. emitted per gram of PFAS treated in relation to the reduction in PFAS concentration (see Figure V in SI). Higher inlet concentrations of PFAS per liter of contaminated water, as well as higher reductions, are associated with lower emissions per gram. These results demonstrate the entropic penalty of treating water with lower PFAS concentrations in terms of the energy and other resources required per unit PFAS removed compared to treating water with high concentrations of PFAS. Put another way, it is easier to "fight entropy" with treatment technologies like GAC and IEX resins when the concentration of PFAS is higher because higher concentrations create a greater driving force for mass transfer, which enhances the rate at which contaminants move from the water to the treatment media. However, pre-treatment steps are important to remove other compounds, such as total organic carbon, that can compete with PFAS for adsorption sites (U.S. EPA, 2024).

On the other hand, the regression analysis indicates that neither plant scale—measured by flow rate—nor the age of the studies significantly influences the variability in climate impact, as evidenced by weak and non-significant correlations between these variables (Figure IV in SI). Regarding the apparent absence of economies of scale, we note that flow rate data was available for only 63 % of the treatment scenarios analyzed. These scenarios were derived from five different studies, which presented identical flow rate values across the different treatment scenarios. So the apparent absence of economies of scale may be due in part to the small data set available for these analyses.

The focus of most LCA studies on the operational phase of treatment technologies, identified earlier in this article (Fig. 1), makes sense since, on average, the operation phase contributed to 87 % of climate impacts. For 63 % of the treatment scenarios, most emissions originate from the operation of technologies to remove PFAS from contaminated waters; for the other 37 % they came mainly from the treatment of PFAS-laden residuals generated by these processes. The contribution to climate change of the construction, assembly or installation and end-of-life phases of treatment technologies was shown to be minimal (Figure II in SI). For the removal of one gram of PFAS, the median climate impact was 80 kg CO_2 eq., whereas for its destruction, the value was 23 kg CO_2 eq. This indicates that, overall, PFAS removal imposes a significantly higher climate burden than its subsequent destruction. In the studies reviewed, destruction processes are implemented after PFAS has been removed from contaminated water, targeting the PFAS-laden residuals generated during treatment. This suggests that removal methods tend to involve more energy-intensive or resource-heavy operations compared

[7] There are two main types of IEX resins: (1) cation exchange resins that remove positively charged ions (cations), and (2) anion exchange resins (AEX), used to remove negatively charged ions (anions), such as PFAS. Some studies used the term AEX. In this study we designated them only as IEX.

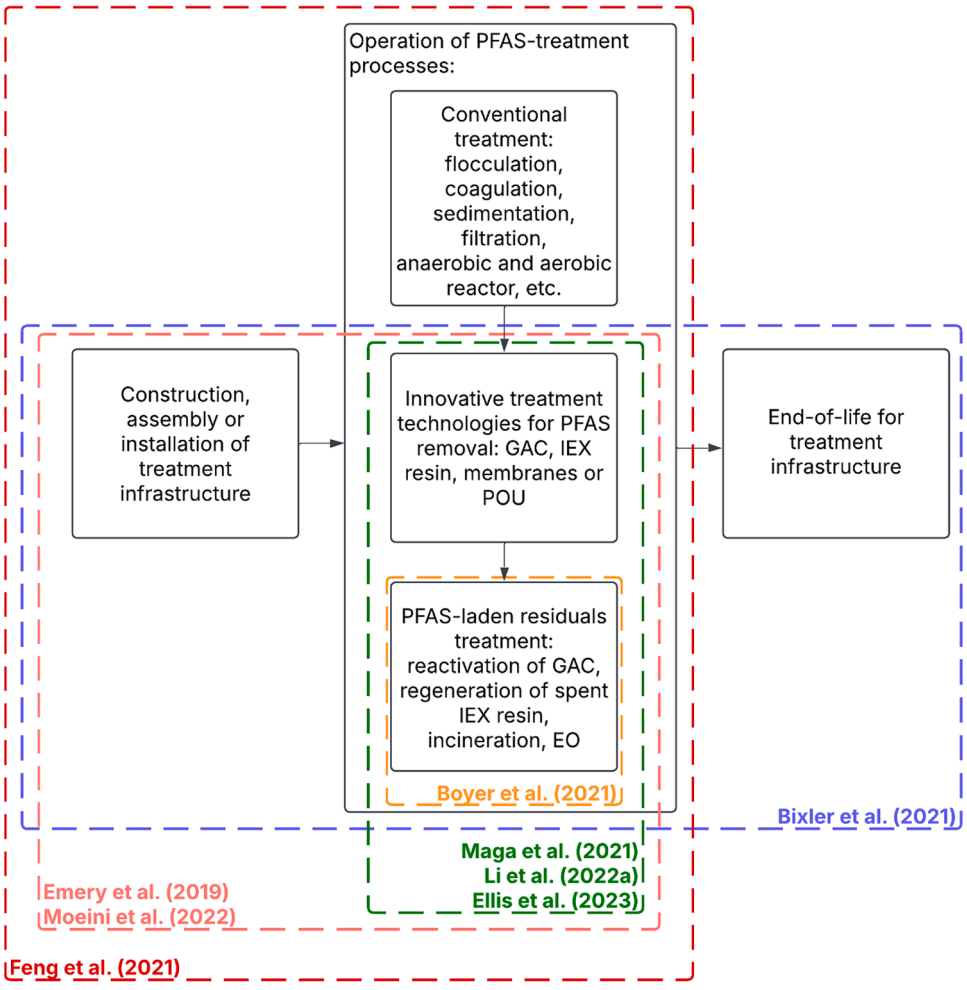

Fig. 1. General system boundaries of LCA studies included in the analysis (life cycle phases of treatment technologies are shown from left to right and PFAS-contaminated water treatment processes are shown from top to bottom).

to the processes used for destroying the concentrated residuals.

