Plasma Chemistry and Plasma Processing (2026) 46:82
https://doi.org/10.1007/s11090-026-10688-y

ORIGINAL PAPER

Unraveling the Role of Gas-Phase Plasma Species in PFAS Degradation During Treatment of Contaminated Water with Submerged Gliding Arc Plasma (GAP) Discharge

Mikaela J. Surace[1] · Mobish A. Shaji[2] · Erich Heinzel[3] · Jimmy Murillo-Gelvez[3] · Jitlekha Tubsuri[1] · Alexander A. Fridman[2] · Alexander Rabinovich[2] · Erica R. McKenzie[3] · Gregory Fridman[4] · Christopher M. Sales[1,2]

Received: 28 February 2026 / Accepted: 10 June 2026
© The Author(s) 2026

Abstract
Per- and polyfluoroalkyl substances (PFAS) are resistant to degradation by typical treatment methods. Various technologies utilizing non-equilibrium plasma, including gliding arc plasma (GAP) discharge, show promise for degrading PFAS in water. This study investigated the mechanism by which air GAP discharge degrades PFAS in water. Experiments scavenging aqueous plasma species indicated that hydrated electrons (e^-_{aq}) are important to degradation, but do not play a significant role in mineralization, while reactive oxygen and nitrogen species (RONS) appeared to play a negligible role. On the other hand, experiments that scavenged gas-phase charged particles decreased PFAS degradation and completely inhibited mineralization. Based on these results, degradation products, and previous literature on PFAS degradation pathways and GAP treatment of PFAS-contaminated water, several significant pathways for PFAS degradation during GAP treatment were hypothesized: (1) thermal mineralization driven by gas-phase charged particles, (2) H/F exchange driven by free electrons (e^-) at the plasma-water interface or e^-_{aq} at the plasma-water interface and in the bulk liquid, and (3) fragmentation via charge transfer from gas-phase charged particles. While the significance of gas-phase charged particles to PFAS degradation in water during non-equilibrium plasma treatment has been previously hypothesized and modeled, this study provides the first experimental evidence of their role.

Keywords Emerging contaminants · Non-equilibrium plasma · Water treatment · Plasma-assisted processes · Reactive species · PFAS destruction · PFAS thermal degradation

Extended author information available on the last page of the article

Published online: 23 June 2026

Introduction

Per- and polyfluoroalkyl substances (PFAS) are a diverse group of manmade organofluorine compounds that have garnered increasing attention from regulatory bodies, researchers, and the general public due to their potential for bioaccumulation and toxicity [1–3]. Much emphasis has been placed on analysis and treatment of PFAS-contaminated drinking water, as its consumption is one main route of human exposure [4–6]. To accommodate PFAS-related drinking water regulations and better protect communities from exposure, researchers are assessing a variety of methods for the treatment of PFAS-contaminated water. A few promising destructive methods, which intend to mineralize PFAS (i.e. convert PFAS to non-toxic inorganic products) in water, include advanced reduction processes (ARPs) [7–10], photocatalytic processes [11–17], supercritical water oxidation (SCWO) [18–22], sonolysis [23–28], electrochemical oxidation (EO) [29–37], activated persulfate oxidation [38–45], and non-equilibrium plasma [46–71], which is the focus of this work.

When assessing a destructive method for treatment of PFAS-contaminated water, it is essential to understand the mechanism by which the method degrades PFAS. Knowledge of degradation mechanism could not only enhance understanding of how to best optimize the method, but also help identify whether the method generates harmful degradation products during treatment. Knowledge of degradation products is of particular importance in this case, as PFAS are known to degrade to shorter chain PFAS, which is undesirable because these are still recalcitrant and have potential for bioaccumulation and toxicity [1, 72, 73]. Despite differences in how the destructive methods under investigation approach PFAS degradation, many follow similar pathways. Generally, these could be divided into four categories. First, (1) decarboxylation-hydroxylation-elimination-hydrolysis (DHEH, also referred to as chain-shortening) in which a PFAS compound one chain-length shorter than the original is formed [10, 74]. Second, (2) hydrodefluorination (H/F exchange) in which a C–F bond is cleaved and a C–H bond is formed in its place [10, 74]. Third, (3) hydroxydefluorination (OH/F exchange) in which a C–F bond is cleaved and a C–OH bond is formed in its place [66, 75]. Fourth, and less commonly proposed, (4) thermal mechanisms (e.g. cleavage of the headgroup via unimolecular bond scission followed by pyrolysis, HF elimination) [76]. General formulas for these pathways can be found in the Supporting Information (Equations S1–S11).

When applied to water, non-equilibrium plasma discharges can create a rich reactive environment of heat, gas-phase charged particles (e.g. free electrons e^-, Ar^+, O_2^+, $\bullet O_2^-$), UV, hydrated electrons (e^-_{aq}), and reactive oxygen and nitrogen species (RONS, e.g. •OH, •NO), all of which can contribute to PFAS degradation and participate in the mechanisms described above [77]. A variety of non-equilibrium plasma discharges have shown promise for treatment of PFAS-contaminated water; these include dielectric barrier discharges (DBD) [47, 48, 57, 78], streamers [54, 55, 57–60, 62, 75, 79], jets [70, 80], corona discharges [46, 57, 79], and gliding arc plasma (GAP) discharge [52, 67, 68], which is the focus of this work. The mechanisms for PFAS degradation via non-equilibrium plasma discharges are not currently well understood, and the dominant pathway likely varies across discharge types investigated, how they are applied (e.g. above the water surface vs. submerged), and the plasma gas used. However, a few studies have gathered evidence to support potentially significant pathways and key reactive species that might facilitate them. The aqueous reactive species e^-_{aq} [62] and •OH[66] have been deemed important to PFAS degradation by

several types of non-equilibrium plasma discharges. Pathways including direct attack of the perfluoroalkyl tail at the plasma-water interface [62, 79], DHEH [62, 75], H/F [66, 75] and OH/F exchange [75] have been hypothesized, and the importance of gas-phase charged particles including Ar^+ [62] and e^- [78] has been suggested and modeled but not experimentally demonstrated.

GAP discharge is unique compared to other discharges studied for this application as it exhibits both thermal and non-thermal properties; the temperature of the gas in GAP discharge (300–1000 K) is elevated compared to other strictly non-thermal plasma discharges (300 K) [81]. Consequently, in studies focused on GAP treatment of PFAS-contaminated water, heat from the discharge in tandem with plasma reactive species [67, 68] have been hypothesized to play key roles in PFAS degradation. RONS have also been hypothesized to play a role [52]. In this work, both the aqueous and gas-phase plasma species that participate in PFAS degradation during GAP treatment of contaminated water were studied using aqueous phase scavengers and a grounded metal mesh, respectively. Results from these experiments were assessed along with degradation products, previous literature on PFAS degradation pathways in water, and previous literature on GAP treatment of PFAS-contaminated water to formulate hypotheses on significant PFAS degradation pathways during GAP treatment.

Experimental

Chemicals

The PFAS used in this study included 6:2 FTS, PFOA, and PFOS. CAS numbers, purities, and manufacturers of the PFAS used can be found in the Supporting Information (Table S1). Air used in GAP experiments was ambient, compressed via an air compressor. Sodium nitrate ($NaNO_3$) manufactured by Sigma-Aldrich (St. Louis, MO, USA) was used in hydrated electron scavenging experiments. In reactive oxygen and nitrogen species (RONS) scavenging experiments, methanol (LCMS grade) manufactured by Oakwood Chemical (Estill, SC, USA), n-acetyl cysteine (NAC) manufactured by NutraBio Labs, Inc. (Middlesex, NJ, USA), and sucrose manufactured by Domino Foods (Yonkers, NY, USA) were used. For fluoride analysis, fluoride standards used for calibration and TISAB II buffer used for sample preparation were manufactured by LabChem (Zelienople, PA, USA). The fluoride electrode filling solution used was manufactured by Thermo Fisher Scientific (Waltham, MA, USA). For LC-QTOF-MS analysis, methanol (LCMS grade) manufactured by Oakwood Chemical (Estill, SC, USA) was used during sample preparation.

GAP Liquid Treatment System

The GAP liquid treatment system used in this work is described in Surace et al. [68]. The capacity of the system is 1 L. In all experiments, air at a flow rate of 40 L/min was used as the plasma gas. Plasma gas flow provided mixing to the liquid in the reactor. Average input plasma power (W) was controlled by supplying a constant current (mAmp) throughout an experiment, and voltage of the arc (kV) was measured with a multimeter; average input plasma power for each experiment was then calculated by multiplying supplied current by

the measured voltage. Samples were collected at predetermined times using the sampling port at the bottom of the reactor.

PFAS Solution Preparation

In all experiments, PFAS solutions were prepared by adding 100 mg of 6:2 FTS, PFOA, or PFOS to 1 L of distilled water in a polypropylene Erlenmeyer flask. Solutions were stirred with gentle heating on a plate for 2 h to promote dissolution.

