Water Environment Research

RESEARCH ARTICLE OPEN ACCESS

Life-Cycle Environmental Impacts of Removing and Destroying PFAS From Wastewater Effluent

Anderson C. Ellis[1,2] | Alison L. Ling[3] | Rebecca Vermace[4] | R. Hapaki Lorenzo Quintana[3] | Scott Kyser[5]

[1]Aquagga, Inc, Tacoma, Washington, USA | [2]Department of Civil and Environmental Engineering, Colorado School of Mines, Golden, Colorado, USA | [3]Department of Civil Engineering, University of St. Thomas, St. Paul, Minnesota, USA | [4]Barr Engineering Co, Minneapolis, Minnesota, USA | [5]Minnesota Pollution Control Agency (MPCA), Minneapolis, Minnesota, USA

Correspondence: Alison L. Ling (aling@stthomas.edu)

Received: 23 February 2026 | **Revised:** 25 June 2026 | **Accepted:** 9 July 2026

Keywords: life-cycle assessment | per- and polyfluoroalkyl substances (PFAS) | PFAS destruction | PFAS treatment | water resource recovery facility (WRRF)

ABSTRACT

A life-cycle assessment (LCA) framework was used to estimate environmental impacts likely to occur during the removal and destruction of per- and polyfluoroalkyl substances (PFAS) in wastewater effluent. Results suggest that upgrades to remove and destroy PFAS from municipal water resource recovery facility (WRRF) effluents could double or triple total greenhouse gas emissions from wastewater facilities, as well as other human and environmental health impacts. Sensitivity of environmental impacts to selected system parameters was also evaluated, highlighting the benefits of granular activated carbon (GAC) reactivation/reuse and use of renewable energy sources. LCA results were translated into "impact curves" to predict environmental impacts of effluent PFAS treatment based on facility size and technologies applied. Overall, these results enable estimation of secondary environmental impacts associated with PFAS treatment of wastewater effluent and can be used to support policy, permitting, and planning by regulatory agencies and utilities.

1 | Introduction

Per- and polyfluoroalkyl substances (PFAS) are a large group of persistent, mobile chemicals. Because of their utility and durability, PFAS uses are widespread across industries and commerce (ECHA 2023a; RTI International 2023), resulting in PFAS detection in water, soils, plants, animals, and air across the globe (Ackerman Grunfeld et al. 2024; Evich et al. 2022). Continued PFAS use and environmental emissions remain in the range of millions of tons per year (Evich et al. 2022), with estimated costs to remove PFAS from the environment at current emission rates exceeding the global economy (Ling 2024). Based on the strong evidence that many individual PFAS compounds have negative human health and environmental impacts (Bălan et al. 2021; Kwiatkowski et al. 2020; National Academies of Sciences, Engineering, and Medicine 2022), many countries and jurisdictions are evaluating PFAS prevention and remediation strategies (ITRC 2022, 2025). However, because complete remediation of environmental PFAS is not economically or logistically feasible (Ling 2024; Ling et al. 2026), most regulatory actions have focused on limiting new emissions and discharges or protecting human health at the point of potential exposure. Individual US and EU member states have implemented PFAS use restrictions to limit future PFAS pollution (Maine State Legislature 2021; State of Minnesota 2023; Zainzinger 2025), with an EU-wide proposal for restricting PFAS as a class still in progress (ECHA 2023b). Similarly, PFAS criteria for drinking water have been established in the United States, Europe, Australia but differ in which PFAS are regulated and on concentration targets for specific compounds (European Union 2020; PFAS National Primary Drinking Water Regulation 2024). In jurisdictions

Summary

- Removing and destroying per- and polyfluoroalkyl substances(PFAS) from municipal water resource recovery facility (WRRF) effluent have both environmental benefits and environmental costs
- Preconcentrating effluent PFAS using reverse osmosis and reusing sorbent media reduced overall environmental impacts of sorptive treatment
- Removing and destroying PFAS from municipal WRRF effluent could double or triple greenhouse gas emissions, challenging wastewater net-zero emission goals
- The social cost of greenhouse gas (GHG) emissions from PFAS treatment is estimated at one-third to one-tenth of previously estimated economic costs.

where water quality limits for PFAS exist, individual PFAS criteria can range widely from 0.01 part per trillion to multiple parts per billion (ITRC 2025).

Municipal WRRFs serve as centralized collection points for PFAS from domestic, commercial, and industrial sources but are not designed to treat PFAS; most PFAS received by WRRFs pass through into discharge effluent or residual biosolids. Commonly regulated perfluoroalkyl acids like PFOA and PFOS make up a relatively small proportion of PFAS in municipal wastewater effluents, with much of the total PFAS mass attributable to fluorinated pharmaceutical compounds or to shorter chain PFAS (Ruyle et al. 2025; Seay et al. 2023). Furthermore, PFAS specifically targeted by analytical methods likely comprise a small portion of total PFAS, with concentrations of 73 targeted PFAS in wastewater effluent only accounting for 10% of extractable organic fluorine (EOF) in one study (Aro et al. 2021) and 30 targeted PFAS accounting for 15% of targeted PFAS after oxidation by total oxidizable precursor (TOP) analysis in another (Önnby et al. 2025). It is also possible for total measured PFAS concentrations to increase through WRRFs due to degradation or biotransformation of nontargeted byproducts or precursors to PFAS readily quantified using targeted analyte lists (Cookson and Detwiler 2022; Thompson et al. 2022). PFAS-laden WRRF effluent is typically discharged to surface waters that are often used downstream for drinking water or fishing (Ruyle et al. 2025; Sun et al. 2024; Thompson et al. 2022). A recent study estimated that PFAS in wastewater effluents could push the drinking water supplies of about 10 million Americans above federal PFAS limits (Ruyle et al. 2025). While municipal WRRFs contribute PFAS to surface waters, they are not a dominant route. Recent work estimated that treating effluent from all large municipal WRRFs in the EU for PFAS would offset less than 2% of estimated EU emissions of short- and long-chain PFAAs (Ling et al. 2026). Primary sources of these and other PFAS to the environment include industrial air emissions and wastewater discharges (U.S. EPA 2021b), existing contaminated soil sources, landfill gas and uncontrolled leachate (De la Cruz et al. 2025), stormwater runoff (Kali et al. 2025), nonpoint use of PFAS in pesticides (Joerss et al. 2024), refrigerants (Arp et al. 2024), building materials, industrial products, and consumer products. However, monitoring and regulating PFAS in WRRF effluents have still been considered or implemented across geographies, partly due to existing effluent regulations frameworks.

Globally, few municipal WRRFs have PFAS discharge limitations. Where discharge limitations do exist, they are mostly passed on to upstream industries that discharge PFAS-containing wastewater to WRRFs, consistent with the U.S. EPA's stated intent to use wastewater regulations primarily as a tool to limit upstream industrial discharges (U.S. EPA 2022a). As of January 2025, at least 65 wastewater discharger permits to surface water have been issued to limit PFAS releases in the United States (U.S. EPA 2025a). The updated EU Urban Wastewater Directive requires PFAS monitoring in WRRF effluent and includes extended producer responsibility, which may drive PFAS management to upstream industrial sources to reduce total costs (European Union 2024).

Removing and destroying PFAS from WRRF effluent is technically achievable but would be expensive and likely to increase wastewater rates beyond affordability metrics in most communities (Ling et al. 2024). The U.S. Clean Water Act recognizes the potential for permit compliance to cause economic or environmental hardship and allows for time-limited variances based on financial unaffordability of treatment or if the source of pollution is human-caused and "would cause more environmental damage to correct than to leave in place" (U.S. 40 CFR Part 131.10 2026). Thus, regulators and policymakers should consider the economic implications of removing PFAS at WRRFs as well as related environmental impacts like greenhouse gas (GHG) emissions goals, nutrient harvesting, and water reuse potential. Wastewater management without PFAS treatment accounts for about 0.7% of global GHG emissions (Ritchie et al. 2020), and implementing PFAS treatment at WRRFs would increase these emissions.

A recent review highlights the urgent need to balance PFAS treatment and destruction efficacy with broader environmental sustainability (Tushar et al. 2024) in evaluating treatment options. The present study aims to fill this gap by estimating environmental impacts associated with treating PFAS in WRRF effluent using multiple treatment trains and across a range of design capacities. While other contaminants would also be removed through this type of treatment train, we assume that installation and upgrades are driven by PFAS-related concerns and thus attribute all estimated externalities to PFAS-driven actions. These results can be used to project environmental impacts of both individual facility choices and regional policies. Multiple studies to date have evaluated environmental impacts of advanced wastewater treatment (Plakas et al. 2015; Rahman et al. 2016) or of treating groundwater for PFAS (Boyer et al. 2021; Ellis et al. 2023; Emery et al. 2019; Smith et al. 2025), but none have addressed PFAS in municipal WRRFs or assessed these impacts on a regional basis. This work aims to contextualize the environmental impacts of WRRF PFAS treatment to provide a quantitative baseline to inform future decision-making and strategic design decisions. We further apply the "social cost of carbon" framework developed by the US EPA to estimate social costs associated with additional GHG emissions.

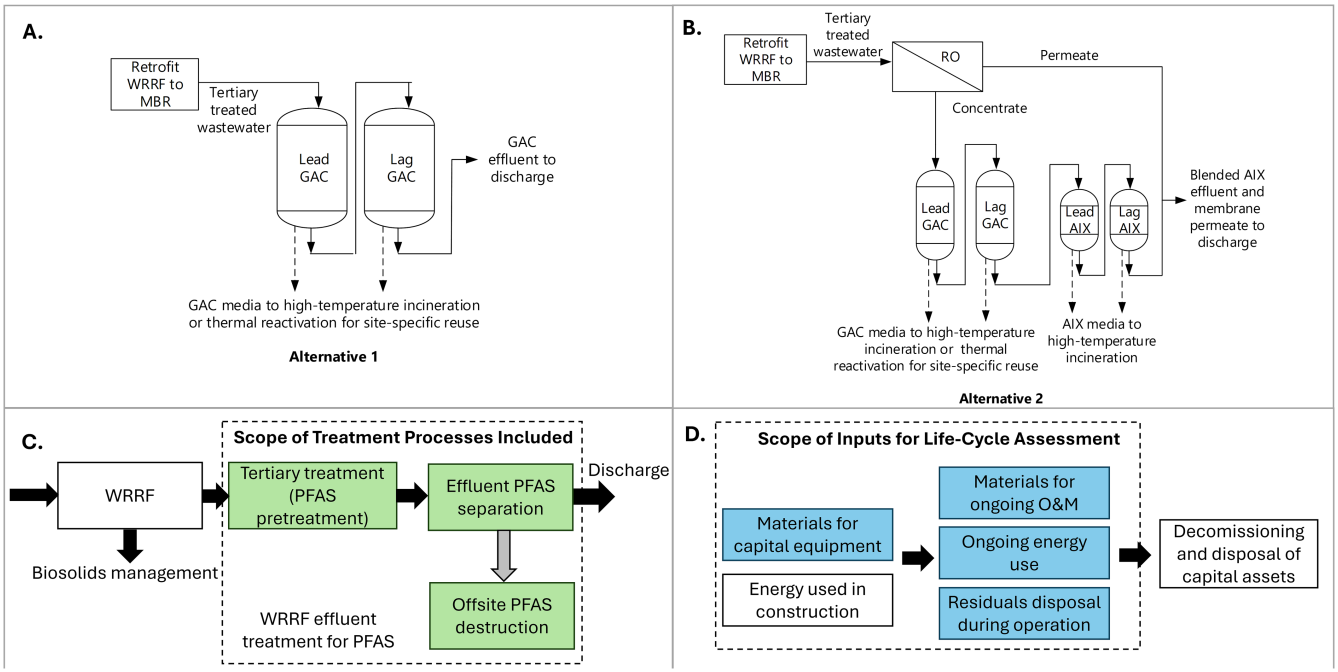

FIGURE 1 | Conceptual process flow diagrams for (A) GAC for tertiary treated WRRF effluent (Alternative 1) and (B) RO with GAC and AIX (Alternative 2) for tertiary treated WRRF effluent, also showing (C) scope considered for treatment processes and (D) inputs for life-cycle assessment models.