The energy required to treat PFAS-contaminated waters—including for system operation, material production, reactivation or regeneration of GAC and IEX resin, and transportation—was identified as a significant contributor to the climate impact. In POU systems where RO is applied, energy consumption is an especially significant contributor to climate change since energy is required to pressurize and push water through the membrane, resulting in greenhouse gas emissions from electricity generation. Most of the LCA studies used an energy mix from the USA, where 60 % of the electricity is generated from fossil fuels—coal, natural gas and petroleum (U.S. EIA, 2023).

In scenarios where the treatment of PFAS-laden residuals led to the highest CO_2 emissions, single-use IEX resin (T26) and the contaminated regeneration solution from regenerable IEX systems (T02, T07, T08, T09) were treated through incineration, while electrochemical oxidation (EO) was used to treat the still bottoms produced during IEX resin regeneration (T26, T27, T28, T29, T30). Additionally, in POU systems incorporating RO (T20, T22), a significant volume of reject water is generated and sent to a wastewater treatment plant. The primary contribution to climate change arises from the additional energy required for pumping and treating this reject water, although in that case the PFAS is expected to pass through such treatment unchanged.

In the EO process, electrical energy is used to directly trigger chemical reactions that break down PFAS in water. The amount of energy needed varies depending on the PFAS species. For instance, destruction of PFOA requires 14 MWh/g, while PFOS removal needs 8.8 MWh/g (Li et al., 2022a). This illustrates how the climate impact of PFAS destruction is dependent on the chemical species.

The use of chemicals in treatment processes for regenerating IEX resins can also significantly contribute to climate change due to emissions from their production and transportation. For example, when methanol and brine solutions containing sodium chloride (NaCl) are substituted by another chemically produced salt, such as ammonium chloride (NH_4Cl), or potassium carbonate (K_2CO_3), the consumption of chemicals can account for up to 25 % of total environmental impacts (Boyer et al., 2021). Ellis et al. (2023) found that PFAS treatment with single-use IEX resin generates less environmental impacts than regenerable IEX because of chemical-intensive processes to regenerate resins and recycle the regenerant solution cosolvent, which can be a major contributor to climate change. Recycling of cosolvent and brine can mitigate some of these impacts, while the choice of chemicals in the regeneration solution plays a critical role in determining the overall environmental footprint. Li et al. (2022a) mentions the need for sustainable disposal methods to avoid environmental harm.

3.3. *Cost assessment of innovative PFAS treatment technologies*

Water treatment investment decisions require a thorough technical and commercial evaluation tailored to the unique conditions of the project. When selecting technology for drinking water treatment plants, the initial capital expenditure (CAPEX) often becomes the primary focus, overshadowing other critical factors (Cherukumilli et al., 2023; Ellis et al., 2023). This approach can lead to suboptimal decisions that may not account for long-term economic sustainability and operational

Table 1
Overview of PFAS treatment technologies considered in the selected LCA studies, and the short names used in this paper (T01-T30).