Mechanistic Studies in the Bulk Liquid

Scavenging Hydrated Electrons

To explore the role that hydrated electrons (e^-_{aq}) play in PFAS degradation in the GAP liquid treatment system, sodium nitrate ($NaNO_3$) was used as a scavenger. PFOA solutions spiked with 0, 1, 10, and 100 mM $NaNO_3$ were treated. All experiments were conducted in air plasma. The calculated average input plasma powers in hydrated electron scavenging experiments ranged from 240 to 300 W; calculated powers for each experiment can be found in the Supporting Information (Table S2). Addition of $NaNO_3$ slightly decreased voltage, which resulted in lower average plasma powers with increasing $NaNO_3$ concentration. Samples were collected at 0 (no plasma), 20, 40, and 60 min of treatment. At each time point, one sample was collected for PFAS analysis, and one sample was collected for fluoride analysis.

Scavenging Reactive Oxygen and Nitrogen Species

To explore the role that aqueous reactive oxygen and nitrogen species (RONS) play in PFAS degradation in the GAP liquid treatment system, methanol, n-acetyl cysteine (NAC), and sucrose were used as scavengers [82–84]. PFOA solutions spiked with 1% (w/v) methanol, NAC, and sucrose (individually) were treated. All experiments were conducted in air plasma. These scavengers did not influence breakdown voltage; thus, calculated average input plasma power, 300 W, was consistent throughout experiments. Samples were collected at 0 (no plasma), 20, 40, and 60 min of treatment. At each time point, one sample was collected for PFAS analysis, and one sample was collected for fluoride analysis.

Mechanistic Studies in the Gas Phase

Trapping/Neutralizing Gas-Phase Charged Particles

To explore the role that gas-phase charged particles play in PFAS degradation in the GAP liquid treatment system, a grounded metal mesh was placed between the plasma source and the PFAS-contaminated liquid. This mesh traps and neutralizes gas-phase charged particles (i.e. free electrons and ions) but allows neutral particles to pass through to interact with the liquid [85, 86]. A schematic of the placement of the mesh and its trapping/neutralizing effect is shown in Fig. 1. All experiments were conducted in air plasma. To ensure the mesh did not significantly affect the temperature of the plasma gas, the reactor was deconstructed and

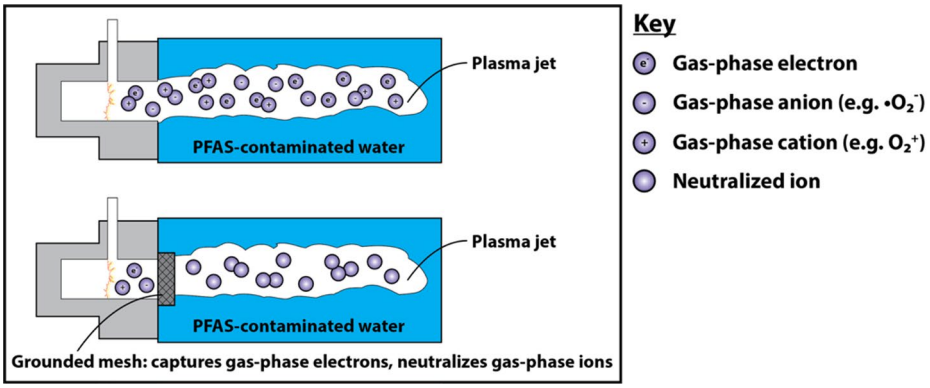


Fig. 1 Schematic illustrating the placement and trapping/neutralizing effect of grounded metal mesh on gas-phase charged particles when added to the GAP liquid treatment system.

temperature of the gas exiting the gliding arc chamber was measured with a thermocouple. Plasma gas temperature was around 400 °C with and without mesh, thus mesh did not significantly affect gas temperature.

6:2 FTS, PFOA, and PFOS solutions (individually) were treated with and without grounded mesh. The calculated average plasma powers in experiments trapping/neutralizing gas-phase charged particles ranged from 200 W with grounded mesh to 300 W without grounded mesh; calculated powers for each experiment can be found in the Supporting Information (Table S3). Throughout these experiments, samples were collected at 0 (no plasma), 20, 40, and 60 min of treatment. At each time point, triplicate samples were collected for PFAS analysis, and duplicate samples were collected for fluoride analysis.

Analytical Methods

Fluoride Analysis Via Ion Selective Electrode (ISE)

Fluoride analysis was conducted using a fluoride ion selective electrode (ORION™ Ionplus Sure-Flow Solid State Combination Fluoride Electrode, Thermo Fisher Scientific, Waltham, MA, USA). Calibration of the electrode ranged from 0.5 mg/L to 10.0 mg/L. Samples were diluted with equal volume TISAB II prior to analysis. All measurements were taken at room temperature.

PFAS Analysis

PFAS were quantified using HPLC-QTOF-MS (SCIEX x500r, Framingham, MA, USA). The quantification method described by Lewis et al. was followed [52]. To account for the potential formation of quantifiable degradation products (e.g. shorter chain PFAS), samples were analyzed for a suite of 24 PFAS, shown in the Supporting Information (Table S4). The limit of quantitation (LOQ) values for this method were analyte specific and spanned 0.01 to 0.1 µg/L.

Data Analysis

Mass of parent compound (i.e. 6:2 FTS, PFOA, or PFOS) removed and mass of fluoride produced were calculated by multiplying measured PFAS or fluoride concentrations by the total water volume in the reactor. For experiments trapping/neutralizing gas-phase charged particles, in which triplicate samples were taken for PFAS analysis and duplicate samples were taken for fluoride analysis, results were averaged, and the standard deviation was calculated at each time point for data presentation.

Because in some cases average plasma power varied between experiments, results were presented as a function of specific energy input (SEI, kJ/L), a metric that combines plasma power and treatment time to normalize these power differences. SEI was calculated using Eq. 1. Results presented as a function of time can be found in the Supporting Information (Figures S1–S3).

$$\text{Specific Energy Input (SEI, kJ/L)} = \frac{\text{Plasma Power (kW)} \times \text{Time (s)}}{\text{Initial Volume (L)}} \quad (1)$$

In all figures that present results as a function of SEI, SEI was calculated at each sampling timepoint in each experimental run. Thus, there is a data point for each sampling timepoint over the course of an experiment.

A fluorine mass balance approach was used to analyze degradation products results. Moles of each product were calculated from the measured concentrations. Total fluorine was calculated based on the quantifiable PFAS and fluoride in the initial sample. Unquantified fluorine was then calculated at each time point by subtracting the moles of quantified fluorine (from PFAS and fluoride analysis results) from the total fluorine.

All graphics for data presentation were generated in RStudio Version 2024.12.1.

Results and Discussion

The Role of Hydrated Electrons

It has been purported that hydrated electrons (e^-_{aq}) play a significant role in PFAS degradation in water during treatment with argon [62] and air [66] plasma streamers over the surface of water. These can partake in a variety of PFAS degradation pathways including DHEH, H/F exchange, and OH/F exchange. Surace et al. hypothesized that e^-_{aq} produced by air GAP discharge might help to decrease the activation energy for thermal degradation of PFAS in water [68]. To investigate the role of e^-_{aq} in PFAS degradation via air GAP discharge, experiments were conducted treating 100 mg/L PFOA solutions spiked with 0, 1, 10, and 100 mM sodium nitrate.

Results from e^-_{aq} scavenging experiments are shown in Fig. 2. Increased e^-_{aq} scavenger concentration correlated with decreased mass of PFOA removed (Fig. 2A) but did not inhibit fluoride production (Fig. 2B). Using fluoride production as a metric for PFOA mineralization, these results suggest that there is a degradation pathway that does not result in significant mineralization, driven in part by e^-_{aq} at the plasma-water interface and/or in the bulk liquid. Discussed in more detail in later sections, no quantifiable shorter chain PFAS

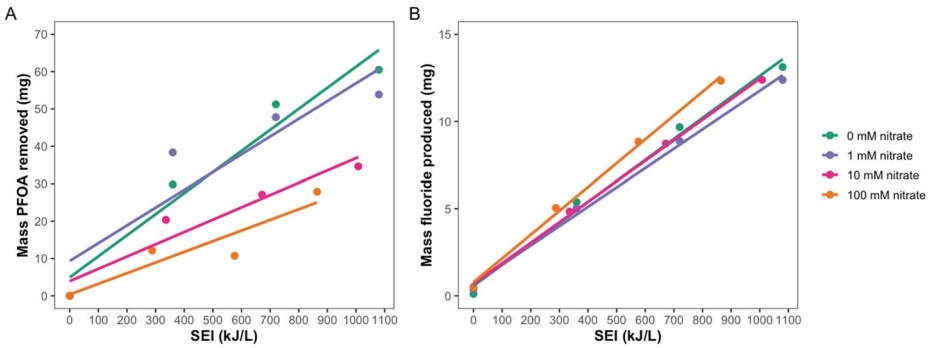


Fig. 2 Mass of (**A**) parent compound removed and (**B**) fluoride produced as a function of specific energy input (SEI) for hydrated electron (e^-_{aq}) scavenging experiments conducted with PFOA in air GAP discharge. Experiments were conducted with 0, 1, 10, and 100 mM scavenger (nitrate). PFAS and fluoride analyses were conducted on single samples at each time point from a single experimental run. A linear regression is displayed for each run. Note that in this figure, linear regressions are intended as a visual aid only to show the general trend for each run and are not meant to be predictive. Results as a function of treatment time can be found in the Supporting Information (Fig. S1).

were formed in these experiments, suggesting that this pathway is not DHEH. Further, because thermal degradation is linked to mineralization during GAP treatment of PFAS-contaminated water [67, 68], it is unlikely that e^-_{aq} participate in PFAS degradation by lowering the energy barrier for thermal degradation, as previously proposed by Surace et al. It is possible that e^-_{aq} drive H/F and/or OH/F exchange pathways, transforming PFOA but resulting in minimal fluoride production.