2 | Methods

2.1 | Conceptual Designs and Treatment Targets

The conceptual design of potential WRRF PFAS treatment systems (Figure 1A,B) used previously established design bases for tertiary effluent water quality (Tables S1-1 and S1-2) and PFAS treatment targets (Table S1-1), consistent with previous work conducted for the Minnesota Pollution Control Agency (Barr Engineering Co. and Hazen and Sawyer 2023; Ling et al. 2024). Additional information on the development of these treatment trains, preliminary design parameters, and relevant assumptions is included in the Supporting Information (Tables S1-3 through S1-8).

Briefly, PFAS separation and destruction technologies were evaluated to identify those viable for large-scale wastewater treatment (> 10 MGD) at the time of the analysis in 2022 and with demonstrated > 90% efficacy in PFAS separation or destruction. Technologies were assembled into treatment trains and further screened for technical and economic feasibility as previously described (Barr Engineering Co. and Hazen and Sawyer 2023), and treatment trains scoring highest across a range of criteria were selected as Alternatives 1 and 2. Treatment targets were established at approximate reporting limits (5 ng/L) for long-chain and short-chain perfluoroalkyl acids PFOA, PFOS, PFHxA, PFHxS, PFBA, and PFBS, aligning with the US drinking water MCLs targeting zero concentrations for PFOA as a known human carcinogen and PFOS as a probable human carcinogen (Drury et al. 2024). It is expected that PFAS of differing chain length and class at similar concentrations (e.g., PFHpS, 6:2 FTS, HFPO-DA) would be removed to nondetect levels prior to PFBA detection, as numerous studies have found PFBA to be one of the lowest affinity structures for adsorption with few exceptions (e.g., TFA) due to its hydrophilicity (Chen et al. 2025; Park et al. 2020). This study did not consider any life-cycle or environmental effects of PFAS in municipal biosolids, as this would require significant assumptions regarding the fate of biosolids (e.g., pyrolysis and land application), and technologies for PFAS mitigation in biosolids are not yet well-defined at the commercial scale.

A PFAS destruction step (either high-temperature incineration or granular activated carbon [GAC] reactivation) was included in both treatment trains to limit rerelease of residuals (Figure 1C). In Alternative 1, tertiary-treated effluent is fed through lead–lag GAC adsorption vessels. In Alternative 2, tertiary-treated effluent is routed through reverse osmosis (RO) membrane separation, with RO concentrate treated by lead–lag GAC adsorption and anion exchange (AIX) vessels. In both alternatives, spent media would be disposed of via high-temperature incineration or GAC reactivation and reuse (10 MGD only). PFAS fate through high-temperature incineration was recently studied by the EPA and found to be over 99% effective at mineralizing PFAS, with no products of incomplete destruction through the most established air testing methods for fluorine tracking: OTM-45, OTM-50, and Method 0010/8270 (Troxler et al. 2025). GAC reactivation has similarly demonstrated over 99% removal efficiency, with slightly lower confidence in full mineralization (U.S. EPA 2026).

Conceptual designs for the two selected treatment alternatives were developed for three design WRRF flows (0.1 million gallons per day (MGD), 1.0 MGD, and 10 MGD). To evaluate the significance of GAC incineration or reactivation and reuse on the overall system environmental impacts, two separate 10 MGD scenarios were evaluated for each alternative: one with reactivation and reuse of GAC and one with incineration or

reactivation without reuse. Reactivation with reuse was not evaluated for smaller facilities because GAC reactivation vendors typically limit sole-source reactivation contracts to facilities changing out > 80,000 pounds of carbon per event (Barr Engineering Co. and Hazen and Sawyer 2023), which corresponds to about 2 MGD for GAC treatment systems. Design bases assumed operation of sorbent media at manufacturer-recommended contact times (described in Tables S1-3 and SI-6), which would be implemented regardless of PFAS profile and concentration. Because any specific WRRF will deviate from these conditions based on site-specific factors, actual unit quantities and life-cycle environmental impacts will also deviate from those estimated here. These conditions reflect minimal industrial loading of PFAS to WRRFs, assuming no significant industrial PFAS discharges through pretreatment agreements to present a baseline case.

2.2 | Estimating Capital and Operational Quantities for Life-Cycle Assessment (LCA)

Capital equipment materials requirements were estimated from previously established conceptual designs (Barr Engineering Co. and Hazen and Sawyer 2023) by selecting pipe size and material, number of valves and material, system configuration, building sizing, and building materials based on standard industry sizing, standard materials used by vendors, and professional judgment. All capital infrastructure models were developed to include an enclosed building, concrete pad, and necessary process equipment. The expanded design assumptions, including assumptions from previous work, are included in Table S1-4 and S1-7. Operational parameters, including baseline media breakthrough rates and change-out frequencies, were also derived from previous work (Barr Engineering Co. and Hazen and Sawyer 2023).

2.3 | Preliminary Design and Quantities for Tertiary Pre-Treatment

Secondary WRRF effluent typically does not meet RO and GAC influent water quality recommended by manufacturers (< 1 mg/L total suspended solids [TSS], < 1 mg/L total organic carbon (TOC), and iron and manganese both < 0.5 mg/L) (Barr Engineering Co. and Hazen and Sawyer 2023; Ling et al. 2024). This study assumed that all WRRFs would require tertiary treatment upgrades prior to PFAS treatment for mitigation of these co-constituents, here assumed to be through installation of or retrofit to a membrane bioreactor (MBR) (Figure 1C).

To consider life-cycle impacts of tertiary MBR treatment as well as PFAS treatment processes, MBR energy use was estimated to increase energy use by 1000 kWh/MG over conventional activated sludge for 1.0 MGD and 10 MGD flow rates (assumed to reflect partial retrofit of activated sludge WRRFs), based on previous studies comparing technologies' energy use (Banti et al. 2020; Lazarova et al. 2012). Installing an MBR system in a small 0.1-MGD WRRF, assumed to be stabilization pond WRRFs, was assumed to be higher at 5000 kWh/MG for 0.1 MGD. Additional material use was considered for membrane replacement and was in the range of 165 square feet of hollow fiber membrane per MG treated, with 5-year membrane life (Glover et al. 2022). Capital equipment for tertiary treatment was not included in this evaluation, assuming that many large WRRFs may already have made these investments and that tertiary treatment is often driven by parameters other than PFAS.

2.4 | LCA

LCA models for this study were developed using SimaPro v9.1 (Pre Sustainability, Amersfoort, NL) and using inventory compilation and impact assessment methodologies consistent with ISO 14040 and 14044 standards for model development (Bare 2011; ISO 2006). For all treatment scenarios, inventory items used in developing models were obtained from industry-relevant databases listed in Table S2-1 in the SI; specific inventory items are presented in Table S2-2. Selection of inventory items in LCA databases was conducted to reflect nationwide environmental externalities (e.g., using national energy matrix data rather than state-level or regional data) to maximize model applicability where possible. Models were developed using materials relevant to and sourced in North America, where possible, as inventory item selection was limited to data aggregated in existing LCA databases. While projected values will vary on a site-specific basis, relative trends among treatment alternatives and WRRF sizes are expected to hold true across a diverse range of WRRF sites and regions. Process flow diagrams highlighting the LCA input subprocesses and refinement steps of these models are shown in Figures S2-1 through S2-4 in the SI.

Environmental impact analysis was performed using the TRACI 2.1 (Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts) assessment method developed by the US EPA (U.S. EPA 2012). Impact assessment enabled estimation of emissions values from the input model inventory (allocated in terms of annual inputs) into new, user-defined models for each treatment scenario using input quantities derived from the aforementioned design bases. Emissions were then categorized into ten distinct impact categories defined by the EPA in the TRACI method to reflect environmental impacts across a range of environmental compartments and human health considerations. Details regarding the LCA methodology, inventories, and the selection process for database entries are available in Section S2 of the SI.

The scope of the LCA included the materials extraction and fabrication of all capital infrastructure, site building and concrete pad, energy production, and chemical inputs, as well as all consumable requirements and transportation, throughout the operational phase of treatment (Figure 1D). The analysis included end-of-life materials management for treatment residuals (e.g., spent GAC) but not for demolition and disposal of capital equipment. The two functional units used throughout the study was "per year of facility treatment," or "per MGD treated," meaning the capacity to remove and destroy PFAS from one million gallons of wastewater per day.

Environmental impact data are represented using scaled characterization values, which standardize results as a

percent of the highest quantity of emissions in each impact category, consistent with existing LCA studies (Crenna et al. 2019; Boyer et al. 2021). Results were also shown in terms of "person-equivalents" or the approximate number of people whose cumulative emissions equal those generated by the treatment system over any given time. This unit was obtained by dividing the raw quantities of emissions (e.g., kg CO_2-eq/year) by the emissions generated by the average US citizen in a given year (e.g., kg CO_2-eq/person-year). As another way to provide context, Minnesota was used as a case study to understand potential environmental impacts of treating PFAS in wastewater effluent at a regional scale. Estimated increases in Minnesota GHG emissions were compared to reported GHG emissions for the state (MPCA 2025).

2.5 | Sensitivity Analysis

After emissions associated with each of the treatment alternatives were estimated over a range of facility sizes (0.1–10 MGD), sensitivity analyses were conducted to evaluate the potential significance that media use rates (change-out frequency) and the potential sources of energy used to drive the treatment processes had on the magnitude of overall environmental impacts incurred from PFAS treatment, focusing on global warming, eutrophication, carcogenicity, and ecotoxicity.

Because media use and disposal contributed most to environmental impacts in Alternative 1, the relationship between media use rates and targeted LCA outcomes was evaluated for this alternative. For this sensitivity analysis, bed volumes (BVs) to breakthrough were varied between 2,000 and 50,000, as these represent the boundary conditions of what might be expected for GAC serviceable lifetimes. This range is representative of previous pilot tests evaluating groundwater and surface waters (Chow et al. 2022; Kempisty et al. 2022; McCleaf et al. 2017), as limited work to date has evaluated PFAS breakthrough rates for tertiary treated wastewaters. For Alternative 2, the impact of different energy sources was evaluated by varying LCA inputs to source energy solely from hydropower, wind, geothermal, photovoltaic (solar), natural gas, and hard coal rather than the national average electricity matrix, highlighting the relative contribution of energy sourcing. Further details regarding the sensitivity analyses can be found in Section S4 of the SI.

2.6 | Social Cost of GHG Emissions

The social cost of carbon (as GHG emissions) associated with removing and destroying PFAS from WRRF effluent under the three facility size scenarios and treatment alternatives was evaluated using the methods detailed in the November 2023 EPA rulemaking (U.S. EPA 2023a), though recent studies suggest that these may be underestimated by a factor of two (Moore et al. 2024). The social cost of carbon was calculated over a 20-year period (2025–2044) with an assumed 2.0% discount rate (U.S. EPA 2023a) and adjusted to 2025 dollars using inflation data from the Bureau of Labor Statistics (US Bureau of Labor Statistics 2025). This analysis carries the baseline assumption that electricity sources used for treatment match the average makeup of energy sources in the United States as a whole.

3 | Results

3.1 | Estimated Environmental Impacts for Treating PFAS in WRRF Effluent

Annual environmental impacts estimates for all treatment alternatives and WRRF flow rates are shown in Table 1 for the ten TRACI impact categories. These estimates are inclusive of both the annual operational phase and the system capital infrastructure divided over a 20-year treatment system lifetime.

While values in Table 1 quantify the magnitude of environmental impacts associated with PFAS treatment, isolation of select impact categories and normalization to standardized units helps to identify primary contributions and forcing factors when assessing trade-offs between PFAS remediation efforts and their environmental impact. Normalization was conducted in two ways: Figure 2A shows emissions data normalized per MGD of throughput, which are then scaled to the percentage of maximum environmental impact for each category. Conversely, Figure 2B shows data normalized to "person-equivalents" or the emissions expected for a given number of people using per capita data for the U.S. collected by Ryberg et al. (2013).