Treatment scenarios	Authors	Treatment methods	Lifespan of treatment technology (years)	PFAS species considered	Inlet concentration of PFAS (μg/L)	PFAS removal efficiency
T01	Ellis et al. (2023)	Single-use IEX resin with off-site incineration of spent media	30	PFOS+PFOA	50.00	100 % [a]
T02		IEX with on-site resin regeneration + still bottoms incineration				
T03		Single-use GAC with off-site incineration of spent media				
T04		GAC with off-site thermal reactivation				
T05	Feng et al. (2021)	Onsite scenario (neutralization, coagulation/ flocculation, UASB, anaerobic and aerobic A/O reactors, ultrafiltration membranes, NF, and RO)	15	Undefined PFAS (average of 29 PCB compounds used as toxicity proxy)	150.704	99 %
T06		Offsite scenario (leachate onsite storage, truck transportation, and leachate treatment in a local WWTP composed of pretreatment (fine screen and grit chamber), 3-stage anaerobic/anoxic/aerobic treatment, ultrafiltration membranes, and UV disinfection)				21 %
T07	Boyer et al. (2021)	Disposal of PFAS-contaminated waste from IEX regeneration solution via off-site incineration	Not mentioned	PFOS+PFOA	70.00	100 % [b]
T08		Partial recycling of IEX waste regeneration solution via onsite recovery and reuse of methanol cosolvent before off-site incineration of PFAS-contaminated distillation bottoms				
T09		Full recycling of IEX waste regeneration solution via onsite recovery and reuse of methanol cosolvent and brine before off-site incineration of PFAS-loaded GAC				
T10	Maga et al. (2021)	Aqueous film-forming foams (AFFF) treatment with precipitation, activated carbon and incineration	Not mentioned	6:2 FTS	23000.00	99 %
T11		AFFF treatment with activated carbon and incineration				
T12		AFFF incineration				
T13	Emery et al. (2019)	GAC with reactivation for 0.7 μg/L PFOS+PFOA	20	PFOS+PFOA	0.70	99 %
T14		GAC with reactivation for 7 μg/L PFOS+PFOA			7.00	
T15		GAC with reactivation for 70 μg/L PFOS+PFOA			70.00	
T16		IEX with regeneration for 0.7 μg/L PFOS+PFOA + GAC for concentrated brine + incineration for GAC			0.70	
T17		IEX with regeneration for 7 μg/L PFOS+PFOA + GAC for concentrated brine + incineration for GAC			7.00	
T18		IEX with regeneration for 70 μg/L PFOS+PFOA + GAC for concentrated brine + incineration for GAC			70.00	
T19	Bixler et al. (2021)	Centralized scenario (GAC filters) for PFOS + PFOA removal + landfill disposal for spent GAC	50	PFOS PFOA	0.00238 [c] 0.02 [c]	85.3 % 98.7 %
T20		Point-of-use (POU) scenario GAC/IEX+RO for PFOS + PFOA removal	10	PFOS PFOA	0.00239 [c] 0.01985 [c]	96 % 96 %
T21		POU scenario GAC/IEX for PFOS + PFOA removal		PFOS PFOA	0.00242 [c] 0.0201 [c]	96 % 96 %
T22		POU scenario GAC+RO+IEX for PFOS + PFOA removal		PFOS PFOA	0.00245 [c] 0.0203 [c]	96 % 96 %
T23	Li et al. (2022a)	Single-use IEX + incineration for PFOS	Not mentioned	PFOS	0.397 [c]	Two orders of removal
T24		Single-use IEX + incineration for PFBA		PFBA	0.114 [c]	
T25		Single-use IEX + EO for still bottoms (PFOS)		PFOS	0.397 [c]	
T26		Single-use IEX + EO for still bottoms (PFBA)		PFBA	0.114 [c]	
T27		IEX with regeneration + EO for still bottoms + incineration for spent resin (PFOS)		PFOS	0.397 [c]	
T28		IEX with regeneration + EO for still bottoms + incineration for spent resin (PFBA)		PFBA	0.114 [c]	
T29		EO for still bottoms 1		∑PFAS (PFHpA, PFOA, PFHxS, PFHpS, PFOS)	3.09 [c]	Two orders of removal
T30		EO for still bottoms 2		∑PFAS (PFBS, PFBA, PFPeA, PFHxA, PFPeS, PFHpA, PFOA, PFHxS, PFHpS, PFOS)	3.29 [c]	Two orders of removal

[a] Life cycle environmental impacts for IEX and GAC systems were evaluated under the baseline scenario (treatment goal of zero effluent PFOS and PFOA).
[b] The functional unit for the study was 1 m^3 of IEX-treated groundwater with combined PFOA and PFOS concentration equal to 70 μg/L in untreated water and nondetectable concentration of PFOA and PFOS in treated water (i.e. 0). Thus, 100 % removal efficiency was assumed.
[c] The study did not provide the inlet concentration of PFAS. It was calculated based on the data available on PFAS concentration removed and the efficiency of the treatment technology.

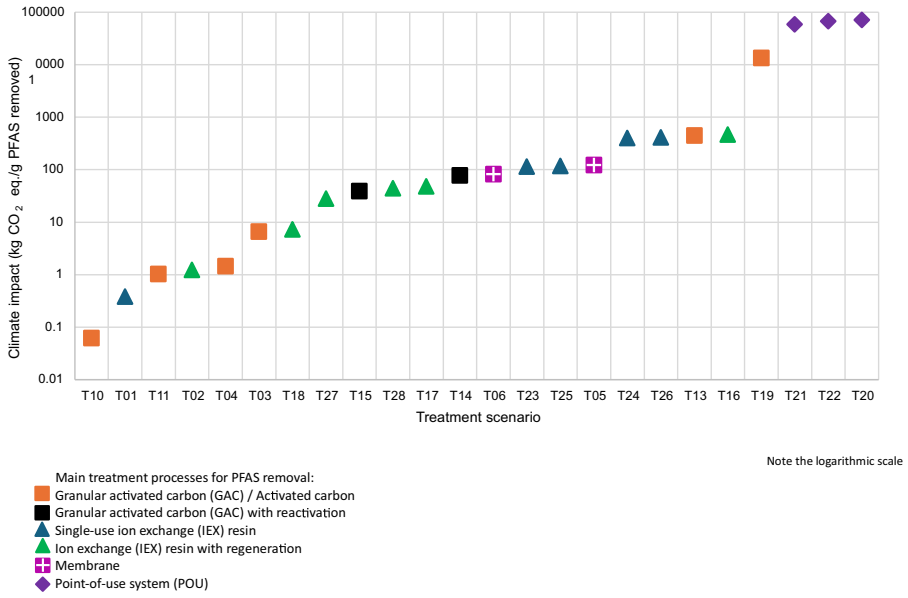

Fig. 2. LCA meta-analysis results: CO_2 eq. emissions per gram of PFAS removed.

