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Research paper

Investigation of perfluoroalkyl acid contamination hotspots in South Africa: a case study of fire outbreak sites

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This study investigated PFAS contamination at selected fire outbreak sites, focusing on PFOA and PFOS detection in soil and surface water samples in South Africa. The findings revealed varying contamination levels across samples. All soil samples contained PFOA and PFOS, at average concentrations of 10.97 ng·g^{-1} dw and 18.58 ng·g^{-1} dw, respectively. Similarly, PFOA and PFOS were found in all surface-water samples, with average levels of 5.16 ng·L^{-1} and 11.48 ng·L^{-1}, respectively. PFOA and PFOS remained detectable in soil at multiple sites up to 2–5 years after documented incidents, indicating persistence at fire-impacted locations. An exploratory Welch's t-test did not indicate statistically detectable differences between soil and surface-water concentrations; however, inference is limited because surface-water sampling was limited (n = 2). A screening-level ingestion scenario was used to contextualize surface-water concentrations (HQ/HI), indicating potential concern only if these surface waters were used as drinking sources. Overall, the findings motivate expanded PFAS profiling (including precursors), improved hydrologic pathway characterization (upstream/downstream designs), and broader spatial sampling to strengthen source attribution and exposure assessment in the African context.

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INTRODUCTION

Per- and polyfluoroalkyl substances (PFAS) are a diverse class of synthetic fluorinated compounds characterised by strong carbon–fluorine bonds that confer thermal stability and resistance to degradation. PFAS are often present in paints, adhesives, cosmetics, electronic devices, cleaning agents, polishes, waxes, and pesticides (Domingo, 2012; La Rocca et al., 2012; Mudumbi et al., 2014). However, persistence and mobility in the environment have raised concerns about long-term contamination of soils and waters near emission sources (Domingo, 2012; Mudumbi et al., 2014). Included among products containing PFAS are aqueous film-forming foams (AFFF), which are widely used to suppress hydrocarbon fuel fires by rapidly forming a film that separates fuel vapour from oxygen and prevents re-ignition (Shinoda and Nomura, 1980; Scheffey et al., 1995; Moody and Field, 2000; Schultz et al., 2003). Early AFFF formulations contained PFOS-based or related chemistries, while later products increasingly incorporated fluorotelomer-based ingredients, meaning PFAS composition can vary by manufacturer, production route, and period of manufacture (Moody and Field, 2000; Place and Field, 2012; Dauchy et al., 2017). Because PFAS resist hydrolysis, photolysis, and biodegradation under many environmental conditions, releases at fire incident sites may persist and migrate into nearby soils and surface waters (Prevedouros et al., 2006; Buck et al., 2011; Cousins et al., 2016).

PFAS are believed to accumulate in living organisms within the environment (Domingo, 2012; Mudumbi et al., 2014a). They have been detected in soil (Gao et al., 2019; Zhang et al., 2019), water (Andersson et al., 2019; Hepburn et al., 2019), air (Wu et al., 2019), and sediments (Cao et al., 2015; Dong et al., 2018). Further studies have shown how PFAS are distributed between plant absorption in soil and water and how they contaminate Kampala's drinking water supplies (Dalahmeh, 2018). Additional research indicates that PFOS and PFOA are present in high concentrations at various levels in Nigerian pore water and surface water (Ololadea et al., 2018). Similarly, high concentrations of PFAS were found in tap and river water (Essumang et al., 2017), as well as in human tissues in the population of a Ghanaian study (Dartey et al., 2021). Compared to the US, Asia, and Europe, PFAS have been found in higher concentrations in South African sediments, fish, and raw water (Groffen, 2018; Booi et al., 2019).

There have been numerous fires and explosions across South Africa over the years. These include a warehouse that was set on fire during civil unrest and looting, resulting in an unknown amount of hazardous agricultural substance seeping into an Indian Ocean estuary. Following the firefighting operations, vivid blue wastewater spilled into a neighbouring river system, killing thousands of fish and other aquatic species in August 2021 (Carnie, 2021). In September 2021, Unit 1 of the coal-fired Kendal Power Station run by South Africa's national electricity provider, Eskom, tripped after the generating transformer caught fire (Bonorchis, 2021). In the same year, in August, the Unit 4 generator at Medupi Power Station in Limpopo Province exploded when hydrogen and air were accidentally allowed to combine (Eskom, 2021). In June 2022, Unit 2 of the Eskom Duvha Power Station in Mpumalanga Province caught fire (Mokgobu, 2022). Additionally, in December 2022,

a massive truck carrying liquid petroleum gas exploded in Boksburg, east of Johannesburg (Burgess, 2022). Furthermore, in July 2023, a breach in the fuel oil pipeline system caused a fire at the Grootvlei Power Station in Mpumalanga, according to Eskom (Mrinmay, 2023). All of these instances involved hydrocarbon (fuel) fires, which may have required the use of firefighting foam to extinguish.

To our knowledge, this study provides one of the first focused case assessments of PFOA and PFOS in environmental media collected at fire outbreak sites in South Africa. Using soil and nearby surface-water sampling within a 500 m radius of incident locations, the study evaluates concentration patterns, considers runoff-driven pathway plausibility, and presents a screening-level exposure interpretation to motivate further targeted monitoring and source-fingerprinting in the African context.