This allows direct comparison among impact categories by standardizing emissions units to a common reference metric. For example, adding Alternative 1 treatment to a single 10 MGD WRRF would emit similar global warming impacts as normal activities of an additional 510 people. This person-equivalent normalization revealed global warming, eutrophication, carcinogens, and ecotoxicity to be among the highest impact categories in magnitude relative to other categories presented in Table 1 (complete normalized data shown in Figure S3-1 in the SI). Because of their disproportionate impacts—as well as their relevance to multiple receptors relevant to Clean Water Act environmental harm language (US 40 CFR Part 131.10 revised 2026) (e.g., atmospheric, freshwater, human health, and environmental health)—these four categories were selected for visualization and discussion in this work.

Figure 2A shows that annual emissions per MGD treated decrease with increasing facility size, highlighting the economies of scale with increasing treatment volume. These reductions were partly due to capital infrastructure scaling but also stem from the nonlinearity of operational-phase input consumption from factors like electricity demands and antiscalant chemical use that exhibited decreases in annual consumption per unit volume with increasing treatment scale. This economy of scale was found to be more pronounced for the RO-based Alternative 2 than for the GAC-based Alternative 1 due to direct linearity associated with sorbent usage rate.

Another major driver of differences shown in Figure 2 was the effect of sorbent disposal and sourcing, with incineration and replacement of spent GAC exhibiting a larger influence on overall environmental impacts than reactivation and reuse of spent GAC (modeled at 10 MGD treatment only). This follows intuitively, as

TABLE 1 | Annual estimated environmental impacts associated with each treatment scenario[a].

Impact category[b]	Unit	Technology alternative 1: GAC + incineration/reactivation				Treatment alternative 2: RO + GAC + AIX + incineration/reactivation			
		0.1 MGD	1 MGD	10 MGD	10 MGD (GAC reuse)	0.1 MGD	1 MGD	10 MGD	10 MGD (GAC reuse)
Ozone depletion	kg CFC-11 eq	8.6E-03	7.6E-02	6.0E-01	2.7E-01	2.0E+00	1.7E+01	1.7E+02	1.7E+02
Global warming	kg CO_2 eq	1.5E+05	1.4E+06	1.2E+07	3.5E+06	1.1E+05	9.0E+05	6.3E+06	4.8E+06
Smog	kg O_3 eq	6.1E+03	5.7E+04	4.4E+05	2.0E+05	4.3E+03	3.4E+04	2.6E+05	2.1E+05
Acidification	kg SO_2 eq	6.5E+02	6.1E+03	5.1E+04	1.8E+04	4.6E+02	3.6E+03	2.4E+04	1.7E+04
Eutrophication	kg N eq	3.6E+02	3.4E+03	3.0E+04	7.9E+03	2.3E+02	1.8E+03	1.2E+04	8.0E+03
Carcinogenics	CTUh	1.5E-02	1.2E-01	1.0E+00	2.0E-01	9.1E-03	5.7E-02	3.6E-01	2.1E-01
Non carcinogenics	CTUh	2.3E-02	2.1E-01	1.7E+00	5.4E-01	2.2E-02	1.7E-01	1.0E+00	8.0E-01
Respiratory effects	kg PM2.5 eq	6.4E+01	6.0E+02	5.1E+03	1.7E+03	4.5E+01	3.4E+02	2.1E+03	1.5E+03
Ecotoxicity	CTUe	6.5E+05	5.9E+06	5.0E+07	1.4E+07	4.7E+05	3.4E+06	2.2E+07	1.5E+07
Fossil fuel depletion	MJ surplus	7.9E+04	6.7E+05	5.0E+06	3.4E+06	9.1E+04	6.8E+05	4.8E+06	4.4E+06

[a]All technology alternatives and associated flow rates assume incineration of solid wastes, excluding the 10 MGD system with GAC reactivation and reuse as specified.
[b]Environmental impacts grouped into "impact categories" as defined by the US EPA's TRACI impact assessment methodology. Cumulative energy demand.

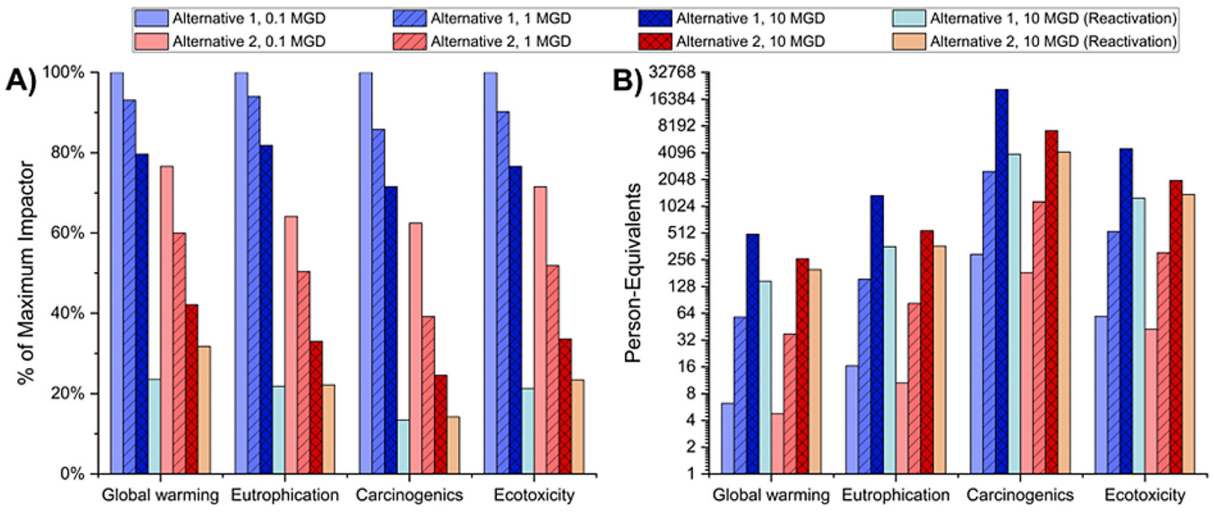

FIGURE 2 | Normalized environmental impacts of treating PFAS in WRRF effluent for eight scenarios. (A) Impacts per MGD scaled to percent of maximum impact and (B) Total Impacts scaled to person equivalents based on average personal environmental impacts of people in the US.

GAC reuse reduces the need for carbon- and energy-intensive production of virgin GAC and associated impacts from production. For Alternative 1 relying primarily on GAC for treatment, reactivation and reuse for the 10 MGD scenario decreased the average impacts from these four categories by an average of 67%. Even in the case of the membrane-focused Alternative 2, which used less than 20% of the GAC mass of Alternative 1, reactivating and reusing the GAC scenario reduced emissions by 24% on average across the categories shown relative to single-use sourcing and incineration.

Another important difference in environmental impacts was observed between treatment alternatives. Alternative 2 generated fewer estimated impacts for all treatment flowrates relative to Alternative 1 (assuming GAC incineration) for 8 of the 10 impact categories, including all four shown in Figure 2. This was driven by reduced GAC usage achieved by RO membranes preconcentration, reducing the flow treated by sorbent media by 85%. While Alternative 2 used more energy and additional types of consumables for membrane treatment, including antiscalants and clean-in-place chemicals, the fixed 5-year lifetime of these membranes

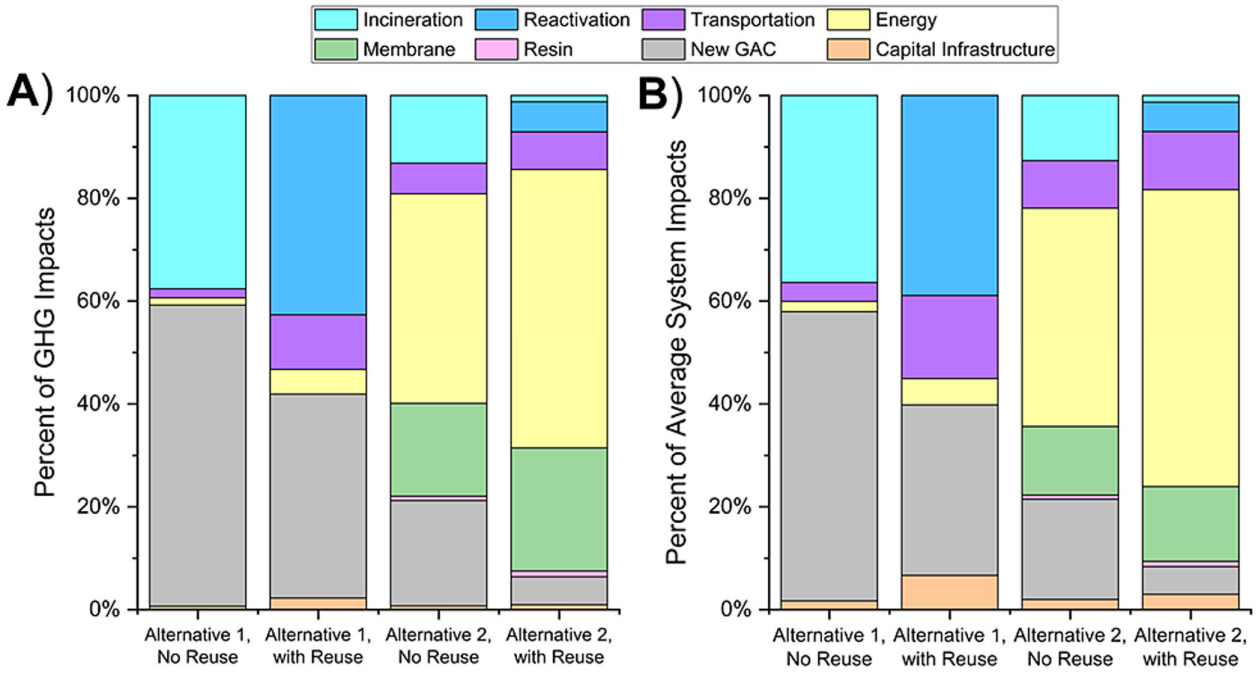

FIGURE 3 | Proportion of environmental impacts attributable to different project components for removing PFAS from WRRF effluent. Panel A shows GHG emissions only, while Panel B shows composite environmental impact fractions across 10 TRACI impact categories.

resulted in a trade-off between increased energy consumption and decreased physical inputs requiring transport and high-temperature disposal. This balance ultimately resulted in lower emissions across the board despite the high energy requirements.

To ascertain which aspects of treatment operations most directly linked to environmental impacts, data plotted in Figure 2A were deconstructed and categorized by project components to identify drivers for future optimization efforts. Figure 3 shows these impact breakdowns, with further deconstruction by alternative and impact categories shown in Figure S3-2 and S3-3 of the SI.

Impacts from the GAC-based Alternative 1 are dominated by virgin media production, while impacts from the RO-based Alternative 2 are dominated by energy use to drive RO membrane flux. Capital infrastructure accounts for less than 5% of GHG emissions and less than 8% of overall environmental impact for all treatment alternatives. However, capital infrastructure accounts for 10%–45% of impacts in the carcinogenics impact category, primarily due to steel production for the building enclosure as well as carbon steel production required for sorbent vessel fabrication. Transportation of virgin, spent, and reactivated media was also a relatively small proportion of environmental impact, with higher relative impacts in reactivation and reuse scenarios due to longer assumed transport distances for reused media than for virgin media due to the need to transport media back to facilities after reactivation.

3.2 | Impact Curves for Targeted Impact Categories

In addition to the technologies used for each treatment alternative, environmental impacts are also highly dependent on the flow rate through the WRRF. Impact curves were created to quantify environmental impacts across a range of flowrates using empirically-fit power functions. Impact curves for both treatment alternatives for the four indicator categories of impacts are shown in Figure 4 below, with panels for each alternative including the three conditions using incineration as well as the 10 MGD condition assuming reactivation of GAC (hollow symbols). Impact curves for all TRACI categories are included as Figure S3-4 through S3-13 in the SI.