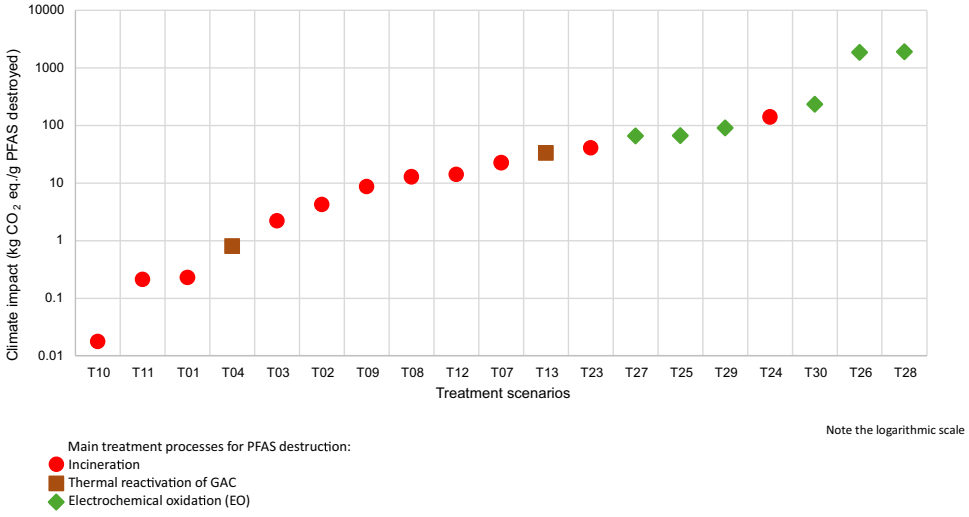

Fig. 3. LCA meta-analysis results: CO_2 eq. emissions per gram of PFAS destroyed.

efficiency. This is apparent from an aggregation of the studies we reviewed that support a life cycle cost perspective, as shown in Fig. 4. According to our analysis, over the 15–30 years lifetime of the innovative technologies to treat PFAS-contaminated waters, the operation costs tend to outweigh the initial capital investment. As a proportion of annual LCC, OPEX of novel technologies used to treat PFAS-contaminated waters tends to be around 20 % higher than CAPEX. Taken together, the annual median cost of treating PFAS-contaminated water is \$1.52/m^3. In 68 % of the treatment scenarios, annual costs fall between \$0.04 and \$1.77 per cubic meter.

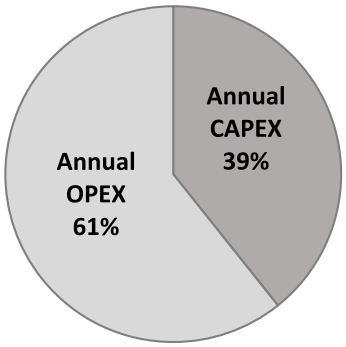

Fig. 4. Components of life cycle cost.

The annual CAPEX for acquiring novel technologies to treat PFAS-contaminated water varies significantly. For IEX and GAC systems, the CAPEX is similar and ranges from \$0.01 to \$0.45 per cubic meter water treated. Membrane technologies, such as reverse osmosis (RO) and nanofiltration (NF), have a CAPEX range of \$0.40 to \$0.51 per cubic meter. While the lower end of the membrane technology range overlaps with the upper end of the IEX and GAC range, membrane technologies generally tend to have higher CAPEX on average.

Additionally, the AquaPRS™ system, which uses a carbon-based micro-adsorbent suspension and a ceramic membrane filter, has a CAPEX between \$0.34 and \$0.46 per cubic meter. These costs are usually considered to have a payback time of between 15 and 30 years at a discount rate of 3 to 7 %, indicating the different financial assumptions and economic conditions considered in distinct projects (Table VI in SI).

The discount rate affects the present value of CAPEX by adjusting for the time value of money, risk, and financing costs. Lower discount rates make future cash flows more valuable today, while higher rates decrease their present value. This variability in CAPEX and financial assumptions highlights the diverse economic impacts and planning requirements associated with different treatment technologies and scenarios.

A regression analysis shows that total CAPEX has a significant and strong negative correlation with both PFAS inlet concentration and the concentration reduction in contaminated waters (Fig. 5). Despite this correlation being based primarily on data from different authors and cases, this relationship is consistent with our previous assessment of the correlation between contributions to climate change (CCC) and PFAS concentrations. This makes sense on account of the underlying entropic factors described previously and the contribution of energy and material consumption to both greenhouse and financial impacts of PFAS treatment processes.