Note that in Fig. 2, linear regressions are intended as a visual aid only to show the general trend for each run and are not meant to be predictive. Additionally, because results are from individual experimental runs that were not replicated, and analytical replicates were not collected, error bars could not be produced for this figure. This set of experiments was conducted sequentially in attempt to minimize variation between runs, but the potential for variation not accounted for due to lack of experimental replicates is of note. Future studies should include experimental replicates to confirm results presented here.

The Role of Reactive Oxygen and Nitrogen Species

Based on experiments conducted in various plasma gases (N_2, O_2, and air, separately), Lewis et al. hypothesized that RONS (e.g. •OH, •NO, $ONOO^-$) play a key role in PFAS degradation in water during treatment with GAP discharge, as PFAS disappeared more rapidly in air discharge than in N_2 or O_2 discharge alone [52]. Another group that used air plasma streamers over the surface of water determined that •OH played a significant role in PFOA degradation in their system, and Surace et al. hypothesized that •OH might help to decrease the activation energy for thermal degradation of PFAS in water [68]. Further, •OH could partake in other steps of a variety of PFAS degradation pathways including DHEH and OH/F exchange [10, 75]. To investigate the role of RONS in PFAS degradation via air GAP discharge, experiments were conducted treating 100 mg/L PFOA solutions spiked with 1% w/v methanol, N-acetyl cysteine (NAC), and sucrose. These scavengers were not intended

to be targeting specific RONS, but RONS in general, as specificity of RONS scavengers has been met with skepticism.

Results from RONS scavenging experiments are shown in Fig. 3. For all scavengers tested, mass of PFOA removed (Fig. 3A) and fluoride produced (Fig. 3B) were not significantly decreased in the presence of scavengers, suggesting that RONS at the plasma-water interface and/or in the bulk liquid are inconsequential to PFAS degradation in water via air GAP discharge. It is important to note that sucrose appeared to slightly increase mass of PFOA removed. Without experimental and analytical replicates, it is difficult to draw conclusions surrounding this observed trend; however, since the purpose of these experiments was to investigate whether RONS scavengers negatively impacted PFAS degradation in this treatment system, this was not further explored in this study and should be investigated in future work.

Combined with results from e^-_{aq} scavenging experiments, which eliminated DHEH as a possible significant PFAS degradation pathway during treatment with air GAP discharge due to minimal formation of shorter chain PFAS, RONS scavenging experiment results eliminate the likelihood of PFAS degradation by OH/F exchange as RONS are expected to participate in this pathway. Further, due to the minimal effect RONS scavengers had on PFOA mineralization (i.e. fluoride production), it is unlikely that RONS participate in PFAS degradation by lowering the energy barrier for thermal degradation, as previously proposed by Surace et al. H/F exchange via e^-_{aq} at the plasma-water interface and/or in the bulk liquid remains as a possible significant pathway for PFAS degradation via air GAP discharge, and there appears to be some other pathway resulting in significant PFAS mineralization that does not involve e^-_{aq} or RONS, which is explored further in the following sections.

Note that, like in Fig. 2, in Fig. 3 linear regressions are intended as a visual aid only to show the general trend for each run and are not meant to be predictive. Additionally, because results are from individual experimental runs that were not replicated, and analytical replicates were not collected, error bars could not be produced for this figure. This set of experiments was conducted sequentially in attempt to minimize variation between runs, but

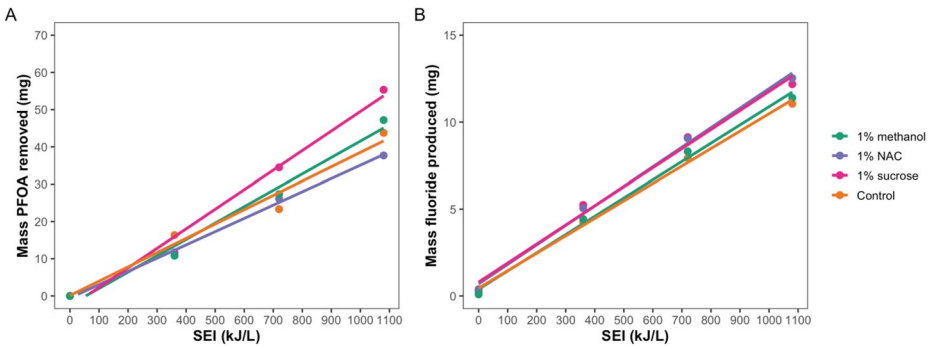


Fig. 3 Mass of (**A**) parent compound removed and (**B**) fluoride produced as a function of specific energy input for reactive oxygen and nitrogen species (RONS) scavenging experiments conducted with PFOA in air GAP discharge. Experiments were conducted with 1% (w/v) scavengers (methanol, NAC, and sucrose). PFAS and fluoride analyses were conducted on single samples at each time point from a single experimental run. A linear regression is displayed for each run. Note that in this figure, linear regressions are intended as a visual aid only to show the general trend for each run and are not meant to be predictive. Results as a function of treatment time can be found in the Supporting Information (Fig. S2).

the potential for variation not accounted for due to lack of experimental replicates is of note. Future studies should include experimental replicates to confirm results presented here.

Importantly, our RONS scavenging experiments appear to contradict previous results from Lewis et al., which suggested that RONS play a key role in PFAS degradation via GAP. However, it is important to note that this study scavenged RONS in the aqueous phase during treatment in air GAP discharge, while Lewis et al. tested different plasma gases (N_2, O_2, and air), which should result in different gas-phase species; for example, one study found that in the gas phase air discharge was higher in nitric oxide radicals (•NO) compared to N_2 and O_2 discharges [87]. It is possible, especially considering results discussed in the following section, that in the work by Lewis et al. species created in the gas phase in air GAP discharge were more conducive to PFAS degradation than those in N_2 and O_2 GAP discharge.

Although the scavengers tested here have been used by other researchers to scavenge RONS [82–84], it is possible that scavenging was unsuccessful in the GAP liquid treatment system. Unsuccessful scavenging of RONS might be due to instability of the scavengers in higher temperature regions of the GAP liquid treatment system (e.g. at the plasma-water interface). Data on the thermal stability of the scavengers used can be found in the Supporting Information (Table S5) [88–90]. Further, while concentrations used in this study were significantly higher than the PFOA concentrations used (1% w/v versus 0.01% w/v) and thus were expected to be sufficient for scavenging, it is possible that the amount of RONS generated overpowered the amount of scavenger used and thus the scavengers were degraded by the plasma. However, if this was the case, we would expect to see at least an initial effect on PFOA degradation; shown in Fig. S2B, samples taken after 10 min of treatment and analyzed for fluoride fit linearly with the data taken at subsequent time points, suggesting no initial hindrance of PFOA degradation by RONS scavengers. Regardless, to directly assess the effectiveness of these scavengers, future studies should be conducted to measure RONS in water with and without scavengers present (e.g. •OH could be measured via spin-trapping with and without scavenger present).

Due to the lack of experimental replicates, it is important to consider that the "Control" in Fig. 3 was conducted under the same experimental conditions as "0 mM nitrate" in Fig. 2, and PFOA removal results differ by ~20%. While these results could suggest variability between experiments that might affect conclusions drawn in this study, it is important to note that general reproducibility of results in this treatment system was considered in experimental design for these mechanistic studies. These sets of experiments (i.e. hydrated electron scavenging set, RONS scavenging set) were conducted several months apart. It is common in this system to see variability in results for replicate experiments conducted months apart, while typically those repeated over shorter timeframes showed better reproducibility. Thus, in these mechanistic studies, experiments were conducted sequentially in sets over short timeframes, always including a control.