As seen in Figure 4, environmental impacts projected as a function of WRRF throughput increased approximately linearly for Alternative 1 (power exponents near 1.0 at 0.92 and 0.94), suggesting a relatively low economy of scale associated with environmental impacts of PFAS treatment. This is due to Alternative 1 impacts being heavily influenced by virgin GAC use and disposal, which scales linearly to flow. RO-based Alternative 2 treatment shows more nonlinearity (lower power exponents of 0.80 to 0.85) than GAC-based Alternative 1 due to the more significant economy of scale for capital membrane infrastructure and smaller sorption vessels.

Impact curves reflect conclusions from previous figures, including differences between alternatives, GAC disposal and reuse, and differences between carcinogenicity and other impact categories but add the additional value of being able to estimate impacts for facilities in between the three modeled sizes. Because treating PFAS in wastewater effluent reduces PFAS mass released to the environment, consideration of environmental impacts per mass of targeted PFAS removed and destroyed was also considered. Impact curves per mass of targeted PFAS removed are included as Figure S3–14, with PFAS mass removal being directly proportional to MGD throughput due to consistent PFAS concentration assumed in the design basis.

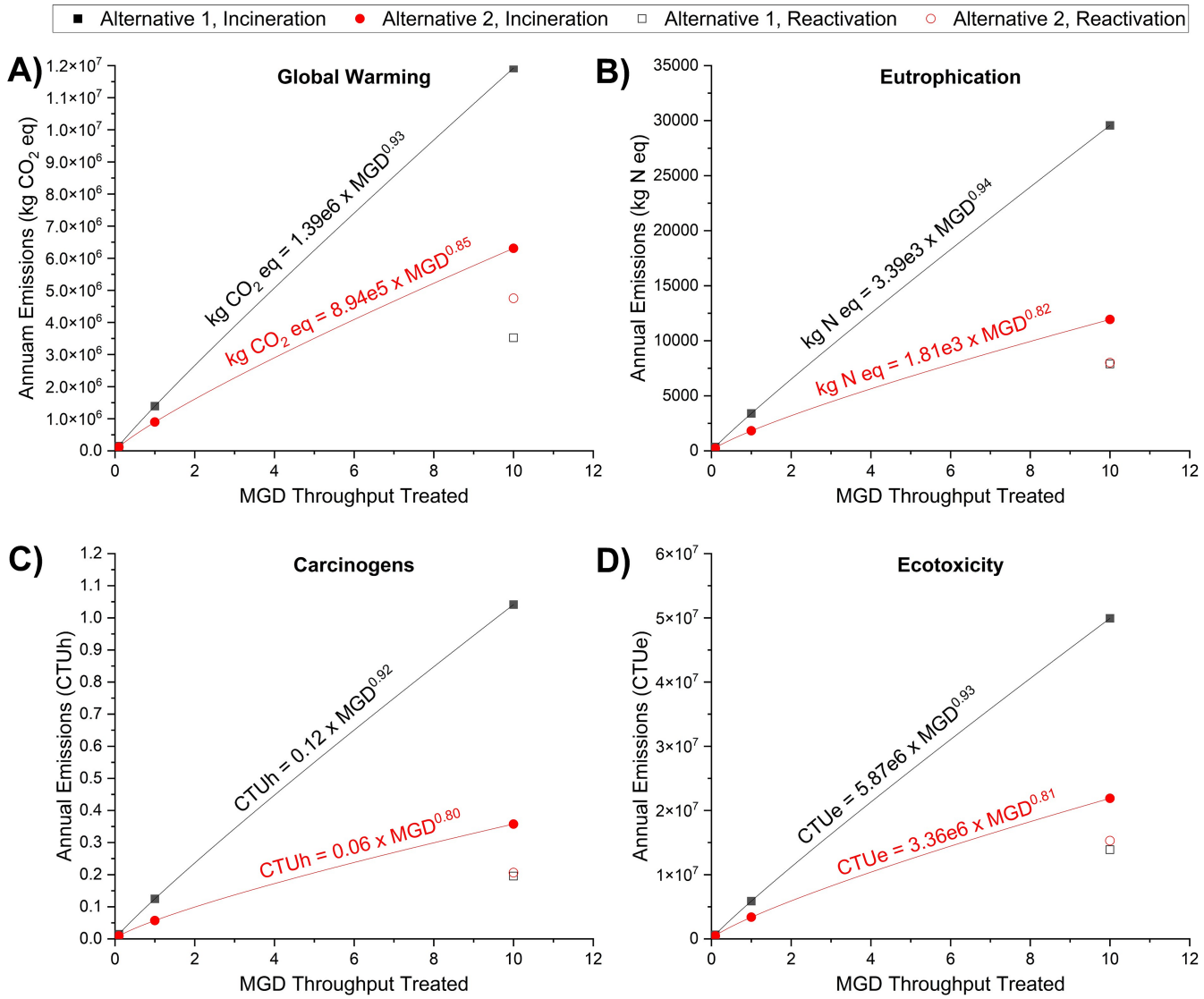

FIGURE 4 | Impact curves for four LCA outcomes across 0.1 to 10 MGD flow range. Data shown for (A) global warming, (B) eutrophication, (C) carcinogens, and (D) ecotoxicity.

3.3 | Sensitivity Analysis on Media Changeout Frequency

For GAC-based Alternative 1, media use was the primary driver of environmental impacts, so the effect of varying BVs to breakthrough beyond the baseline assumption of 10,000 was evaluated, spanning 2,000 to 50,000 BV to breakthrough to represent practical boundary conditions. These boundary conditions were selected to account for the wide range of WRRF matrices and PFAS profiles, as varied co-constituents or PFAS chemistries are likely to result in differing media use rates in practice. Users looking to estimate impacts of removing additional PFAS or PFAS at higher concentrations can conduct this sensitivity analysis to estimate site- or matrix-specific emissions associated with a given sourcewater. Because potential regulatory limits for PFAS in wastewater effluent remain uncertain, the applicability of impact curves to all WRRFs remains uncertain. Acknowledging that real-world, tertiary-treated water quality and PFAS concentrations may vary significantly by site, users can judge whether impact curves overestimate or underestimate site-specific environmental impacts by comparing actual tertiary water quality to the assumed design basis in Tables S1-1 and S1-2. Alternately, users can estimate media changeout rates using modeling, RSSCT, or pilot testing, then use sensitivity analyses results provided in Figure 5 to more quantitatively adjust estimated impacts.

Environmental impacts to global warming, ecotoxicity, carcinogens, and eutrophication increased linearly with media use rates and decreased inversely with increasing BVs to changeout, as illustrated in Figure 5. Impacts from single-use GAC with incineration were estimated to be 3× to 7× higher than for GAC reactivation and reuse, with this difference greatest for carcinogenic impacts and lowest for global warming. The single-use GAC system was found to be far more sensitive to variability in media changeout frequency relative to the GAC system with reactivation and reuse. Despite capital infrastructure and electricity being two facets of this system that do not linearly scale with media changeout frequency, their small relative contributions to

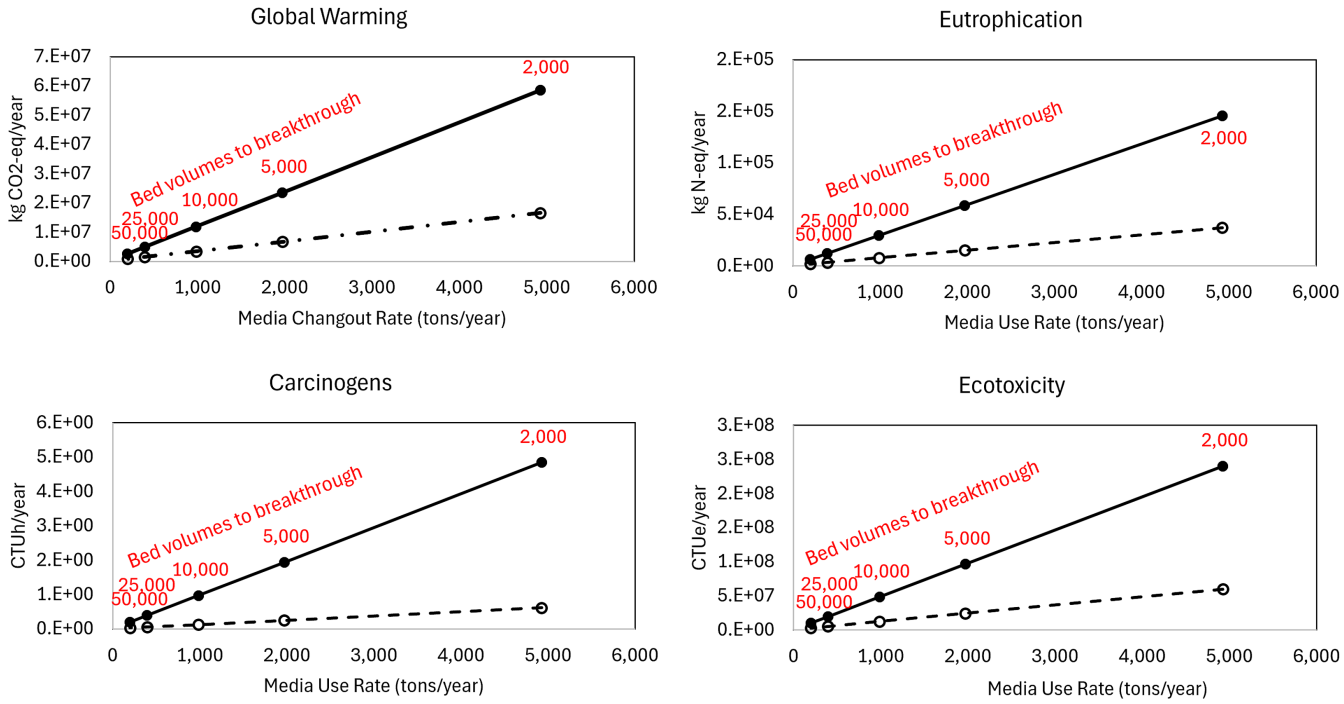

FIGURE 5 | Sensitivity analysis on impact of media use rate (*y*-axis) and bed volumes to breakthrough (in red) on environmental impacts for WRRF effluent PFAS treatment, 10-MGD WRRFs with Alternative 1 single-use GAC (solid line) or GAC with reactivation and reuse (dashed line).

overall system emissions resulted in a *y*-intercept near zero for all relationships shown in Figure 5.

3.4 | Sensitivity Analysis on Energy Sources

While environmental impacts in Alternative 1 are driven by sorbent media production and disposal, energy inputs serve as the primary driver for Alternative 2 due to the electricity requirements of RO membranes (Figure 3). Thus, environmental impacts of the overall treatment system can be influenced by the type(s) of electricity used in the WRRF's energy matrix. To assess the effect, differing energy matrices on environmental impacts for Alternative 2 were evaluated for a range of electricity sources.

As shown in Figure 6, environmental impacts associated with Alternative 2 vary significantly as a function of the energy source used in system models. Use of a renewable energy source (e.g., hydropower and wind) can reduce overall system emissions by >50% for certain categories. Among renewable sources, hydropower and wind power were found to be the least emissions-intensive energy sources, supported by previous literature (Amponsah et al. 2014; IPCC 2011). Conversely, hard coal is significantly more emissions-intensive than its renewable counterparts, with coal-based energy increasing overall system emissions by nearly 300% on average relative to the national average consumer energy mix. Environmental impacts for specific locations can be estimated using weighted averages based on the energy source profile of that region. The sensitivity of all ten impact categories to different energy sources are detailed in Table S4-1.