In contrast to the environmental analysis, the cost analysis shows a moderate correlation between the annual CAPEX and the volume of water treated (Fig. 6). The correlation has a less than linear exponent on the volume term, suggesting the existence of economies of scale in the treatment processes. This makes sense given that the underlying drivers of CAPEX would typically include, for example, the construction of large vessels or basins for contact between contaminated water and GAC or IEX resins. Consider a rectangular basin with each side length some factor of a characteristic basin length L. The bed design volume would depend on the operational flowrate f, thus f is proportional to $(\propto) L^3$. The material required for the walls m would vary primarily with the surface area of the walls i.e. $m \propto L^2$ (thickness increases at a lower rate) (Lipsey, 2018). From this it may be deduced that $m \propto f^{2/3}$. Other elements of the capital cost may have different relationships with the flowrate, but given the variability in the underlying data, this theoretical value (0.67) is reasonably consistent with the calculated value in Fig. 6 (0.63).

The annual operation and maintenance costs for PFAS-contaminated water treatment processes exhibit a substantial range, from \$0.03/m^3 (T38) to \$28/m^3 of water treated (T53). T38's costs are derived from a laboratory study and encompass costs for operating a GAC system with reactivation conducted using steam at approximately 800 °C. In this scenario, PFAS-contaminated groundwater with inlet concentration of 1.92 μg/L of PFOS and PFOA is treated, achieving 50 % removal. Although T53 also uses GAC (specifically Norit 1240 W), it manages a lower raw water concentration (0.57 μg/L for the sum of 11 PFAS) and targets an exceptionally low outlet concentration of 0.004 μg/L. The expenses for T53 include the purchase of virgin GAC, regeneration, transportation, and electrical costs for pumping the NF concentrate through the GAC. This comparison demonstrates that even when comparing scenarios utilizing the same treatment technology (i.e. GAC), the annual operation and maintenance costs vary widely.

As illustrated in Fig. 7, the annual OPEX for the three primary

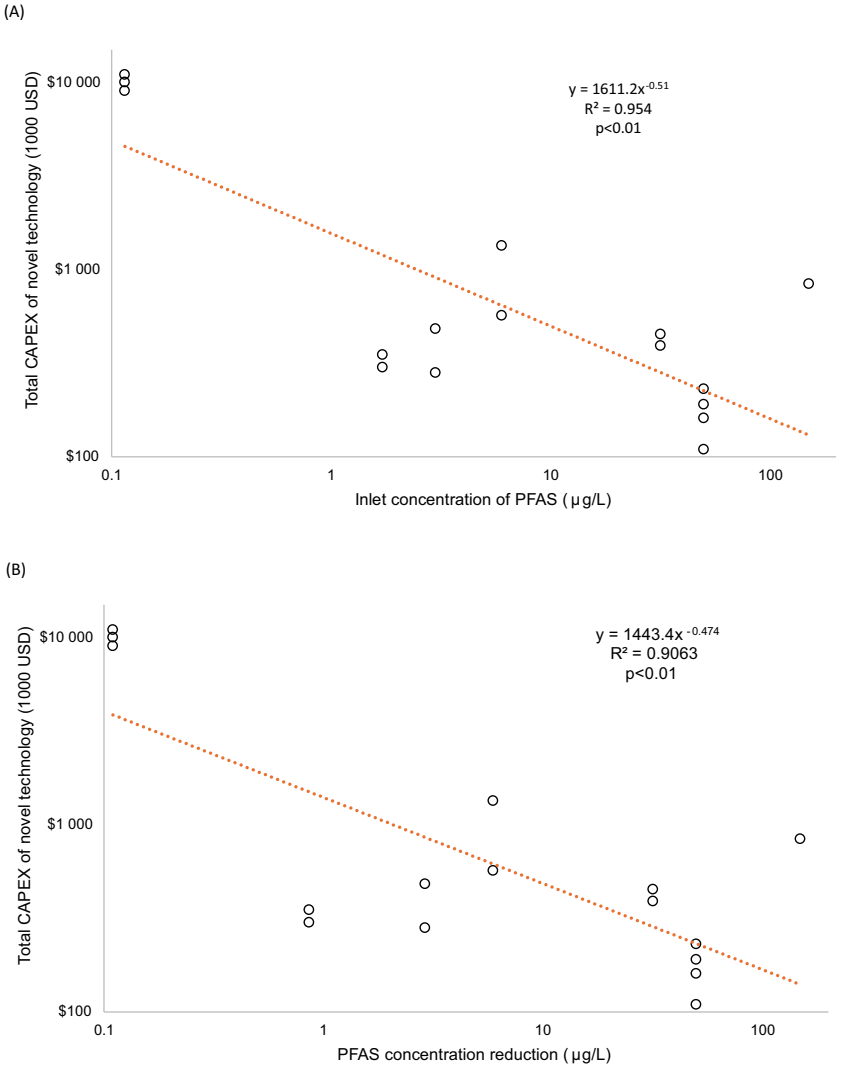

Fig. 5. Relationship between CAPEX and (A) PFAS inlet concentration and (B) concentration of PFAS treated on a double logarithmic scale.