MATERIALS AND METHODS

Sampling site identification

The sampling sites were selected based on information gathered on fire explosion locations where aqueous film-forming foam (AFFF) was likely used as a fire extinguisher. Areas potentially contaminated with PFAS were identified as sampling sites in the Gauteng Province of South Africa, with Johannesburg being the primary focus (Fig. 1). The analysis of potential PFAS contamination involved extracting soil and surface water samples from the identified areas. Additional information on fire incidents, such as cause, location, year, land cover, and geographical coordinates, can be found in Table A1 (Appendix).

Study area and surface-water pathways

The study was conducted in Gauteng Province, South Africa, with sampling sites located within the greater Johannesburg metropolitan area (Fig. 1). Johannesburg contains dense industrial, transport, and energy infrastructure where hydrocarbon fires may occur and where firefighting activities can generate contaminated runoff. From an exposure-pathway perspective, surface-water transport is plausible where firefighting foam and impacted wash-water enter stormwater drains, roadside drainage lines, or natural channels, particularly on sloping terrain that promotes overland flow during rainfall. Therefore, for each fire-incident site we documented (i) local drainage direction/topographic gradient, (ii) the nearest surface-water body or stormwater conveyance, and (iii) whether sampling points were located up-gradient, at-site, or down-gradient relative to the incident location (Table A1, Appendix). This pathway framing guided water-sampling placement within a 500 m radius, while acknowledging that urban catchments may also receive PFAS inputs from wastewater effluent, landfill leachate, and diffuse urban runoff.

Sample collection

Samples of soil (n = 6 sites) and surface water (n = 2 sites) were collected in November 2023. Soil was sampled at all 6 fire-incident locations; however, surface water could only be sampled at 2 sites, because an accessible receiving stream or drainage channel was not present within the 500 m search radius at every fire location, and in some cases the surrounding drainage infrastructure did not provide a safe or representative open-water sampling point during the field campaign. Accordingly, the surface-water component was designed as a targeted pathway-screening exercise rather than a full comparative survey across all incident sites. This limitation is important for interpretation and is the reason why all inferential comparisons involving surface water are treated as exploratory. The location of Sampling Sites JB 1 through JB 5 are shown in Fig. 1.