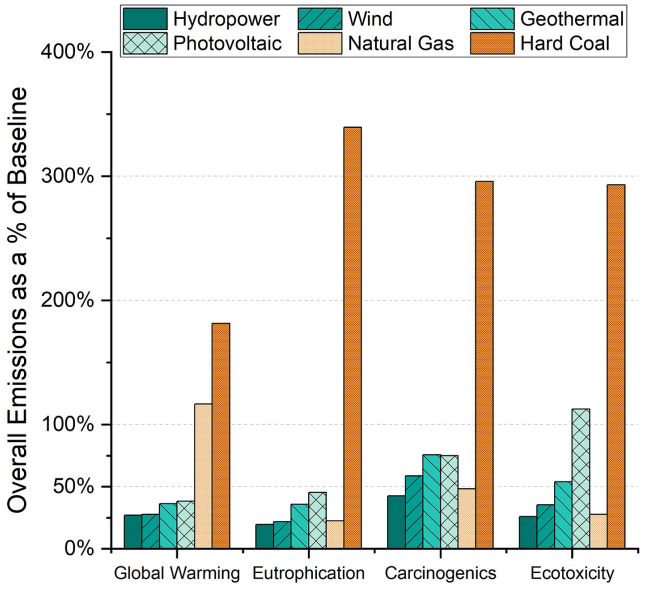

FIGURE 6 | Sensitivity analysis on impact of energy sources on environmental impacts for WRRF effluent PFAS treatment; 10-MGD WRRFs with Alternative 2, assuming incineration of spent GAC.

3.5 | Social Cost of GHG Emissions

The "social cost of carbon" is an established framework used to quantify net harm to society resulting from GHG emissions, including impacts to agricultural productivity, flood risk, human health, conflict risk, and ecosystem services (U.S. EPA 2023a). The estimated, 20-year net-present-value

TABLE 2 | Annualized PFAS treatment costs and social costs of carbon emissions from PFAS treatment and destruction at WRRF effluent (in millions of USD, 2023 basis).

	Design	PFAS treatment costs (20-year NPE, $Mil)	Social cost of carbon (20-year NPE, $Mil)	Ratio of treatment costs to social cost of carbon
Alternative 1	0.1 MGD	$8.5	$0.69	12
	1.0 MGD	$33	$6.5	5.1
	10 MGD	$217	$55	3.9
	10 MGD with reuse	$243	$16	15
Alternative 2	0.1 MGD	$14	$0.53	27
	1.0 MGD	$31	$4.2	7.5
	10 MGD	$126	$29.25	4.3
	10 MGD with reuse	$215	$22.03	9.8

social costs of GHG emissions are presented below in Table 2 and compared with previously estimated financial costs of PFAS treatment and destruction (Ling et al. 2024), adjusted to match assumptions used in this paper. The social cost of GHG emissions for the scenarios evaluated was estimated to be between 4× and 27× lower than annualized capital and operations costs for WRRFs between 0.1 and 10 MGD. This ratio is lower at higher flow rates, reflecting the difference in power function for costs (lower power coefficient, more economy of scale) versus environmental impacts (higher power coefficient, less economy of scale, especially for Alternative 1). Thus, large facilities on the order of hundreds of MGD could see social costs of GHG emissions approaching financial costs. Scenarios with GAC reuse have higher ratios, reflecting more environmental impact savings than financial savings from this operational choice. The scenario with the highest ratio of financial costs to social cost of carbon was Alternative 2 at 0.1 MGD, which has relatively high capital costs per volume treated due to many treatment processes with high economies of scale.

4 | Discussion

4.1 | Estimated GHG Emissions for Minnesota as a Case Study

Minnesota reported statewide 2022 total GHG emissions as 127 million metric tons/year of CO_2 equivalents, 400,000 of which were emitted by the wastewater sector (MPCA 2025). The additional emissions estimated here for adding PFAS treatment are 620,000 metric tons CO_2 equivalents/year if including all WRRFs greater than 10 MGD and 840,000 metric tons/year if including all WRRFs greater than 1 MGD. This would correspond to a 2.3- to 3.1-fold increase in GHG emissions from the wastewater sector and a 0.5%–0.7% increase in total statewide emissions (MPCA 2025). This projected increase in GHG emissions from the wastewater sector would make it more difficult for WRRFs to achieve GHG-neutral operation. For context, 620,000–840,000 metric tons/year from PFAS treatment in WRRF effluent is equivalent to the annual emissions of about 160,000 surplus conventional cars (U.S. EPA 2023b).

4.2 | Media Changeout Frequency Considerations

Media production and disposal accounted for a majority of the environmental impacts estimated for Alternative 1, consistent with findings from previous studies (Ling et al. 2025; Smith et al. 2025), so differences in media use rates between WRRFs of similar size significantly change environmental impacts (Figure 5), Media use rates depend on a combination of permit conditions like averaging period, limit value, and PFAS included; water quality conditions like PFAS concentrations, nontarget co-constituents that compete for adsorption sites, solution pH, organic matter makeup; and operational choices and risk preferences. Which individual PFAS are assigned discharge permit limits also strongly affects media changeout, as breakthrough can occur between 2 and 10 times faster for four-carbon PFAS compared to eight-carbon analogs when using GAC (Westreich et al. 2018; J. Burkhardt et al. 2019; Franke et al. 2021). This means that removal of short-chain compounds will use more media for a given amount of water treated and thus incur more environmental externalities for the overall system. Approaches to predict media use rates, in order of increasing complexity and accuracy, include computational modeling (Burkhardt et al. 2022), rapid small-scale column tests (RSSCTs), and on-site pilot tests. Complex water quality is difficult to account for in modeling, contributing uncertainty to model results. RSSCTs use smaller media and columns and can achieve breakthrough faster than pilot tests, but commonly underpredict media use rates (Kempisty et al. 2022).

Another complication impacting media changeout frequency is that current PFAS analytical results are expensive and can take weeks to receive (U.S. EPA 2021a), limiting the ability of WRRFs to rely solely on PFAS analysis to initiate changeout. In addition, PFAS profiles entering WRRFs can vary significantly with time (Szabo et al. 2023), which complicates breakthrough modeling and makes it difficult to guarantee compliance with low-level PFAS effluent limits for individual WRRFs.

4.3 | Implications for Media Reactivation and Reuse

Two separate 10 MGD scenarios were evaluated in this study to elucidate the impact of reusing reactivated carbon. The first 10

MGD scenario matched the 1 MGD and 0.1 MGD scenarios in that spent GAC is sent to off-site incineration without reuse. The second scenario assumed that spent GAC was thermally reactivated and returned to the site for reuse, with 20% makeup of virgin carbon due to GAC mass lost during reactivation. Sending spent GAC to reactivation without a site-specific contract and purchasing reactivated GAC through the general pool is an alternate disposal method that was not evaluated here but could be a lower impact option for smaller facilities. Reactivating and reusing GAC for purposes other than PFAS removal, reflecting downgraded GAC quality, is another alternate disposal option not evaluated here.

Reuse of reactivated media significantly reduced environmental impacts of PFAS treatment, with 70% and 25% reductions in GHG emissions relative to single-use GAC for Alternatives 1 and 2, respectively. In fact, GAC reuse in Alternative 1 is associated with lower GHG emissions than Alternative 2 with either GAC reuse or virgin GAC use, suggesting that reusing GAC reduces secondary environmental impacts more than using RO to reduce flowrates requiring sorbent treatment.

Considerations for reactivated media reuse include performance and logistics. Previous work has found that PFAS adsorption with reactivated media is effective when reusing media at the same site but less so when using general-pool reactivated carbon from other sources (Westreich et al. 2018). Same-site reuse requires a specific contract with a reactivation facility to ensure separation throughout the process and is typically limited to facilities changing out at least 80,000 lbs. of media per event (Barr Engineering Co. and Hazen and Sawyer 2023). Based on the design empty-bed contact time of 15 min per vessel used in Alternative 1, 80,000 lbs. of media per changeout corresponds to facilities treating greater than about 2 MGD. Facilities reactivating and reusing GAC will also need to purchase a "swing load" of carbon to use while GAC is being reactivated offsite and 15%–30% virgin carbon after each reactivation cycle to make up for GAC lost during reactivation (U.S. EPA 2026; Redding 2024). While GAC reactivation has been validated for restoration of PFAS sorption capacity on the media itself, questions still remain regarding the ultimate fate of all PFAS during reactivation, as current analytical methods limit researchers' ability to close fluorine mass balances through the process (DiStefano et al. 2022).

4.4 | Benefits of PFAS Treatment and Destruction

This and previous work outlined financial, social, and environmental costs of treating and destroying PFAS in WRRF effluent. There are also clear benefits, but these are more diffuse in impact and harder to quantify.

While total human health impacts from PFAS exposure are estimated to incur costs between $6 and $60 billion each year in the United States (Obsekov et al. 2023), estimated human health benefit of treating PFAS in WRRF effluent is difficult to estimate. The US EPA estimates that drinking water only accounts for 20% of total PFAS exposure (U.S. EPA 2022b), with most exposure likely coming from food ingestion and dust inhalation (De Silva et al. 2021), which are not directly impacted by water quality improvements. Public drinking water systems in the United States are required to have PFOA and PFOS levels below 4 ng/L by 2031 (PFAS National Primary Drinking Water Regulation 2024; US EPA 2025b), so PFAS-associated human health risk from the US drinking water will be lower then, even in the case of ongoing wastewater discharges containing PFAS. And while some 10% of Americans get drinking water from private systems not affected by drinking water regulations; those groundwater systems are unlikely to be directly impacted by wastewater discharges. A recent study found that treating low-level PFAS concentrations in drinking water would not provide a net human health benefit (Smith et al. 2025). Treating an equivalent volume of wastewater would likely provide even less health benefit, as WRRF effluent is not directly tied to exposure beyond the currently limited instances of direct potable reuse. Thus, treating WRRF effluent for PFAS is not expected to provide meaningful health benefits through drinking water.

Human PFAS exposures from surface waters can also occur through nonpoint sources like consumption of contaminated fish or wildlife in the riparian food web (Barbo et al. 2023; Kotalik et al. 2025). Thus, treating upstream wastewater discharges for PFAS is more likely to improve health outcomes for subsistence fishermen who consume substantially more locally caught fish than the typical person.

The framing of this study assigns estimated treatment externalities to PFAS treatment, assuming that regulatory need or proactive action to treat PFAS drives installation of the discussed technologies. PFAS treatment at WRRFs is also likely to have environmental benefits related to improved water quality in immediate receiving waters for non-PFAS parameters. Compared to conventional activated sludge treatment, adding PFAS treatment to WRRFs would significantly improve discharge quality for conventional pollutants and many contaminants of emerging concern. Both alternatives require tertiary treatment to protect RO membranes or media bed life, resulting in substantially improved discharge water quality than conventional secondary WRRFs for conventional pollutants like solids (TSS) and organics (BOD). Wastewater treatment technologies similar to those evaluated in this paper have significantly reduced overall aquatic toxicity in downstream receiving waters (Alzahrani et al. 2013; Völker et al. 2019). In drinking water treatment, one recent study found that adding PFAS treatment reduced disinfection byproduct concentrations by about half (Evans et al. 2025), and similar reductions in WRRF effluent could benefit downstream aquatic life. Wastewater effluent that has been treated to remove PFAS is more likely to be suitable for direct, indirect, and de facto potable water reuse due to improved water quality across the range of parameters mentioned here, thus providing additional water supply security and potential for a more circular water economy. A recent study estimates that wastewater discharges impair drinking water for 10 million Americans due to PFAS (Ruyle et al. 2025), showing how much "de facto" reuse is occurring.

Reducing PFAS loading to downstream waters can have political benefits for wastewater discharges that eventually flow into another legal jurisdiction. For example, in the United States, many sovereign Tribal Nations retain reserved treaty rights that

guarantee Tribal Nations the right to hunt, fish, and gather in treaty protected areas. Limiting fish PFAS exposures and accumulation, notably for PFAS that did not exist at the signing of those treaties, is one way that intergovernmental commitments could be honored.