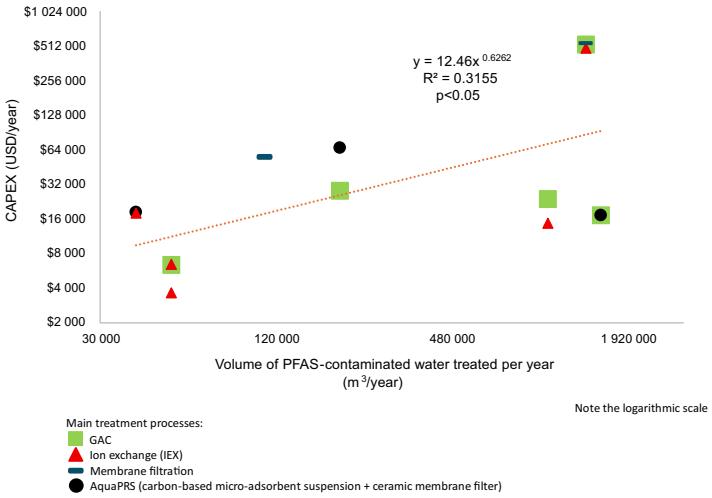

Fig. 6. Relationship between annual CAPEX and volume of PFAS-contaminated water treated per year.

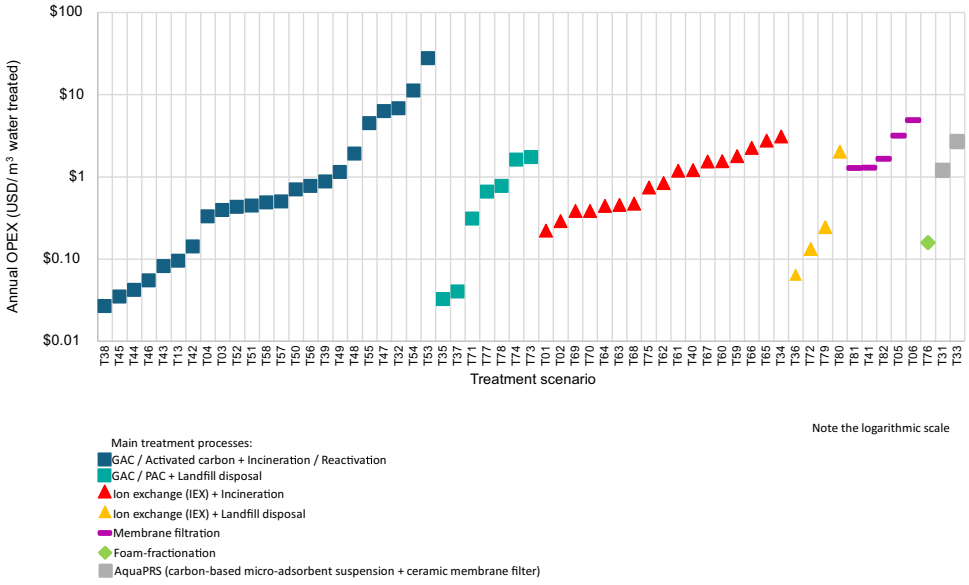

Fig. 7. Annual OPEX of innovative treatment technologies.

technologies assessed in the studies vary from \$0.03 to 28/m^3 of water treated with GAC, \$0.06 to 3.20/m^3 for IEX and \$1.30 to 4.90/m^3 for membrane technologies. A regression analysis (Figure III in SI) suggests that the inlet concentration of PFAS, the amount of PFAS removed, and the volume of water treated do not significantly impact annual operational expenses. The reviewed studies do not permit detailed comparisons of OPEX but we can hypothesize that differences in water hardness, local electricity process and treatment plant topography can cause differences between studies. We could nevertheless extract some comparative unit cost data. For instance, the initial purchase costs of virgin GAC range from \$1.25 to 3.20/kg of adsorbent, while IEX resin prices range from \$8.43 to 17.60/kg of resin (Jiang et al., 2024; Ellis et al., 2023; Franke et al., 2021; Murray et al., 2021). These differences can significantly impact the overall budget.

The media replacement is driven by the number of bed volumes before a target concentration is breached (Quinnan et al., 2023; Murray et al., 2021). Stricter PFAS treatment goals drive higher media usage rates (i.e. GAC), resulting in more frequent reactivation cycles, which increase operational costs. For example, Ellis et al. (2023) found that GAC adsorbents require replacement in half of the time of IEX resins. Energy costs also add up, particularly for energy-intensive processes, as electricity is required to pump water through GAC, IEX beds or membranes. Assumptions about disposal costs for incinerating spent IEX and disposing of used GAC can be a considerable variable, approximately \$90/tonne if the adsorbent media are incinerated as non-hazardous solid waste but \$870 if they are considered hazardous solid waste (Malovanyy et al., 2023). By effectively managing these unit costs, treatment facilities can improve cost efficiency while maintaining high PFAS removal performance.

While the regression analyses performed in this study provide valuable empirical insights into the relationships between PFAS concentrations and treatment costs, we acknowledge that these models do not consider potential confounding factors or fully capture the complex process mechanisms affecting PFAS treatment efficiency. The same can be related to climate impact. Future work could benefit from incorporating mechanistic modeling to account for operational parameters, system design, and other key factors influencing treatment performance. This approach would enhance the explanatory power of the analysis, providing a more detailed understanding of the processes that drive PFAS removal efficiency in different treatment technologies.