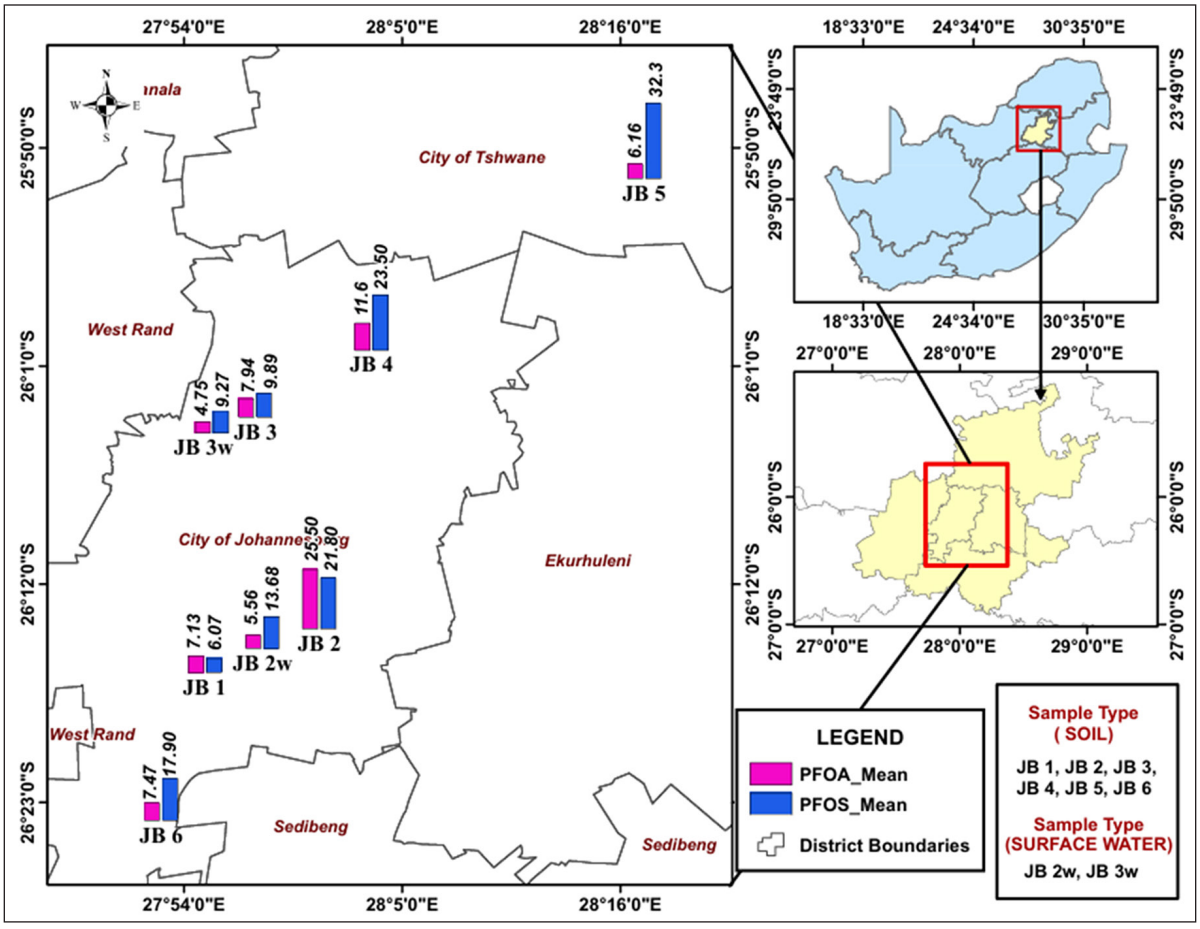

Figure 1. Study area and sampling locations in Gauteng Province, South Africa

For each soil and surface-water sampling point, the horizontal distance to the reported fire epicentre was measured using GIS and recorded (Table A1, Appendix). Soil sampling locations were selected as close as practicable to the incident footprint while remaining accessible and safe; water sampling points were located along the nearest receiving drainage line or channel, prioritizing down-gradient locations likely to intercept runoff.

Sampling and preparation

Soil samples

Soil analysis was conducted following the method of Carter and Gregorich (2007). In brief, a bucket auger was used to extract soil samples at a depth of 0–15 cm; to avoid cross-contamination, the bucket auger was thoroughly cleaned with methanol after each use. Five samples, collected from the centre and 4 corners of a 5×5 m² area, were obtained at each designated location within a 500 m radius of the fire location. A soil sample of about 1.0 kg was gathered and stored in clean 500 mL zip-lock polypropylene bags. Field blanks were incorporated into the batch and subjected to the same environmental conditions as actual samples. The soil samples were transported to the laboratory, where they were air-dried to get rid of any moisture, crushed, and blended with a silica mortar and pestle after removing large stones and plant and animal remnants. The materials were stored at ambient temperature in pre-cleaned glass jars pending further analysis, after being screened through a 2 mm mesh.

Surface-water sampling

Surface-water samples were collected only at fire sites where a plausible receiving water body or open drainage channel could be identified and safely accessed within 500 m of the incident footprint. Sampling prioritized down-gradient positions along the local drainage direction. Because not all incident sites had accessible surface water within the defined search radius, only 2 surface-water locations were sampled. The resulting water dataset is therefore limited and was used primarily to assess pathway plausibility and screening-level exposure context rather than to support strong between-site statistical inference. Samples were collected into pre-cleaned polypropylene bottles that had been rinsed 3 times with water from the sampling location. Bottles were submerged approximately 10 cm below the surface, facing upstream to avoid disturbance and capture representative flow. Samples were stored on ice in the field and transported to the laboratory for refrigerated storage prior to extraction.

Chemical reagents

PFOA (98%) and PFOS (98%) were purchased from ACCU Standard, USA, while labelled PFOA ([$^{13}C_8$, 99%]) and PFOS ([$^{13}C_8$, 99%]) were obtained from Cambridge Isotope Laboratories, USA. Formic acid (99+%) and ultra-pure methanol (Romil-UpSTM) were purchased from Thermo Scientific and Microsep, respectively, in South Africa. Sigma-Aldrich provided the 25% ammonium hydroxide, which was used without additional purification. To create the working solutions, Milli-Q system deionised water (resistivity >18.5 MΩ·cm) was used.

Sample extraction methods

Soil extraction procedure

The procedure outlined by Lorenzo et al. (2015) for methanol extraction was employed for the soil extraction. Internal standards ((20 µL of 200 µg·L^{-1} [$^{13}C_8$] PFOA and [$^{13}C_8$] PFOS)) were added to all samples (field samples, laboratory blanks, and spiked recovery samples) prior to extraction. The soil samples underwent triple methanol extractions (10 mL each), followed by centrifugation (965 r·min^{-1}, 15 min), vortexing, and sonication (15 min each). The resultant supernatant was diluted with 18 MΩ ultra-pure water to reach 100 mL after being condensed to 5 mL under a nitrogen stream. The reconstituted supernatant was run at a flow rate of roughly 5 mL·min^{-1} through a HyperSep SPE cartridge that had been pre-conditioned. Following a 20 min nitrogen gas purge, the cartridge was connected to an Envicarb system for elution using 5 mL of 0.1% NH_4OH in methanol. The ultimate volume of the eluents was 1 mL after it was evaporated under nitrogen. The spiked extracts were filtered after going through a 0.22 µm syringe filter and then put into auto-sampler vials for analysis.

Water extraction

Prior to extraction, refrigerated water samples were equilibrated to ambient temperature. To guarantee precision and uniformity, a 20 µL volume of internal standard solution, containing 200 µg·L^{-1} each of [$^{13}C_8$] PFOA and [$^{13}C_8$] PFOS was added to each sample, followed by thorough mixing using a Vortex mixer for 1 min. The extraction method employed a Hypersep SPE (150 mg, 6 mL, WAX) cartridge, adhering to the procedure established by Taniyasu et al. (2005). The cartridges were preconditioned by washing them with 5 mL of methanol containing 0.1% NH_4OH, followed by 5 mL of 18 MΩ deionized water. Polypropylene tubing was used to pump 500 mL of water sample through the C18 column cartridges at a flow rate of 5 mL·min^{-1}, effectively capturing the desired compounds. To eliminate impurities and inorganic ions, the cartridges were rinsed with 5 mL of 18 MΩ deionized water. For elution, 5 mL of 0.1% NH_4OH in methanol was passed through cartridges connected to Envicarb. The spiked extracts were then evaporated using nitrogen until the final volume reached 0.5 mL. After being filtered through a 0.22 µm syringe filter to ensure clarity, the extracts were moved to auto-sampler vials for further analysis.