Synergies of Gas-Phase Charged Particles and Thermal Energy

As discussed in previous sections, results from experiments scavenging reactive species in the aqueous phase (e^-_{aq} and RONS) suggest that these species do not play a significant role in PFOA mineralization (i.e. fluoride production) via air GAP discharge. Surace et al. determined that PFOA mineralization in water is endothermic, and thus hypothesized that in

GAP discharge heat drives the mineralization reaction, while plasma reactive species may help to decrease the activation energy for this thermal reaction [68]. Shaji et al. experimentally demonstrated that increased average plasma gas temperature led to increased PFOS mineralization in water, and that gas heated to the same temperature without plasma discharge did not lead to PFOS mineralization, supporting the hypothesis made by Surace et al. that plasma reactive species and heat play a synergistic role in PFAS mineralization via GAP [67]. Considering the seemingly inconsequential effect that aqueous reactive species have on PFAS mineralization during GAP treatment along with the hypothesis that heat drives mineralization, experiments were designed to explore the role that gas-phase neutral and charged particles, which exist in a higher-temperature region of the GAP liquid treatment system than aqueous species, play. A grounded metal mesh, which captures gas-phase free electrons, neutralizes gas-phase ions, and allows neutral particles to pass through to the bulk solution, was placed in the GAP liquid treatment system just past the source of plasma discharge [85, 86].

PFAS and fluoride results from these experiments and their controls are shown in Fig. 4. Trapping/neutralizing gas-phase charged particles with grounded mesh significantly decreased the mass of parent compound removed (Fig. 4A) and completely inhibited fluoride production (Fig. 4B) for all PFAS tested (6:2 FTS, PFOA, and PFOS). Using fluoride production as a metric for PFAS mineralization, these results suggest that gas-phase charged particles are essential to initiating PFAS mineralization in water during GAP treatment. Further, results suggest that neutral particles do not play a role in initiating mineralization, however gas-phase charged particles may react at the plasma-water interface to generate neutral species that are significant to mineralization. Other non-equilibrium plasma studies have also suggested that gas-phase charged particles, including Ar^+ [62] and e^- [62, 78], are important to PFAS degradation in water. Stratton et al. proposed that, due to the surfactant nature of PFAS, their hydrophobic perfluoroalkyl tails might protrude into the gas phase to react with Ar^+ via charge transfer, causing rapid fragmentation, or e^- via excitation or ionization [62]. Meanwhile, Song et al. proposed that e^- penetrate the plasma-water interface and react directly with PFAS there to initiate DHEH, H/F, and OH/F exchange pathways [78]. While previously hypothesized and modeled, to the best of our knowledge this is the first study to experimentally demonstrate the role that gas-phase charged particles play in PFAS degradation in water during treatment with non-equilibrium plasma.

Based on the thermal characteristics of GAP, prior evidence that temperature plays a significant role in PFAS mineralization via GAP [67, 68], and the distribution of observed degradation products during treatment discussed in the following section, we hypothesize that gas-phase charged particles react with PFAS in the gas phase and at the plasma-water interface in ways similar to those proposed by Song et al. and Stratton et al. (i.e. charge transfer, excitation, or ionization), initiating a thermal mineralization reaction, while also potentially causing fragmentation and/or H/F exchange. Though plasma diagnostics were not conducted in these experiments, air plasma is expected to produce a variety of gas-phase ions such as O^+, O_2^+, N_2^+, H_2O^+, H^+, O_3^-, and O_2^- [77, 91]. Using PFOA as an example, based on its ionization potential estimated by Stratton et al. (~ 11 eV) [62], if charge exchange were to occur several of these gas-phase ions that might be present in air GAP discharge have significantly higher ionization potentials and would provide the molecule with ample energy to initiate a reaction. Ionization potentials of several of the gas-phase ions that might be present in air GAP discharge can be found in the Supporting Information

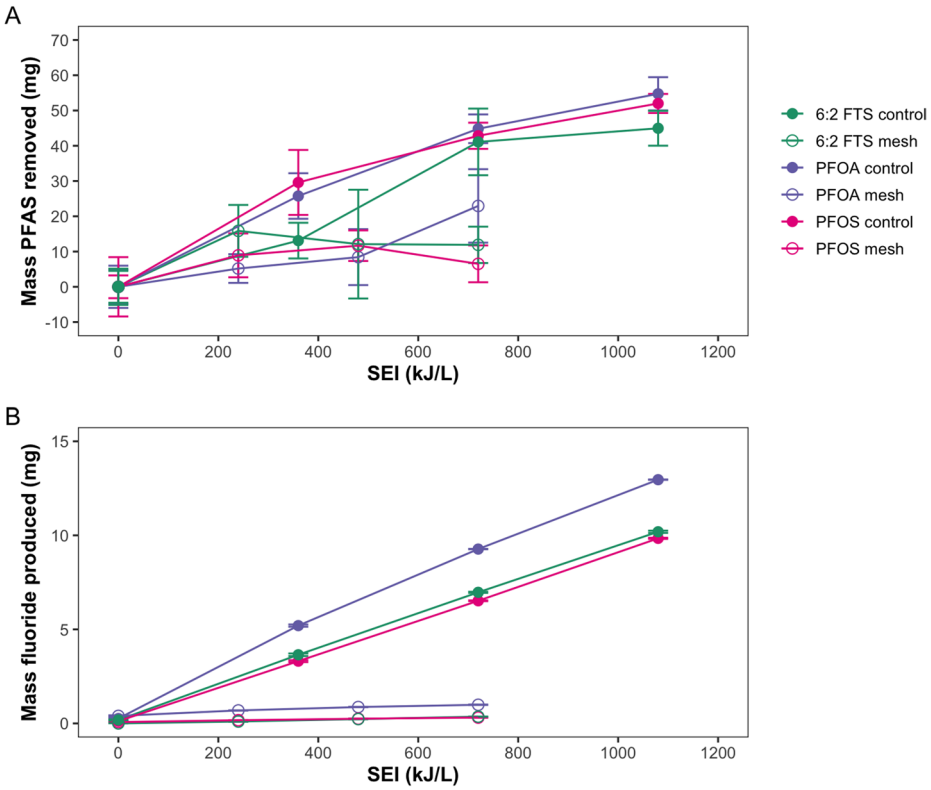


Fig. 4 PFAS and fluoride results from gas-phase charged particle trapping/neutralizing experiments. Mass of (**A**) parent compound removed and (**B**) fluoride produced as a function of specific energy input for experiments conducted with 6:2 FTS, PFOA, and PFOS in air GAP discharge. PFAS and fluoride analyses were conducted on triplicate and duplicate samples, respectively, at each time point from a single experimental run. Average PFAS and fluoride results with error bars representing standard deviation for each time point are shown. Results as a function of treatment time can be found in the Supporting Information (Fig. S3).

(Table S6) [92–96]. Future work on the GAP liquid treatment system should include plasma diagnostics (e.g. optical emission spectroscopy) to help identify which gas-phase charged particles play a key role in PFAS degradation.

Since we propose that thermal mineralization initiated by gas-phase charged particles is a significant PFAS degradation pathway during GAP treatment, it is important to note that while GAP has thermal characteristics, it is not expected to provide temperatures conducive to PFAS degradation by heat alone. The estimated average air discharge temperature at the powers used in this work is typically around 300 °C, calculated using an energy balance following the process described by Shaji et al. [67]. When this air comes into contact with water, the temperature dissipates throughout the bulk, warming but not boiling it. The temperature at the plasma-water interface and in the bulk is not expected to be high enough for thermal degradation. Purely thermal destructive techniques for PFAS treatment typically run at much higher temperatures; for example, incineration and smoldering combustion of PFAS-contaminated matrices are typically conducted at temperatures greater than 900 °C

[97–99]. Further, while some other non-equilibrium discharges produce high temperature filaments that might come into contact with water at the surface and participate in thermal degradation of PFAS [62, 100], we do not expect direct contact between the arc (the filament in GAP discharge) and water in the GAP liquid treatment system because of the high plasma gas flow rate. Thus, thermal degradation via direct contact with high temperature filaments is not expected in the GAP liquid treatment system. However, based on previous evidence that temperature plays a role in PFAS mineralization during GAP treatment [67, 68], in combination with results from this work in which trapping/neutralizing gas-phase charged particles completely halted mineralization, we hypothesize that these charged particles react with PFAS via charge transfer, excitation, and/or ionization to create a less stable intermediate that thermally degrades at the temperatures provided by GAP. This hypothesized process in which gas-phase plasma species have a catalytic effect is similar to plasma-assisted combustion processes [101].

Notably, in this set of experiments it appears that 6:2 FTS, PFOA, and PFOS behave similarly during GAP treatment. For each compound, comparable levels of parent compound degradation and fluoride production were achieved for the same SEI. Further, each compound behaved similarly in the presence of mesh (i.e. mass of parent compound was significantly decreased, and fluoride production was completely inhibited). These results suggest that the major pathway by which PFAS are mineralized during GAP treatment may not be compound-specific. However, other degradation pathways might occur during GAP treatment that are compound-specific; for example, 6:2 FTS is known to be susceptible to oxidation by •OH, which might be revealed by aqueous phase RONS scavenging experiments like those conducted for PFOA in this study. Future studies should be conducted to further explore compound-specific degradation pathways during GAP treatment.