3.4. Shortcomings of LCAs on innovative PFAS treatment technologies

ISO 14044 says that LCA practitioners should use indicators that reflect the environmental issues associated with the system under study and the goal of the LCA. In line with this guidance, the LCA studies we reviewed collectively addressed 13 impact categories (Table IV in SI), including climate change, ozone depletion, photochemical ozone formation, acidification, eutrophication, fossil fuel depletion, ecotoxicity, respiratory effects, cumulative energy demand, carcinogenic and noncarcinogenic disability-adjusted life years, ionizing radiation and resource use.

Among these, indicators related to toxicity—specifically human toxicity (cancer and non-cancer) and ecotoxicity—are particularly relevant for technologies targeting the removal of harmful substances such as PFAS. Since these technologies aim to reduce toxicological threats to human and ecological health, it is relevant to discuss how well toxicity-related impacts are captured in LCAs. Indeed, all but one (Maga et al., 2021) of the LCAs we reviewed included cancer and non-cancer indicators, and only two (Bixler et al., 2021; Maga et al., 2021) omitted ecotoxicity (see Table IV in SI). However, these indicators are only meaningful if the studies include relevant toxicant flows and applicable characterization factors. Given that mitigating toxic effects is the central objective of treating PFAS-contaminated water, we focus our analysis on shortcomings of LCAs on how toxicity-related impacts were assessed in the reviewed studies.

PFAS treatment technologies such as IEX and GAC, whether used in single-use or regeneration/reactivation scenarios, are primarily separation methods rather than destruction technologies. The question is: what happens to the PFAS that is removed from water and adsorbed to the media, and how does the analyst handle the PFAS that remains in water? Two studies apparently allow spent media containing “forever chemicals” to be disposed to ordinary landfills (Moeini et al., 2022; Bixler et al., 2021). Most studies on GAC make the assumption that incineration or regeneration of GAC will eventually destroy PFAS (Ellis et al., 2023; Moeini et al., 2022; Bixler et al., 2021; Maga et al., 2021; Emery et al., 2019). Some studies of IEX methods (Ellis et al., 2023; Li et al., 2022a; Boyer et al., 2021) make the same assumption—that incineration of spent resin and/or resin regeneration concentrates will destroy PFAS. This assumption of PFAS destruction is nevertheless criticized by Boyer et al. (2021) who emphasized that PFAS incineration is still an evolving field of research, meaning that comprehensive knowledge about the complete destruction of PFAS is lacking, particularly in terms of its environmental impacts and byproducts. If incineration is incomplete, it could lead to the formation and release of smaller, potentially harmful PFAS compounds into the atmosphere. Electrochemical oxidation (EO) has also been used for PFAS destruction (Li et al., 2022a). However, similar concerns apply to this emerging technology, as it is still in development and under investigation for its effectiveness in PFAS treatment.

The study by Feng et al. (2021) had to deal with a large flow of PFAS from treatment plants back to a river, and calculated the associated health impact explicitly. In contrast, Bixler et al. (2021) focused on health effects of PFAS in consumed drinking water and described the impacts of reject water (from point-of-use systems with RO) treatment via conventional systems without describing the toxicity of the PFAS that would be expected to mostly pass through them. They suggested possibly repurposing this reject water for applications such as lawn irrigation but noted that this could mean polluting the environment. Omitting ecotoxicity impacts as an impact category, as did Bixler et al. (2021), limits the ability to fully evaluate the trade-offs associated with using PFAS-contaminated water for irrigation purposes. In fact, most of the other reviewed LCAs apparently do not consider the residual PFAS in the treated water as having any health effects, either because the user of the water is not specified, or because a legally “safe” concentration is reached by the treatment. One may argue that LCAs on management of suspected persistent carcinogens should nevertheless consider the health effects associated with PFAS consumption via treated tap water or indirect exposure. By not considering the fate of the treated water, the studies may overlook the impacts associated with its use or release. Another example of this matter is the study by Emery et al. (2019) in which two scenarios treat groundwater for consumption and a third supplies bottled water from elsewhere. These are functionally equivalent for the delivery of drinking water, but the function of contaminated aquifer remediation that the third option lacks ought to be accounted for via the ecological impacts of additional PFAS-contaminated groundwater.

Accounting for ecological and human health impacts of PFAS has been constrained by a lack of characterization factors (CFs) (Boyer et al., 2021). In the assessment of toxicity impacts, the methodology predominantly utilized for evaluating toxicity effects is USEtox (whether modified for the TRACI-suite or otherwise) which has some PFAS factors but not for the main toxicants of concern i.e. PFOS and PFOA. This omission results in data gaps and underestimation of toxicity-related impacts, as the absence of PFAS CFs means their impact cannot be accurately reflected in LCA calculations (Aggarwal and Peters, 2024). Feng et al. (2021) acknowledged this limitation by using polychlorinated biphenyls (PCBs) as proxies to estimate the environmental and health impacts of PFAS, given their similar health effects. The impact of each PCB compound was calculated using the TRACI method, with the average impact values of PCB compounds representing the potential impacts of direct PFAS emissions in water through treated effluent discharge. Bixler et al. (2021) overcame the problem for direct water consumption by applying a PFAS-based, nearfield risk assessment methodology to assess the endpoint of carcinogenic and non-carcinogenic impacts on human health, complementing a more traditional LCA approach for indirect emissions and effects. At this point in time, PFAS CFs for both human toxicity and aquatic ecotoxicity are available and published by Aggarwal et al. (2024), Aggarwal & Peters (2024) and Holmquist et al. (2020) and can be applied in future LCA studies.