Liquid chromatography–mass spectrometry analysis

Quantification of PFOA and PFOS was performed using an ultra-high-performance liquid chromatography system coupled to a triple quadrupole mass spectrometer operated in negative electrospray ionization (ESI–) with targeted MS/MS with exact mass. A BEH C18 column (ACQUITY UPLC, 1.7 µm, 2.1 × 50 mm) with a BEH C18 VanGuard pre-column was used with a gradient mobile phase of water and acetonitrile containing 20 mM ammonium acetate at a flow rate of 0.3 mL·min^{-1} and an injection volume of 1 µL. Mass spectrometer settings were capillary voltage 0.5 kV, source temperature 100°C, desolvation temperature 350°C, desolvation gas flow 900 L·h^{-1}, and cone gas flow 100 L·h^{-1}. Quantification used isotope dilution with $^{13}C_8$-labelled internal standards and a 6-point calibration curve (0.01–50 ng·L^{-1}) prepared in solvent/extract matrix

Quality assurance and control

Every possible source of instrumental and procedural contamination was carefully removed to reduce background contamination. To ensure the highest level of purity and accuracy, polytetrafluoroethylene (PTFE) components were strictly avoided, and all containers and laboratory disposables were carefully cleaned and rinsed with methanol before use. The performance of the method was comprehensively assessed utilising a number of factors, such as linearity, accuracy, relative recovery, limit of quantification (LOQ), limit of detection (LOD), and blank effects. In Table 1 these parameters for PFOA and PFOS are comprehensively summarised. All tests were carried out in triplicate to guarantee dependability.

Statistical analysis

All statistical analyses were performed using Python 3.11 (SciPy). Concentrations are reported as mean ± standard deviation. Because surface-water sampling was limited (n = 2 sites),

Table 1. Method validation summary for PFOA and PFOS measurement in soil (ng·g^{-1} dw) and surface water (ng·L^{-1})

Compound	LoD	LoQ	R^2	Recovery (%)	Relative percentage difference (%)
			Soil (ng·g^{-1} dw)		
PFOA	0.10	0.22	0.997	83.00	14.85
PFOS	0.19	0.33	0.998	89.00	15.80
			Surface water (ng·L^{-1})		
PFOA	0.10	0.22	0.997	89.00	16.00
PFOS	0.19	0.33	0.998	95.00	15.00

inferential comparisons between soil (n = 6 sites) and water are interpreted cautiously and treated as exploratory. Group comparisons were evaluated using Welch's two-sample t-test (α = 0.05) due to unequal variances and unequal sample sizes; no claim of equivalence between matrices is made.

Risk assessment (surface-water ingestion scenario)

A screening-level, non-cancer risk assessment was conducted to contextualize measured surface-water concentrations. This scenario is conservative and may not represent actual drinking-water behaviour in urban Johannesburg where municipal treated water is typically used, although residents of informal settlements in some locations do make use of streamwater for domestic purposes. Daily intake (E, mg·kg^{-1}·day^{-1}) was estimated for adults assuming body weight (BW) = 70 kg and water ingestion rate (IR) = 2 L·day^{-1}.

Exposure was calculated as:

$$E = (C \times IR) / BW, \text{ where } C \text{ is measured concentration (mg·L}^{-1}) \quad (1)$$

Hazard Quotient (HQ) was calculated as:

$$HQ = E/RfD \quad (2)$$

Hazard Index (HI) was calculated as:

$$HI = \Sigma HQ \text{ for PFOS and PFOA} \quad (3)$$

Oral reference doses were taken from USEPA (2022). RfD (PFOA) = 0.03 ng·kg^{-1}·day^{-1} and RfD (PFOS) = 0.1 ng·kg^{-1}·day^{-1}.

Results are interpreted as screening indicators, not definitive population risk estimates. A terrestrial ecological risk quotient was not computed because receptor-specific soil benchmarks for PFOA/PFOS vary widely and site-specific exposure pathways were not measured; this is recommended for follow-up work.

Time-since-incident analysis

Sampling was conducted once (November 2023). 'Time since incident' (years) does not represent repeated sampling over 5 years; it represents the elapsed time between the most recent documented fire incident at each site and the single sampling event. Time since incident for each site was derived from incident records (Appendix). Associations between time since incident and log-transformed PFOA/PFOS concentrations were explored using Spearman correlation and simple linear regression as descriptive analyses, recognizing limited sample size.