Finally, although results from experiments scavenging reactive species in the aqueous phase suggest that aqueous reactive species do not play a role in PFAS mineralization during GAP treatment, it is possible that gas-phase charged particles generate aqueous reactive species conducive to mineralization that were not scavenged by experiments conducted here. In this case, gas-phase charged particles would still play a significant role in initiating PFAS mineralization via GAP discharge, but indirectly. While it is possible that scavenging experiments conducted in this work did not cover some aqueous reactive species produced by gas-phase charged particles that are important to PFAS degradation during GAP treatment, aqueous phase scavenging was conducted for the species that typically partake in PFAS degradation in other non-equilibrium plasma systems (i.e. e^-_{aq} and •OH), and no effect on mineralization was observed.

Hypothesized PFAS Degradation Mechanisms

Knowledge of the degradation products formed during treatment of PFAS-contaminated water could provide support and insight into the hypothesized mechanisms. For example, formation of shorter chain PFAS compounds is typically used as an indicator for the DHEH (chain-shortening) pathway. A fluorine mass balance analysis of degradation products quantifiable by the methods used in this work was conducted on samples from the gas-phase charged particle trapping/neutralizing control experiments (i.e. air GAP discharge with no grounded mesh present). Degradation of parent compound (6:2 FTS, PFOA, or PFOS) along with production of fluoride (F^-) and other shorter chain FTS, PFCA, and PFSA compounds

(listed in the Supporting Information, Table S4) were tracked, and unquantified fluorine was calculated based on these results.

Figure 5 shows the products that make up a majority of the fluorine mass balance in these experiments which, for all PFAS tested, were fluoride (Fig. 5A, C, and E), indicative of mineralization, and unquantified fluorine (Fig. 5B, D, and F). In Fig. 5, the "Theoretical" line represents a 1:1 fluorine mass balance (i.e., complete conversion of moles fluorine in parent compound to product of interest). Based on our hypothesis that heat, assisted by gas-phase charged particles, drives PFAS mineralization during GAP treatment, we suspect that the fluoride produced is, at least in part, a product of this mechanism. Unquantified fluorine could be indicative of various PFAS degradation pathways. First, unquantified fluorine

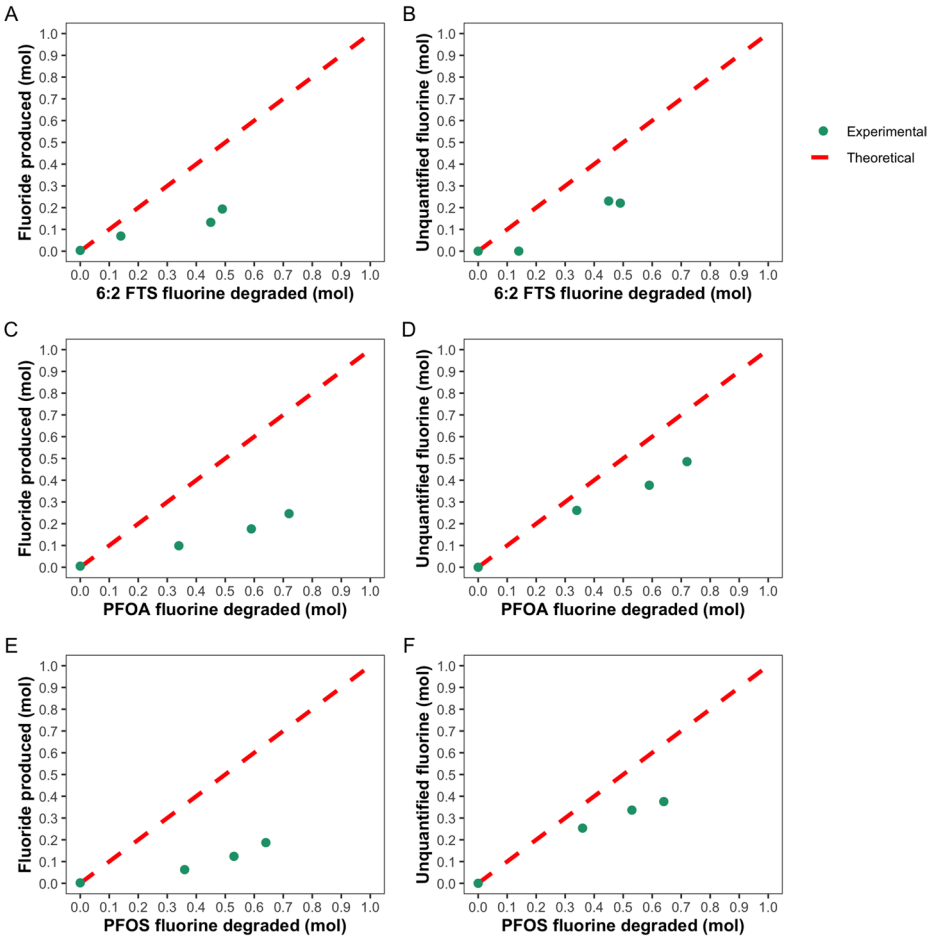


Fig. 5 Fluoride and unquantified fluorine make up a majority of the fluorine mass balance during treatment of (**A, B**) 6:2 FTS, (**C, D**) PFOA, and (**E, F**) PFOS in air GAP discharge. The "Theoretical" line represents a 1:1 fluorine mass balance (i.e., complete conversion of moles fluorine in parent compound to product of interest). PFAS and fluoride results were taken from control experiments conducted alongside experiments trapping/neutralizing gas-phase charged particles. PFAS and fluoride analyses were conducted on triplicate and duplicate samples, respectively, at each time point from a single experimental run. Average results for each time point are shown.

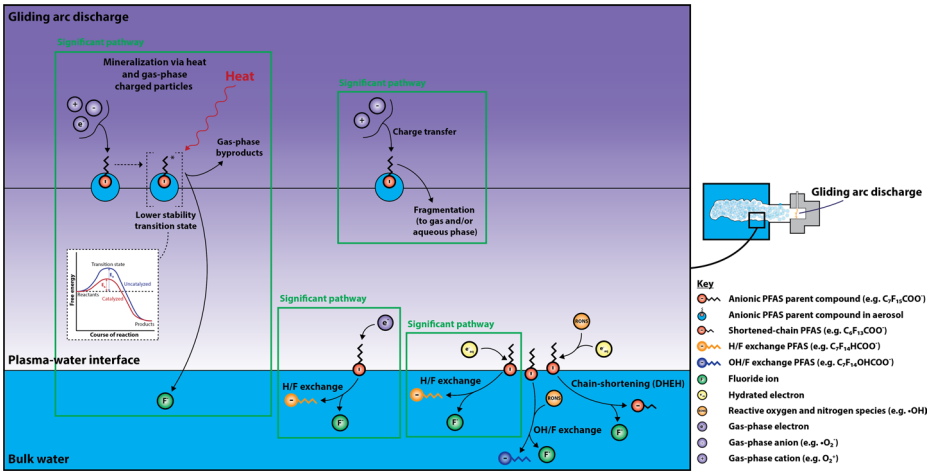


Fig. 6 Schematic of potential PFAS degradation pathways during GAP treatment. Hypothesized significant pathways (thermal mineralization driven by gas-phase charged particles, H/F exchange driven by e^- and/or e^-_{aq}, and fragmentation via charge transfer from gas-phase charged particles) are indicated by a green outline. Black horizontal lines in the schematic represent boundaries between the bulk water, plasma-water interface, and gliding arc discharge. Please refer to the key for details on symbols used in the schematic.

might be partially hydrogenated PFAS, products of H/F exchange that are unquantifiable by methods used in this work. Based on results from gas-phase charged particle trapping/neutralizing experiments, H/F exchange during GAP treatment might be driven by e^- at the plasma-water as described by Song et al. [78]. Further, based on results from e^-_{aq} scavenging experiments, H/F exchange during GAP treatment might also be driven by e^-_{aq} at the plasma-water interface and/or in the bulk liquid [66]. Unquantified fluorine could also be (1) fragments created from reactions between PFAS and gas-phase charged particles via charge transfer, as described by Stratton et al. [62] and supported by results from gas-phase charged particle trapping/neutralizing experiments in this work; (2) products of incomplete thermal degradation, such as perfluoroacyl fluoride compounds and volatile organofluorine compounds, perfluoroalkanes and/or olefins [76]; and/or (3) gaseous HF generated via thermal mineralization or H/F exchange—though $HF_{(g)}$ is highly soluble in water, it might vaporize when in high temperature regions of the GAP liquid treatment system. In future work, non-targeted PFAS analysis of aqueous samples along with collection and analysis of effluent gas (for fluoride and PFAS) should be conducted to better understand which of the proposed pathways might be most dominant: thermal mineralization, H/F exchange (via e^- and/or e^-_{aq}), or fragmentation. A schematic of these hypothesized pathways is shown in Fig. 6.