Reducing PFAS loading to the environment through various means also has abstract but important, potential benefits to the rights of nature and to future humanity. Because of their persistence, ambient processes to fully defluorinate PFAS have not been identified, meaning that ongoing PFAS emissions cause continued, irreversible increases in environmental PFAS mass (Arp et al. 2024; Ling 2024), increasing future exposures and compounding intergenerational injustice issues. Furthermore, the rights of nature is a legal concept that recognizes that the earth and living creatures have an inherent right to exist without being harmed by humans (Epstein et al. 2023). These rights could extend to the right to exist without anthropogenic contamination of persistent, toxic compounds.

4.5 | Regulatory Implications

Treating and destroying PFAS in municipal WRRF effluents have cross-media regulatory implications that involve tradeoffs between competing environmental and economic goals. For example, the state of Minnesota aims to achieve net-zero GHG emissions by 2050; however, increased GHG emissions from PFAS treatment and destruction at WRRFs would make meeting the net-zero goal more difficult. Many regulatory programs addressing wastewater discharges worldwide do not have straightforward regulatory mechanisms to simultaneously weigh competing large-scale environmental tradeoffs, especially when the environmental effects of wastewater treatment affect another environmental media. In many jurisdictions, wastewater effluent, wastewater biosolids, and air emissions are regulated under separate permitting authorities, making cumulative impact assessment of PFAS treatment at WRRFs regulatorily difficult. For example, the U.S. Clean Water Act allows for consideration of whether wastewater treatment would "cause more environmental damage to correct than to leave in place" (U.S. 40 CFR Part 131.10 2026) but only allows this consideration under a temporary permitting variance with complex regulatory approvals independent of air emission or solid waste permitting approvals. When evaluating the costs and benefits of PFAS treatment and destruction at WRRFs, care should be taken to incorporate the legal and regulatory mechanisms that control pollution and the cross-media impacts of those mechanisms.

4.6 | Considerations for Other Treatment Technologies

Because of their inherent persistence and mobility and low concentrations in wastewater, the environmental impacts associated with removing and destroying PFAS are greater than those for conventional pollutants like solids, organic matter, and nutrients. While this work evaluated impacts of technologies currently available at the commercial scale, other technologies undergoing development and scale-up could potentially be used in the future with lower environmental impacts. For example, foam fractionation and regenerable ion exchange (IXR) have both been recently applied to treat wastewater at WRRFs with design flows exceeding 1 MGD (Woodard et al. 2024; Wu et al. 2025) and have the potential for lower environmental impacts compared with the alternatives evaluated here (Ellis et al. 2023). Foam fractionation and regenerable media technologies produce liquid residuals with concentrated PFAS. Studies to date have evaluated supercritical water oxidation (SCWO), hydrothermal alkaline treatment (HALT), electrochemical oxidation (ECO), plasma vortex technology, and others for destruction of PFAS in concentrated residuals (Higgins et al. 2025; Woodard 2024). Further research is recommended regarding application of these technologies for complex WRRF effluent matrices, as well as for other stores of PFAS in WRRFs like sludge and biosolids.

5 | Conclusions

Removing and destroying PFAS in WRRF effluent would have economic, social, and environmental costs and benefits. This study estimates environmental impacts of PFAS treatment for wastewater effluent and translates estimated GHG emissions into associated social costs using a previously established framework. When paired with a previous study on financial costs (Ling et al. 2024), these results show a range of potential drawbacks to treating wastewater effluent for PFAS. Additional research is needed to better characterize and quantify the potential benefits of removing and destroying PFAS from WRRF effluent to support policy makers in conducting comprehensive cost–benefit analysis of potential regulations.

Environmental impacts from treating PFAS in WRRF effluent have the potential to affect aquatic, atmospheric, biological, and human systems in numerous ways. Such impacts are not limited to local regions or receiving bodies alone but rather affect a much broader scope of the environment due to a wide array of inputs from segments like coal mining, chemicals synthesis, and transportation methods with a global scope of materials production and environmental degradation. For adsorption-dominated PFAS treatment trains, use of new media and thermal treatment/destruction of spent media has the greatest environmental impacts while for high-pressure membrane PFAS treatment trains, electricity requirements exert the greatest environmental impact. Overall, RO-based Alternative 2 is projected to be a more sustainable approach to PFAS treatment in WRRF effluent than GAC-based Alternative 1 despite its higher energy demands and more complex process configuration. Similarly, GAC-based technologies with reactivation and reuse are associated with fewer environmental externalities than a single-use GAC operation. The heavy contribution from ongoing consumable inputs means that environmental impacts have limited economies-of-scale for larger facilities when using Alternative 1.

Wastewater utilities and regulators need tools that can be used to estimate environmental impacts and environmental externalities of PFAS treatment to inform permitting and long-term planning. This study presents impact curves to predict GHG

emissions, carcinogenicity, ecotoxicity, and eutrophication impacts of removing and destroying PFAS in WRRF effluent using two treatment trains with varied waste disposal methods. These curves can be used in conjunction with previously published cost curves (Ling et al. 2024) to evaluate economic and environmental impacts of potential future treatment systems and to compare the costs and benefits of PFAS treatment. Decision-making around advanced water and wastewater treatment should consider environmental and community impacts in addition to other costs and benefits.

Author Contributions

Anderson C. Ellis: data curation, formal analysis, visualization, writing – original draft, writing – review and editing, methodology, software, investigation. **Alison L. Ling:** conceptualization, investigation, formal analysis, data curation, methodology, visualization, writing – review and editing, writing – original draft, supervision. **Rebecca Vermace:** methodology, writing – original draft, writing – review and editing, data curation, formal analysis, funding acquisition, investigation. **R. Hapaki Lorenzo Quintana:** writing – original draft, investigation. **Scott Kyser:** conceptualization, methodology, writing – review and editing, writing – original draft, investigation.

Acknowledgments

The authors are grateful to the broader Barr Engineering, Hazen and Sawyer, and MPCA team that developed conceptual designs for PFAS treatment in wastewater effluent as part of the 2023 study "Evaluation of Current Alternatives and Estimated Cost Curves for PFAS Removal and Destruction from Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water" and to Don Richard for review and suggestions.

Conflicts of Interest

Anderson Ellis is an employee of a company that develops, demonstrates, and sells PFAS treatment technologies and services. Rebecca Vermace works at a company providing PFAS treatment consulting services.

Data Availability Statement

The data that support the findings of this study are available in the Supporting Information of this article.

References

Ackerman Grunfeld, D., D. Gilbert, J. Hou, et al. 2024. "Underestimated Burden of Per- and Polyfluoroalkyl Substances in Global Surface Waters and Groundwaters." *Nature Geoscience* 17: 340–346. https://doi.org/10.1038/s41561-024-01402-8.

Alzahrani, S., A. W. Mohammad, N. Hilal, P. Abdullah, and O. Jaafar. 2013. "Comparative Study of NF and RO Membranes in the Treatment of Produced Water—Part I: Assessing Water Quality." *Desalination* 315: 18–26. https://doi.org/10.1016/j.desal.2012.12.004.

Amponsah, N. Y., M. Troldborg, B. Kington, I. Aalders, and R. L. Hough. 2014. "Greenhouse Gas Emissions From Renewable Energy Sources: A Review of Lifecycle Considerations." *Renewable and Sustainable Energy Reviews* 39: 461–475. https://doi.org/10.1016/j.rser.2014.07.087.

Aro, R., U. Eriksson, A. Kärrman, F. Chen, T. Wang, and L. W. Y. Yeung. 2021. "Fluorine Mass Balance Analysis of Effluent and Sludge From Nordic Countries." *ACS ES&T Water* 1, no. 9: 2087–2096. https://doi.org/10.1021/acsestwater.1c00168.

Arp, H. P. H., A. Gredelj, J. Glüge, M. Scheringer, and I. T. Cousins. 2024. *The Global Threat From the Irreversible Accumulation of Trifluoroacetic Acid (TFA)*. ChemRxiv. https://doi.org/10.26434/chemrxiv-2024-0djqt.

Bălan, S. A., V. C. Mathrani, D. F. Guo, and A. M. Algazi. 2021. "Regulating PFAS as a Chemical Class Under the California Safer Consumer Products Program." *Environmental Health Perspectives* 129: 25001. https://doi.org/10.1289/EHP7431.

Banti, D. C., M. Tsangas, P. Samaras, and A. Zorpas. 2020. "LCA of a Membrane Bioreactor Compared to Activated Sludge System for Municipal Wastewater Treatment." *Membranes* 10, no. 12: 421. https://doi.org/10.3390/membranes10120421.

Barbo, N., T. Stoiber, O. V. Naidenko, and D. Q. Andrews. 2023. "Locally Caught Freshwater Fish Across the United States Are Likely a Significant Source of Exposure to PFOS and Other Perfluorinated Compounds." *Environmental Research* 220: 115165. https://doi.org/10.1016/j.envres.2022.115165.

Bare, J. 2011. "TRACI 2.0: The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts 2.0." *Clean Technologies and Environmental Policy* 13, no. 5: 687–696. https://doi.org/10.1007/s10098-010-0338-9.

Barr Engineering Co, and Hazen and Sawyer. 2023. *Evaluation of Current Alternatives and Estimated Cost Curves for PFAS Removal and Destruction From Municipal Wastewater, Biosolids, Landfill Leachate, and Compost Contact Water*. Minnesota Pollution Control Agency (MPCA). https://www.pca.state.mn.us/sites/default/files/c-pfc1-26.pdf.

Boyer, T. H., A. Ellis, Y. Fang, C. E. Schaefer, C. P. Higgins, and T. J. Strathmann. 2021. "Life Cycle Environmental Impacts of Regeneration Options for Anion Exchange Resin Remediation of PFAS Impacted Water." *Water Research* 207: 117798. https://doi.org/10.1016/j.watres.2021.117798.

Burkhardt, J. B., N. Burns, D. Mobley, J. G. Pressman, M. L. Magnuson, and T. F. Speth. 2022. "Modeling PFAS Removal Using Granular Activated Carbon for Full-Scale System Design." *Journal of Environmental Engineering* 148, no. 3: 1–11. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001964.

Burkhardt, J., C. Vandermeyden, N. Burns, et al. 2019. *Modeling Pilot-Scale GAC PFAS Adsorption for the Simulation of Full-Scale Performance and Costs*. AWWA Water Quality and Technology Conference.

Chen, J., S. H.-A. Samaei, L. Zhang, et al. 2025. "Per- and Poly-Fluoroalkyl Substances (PFAS) in Water Matrices Impacted by Industrial Activities: Challenges and Treatment Strategies." *Journal of Hazardous Materials Advances* 20: 100920. https://doi.org/10.1016/j.hazadv.2025.100920.

Chow, S. J., H. C. Croll, N. Ojeda, et al. 2022. "Comparative Investigation of PFAS Adsorption Onto Activated Carbon and Anion Exchange Resins During Long-Term Operation of a Pilot Treatment Plant." *Water Research* 226: 119198. https://doi.org/10.1016/j.watres.2022.119198.

Cookson, E. S., and R. L. Detwiler. 2022. "Global Patterns and Temporal Trends of Perfluoroalkyl Substances in Municipal Wastewater: A Meta-Analysis." *Water Research* 221: 118784. https://doi.org/10.1016/j.watres.2022.118784.

Crenna, E., M. Secchi, L. Benini, and S. Sala. 2019. "Global Environmental Impacts: Data Sources and Methodological Choices for Calculating Normalization Factors for LCA." *International Journal of Life Cycle Assessment* 24, no. 10: 1851–1877. https://doi.org/10.1007/s11367-019-01604-y.

De la Cruz, F. B., I. A. Titaley, Y. Wang, J. A. Field, and M. A. Barlaz. 2025. "Nationwide Estimate of Volatile Per- and Polyfluoroalkyl Substance (PFAS) Emissions From U.S. Landfills via Landfill Gas." *Environmental Science & Technology* 59: 24899–24908. https://doi.org/10.1021/acs.est.5c08763.

De Silva, A. O., J. M. Armitage, T. A. Bruton, et al. 2021. "PFAS Exposure Pathways for Humans and Wildlife: A Synthesis of

Current Knowledge and Key Gaps in Understanding." *Environmental Toxicology and Chemistry* 40, no. 3: 631–657. https://doi.org/10.1002/etc.4935.