Quality control and quality assurance

To assess instrument linearity, a 6-point calibration was performed across a 0.01–50 ng·L^{-1} concentration range for both PFOA and PFOS, each analysed in triplicate. Calibration standards (0.01, 0.1, 5, 10, 30, and 50 ng·L^{-1}) were dissolved in methanol and regression coefficients (R^2) from plotting area ratio vs. concentration yielded an average R^2 of 0.998 for PFOA and PFOS, indicating excellent linearity. Background correction was performed by analysing blanks (n = 3) and subtracting the mean concentrations of each analyte from the test sample concentrations to minimise instrumental bias.

To validate measurement accuracy and precision, deionised water was spiked with internal standards (20 µg·L^{-1}) and subjected to identical extraction conditions as the real samples. Results showed PFOA and PFOS recoveries between 83% and 89% (Table 1), indicating satisfactory trueness. Intra-laboratory reproducibility was assessed using relative percentage difference (RPD), yielding generally acceptable results (Table 1). These findings meet recommended guidelines (Pihlström, 2011), confirming good precision.

RESULTS AND DISCUSSION

Concentrations of PFOA and PFOS in soil and surface-water samples are summarized in Table 2 and visualized using boxplots in Figs 2 and 3.

Table 2. Concentration of PFOA and PFOS in soil (ng·g^{-1} dw) and surface water (ng·L^{-1}) found at some selected outbreak sites in South Africa (with sampling points marked in Figure 1 and description of the sampling sites in Table A1, Appendix).

Concentrations of PFOA and PFOS in soil samples

The analysis of 6 soil samples found PFOA and PFOS in every sample, with concentrations ranging from 6.16–25.50 ng·g^{-1} dw for PFOA and 6.07–32.30 ng·g^{-1} dw for PFOS. The corresponding mean concentrations were 10.97 ng·g^{-1} and 18.58 ng·g^{-1}.

PFOA contamination levels varied across the sites, with JB2 recording the highest concentration (25.5 ng·g^{-1} dw) and JB5 the lowest (6.16 ng·g^{-1} dw). PFOS contamination also showed spatial variability, with JB5 registering the highest concentration (32.3 ng·g^{-1} dw) and JB1 the lowest (6.07 ng·g^{-1} dw). Although JB2 and JB5 are both electricity substation-related fire sites, the present dataset does not permit a source-specific explanation for why one site shows relatively higher PFOA while another shows higher PFOS. Such differences may reflect a combination of incident history, amount of foam applied, local drainage and retention conditions, and site-specific soil properties affecting PFAS sorption and redistribution. Because only two target analytes were measured and no formulation, precursor, or hydrologic fingerprinting data were available, these site-to-site differences should be interpreted descriptively rather than as evidence of distinct foam chemistries or a single confirmed source mechanism. Studies in Sweden (Niarchos et al., 2022) and China (Li et al., 2019), found similar levels, with maximum concentrations ranging from 1.2 to 17 ng·g^{-1} for PFOA and 3.1 to 10 ng·g^{-1} for PFOS. A site contaminated with AFFF in the United States, which was studied by Xiao et al. (2015), was found to have PFOA concentrations of 26.0 ± 0.9 ng·g^{-1} and PFOS concentrations of 20.4 ± 0.2 ng·g^{-1}.

Table 2. PFPA and PFOS concentrations in soil and surface water

Sampling point	Concentration of PFOA	Concentration of PFOS
Soil concentrations (mean: ng·g^{-1} dw)		
JB 1	7.13	6.07
JB 2	25.50	21.80
JB 3	7.94	9.89
JB 4	11.6	23.50
JB 5	6.16	32.30
JB 6	7.47	17.90
Surface-water concentrations (mean: ng·L^{-1})		
JB 2W	5.56	13.68
JB 3W	4.75	9.27

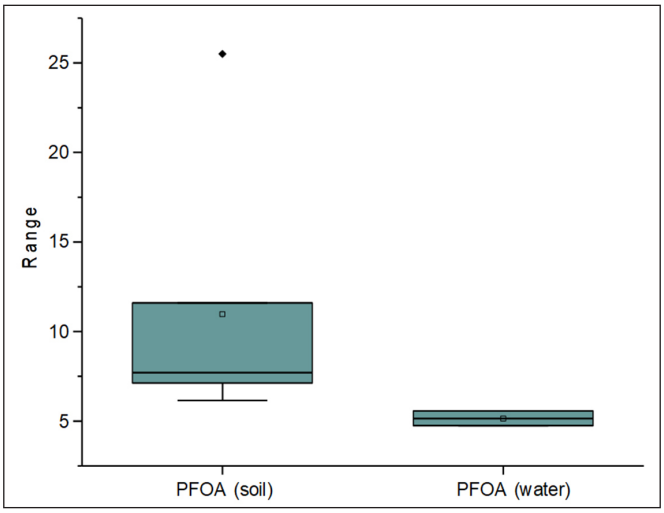

Figure 2. Box-plot of PFOA distribution profile in soil and surface water from Gauteng (November 2023)