Non-dominant degradation products formed in these experiments, all of which are shorter-chain PFAS, can be found in the Supporting Information (Fig. S4). These results indicate that DHEH (chain-shortening) is present but not a significant pathway for PFAS degradation in water during GAP treatment. This pathway might be driven by e^- at the plasma-water interface as described by Song et al. [78] or by aqueous reactive species (e.g. e^-_{aq}, •OH) as described in other non-equilibrium plasma work [66, 102]. DHEH and OH/F exchange, both pathways proposed in other non-equilibrium plasma work that do not appear

significant during GAP treatment, are shown with the hypothesized significant pathways in Fig. 6.

Of note, as mentioned throughout the manuscript, we acknowledge that lack of experimental replicates is a shortcoming of this study, and future studies should include experimental replicates to confirm results presented here. However, we contend that the observed trends still support the interpretations made here, which are drawn based on broader trends in the data (e.g. complete inhibition of fluoride production in the presence of mesh) rather than slight differences in results between runs. Further, the similarities in fluoride production trends among all runs aside from those with mesh present, as well as the similar and repeated trends observed in experiments conducted under the same experimental conditions with different PFAS parent compounds can help to support reproducibility. Experiments treating PFOA, PFOS, and 6:2 FTS solutions were conducted under the same conditions, changing only the PFAS parent compound and the presence of mesh, and similar trends were observed for each compound (i.e. similar levels of parent compound removal and fluoride production without mesh, and decreased degradation of parent compound and complete inhibition of fluoride production in the presence of mesh, shown in Fig. 4).

Over the course of GAP treatment a significant portion of fluorine becomes unquantifiable by methods used in this study. As discussed, it is possible that this is the result of formation of partially hydrogenated PFAS, which was observed by Tomei et al. and Ganzallo et al. in their works studying the mechanism of PFAS degradation in water during treatment with other plasma systems [103, 104]. As discussed by Ganzallo et al., these byproducts are more likely to accumulate in the bulk liquid than PFAS as they are less surface active; they are also more easily oxidizable. Ganzallo et al. found that generating additional •OH in the bulk liquid via a plasma-Fenton process significantly aided in mineralization of PFBS in their plasma system. While RONS scavenging conducted in this work suggested •OH do not play a significant role in PFAS mineralization in the GAP system, it is possible that not all •OH in the bulk liquid were scavenged. Along with additional analyses to confirm the formation of partially hydrogenated PFAS, future studies should be conducted similar to the plasma-Fenton study conducted by Ganzallo et al. to explore the role that •OH might play in mineralizing these products in the GAP system.

Notably, the influence of VUV was not explored in this study. VUV is known to be produced in plasma discharges, and recent evidence suggests that VUV may play a role in breaking C–F bonds [105]. Because experiments were conducted exposed to ambient air, at atmospheric conditions with air as the plasma gas, VUV is not expected to play a significant role in PFAS degradation in this system; air is known to absorb VUV. However, future studies should be conducted to confirm the role of VUV in this system. For instance, VUV plasma diagnostics could be conducted to measure the radiation of the plasma used in the VUV.

Finally, while GAP discharge is unique compared to other discharges studied for this application as it exhibits both thermal and non-thermal properties, it is possible that the degradation mechanisms hypothesized here, especially those involving gas-phase charged particles, are significant during PFAS treatment in other plasma systems as well. As discussed, other groups have proposed that gas-phase charged particles play a role in PFAS degradation in their plasma systems [62, 78]. While this study provides experimental evidence suggesting that gas-phase charged particles generally contribute to PFAS mineralization via

GAP, future studies should be conducted explore more specifically which gas-phase charged particles are driving this, and whether this translates to other plasma systems.

Conclusions

The results of this study indicated several potential dominant pathways by which PFAS degrade during GAP treatment of contaminated water: (1) thermal mineralization initiated by gas-phase charged particles, (2) H/F exchange driven by free electrons (e^-) at the plasma-water interface or e^-_{aq} at the plasma-water interface/in the bulk liquid, and (3) fragmentation via charge transfer from gas-phase charged particles. To the best of our knowledge this is the first study to experimentally suggest the role that gas-phase charged particles play in PFAS degradation in water during treatment with non-equilibrium plasma. These results have implications for the optimal design and scale-up of the GAP liquid treatment system, as they suggest that to achieve significant mineralization contact between gas-phase charged particles and PFAS should be maximized.

As indicated throughout this work, future studies should be conducted with experimental replicates to confirm the results in this study. Further, future studies should be conducted to confirm the effectiveness of the scavengers used to quench aqueous RONS. For example, •OH could be measured via spin-trapping with and without scavenger present. Further, plasma diagnostics, for example via optical emission spectroscopy (OES), would be useful for determining the major species produced by the discharge used. Finally, more robust analyses of degradation products, including non-targeted PFAS analysis, total organofluorine (TOF) analysis, and analysis of gas-phase products, are necessary to identify which of the proposed PFAS degradation pathways are most dominant.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s11090-026-10688-y.

Author Contributions Conceptualization, A.A.F., A.R., E.R.M., G.F. and C.M.S.; Formal analysis, M.J.S.; Investigation, M.J.S., M.A.S., E.H., J.M.G. and J.T.; Project administration, A.A.F., A.R., E.R.M., G.F. and C.M.S.; Supervision, A.A.F., A.R., E.R.M., G.F. and C.M.S.; Writing—original draft preparation, M.J.S.; Writing—review and editing, J.M.G., A.A.F., A.R., E.R.M., G.F. and C.M.S.; Funding acquisition, C.M.S.

Funding This research was funded by the Strategic Environmental Research and Development Program (SERDP), project number ER18-1570.

Data Availability The data that support the findings of this study are available for the corresponding author, M.J.S., upon request.

Declarations

Competing interest The authors declare no competing interests.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the

copyright holder. To view a copy of this licence, visit http://creativecommons.org/licenses/by/4.0/.

References

1. Lindstrom AB, Strynar MJ, Libelo EL (2011) Polyfluorinated Compounds: Past, Present, and Future. Environmental Science & Technology 45(19):7954–7961
2. Lau C et al (2007) Perfluoroalkyl Acids: A Review of Monitoring and Toxicological Findings. Toxicol Sci 99(2):366–394
3. Fenton SE et al (2021) Per- and Polyfluoroalkyl Substance Toxicity and Human Health Review: Current State of Knowledge and Strategies for Informing Future Research. Environ Toxicol Chem 40(3):606–630
4. Ericson I et al (2008) Levels of perfluorochemicals in water samples from Catalonia, Spain: is drinking water a significant contribution to human exposure? Environ Sci Pollut Res 15(7):614–619
5. Domingo JL, Nadal M (2019) Human exposure to per- and polyfluoroalkyl substances (PFAS) through drinking water: a review of the recent scientific literature. Environ Res 177:108648
6. Stoiber T et al (2020) PFAS in drinking water: an emergent water quality threat. Water Solutions 1(40):e49
7. Yu K et al (2018) Mechanism and efficiency of contaminant reduction by hydrated electron in the sulfite/iodide/UV process. Water Res 129:357–364
8. Song Z et al (2013) Reductive defluorination of perfluorooctanoic acid by hydrated electrons in a sulfite-mediated UV photochemical system. J Hazard Mater 262:332–338
9. O'Connor N et al (2023) Forever no more: Complete mineralization of per- and polyfluoroalkyl substances (PFAS) using an optimized UV/sulfite/iodide system. Sci Total Environ 888:164137
10. Bentel MJ et al (2019) Defluorination of Per- and Polyfluoroalkyl Substances (PFASs) with Hydrated Electrons: Structural Dependence and Implications to PFAS Remediation and Management. Environmental Science & Technology 53(7):3718–3728
11. Chen C et al (2021) Photocatalytically reductive defluorination of perfluorooctanoic acid (PFOA) using Pt/La2Ti2O7 nanoplates: Experimental and DFT assessment. J Hazard Mater 419:126452
12. Gomez-Ruiz B et al (2018) Photocatalytic degradation and mineralization of perfluorooctanoic acid (PFOA) using a composite TiO_2–rGO catalyst. J Hazard Mater 344:950–957
13. Song Z et al (2017) Efficient photocatalytic defluorination of perfluorooctanoic acid over BiOCl nanosheets via a hole direct oxidation mechanism. Chem Eng J 317:925–934
14. Sun Y et al (2019) Photocatalytic defluorination of perfluorooctanoic acid by surface defective BiOCl: Fast microwave solvothermal synthesis and photocatalytic mechanisms. J Environ Sci 84:69–79
15. Verma S et al (2021) Photooxidative decomposition and defluorination of perfluorooctanoic acid (PFOA) using an innovative technology of UV–vis/$Zn_xCu_{1-x}Fe_2O_4$/oxalic acid. Chemosphere 280:130660
16. Wen Y et al (2022) Integrated Photocatalytic Reduction and Oxidation of Perfluorooctanoic Acid by Metal–Organic Frameworks: Key Insights into the Degradation Mechanisms. J Am Chem Soc 144(26):11840–11850
17. Zhao B, Zhang P (2009) Photocatalytic decomposition of perfluorooctanoic acid with β-Ga_2O_3 wide bandgap photocatalyst. Catal Commun 10(8):1184–1187
18. Austin C et al (2023) Destruction and defluorination of PFAS matrix in continuous-flow supercritical water oxidation reactor: Effect of operating temperature. Chemosphere 327:138358
19. Krause MJ et al (2022) Supercritical Water Oxidation as an Innovative Technology for PFAS Destruction. J Environ Eng 148(2):05021006
20. Li J et al (2023) PFOS destruction in a continuous supercritical water oxidation reactor. Chem Eng J 451:139063
21. Pinkard BR et al (2021) Destruction of perfluorooctanesulfonate (PFOS) in a batch supercritical water oxidation reactor. Chemosphere 279:130834
22. Scheitlin CG et al (2023) Application of Supercritical Water Oxidation to Effectively Destroy Per- and Polyfluoroalkyl Substances in Aqueous Matrices. ACS ES&T Water
23. Campbell T, Hoffmann MR (2015) Sonochemical degradation of perfluorinated surfactants: power and multiple frequency effects. Sep Purif Technol 156:1019–1027
24. James Wood R et al (2020) Ultrasonic degradation of perfluorooctane sulfonic acid (PFOS) correlated with sonochemical and sonoluminescence characterisation. Ultrason Sonochem 68:105196
25. Lee Y-C et al (2016) Efficient sonochemical degradation of perfluorooctanoic acid using periodate. Ultrason Sonochem 31:499–505