DiStefano, R., T. Feliciano, R. A. Mimna, A. M. Redding, and J. Matthis. 2022. "Thermal Destruction of PFAS During Full-Scale Reactivation of PFAS-Laden Granular Activated Carbon." *Remediation Journal* 32, no. 4: 231–238. https://doi.org/10.1002/rem.21735.

Drury, N. L., R. L. Prueitt, and B. D. Beck. 2024. "Understanding IARC's PFOA and PFOS Carcinogenicity Assessments." *Regulatory Toxicology and Pharmacology* 154: 105726. https://doi.org/10.1016/j.yrtph.2024.105726.

ECHA. 2023a. *Annex A: Manufacture and Uses, Annex XV Proposal to Restrict PFAS, v2*. ECHA. https://echa.europa.eu/documents/10162/d2f7fce1-b089-c4fd-1101-2601f53a07d1.

ECHA. 2023b. *Annex XV Restriction Report: Per- and Polyfluoroalkyl Substances (PFASs)*. European Chemicals Agency. https://echa.europa.eu/documents/10162/f605d4b5-7c17-7414-8823-b49b9fd43aea.

Ellis, A. C., T. H. Boyer, Y. Fang, C. J. Liu, and T. J. Strathmann. 2023. "Life Cycle Assessment and Life Cycle Cost Analysis of Anion Exchange and Granular Activated Carbon Systems for Remediation of Groundwater Contaminated by Per- and Polyfluoroalkyl Substances (PFASs)." *Water Research* 243: 120324. https://doi.org/10.1016/j.watres.2023.120324.

Emery, I., D. Kempisty, B. Fain, and E. Mbonimpa. 2019. "Evaluation of Treatment Options for Well Water Contaminated With Perfluorinated Alkyl Substances Using Life Cycle Assessment." *International Journal of Life Cycle Assessment* 24, no. 1: 117–128. https://doi.org/10.1007/s11367-018-1499-8.

Epstein, Y., A. M. Ellison, H. Echeverría, and J. K. Abbott. 2023. "Science and the Legal Rights of Nature." *Science* 380, no. 6646: eadf4155. https://doi.org/10.1126/science.adf4155.

European Union. 2020. *Directive - 2020/2184 - EN - EUR-Lex*, No. 2020/2184. Publications Office of the European Union. https://eur-lex.europa.eu/eli/dir/2020/2184/oj/eng.

European Union. 2024. *Directive 2024/3019 of the European Parliament and of the Council of 27 November 2024 Concerning Urban Wastewater Treatment (Recast) (Text With EEA Relevance)*, No. 2024/3019. Official Journal of the European Union. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=OJ:L_202403019.

Evans, S. S., V. Subramaniam, A. Cullen, C. Campbell, O. V. Naidenko, and D. Q. Andrews. 2025. "PFAS Treatment as an Opportunity for Broader Drinking Water Improvements: Evidence From U.S. Water Systems." *ACS ES&T Water* 5: 5447–5459. https://doi.org/10.1021/acsestwater.5c00519.

Evich, M. G., M. J. B. Davis, J. P. McCord, et al. 2022. "Per- and Polyfluoroalkyl Substances in the Environment." *Science* 375, no. 6580: eabg9065. https://doi.org/10.1126/science.abg9065.

Franke, V., M. Ullberg, P. McCleaf, M. Wålinder, S. J. Köhler, and L. Ahrens. 2021. "The Price of Really Clean Water: Combining Nanofiltration With Granular Activated Carbon and Anion Exchange Resins for the Removal of Per- And Polyfluoroalkyl Substances (PFASs) in Drinking Water Production." *ACS ES&T Water* 1, no. 4: 782–795. https://doi.org/10.1021/acsestwater.0c00141.

Glover, C. J., J. A. Phillips, E. A. Marchand, and S. R. Hiibel. 2022. "Modeling and Life Cycle Assessment of a Membrane Bioreactor–Membrane Distillation Wastewater Treatment System for Potable Reuse." *Separations* 9, no. 6: 151. https://doi.org/10.3390/separations9060151.

Higgins, R. J., H. Kim, E. Houtz, P. Newman, D. Casey, and Y. I. Cho. 2025. "PFAS Destruction in IX Still Bottoms With Plasma Vortex Technology." *ACS ES&T Water* 5, no. 5: 2110–2119. https://doi.org/10.1021/acsestwater.4c00863.

IPCC. 2011. "Renewable Energy Sources and Climate Change Mitigation: Special Report of the Intergovernmental Panel on Climate Change." *Choice Reviews Online* 49, no. 11: 49–6309. https://doi.org/10.5860/choice.49-6309.

ISO. 2006. *ISO 14040:2006 Environmental Management—Life Cycle Assessment—Principles and Framework*. ISO. https://www.iso.org/standard/37456.html.

ITRC. 2022. *PFAS Treatment Technologies* PFAS Technical and Regulatory Guidance Document and Fact Sheets. ITRC. https://pfas-1.itrcweb.org/12-treatment-technologies/.

ITRC. 2025. *PFAS Environmental Media Values Table*. ITRC. https://pfas-1.itrcweb.org/wp-content/uploads/2025/05/ITRCPFASEnvironmentalMediaValuesTables_Apr-FINAL.xlsx.

Joerss, H., F. Freeling, S. van Leeuwen, et al. 2024. "Pesticides Can Be a Substantial Source of Trifluoroacetate (TFA) to Water Resources." *Environment International* 193: 109061. https://doi.org/10.1016/j.envint.2024.109061.

Kali, S. E., H. Österlund, M. Viklander, and G.-T. Blecken. 2025. "Stormwater Discharges Affect PFAS Occurrence, Concentrations, and Spatial Distribution in Water and Bottom Sediment of Urban Streams." *Water Research* 271: 122973. https://doi.org/10.1016/j.watres.2024.122973.

Kempisty, D., E. Arevalo, A. Spinelli, et al. 2022. "Granular Activated Carbon Adsorption of Perfluoroalkyl Acids From Ground and Surface Water." *AWWA Water Science* 4, no. 1: e1269. https://doi.org/10.1002/aws2.1269.

Kotalik, C. J., L. E. Hubbard, B. G. Perrotta, et al. 2025. "Bioaccumulation and Transfer of Per- and Polyfluoroalkyl Substances (PFAS) in a Stream and Riparian Food Web Contaminated by Food Processing Wastewater." *Environmental Science & Technology* 59, no. 36: 19444–19456. https://doi.org/10.1021/acs.est.5c04867.

Kwiatkowski, C. F., D. Q. Andrews, L. S. Birnbaum, et al. 2020. "Scientific Basis for Managing PFAS as a Chemical Class." *Environmental Science & Technology Letters* 7, no. 8: 532–543. https://doi.org/10.1021/acs.estlett.0c00255.

Lazarova, V., S. Martin Ruel, B. Barillon, and P. Dauthuille. 2012. "The Role of MBR Technology for the Improvement of Environmental Footprint of Wastewater Treatment." *Water Science and Technology* 66, no. 10: 2056–2064. https://doi.org/10.2166/wst.2012.413.

Ling, A. L. 2024. "Estimated Scale of Costs to Remove PFAS From the Environment at Current Emission Rates." *Science of the Total Environment* 918: 170647. https://doi.org/10.1016/j.scitotenv.2024.170647.

Ling, A. L., R. Aubert, E. Bersi, S. Horel, and H. P. H. Arp. 2026. "PFAS Remediation Across Europe: Costs and Limited Impacts." *Environmental Science: Processes & Impacts* 26, no. 7: 2092–2901. https://doi.org/10.1039/D5EM00870K.

Ling, A. L., T. Stegner, M. Thompson, et al. 2025. "Spent Media Management Pathways for PFAS Treatment Applications." *Water Environment Research* 97, no. 7: e70130. https://doi.org/10.1002/wer.70130.

Ling, A. L., R. R. Vermace, A. J. McCabe, K. M. Wolohan, and S. J. Kyser. 2024. "Is Removal and Destruction of Perfluoroalkyl and Polyfluoroalkyl Substances From Wastewater Effluent Affordable?" *Water Environment Research* 96, no. 1: e10975. https://doi.org/10.1002/wer.10975.

Maine State Legislature. 2021. *An Act to Stop Perfluoroalkyl and Polyfluoroalkyl Substances Pollution*. Maine State Legislature, 130LR0942(05). http://www.mainelegislature.org/legis/bills/getPDF.asp?paper=HP1113&item=5&snum=130.

McCleaf, P., S. Englund, A. Östlund, K. Lindegren, K. Wiberg, and L. Ahrens. 2017. "Removal Efficiency of Multiple Poly- and

Perfluoroalkyl Substances (PFASs) in Drinking Water Using Granular Activated Carbon (GAC) and Anion Exchange (AE) Column Tests." *Water Research* 120: 77–87. https://doi.org/10.1016/j.watres.2017.04.057.

Moore, F. C., M. A. Drupp, J. Rising, S. Dietz, I. Rudik, and G. Wagner. 2024. "Synthesis of Evidence Yields High Social Cost of Carbon due to Structural Model Variation and Uncertainties." *Proceedings of the National Academy of Sciences of the United States of America* 121, no. 52: e2410733121. https://doi.org/10.1073/pnas.2410733121.

MPCA. 2025. *Greenhouse Gas Emissions in Minnesota 2005–2022*. MPCA. https://www.pca.state.mn.us/sites/default/files/lraq-3sy25.pdf.

National Academies of Sciences, Engineering, and Medicine. 2022. "Chapter 3: Potential Health Effects of PFAS." In *Guidance on PFAS Exposure, Testing, and Clinical Follow-Up*. National Academies Press. http://nap.nationalacademies.org/26156.

Obsekov, V., L. G. Kahn, and L. Trasande. 2023. "Leveraging Systematic Reviews to Explore Disease Burden and Costs of Per- and Polyfluoroalkyl Substance Exposures in the United States." *Exposure and Health* 15, no. 2: 373–394. https://doi.org/10.1007/s12403-022-00496-y.

Office of the Federal Register (OFR). Revised 2026. "Establishment of Water Quality Standards, 40 CFR Part 131.10." https://www.ecfr.gov/current/title-40/chapter-I/subchapter-D/part-131/subpart-B/section-131.10.

Önnby, L., I. Liagkouridis, C. Baresel, and L. W. Y. Yeung. 2025. "PFAS in Swedish Wastewater: Occurrence, Transformation, and Fate in Biological Treatment Processes." *Environmental Chemistry and Ecotoxicology* 7: 2744–2755. https://doi.org/10.1016/j.enceco.2025.10.024.

Park, M., S. Wu, I. J. Lopez, J. Y. Chang, T. Karanfil, and S. A. Snyder. 2020. "Adsorption of Perfluoroalkyl Substances (PFAS) in Groundwater by Granular Activated Carbons: Roles of Hydrophobicity of PFAS and Carbon Characteristics." *Water Research* 170: 115364. https://doi.org/10.1016/j.watres.2019.115364.

PFAS National Primary Drinking Water Regulation. 2024. *Federal Register*. PFAS National Primary Drinking Water Regulation. https://www.federalregister.gov/documents/2024/04/26/2024-07773/pfas-national-primary-drinking-water-regulation.

Plakas, K. V., A. A. Georgiadis, and A. J. Karabelas. 2015. "Sustainability Assessment of Tertiary Wastewater Treatment Technologies: A Multi-Criteria Analysis." *Water Science and Technology* 73, no. 7: 1532–1540. https://doi.org/10.2166/wst.2015.630.

Rahman, S. M., M. J. Eckelman, A. Onnis-Hayden, and A. Z. Gu. 2016. "Life-Cycle Assessment of Advanced Nutrient Removal Technologies for Wastewater Treatment." *Environmental Science & Technology* 50, no. 6: 3020–3030. https://doi.org/10.1021/acs.est.5b05070.