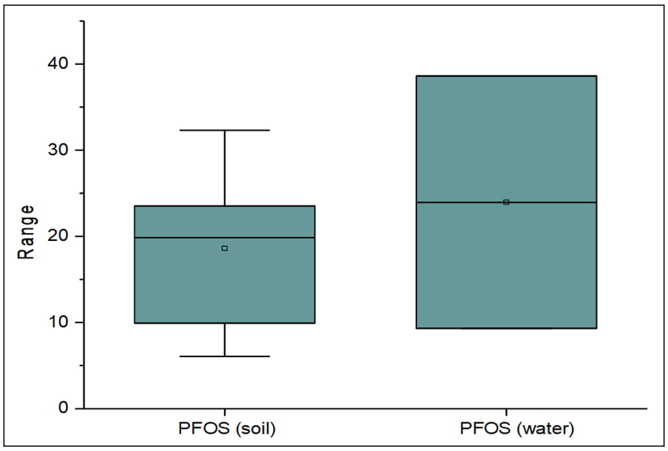

Figure 3. Box-plot of PFOS distribution profile in soil and surface water from Gauteng (November 2023)

The inconsistent concentrations of PFAS observed across different sites can be attributed to the non-uniform formulation and application of AFFF, as well as the inherent properties of the soil. The soil's type, structure, and composition significantly impact PFAS contamination patterns, as established by previous studies (Houtz et al., 2013; Anderson et al., 2018).

The PFOA and PFOS concentrations measured in this study are best interpreted as being within the lower-to-moderate range reported for AFFF-impacted soils internationally, rather than uniformly similar to all contaminated sites worldwide. For example, the concentrations overlap with some values reported for selected impacted locations in Sweden, China, and the United States, but they are substantially lower than the much higher concentrations reported for heavily impacted manufacturing sites and airports in Belgium (114 ng·g^{-1} for PFOA and 7 800 ng·g^{-1} for PFOS), Norway (75 ng·g^{-1} for PFOA and 3000 ng·g^{-1} for PFOS), and Australia (55 ng·g^{-1} for PFOA and 13 400 ng·g^{-1} for PFOS) (Braunig et al., 2019; Groffen et al., 2019; Hoisaeter et al., 2019). This distinction is important because PFAS concentrations at fire-affected locations are strongly influenced by site history, intensity and frequency of AFFF use, distance from the release area, and environmental retention conditions.

Although soil properties are known to influence PFAS retention and mobility, the present study did not include direct

characterization of site-specific soil texture, organic carbon content, pH, mineralogy, or permeability (Bräunig et al., 2017; Li et al., 2018) and therefore cannot determine whether South African soils at these locations behave differently from soils reported in other countries. Previous studies have shown that PFAS sorption and transport can vary with soil composition and physical structure, including organic matter content, particle-size distribution, and hydraulic properties (Kissa, 2001; Wang et al., 2017; Anderson, 2018; Zhou et al., 2022). Accordingly, the relatively lower concentrations observed here, compared with some international hotspot sites, should not be attributed to intrinsic differences in South African soils without dedicated soil-property measurements.

Concentrations of PFOA and PFOS in surface water

Analysis of the two surface-water samples showed measurable concentrations of both PFOA (4.75 to 5.36 ng·L^{-1}; average: 5.16 ng·L^{-1}) and PFOS (9.27 to 13.68 ng·L^{-1}; average: 11.48 ng·L^{-1}). JB 2W, a stream that flows through Roosevelt Park, located approximately 50 m down-gradient of the Roosevelt Park Power Substation, showed higher concentrations (PFOA 5.54 ng·L^{-1} and PFOS 13.68 ng·L^{-1}) than JB3W. While incident-related runoff from firefighting activities is a plausible contributor, the present study did not assess whether runoff was contained, diverted, or managed under site-specific emergency or environmental control procedures. In principle, firefighting response protocols may seek to minimize discharge of foam-impacted water to the natural environment, for example, through containment, recovery, or diversion to controlled systems; however, actual implementation can vary by incident setting and was not investigated here. Accordingly, the observed surface-water concentrations should be interpreted as evidence of PFAS presence in nearby receiving waters, not as proof of a specific uncontrolled release pathway from firefighting operations.

Although incident runoff is a plausible contributor, urban surface waters may also reflect upstream wastewater and diffuse inputs; therefore, source attribution cannot be confirmed without broader PFAS/precursor profiling and hydrologic tracing. In addition to potential AFFF runoff from the incident site, PFAS in urban surface waters can also reflect mixed upstream inputs including wastewater effluent, storm-water runoff from urban surfaces, and landfill leachate (Ahrens et al., 2011; Houtz et al., 2016; Andrews et al., 2021; O'Connor et al., 2022). Therefore, the observed concentrations at JB2W and JB3W may represent combined contributions rather than a single source.