26. Moriwaki H et al (2005) Sonochemical Decomposition of Perfluorooctane Sulfonate and Perfluorooctanoic Acid. Environmental Science & Technology 39(9):3388–3392
27. Rodriguez-Freire L et al (2015) Effect of sound frequency and initial concentration on the sonochemical degradation of perfluorooctane sulfonate (PFOS). J Hazard Mater 300:662–669
28. Vecitis CD et al (2008) Kinetics and Mechanism of the Sonolytic Conversion of the Aqueous Perfluorinated Surfactants, Perfluorooctanoate (PFOA), and Perfluorooctane Sulfonate (PFOS) into Inorganic Products. J Phys Chem A 112(18):4261–4270
29. Le TXH et al (2019) Energy-Efficient Electrochemical Oxidation of Perfluoroalkyl Substances Using a Ti4O7 Reactive Electrochemical Membrane Anode. Environmental Science & Technology Letters 29(8):504–510
30. Liang S et al (2018) Electrochemical oxidation of PFOA and PFOS in concentrated waste streams. Remediation J 28(2):127–134
31. Liao Z, Farrell J (2009) Electrochemical oxidation of perfluorobutane sulfonate using boron-doped diamond film electrodes. J Appl Electrochem 39(10):1993–1999
32. Lin H et al (2021) Defect Engineering on a Ti4O7 Electrode by Ce3 + Doping for the Efficient Electrooxidation of Perfluorooctanesulfonate. Environmental Science & Technology 55(4):2597–2607
33. Niu J et al (2013) Theoretical and Experimental Insights into the Electrochemical Mineralization Mechanism of Perfluorooctanoic Acid. Environmental Science & Technology 47(24):14341–14349
34. Sukeesan S, Boontanon N, Boontanon SK (2021) Improved electrical driving current of electrochemical treatment of Per- and Polyfluoroalkyl Substances (PFAS) in water using Boron-Doped Diamond anode. Environ Technol Innov 23:101655
35. Wang Y et al (2020) Electrochemical degradation of perfluoroalkyl acids by titanium suboxide anodes. Environ Science: Water Res Technol 6(1):144–152
36. Xu Z et al (2017) Highly efficient and stable Zr-doped nanocrystalline PbO2 electrode for mineralization of perfluorooctanoic acid in a sequential treatment system. Sci Total Environ 579:1600–1607
37. Yang B et al (2017) Electrochemical mineralization of perfluorooctane sulfonate by novel F and Sb co-doped Ti/SnO2 electrode containing Sn-Sb interlayer. Chem Eng J 316:296–304
38. Bruton TA, Sedlak DL (2018) Treatment of perfluoroalkyl acids by heat-activated persulfate under conditions representative of in situ chemical oxidation. Chemosphere 206:457–464
39. Carre-Burritt AE, Van Hoomissen DJ, Vyas S (2021) Role of pH in the Transformation of Perfluoroalkyl Carboxylic Acids by Activated Persulfate: Implications from the Determination of Absolute Electron-Transfer Rates and Chemical Computations. Environmental Science & Technology 55(13):8928–8936
40. Liu Z et al (2021) Near-Quantitative Defluorination of Perfluorinated and Fluorotelomer Carboxylates and Sulfonates with Integrated Oxidation and Reduction. Environmental Science & Technology 55(10):7052–7062
41. Manz KE et al (2023) Low-temperature persulfate activation by powdered activated carbon for simultaneous destruction of perfluorinated carboxylic acids and 1,4-dioxane. J Hazard Mater 442:129966
42. Parenky Akshay C et al (2020) Decomposition of Carboxylic PFAS by Persulfate Activated by Silver under Ambient Conditions. J Environ Eng 146(10):06020003
43. Tran T et al (2021) Iron-activated persulfate oxidation degrades aqueous Perfluorooctanoic acid (PFOA) at ambient temperature. Chemosphere 281:130824
44. Uwayezu JN et al (2023) Validation of UV/persulfate as a PFAS treatment of industrial wastewater and environmental samples. J Water Process Eng 53:103614
45. Yang S et al (2013) Defluorination of Aqueous Perfluorooctanesulfonate by Activated Persulfate Oxidation. PLoS ONE 8(10):e74877
46. Aziz KHH et al (2021) Degradation of perfluorosurfactant in aqueous solution using non-thermal plasma generated by nano-second pulse corona discharge reactor. Arab J Chem 14(10):103366
47. Gao Z et al (2022) Theoretical and experimental insights into the mechanisms of C6/C6 PFPiA degradation by dielectric barrier discharge plasma. J Hazard Mater 424:127522
48. Groele JR et al (2021) An investigation of plasma-driven decomposition of per- and polyfluoroalkyl substances (PFAS) in raw contaminated ground water. J Appl Phys 130(5):053304
49. Guo J et al (2022) Theoretical and experimental insights into electron-induced efficient defluorination of perfluorooctanoic acid and perfluorooctane sulfonate by mesoporous plasma. Chem Eng J 430:132922
50. Hayashi R et al (2015) Decomposition of Perfluorinated Compounds in Water by DC Plasma within Oxygen Bubbles. Electr Eng Japan 190(3):9–16
51. Jovicic V et al (2018) Degradation of Low Concentrated Perfluorinated Compounds (PFCs) from Water Samples Using Non-Thermal Atmospheric Plasma (NTAP). Energies 11(5):1290
52. Lewis AJ et al (2020) Rapid degradation of PFAS in aqueous solutions by reverse vortex flow gliding arc plasma, vol 6. Water Research & Technology, Environmental Science, pp 1044–1057. 4