Redding, A. 2024. *Best Practices for PFAS Removal and Reactivation*. Calgon Carbon Webinar. https://event.eu.on24.com/wcc/r/8000074185/2E91C054138F3ADEF26A18D1EAC82C1B.

Ritchie, H., P. Rosado, and M. Roser. 2020. *Breakdown of Carbon Dioxide, Methane and Nitrous Oxide Emissions by Sector*. Our World in Data. https://ourworldindata.org/emissions-by-sector.

RTI International. 2023. *Characterizing PFAS Chemistries, Sources, Uses, and Regulatory Trends in U*. Consumer Product Safety Commission. https://www.cpsc.gov/s3fs-public/CPSC-PFAS-WhitePaper.pdf?VersionId=HYcrhWL_cSeA61QrR5bzoVOHRV0nw9_f.

Ruyle, B. J., E. H. Pennoyer, S. Vojta, et al. 2025. "High Organofluorine Concentrations in Municipal Wastewater Affect Downstream Drinking Water Supplies for Millions of Americans." *Proceedings of the National Academy of Sciences* 122, no. 3: e2417156122. https://doi.org/10.1073/pnas.2417156122.

Ryberg, M., M. D. M. Vieira, M. Zgola, J. Bare, and R. K. Rosenbaum. 2013. "Updated US and Canadian Normalization Factors for TRACI 2.1." *Clean Technologies and Environmental Policy* 16, no. 2: 329–339. https://doi.org/10.1007/s10098-013-0629-z.

Seay, B. A., K. Dasu, I. C. MacGregor, et al. 2023. "Per- and Polyfluoroalkyl Substances Fate and Transport at a Wastewater Treatment Plant With a Collocated Sewage Sludge Incinerator." *Science of the Total Environment* 874: 162357. https://doi.org/10.1016/j.scitotenv.2023.162357.

Smith, S. J., É. Sylvestre, M. Motelica-Wagenaar, B. Cantoni, P. Suresh Nair, and M. P. Parada. 2025. "PFAS Drinking Water Treatment Trade-Offs: Comparing the Health Burden of GAC Treatment to the Health Benefits of Reduced PFAS Exposure." *Environmental Science Processes & Impacts* 27: 1796–1809. https://doi.org/10.1039/D5EM00238A.

State of Minnesota. 2023. *Minnesota PFAS Ban, MN HF2310*. State of Minnesota.

Sun, S., M. Liang, D. Fan, et al. 2024. "Occurrence and Profiles of Perfluoroalkyl Substances in Wastewaters of Chemical Industrial Parks and Receiving River Waters: Implications for the Environmental Impact of Wastewater Discharge." *Science of the Total Environment* 945: 173993. https://doi.org/10.1016/j.scitotenv.2024.173993.

Szabo, D., J. Marchiandi, S. Samandra, et al. 2023. "High-Resolution Temporal Wastewater Treatment Plant Investigation to Understand Influent Mass Flux of Per- and Polyfluoroalkyl Substances (PFAS)." *Journal of Hazardous Materials* 447: 130854. https://doi.org/10.1016/j.jhazmat.2023.130854.

Thompson, K. A., S. Mortazavian, D. J. Gonzalez, et al. 2022. "Poly- and Perfluoroalkyl Substances in Municipal Wastewater Treatment Plants in the United States: Seasonal Patterns and Meta-Analysis of Long-Term Trends and Average Concentrations." *ACS ES&T Water* 2, no. 5: 690–700. https://doi.org/10.1021/acsestwater.1c00377.

Troxler, W., W. Anderson, C. McBride, et al. 2025. *PFAS Destruction by a Hazardous Waste Incinerator: Testing Results*. U.S. EPA Office of Research and Development. https://cfpub.epa.gov/si/si_public_record_Report.cfm?dirEntryId=367138&Lab=CEMM.

Tushar, M. M. R., Z. A. Pushan, N. Aich, and L. S. Rowles. 2024. "Balancing Sustainability Goals and Treatment Efficacy for PFAS Removal From Water." *NPJ Clean Water* 7, no. 1: 130. https://doi.org/10.1038/s41545-024-00427-1.

U.S. Bureau of Labor Statistics. 2025. *CPI Inflation Calculator*. U.S. Bureau of Labor Statistics. https://www.bls.gov/data/inflation_calculator.htm.

U.S. EPA. 2012. "Tool for the Reduction and and Assessment of Chemicals and Other Environmental Impacts (TRACI): User's Manual v2.1." EPA/600/R-12/554 https://nepis.epa.gov/Adobe/PDF/P100HN53.pdf.

U.S. EPA. 2021a. *Draft Method 1633, Analysis of Per- and Polyfluoroalkyl Substances (PFAS) in Aqueous, Solid, Biosolids, and Tissue Samples by LC-MS/MS, EPA 821-R-21-001* United States Environmental Protection Agency PFAS Innovative Treatment Team. United States Environmental Protection Agency. https://www.epa.gov/system/files/documents/2021-09/method_1633_draft_aug-2021.pdf.

U.S. EPA. 2021b. *Multi-Industry Per- and Polyfluoroalkyl Substances (PFAS) Study – 2021 Preliminary Report*. U.S. EPA (p. 81). https://www.epa.gov/system/files/documents/2021-09/multi-industry-pfas-study_preliminary-2021-report_508_2021.09.08.pdf.

U.S. EPA. 2022a. *Addressing PFAS Discharges in NPDES Permits and Through the Pretreatment Program and Monitoring Programs*. U.S. EPA. https://www.epa.gov/system/files/documents/2022-12/NPDES_PFAS_State%20Memo_December_2022.pdf.

U.S. EPA. 2022b. *Technical Fact Sheet: Drinking Water Health Advisories for Four PFAS (PFOA, PFOS, GenX Chemicals, and PFBS) (EPA 822-F-22-002)*. U.S. EPA. https://www.epa.gov/system/files/documents/2022-06/technical-factsheet-four-PFAS.pdf.

U.S. EPA. 2023a a. *EPA Report on the Social Cost of Greenhouse Gases: Estimates Incorporating Recent Scientific Advances [EPA-HQ-OAR-2021-0317]*. U.S. EPA. https://www.epa.gov/system/files/documents/2023-12/epa_scghg_2023_report_final.pdf.

U.S. EPA. 2023b. *Tailpipe Greenhouse Gas Emissions From a Typical Passenger Vehicle*. Office of Transportation and Air Quality. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1017FP5.pdf.

U.S. EPA. 2025a. *Implementing Case-by-Case Technology-Based Effluent Limitations in NPDES Permits for Pollutants of Emerging Concern*. U.S. EPA. https://www.epa.gov/system/files/documents/2025-01/case-by-case-tbel-factsheet.pdf.

U.S. EPA. 2025b. *EPA Announces It Will Keep Maximum Contaminant Levels for PFOA, PFOS [News Release]*. US EPA [News Release]. https://www.epa.gov/newsreleases/epa-announces-it-will-keep-maximum-contaminant-levels-pfoa-pfos.

U.S. EPA. 2026. *Interim Guidance on the Destruction and Disposal of Perfluoroalkyl and Polyfluoroalkyl Substances and Materials Containing Perfluoroalkyl and Polyfluoroalkyl Substances—2026 Version*. U.S. EPA.

Völker, J., M. Stapf, U. Miehe, and M. Wagner. 2019. "Systematic Review of Toxicity Removal by Advanced Wastewater Treatment Technologies via Ozonation and Activated Carbon." *Environmental Science & Technology* 53, no. 13: 7215–7233. https://doi.org/10.1021/acs.est.9b00570.

Westreich, P., R. Mimna, J. Brewer, and F. Forrester. 2018. "The Removal of Short-Chain and Long-Chain Perfluoroalkyl Acids and Sulfonates via Granular Activated Carbons: A Comparative Column Study." *Remediation Journal* 29, no. 1: 19–26. https://doi.org/10.1002/rem.21579.

Woodard, S. 2024. *Destruction of PFAS in IX Still Bottoms and Foamate: Pairing Specific Technologies for PFAS Treatment*. NEWMOA The Science of PFAS. https://www.newmoa.org/wp-content/uploads/2024/08/WoodardDestructionApril2024.pdf.

Woodard, S., P. Rodriguez, and C. Hutchins. 2024. *Treatment of a 1 MGD Lagoon System POTW Effluent to PFAS Drinking Water Standards Using Foam Fractionation*. ECT2 (Emerging Compounds Treatment Technologies), Battelle Chlorinated 2024, Denver CO. https://xcdacademy.s3.amazonaws.com/battelle/2024_Chlorinated/I8_1055_640_Woodard.pdf.

Wu, A., T. Zweifel, D. Kaczynski, et al. 2025. *Regenerative Anion Exchange for PFAS Remediation of Water: From Bench-Scale to Full-Scale Implementation*. AWMA Advancing PFAS Treatment and Remediation Solutions.

Zainzinger, V. 2025. *France Adopts Ban on PFAS in Consumer Products*. Chemical & Engineering News. https://cen.acs.org/regions/europe/France-adopts-ban-PFAS-consumer/103/web/2025/02.

Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Table S1-1**. Assumed influent concentrations and treatment goals for target PFAS in municipal wastewater (all units in ng/L). **Table S1-2**. Assumed initial and pretreated municipal wastewater quality. **Table S1-3**. Summary of design basis assumptions for GAC adsorption with HTI (Alternative 1). **Table S1-4**. Summary of capital materials use modeled for GAC adsorption with HTI (Alternative 1). **Table S1-5**. Annual unit quantities used for GAC adsorption with HTI (Alternative 1). **Table S1-6**. Summary of design basis assumptions for RO/GAC/AIX (Alternative 2). **Table S1-7**. Summary of capital materials use modeled for RO/GAC/AIX (Alternative. 2). **Table S1-8**. Annual unit quantities used for RO/GAC/AIX (Alternative 2). **Table S2-1**. Industry-relevant databases. **Table S2-2**. Specific itemized inventory item table. **Figure S2-1**. Process flowchart of Alternative 1, incineration (10 MGD). **Figure S2-2**. Process flowchart of Alternative 1, reactivation (10 MGD). **Figure S2-3**. Process flowchart of Alternative 2, incineration (10 MGD). **Figure S2-4**. Process flowchart of Alternative 2, reactivation (10 MGD). **Figure S3-1**. Normalized emissions for all impact categories shown for all treatment alternatives and design treatment throughputs. **Figure S3-2**. Emissions breakdown for Alternative 1, assuming incineration unless otherwise noted. Panel (A) shows 0.1 MGD throughput, panel (B) shows 1 MGD throughput, panel (C) shows 10 MGD throughput, and panel (D) shown 10 MGD throughput using GAC reactivation. Emissions are shown using the 10 TRACI-defined emissions categories. **Figure S3-3**. Emissions breakdown for Alternative 2, assuming incineration unless otherwise noted. Panel (A) shows 0.1 MGD throughput, panel (B) shows 1 MGD throughput, panel (C) shows 10 MGD throughput, and panel (D) shown 10 MGD throughput using GAC reactivation. Emissions are shown using the 10 TRACI-defined emissions categories. **Figure S3-4**. Impact curves for ozone depletion (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-5**. Impact curves for global warming (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-6**. Impact curves for smog (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-7**. Impact curves for acidification (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-8**. Impact curves for Eutrophication (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-9**. Impact curves for carcinogens (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-10**. Impact curves for noncarcinogens (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-11**. Impact curves for respiratory effects (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-12**. Impact curves for ecotoxicity (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-13**. Impact curves for fossil fuel depletion (left panel showing Alternative 1 with right panel showing Alternative 2). Reactivation condition at 10 MGD shown in red. **Figure S3-14**. Impact curves for the four indicator categories shown in Figure 4, presented in terms of annual PFAS mass removed rather than MGD throughput, assuming influent PFAS concentrations listed in S1-1 of 115 ng/L. Impacts shown for the EPA-defined categories of (A) global warming, (B) eutrophication, (C) carcinogens, and (D) ecotoxicity. **Table S4-1**. Electricity sources used in the sensitivity analysis for Alternative 2.