The findings from this study align with previous research indicating an increased level of PFOS in surface water close to AFFF treatment locations in other countries (Awad et al., 2011; Ahrens et al., 2015; Dauchy et al., 2017). Furthermore, variations in PFOA and PFOS proportions in firefighting foams, as noted by Houtz et al. (2013), may also contribute to the observed concentrations. This study's surface water PFOA and PFOS values were broadly comparable in order of magnitude to those observed in other African nations while, even in locations without documented AFFF point sources, PFAS can occur due to diffuse urban and industrial inputs (Ahrens et al., 2011). This is confirmed by the detection of PFOS and PFOA in surface water samples across Africa: in Mozambique, where concentrations of 0.141 ng·L^{-1} for PFOS and 0.025 ng·L^{-1} for PFOA were reported (Hope et al., 2020); in Zambia, with concentrations of 0.13 ng·L^{-1} for PFOS and 0.10 ng·L^{-1} for PFOA (Baabish et al., 2021); in Kenya's Sabaki River, with PFOA levels of 2.13 ng·L^{-1} and PFOS levels of 1.94 ng·L^{-1} (Fiedler et al., 2021); and in a river in Nigeria, with PFOS levels ranging from 3.90 to 10.10 ng·L^{-1} and PFOA levels from 0.80 to 2.80 ng·L^{-1} (Ololade et al., 2018). These observations may result from contributions which potentially include industrial discharges from manufacturing, wastewater treatment plants, hospital waste, surface water runoff from contaminated areas, and landfill leachates (Ahrens et al., 2011; Houtz et al., 2016; Sedlak et al., 2017; Dalahmeh et al., 2018).

The levels of PFOA and PFOS in the surface water surrounding the AFFF-contaminated sites in this investigation were lower than worldwide values. This finding contrasts with reports of elevated levels of these chemicals in waterways worldwide. For instance, the Rhine River and its tributaries in Germany have recorded concentrations ranging from 2.0–33 900 ng·L^{-1} of PFOA and 2.0–5 900 ng·L^{-1} of PFOS, as reported by Skutlarek et al. (2006). Similarly, researchers have found significant contamination in other regions. In Korea's Nakdong River Basin, Son et al. (2013) detected PFOA levels of 3.8–329 ng·L^{-1} and PFOS levels of 4.1–302.8 ng·L^{-1}. Meanwhile, surface water in Sweden recorded levels of 522 ng·L^{-1} of PFOA and 2 280 ng·L^{-1} of PFOS, according to Gobelius et al. (2018). Furthermore, Zhou et al. (2013) reported PFOA and PFOS concentrations that reached a maximum of 1 390 ng·L^{-1} and 1 650 ng·L^{-1}, respectively, in China's Tangxun Lake. Several factors may have contributed to the disparity between previous findings and our study. Firstly, the extent of AFFF exposure to soil significantly impacts nearby surface water contamination. Research has consistently shown that areas with high AFFF application exhibit elevated PFAS levels, as the foam can leach or run off into adjacent water bodies (Kissa, 2001; Anderson, 2018).

In addition to potential AFFF runoff from the incident site, PFAS in urban surface waters can also reflect mixed upstream inputs including wastewater effluent, storm-water runoff from urban surfaces, and landfill leachate (Ahrens et al., 2011; Houtz et al., 2016; Andrews et al., 2021; O'Connor et al., 2022). Therefore, the observed concentrations at JB2W and JB3W may represent combined contributions rather than a single source. Figure 4 shows the PFOA and PFOS contamination profile in soil and surface water across the sampling sites.

Statistical analysis

Welch's 2-sample t-tests (α = 0.05) were used as exploratory comparisons of mean concentrations between soil (n = 6) and surface water (n = 2). No statistically detectable difference was observed for PFOA or PFOS (p > 0.05); however, inference is limited by the small number of surface-water sites, and these results should not be interpreted as evidence of equivalence between matrices. The primary value of the comparison is descriptive, highlighting that both media contained measurable concentrations at sampled locations.

Time-since-incident patterns

Concentrations varied substantially across sites and did not show a simple monotonic relationship with time since the most recent documented incident. Because sampling was conducted once (November 2023), 'years since incident' represents elapsed time derived from records rather than repeated sampling over time; therefore, patterns are descriptive and should not be interpreted as degradation-rate estimates. Site-to-site variability is consistent with differences in incident magnitude and response, heterogeneous foam application, site drainage, and soil sorption properties, and possible mixed PFAS inputs, particularly for urban surface waters. Follow-up work using repeated sampling, expanded PFAS panels (including precursors), and hydrologic tracing would be required to quantify temporal trends and strengthen source attribution.

The distribution pattern of PFOA shows marked site-to-site heterogeneity, ranging from 25.5 ng·g^{-1} dw at JB2 to 6.16 ng·g^{-1} dw at JB5, with intermediate values at the remaining sites. Because the study is based on one sampling campaign and a limited analyte