53. Mahyar A et al (2019) Development and Application of Different Non-thermal Plasma Reactors for the Removal of Perfluorosurfactants in Water: A Comparative Study. Plasma Chem Plasma Process 39:531–544
54. Nau-Hix C et al (2021) Field Demonstration of a Pilot-Scale Plasma Reactor for the Rapid Removal of Poly- and Perfluoroalkyl Substances in Groundwater. ACS ES&T Water 1(3):680–687
55. Palma D et al (2021) PFAS Degradation in Ultrapure and Groundwater Using Non-Thermal Plasma. Molecules 26(4):924
56. Palma D, Richard C, Minella M (2022) State of the art and perspectives about non-thermal plasma applications for the removal of PFAS in water. Chem Eng J Adv 10:100253
57. Saleem M et al (2020) Comparative performance assessment of plasma reactors for the treatment of PFOA; reactor design, kinetics, mineralization and energy yield. Chem Eng J 382:123031
58. Saleem M et al (2022) Highly efficient degradation of PFAS and other surfactants in water with atmospheric RAdial plasma (RAP) discharge. Chemosphere 307:135800
59. Singh RK et al (2020) Treatment of PFAS-containing Landfill Leachate using an Enhanced Contact Plasma Reactor. J Hazard Mater 124452
60. Singh RK et al (2019) Rapid Removal of Poly- and Perfluorinated Compounds from Investigation-Derived Waste (IDW) in a Pilot-Scale Plasma Reactor. Environmental Science & Technology 53(19):11375–11382
61. Singh RK et al (2020) Removal of Poly- and Per-Fluorinated Compounds from Ion Exchange Regenerant Still Bottom Samples in a Plasma Reactor. Environmental Science & Technology 54(21):13973–13980
62. Stratton GR et al (2017) Plasma-Based Water Treatment: Efficient Transformation of Perfluoroalkyl Substances in Prepared Solutions and Contaminated Groundwater, Environmental Science & Technology, 51(3):1643–1648.
63. Tachibana K, Takeuchi N, Yasuoka K (2014) Reaction Process of Perfluorooctanesulfonic Acid (PFOS) Decomposed by DC Plasma Generated in Argon Gas Bubbles. IEEE Trans Plasma Sci 42(3):786–793
64. Yang L et al (2020) Persulfate-based degradation of perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) in aqueous solution: Review on influences, mechanisms and prospective. J Hazard Mater 393:122405
65. Zhan J et al (2020) Remediation of perfluorooctanoic acid (PFOA) polluted soil using pulsed corona discharge plasma. J Hazard Mater 387:121688
66. Zhang H et al (2021) Enhancing Interface Reactions by Introducing Microbubbles into a Plasma Treatment Process for Efficient Decomposition of PFOA, Environmental Science & Technology, 55(23):16067–16077.
67. Shaji MA et al (2024) Investigating the Effects of Gliding Arc Plasma Discharge's Thermal Characteristic and Reactive Chemistry on Aqueous PFOS Mineralization. Plasma 7(3):705–720
68. Surace MJ et al (2023) Plasma-Assisted Abatement of Per- and Polyfluoroalkyl Substances (PFAS): Thermodynamic Analysis and Validation in Gliding Arc Discharge. Plasma 6(3):419–434
69. Chen C et al (2024) Degradation of perfluoroalkyl and polyfluoroalkyl substances (PFAS) in water by use of a nonthermal plasma-ozonation cascade reactor: Role of different processes and reactive species. Chem Eng J 486:150218
70. Topolovec B et al (2024) Plasma water treatment for PFAS: Study of degradation of perfluorinated substances and their byproducts by using cold atmospheric pressure plasma jet. J Environ Chem Eng 12(3):112979
71. Gyaljen Tamang S et al (2024) Degradation of PFOA solutions and PFAS-contaminated groundwater using atmospheric non-thermal plasma treatment. Water Pract Technol 19(7):2645–2654
72. Shi Y et al (2018) Probing the Differential Tissue Distribution and Bioaccumulation Behavior of Per- and Polyfluoroalkyl Substances of Varying Chain-Lengths. Isomeric Structures and Functional Groups in Crucian Carp, Environmental Science & Technology 52(8):4592–4600
73. Brunn H et al (2023) PFAS: forever chemicals—persistent, bioaccumulative and mobile. Reviewing the status and the need for their phase out and remediation of contaminated sites. Environ Sci Eur 35(1):20–50
74. Arana Juve J-M et al (2023) Complete defluorination of per- and polyfluoroalkyl substances — dream or reality? Curr Opin Chem Eng 41:100943
75. Biondo O et al (2023) Products, reactive species and mechanisms of PFOA degradation in a self-pulsing discharge (SPD) plasma reactor. Chemosphere 341:139972
76. Etz BD, Shukla MK (2023) Per- and polyfluoroalkyl substances chemical degradation strategies: insights into the underlying reaction mechanisms. Curr Opin Chem Eng 42:100956
77. Bruggeman PL, Christopher (2009) Non-thermal plasmas in and in contact with liquids. J Phys D: Appl Phys 42
78. Song C et al (2025) Plasma-Generated Free Electrons Induced Perfluorooctanoic Acid Efficient Degradation at the Gas–Liquid Interface. Environmental Science & Technology

79. Mahyar A et al (2019) Development and Application of Different Non-thermal Plasma Reactors for the Removal of Perfluorosurfactants in Water: A Comparative Study. Plasma Chem Plasma Process 39(3):531–544
80. Jovicic V et al (2018) Degradation of Low Concentrated Perfluorinated Compounds (PFCs) from Water Samples Using Non-Thermal Atmospheric Plasma (NTAP). Energies 11
81. Rabinovich A et al (2021) Scaling Up of Non-Thermal Gliding Arc Plasma Systems for Industrial Applications. Plasma Chem Plasma Process 42(1):35–50
82. Morelli R et al (2003) Fenton-Dependent Damage to Carbohydrates: Free Radical Scavenging Activity of Some Simple Sugars. J Agric Food Chem 51(25):7418–7425
83. Halasi M et al (2013) ROS inhibitor N-acetyl-L-cysteine antagonizes the activity of proteasome inhibitors. Biochem J 454(2):201–208
84. Pelaez M et al (2016) Use of selected scavengers for the determination of NF-TiO2 reactive oxygen species during the degradation of microcystin-LR under visible light irradiation. J Mol Catal A: Chem 425:183–189
85. Li Y et al (2021) Cold atmospheric pressure plasma-activated medium induces selective cell death in human hepatocellular carcinoma cells independently of singlet oxygen, hydrogen peroxide, nitric oxide and nitrite/nitrate. Int J Mol Sci 22(11):5548
86. Lee HY et al (2018) Comparative study of the Ar and He atmospheric pressure plasmas on E-cadherin protein regulation for plasma-mediated transdermal drug delivery. Journal of Physics D: Applied Physics 51(21):215401
87. Takamatsu T et al (2014) Investigation of reactive species using various gas plasmas. RSC Adv 4(75):39901–39905
88. *PubChem Compound Summary for CID 5988, Sucrose*. (2025) [cited 2025 April 15]. Available from: https://pubchem.ncbi.nlm.nih.gov/compound/Sucrose
89. *PubChem Compound Summary for CID 12035, Acetylcysteine*. (2025) [cited 2025 April 15]. Available from: https://pubchem.ncbi.nlm.nih.gov/compound/Acetylcysteine
90. *PubChem Compound Summary for CID 887, Methanol*. (2025) [cited 2025 April 15]. Available from: https://pubchem.ncbi.nlm.nih.gov/compound/Methanol
91. Burlica R, Kirkpatrick M, Locke B (2006) Formation of reactive species in gliding arc discharges with liquid water. J Electrostat 64:35–43
92. *Ionization Energy in the Periodic Table of Elements*. (2025). Available from: https://pubchem.ncbi.nlm.nih.gov/periodic-table/ionization-energy
93. Oxygen [cited 2025 April 15]. Available from: https://webbook.nist.gov/cgi/cbook.cgi?ID=C7782447&Mask=20
94. Nitrogen [cited 2025 April 15]. Available from: https://webbook.nist.gov/cgi/cbook.cgi?ID=C7727379&Mask=20
95. Water [cited 2025 April 15]. Available from: https://webbook.nist.gov/cgi/inchi?ID=C7732185&Mask=20
96. Hydrogen *atom*. [cited 2025 April 15]. Available from: https://webbook.nist.gov/cgi/cbook.cgi?ID=C12385136&Mask=20
97. Xiao F et al (2020) Thermal Stability and Decomposition of Perfluoroalkyl Substances on Spent Granular Activated Carbon. Environmental Science & Technology Letters 7(5):343–350
98. Watanabe N et al (2018) Thermal mineralization behavior of PFOA, PFHxA, and PFOS during reactivation of granular activated carbon (GAC) in nitrogen atmosphere. Environmental Science and Pollution Research 25(8):7200–7205
99. Duchesne AL et al (2020) Remediation of PFAS-Contaminated Soil and Granular Activated Carbon by Smoldering Combustion,Environmental Science & Technology, 54(19):12631–12640.
100. Takeuchi N et al (2014) Plasma–liquid interfacial reaction in decomposition of perfluoro surfactants. J Phys D 47(4):045203
101. Ju Y, Sun W (2015) Plasma assisted combustion: dynamics and chemistry. Prog Energy Combust Sci 48:21–83
102. Singh RK et al (2019) Breakdown Products from Perfluorinated Alkyl Substances (PFAS) Degradation in a Plasma-Based Water Treatment Process, Environmental Science & Technology, 53(5):2731–2738.
103. Ganzallo FJ et al (2026) Plasma-assisted degradation and defluorination of perfluorobutane sulfonate (PFBS) in a plasma spinning disk reactor: The critical role of OH radicals. Chem Eng J 531:174175
104. Tomei G et al (2025) Fluorine Substitution and CO2 Production in the Treatment of Perfluorooctanoic Acid in a Radial Plasma Discharge Reactor. Plasma Chem Plasma Process 46(1):10
105. Umar M (2021) Reductive and Oxidative UV Degradation of PFAS—Status, Needs and Future Perspectives. Water 13(22):3185

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Authors and Affiliations

Mikaela J. Surace[1] · Mobish A. Shaji[2] · Erich Heinzel[3] · Jimmy Murillo-Gelvez[3] · Jitlekha Tubsuri[1] · Alexander A. Fridman[2] · Alexander Rabinovich[2] · Erica R. McKenzie[3] · Gregory Fridman[4] · Christopher M. Sales[1,2]

✉ Mikaela J. Surace
ms4966@drexel.edu

[1] Department of Civil, Architectural and Environmental Engineering, Drexel University, Philadelphia, PA 19104, USA

[2] C and J Nyheim Plasma Institute, Drexel University, Camden, NJ 08103, USA

[3] Department of Civil and Environmental Engineering, Temple University, Philadelphia, PA 19122, USA

[4] AAPlasma LLC, Grove City, OH 43123, USA