Synthesizing these methodological observations, we can see that practitioners facing data gaps in toxicity assessment during LCA use several options to address or bypass the lack of CFs. First, in the context of consequential LCA (CLCA), one may assume that all compared options influence toxicity to the same extent, allowing it to be excluded from comparative results—though this assumption should be justified and used cautiously. Second, practitioners may turn to near-field risk assessment approaches, which estimate exposure and risk directly, outside the LCA framework, offering a more localized and context-specific evaluation of toxic effects. Third, "read-across" methods can be used to approximate missing CFs by borrowing data from structurally or functionally similar substances, leveraging existing knowledge to fill in gaps. Finally, when no adequate proxies exist, new CFs can be generated using fate, exposure, and effect modeling tools. Although this is data- and resource-intensive, it can significantly enhance the robustness of the toxicity-related impact assessment. These strategies represent a spectrum from simplified assumptions to detailed modeling, and each of the first three were applied in the LCA studies we reviewed.

The reliance on these workaround strategies underscores a broader issue within the field. As Song et al. (2024) also observe, current LCA applications in PFAS remediation remain fragmented and inconsistent. They emphasize the need for methodological harmonization and improved data quality to enhance the robustness and reliability of PFAS-related LCAs. In line with our findings, Song et al. (2024) highlight ongoing challenges in capturing the environmental impacts of PFAS remediation technologies, attributing many of these difficulties to incomplete or low-quality data. To move the field forward, both their study and ours call for targeted research to support the development of high-quality life cycle inventory datasets—particularly for the final disposal phase of PFAS-laden sorbents and other residual materials.

4. Conclusions

This study offers new insights into the climate impacts per gram of PFAS treated and the annual capital and operational costs per volume of water treated with innovative technologies. Additionally, we provide information about methodological limitations in LCAs of emerging technologies for PFAS removal, focusing on toxicity-related impact assessment.

The meta-analysis showed that PFAS removal technologies exhibit wide variations in climate impact, measured in CO_2 eq. emissions per gram of PFAS treated. The emissions varied from 0.1 to 70 190 kg CO_2 eq. Technologies utilizing activated carbon paired with incineration or reactivation tend to result in lower greenhouse gas emissions, whereas point-of-use systems with combined GAC, IEX, and RO membranes tend to generate higher emissions.

On PFAS treatment costs, our meta-analysis showed that the capital and operational expenses associated with these technologies also vary considerably, with total annual costs usually ranging from $0.04 and $1.77/m^3. OPEX tends to be lower than CAPEX, but higher PFAS concentrations in water can reduce total CAPEX due to increased treatment efficiency and economies of scale, while annual OPEX fluctuates based on factors such as media replacement and choice of disposal methods.

For the application of LCA and LCC in PFAS removal, we found that challenges and data gaps pose limitations to it—a reflection of this field's emerging status. In LCA, many studies focus solely on treatment processes, often neglecting the downstream implications of treated water applications, which may overlook critical associated impacts, especially related to toxicity of PFAS. Many are dependent on assumptions about PFAS destruction from incineration, which may lead to underestimated impacts. Additionally, PFAS-specific characterization factors were typically absent in toxicity assessment methods, resulting in gaps and potentially underestimating toxicity-related impacts. However, these factors are now available for future studies. On the economic side, LCC analyses are still a few, they omit decommissioning costs and may fail to capture variations in financial assumptions across projects, impacting the accuracy of cost estimations.

5. Recommendations

Moving forward, LCA practitioners assessing the impacts of PFAS-contaminated water treatment should employ indicators that align with the environmental concerns specific to the system and the study's objectives—particularly those addressing ecotoxicity and human toxicity—including toxicant flows and relevant characterization factors. These indicators are especially pertinent for evaluating technologies aimed at removing hazardous substances like PFAS. At the same time, utility managers should consider both environmental impacts and financial costs to comprehensively evaluate PFAS removal technologies when selecting treatment options. Technology developers, in turn, should focus on creating more efficient and cost-effective solutions that balance environmental and economic considerations, thereby contributing to sustainable water management. However, given the substantial costs and environmental burdens associated with PFAS removal and destruction, policymakers should prioritize preventive strategies, including comprehensive bans on these persistent and mobile chemicals, rather than relying solely on remediation to ensure water safety.

CRediT authorship contribution statement

Sabrina Altmeyer Mendes: Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization, Writing – review & editing. **Rahul Aggarwal:** Writing – original draft, Data curation. **Magdalena Svanström:** Validation, Supervision, Writing – review & editing. **Gregory Peters:** Supervision, Funding acquisition, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.resconrec.2025.108524.

Data availability

Data will be made available on request.

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