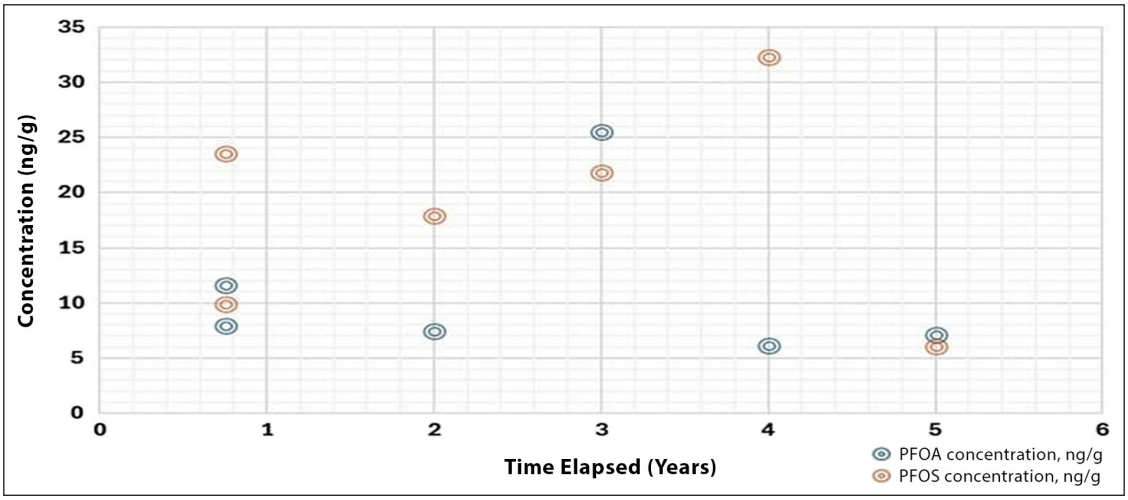

Figure 4. Concentration of PFOA and PFOS in soil and surface water from 6 sites (with sampling points marked in Figure 1)

Table 3. Screening risk results (Hazard Quotient and Hazard Index) for PFOS and PFOA in surface-water samples

Sampling site	Concentration (ng·L^{-1})		Hazard Quotient		Hazard Index
	PFOA	PFOS	PFOA	PFOS	(PFOA, PFOS)
JB 2W	5.56	13.68	5.30	3.91	9.21
JB 3W	4.75	9.27	4.52	2.65	7.17

panel, these differences are interpreted descriptively and should not be attributed to a single controlling factor. In particular, the present data do not allow us to separate the relative influence of incident history, runoff behaviour, local retention conditions, or possible non-fire-related PFAS inputs.

Similarly, PFOS concentrations varied across sites, with relatively high values at JB5 and JB2 and lower values at JB1 and JB3. A key observation is that both PFOA and PFOS remained detectable in soils at sites where the most recent documented fire incident occurred several years before sampling, indicating persistence of these compounds in the local environment. However, the concentration differences among sites should not be interpreted as direct evidence of degradation rates or a single dominant source. For nearby surface waters in particular, mixed upstream inputs such as wastewater, diffuse urban runoff, and other urban PFAS sources remain plausible and cannot be excluded without broader analyte coverage and hydrologic tracing.

Potential exposure and risk assessments

Under the conservative ingestion scenario, HQ and HI values exceeded unity for both surface-water sites (Table 3), indicating potential concern only if these waters were used as drinking sources. Because this is an urban setting where municipal supply is common, the assessment is intended as screening-level context rather than a definitive estimate of community risk. Additional exposure pathways (recreational contact, irrigation, livestock watering) and actual use patterns should be assessed in follow-up work

CONCLUSION

This study provides evidence of PFOA and PFOS presence in soils (n = 6 sites) and nearby surface waters (n = 2 sites) associated with fire-impacted locations in Gauteng Province, South Africa. Soil concentrations (6.16–25.50 ng·g^{-1} dw for PFOA; 6.07–32.30 ng·g^{-1} dw for PFOS) indicate persistent residues that remain detectable up to several years post-incident at multiple sites. Surface-water concentrations were measurable at 2 sites and, under a conservative screening ingestion scenario, produced HQ/HI values > 1, motivating follow-up exposure assessment. However, surface-water inference is limited by small sample size and the possibility of mixed PFAS sources in urban catchments (e.g., wastewater effluent, storm-water runoff, landfill leachate). Future work should expand surface-water sampling (including upstream/downstream designs), include additional PFAS analytes/precursors for source fingerprinting, and integrate site hydrology and soil properties to clarify transport pathways and exposure relevance.

DATA AVAILABILITY

Data for the study will be made available from the corresponding author on reasonable request.

STATEMENTS AND DECLARATIONS

The authors affirm that there was no possibility of a conflict of interest during the creation of this work.

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AUTHOR CONTRIBUTIONS

All authors contributed to the study conception and design. Material preparation, sample collection and analysis were performed by Bernard Barnor, Albert Eshun, Christopher Curtis and Kriveshini Pillay. The first draft of the manuscript was written by Bernard Barnor and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

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APPENDIX

Table A1. Sampling sites and their geographical coordinates

Sampling site	Type, place and year of fire	Land cover	Coordinates
JB 1	Power plant substation explosion, Florida Glen (2017)	Commercial	26° 09.850″ S 27° 56.710″ E
JB 2	Roosevelt Park power substation (RPPS) explosion, Roosevelt Park (2015)	Commercial	26° 09.091″ S 27° 59.345″ E
JB 2W	A river 50 m away from the Roosevelt Park power substation (RPPS)	Surface water	26° 09.161″ S 27° 59.447″ E
JB 3	Wilgeheuwel park (WP) (2023)	Residential	26° 06.467″ S 27° 53.483″ E
JB 3W	A stream by Wilgeheuwel park (WP)	Surface water	26° 06.465″ S 27° 53.502″ E
JB 4	Gas tanker explosion on N1 highway near William Nicol (2023)	Commercial	26° 03.528″ S 27° 59.219″ E
JB 5	Power plant explosion, Wapadrand substation (2021)	Commercial	25° 46.968″ S 28° 20.637″ E
JB 6	Power plant substation explosion, Lenasia (2023)	Commercial	26° 23.148″ S 27° 52.634″ E