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Volatile per- and polyfluoroalkyl substances (PFAS) and other semi-volatile organic chemicals in indoor air of fire stations: the influence of gear storage conditions

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Previous studies have examined firefighters' exposure to chemicals associated with fire calls and per- and polyfluoroalkyl substances (PFAS); however, little attention has been given to exposures in the fire station itself. This study measured volatile PFAS and other semi-volatile organic chemicals (SVOCs) in indoor air of fire stations in North Carolina, USA. Eleven fire stations and one turnout gear cleaning and supply station across three different counties in North Carolina were sampled between May and September of 2024. In each station, air was sampled both in the day room (area where firefighters congregate during the day) and in the area where the turnout gear was stored. Samples were extracted and analyzed for 14 different PFAS and 120 different SVOCs using GC-HRMS. The most frequently detected chemicals were PFASs, combustion byproducts, and phthalates. 8 : 2 FTOH was measured at the highest level at 1067 ng m⁻³. Total SVOC levels, and particularly PFAS, were significantly higher in closed rooms with turnout gear relative to storage areas open to the apparatus bay or the day room. Exposure estimates indicate that some turnout gear storage areas may pose a health risk for PFOA based on inhalation exposure to 8 : 2 FTOH and conversion to PFOA in the blood. These data highlight a need to consider station design and ventilation needs to mitigate risks.

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Environmental significance

This study provides insight into environmental processes governing indoor chemical transport, transformation, and exposure to PFAS in occupational settings, and namely fire stations. Elevated concentrations of volatile PFAS and SVOCs in enclosed turnout gear storage areas demonstrate how building design and ventilation influence contaminant accumulation and distribution. The detection of elevated 8 : 2 FTOH levels in air, and its potential biotransformation to PFOA, highlights an important pathway linking indoor air chemistry to human health risk. These results underscore the role of fire stations as underrecognized microenvironments of PFAS exposure and emphasize the need to integrate environmental engineering controls to reduce inhalation risks indoors.

Introduction

Firefighters play a vital role in protecting public safety but face a variety of occupational hazards that place them at increased risk for certain chronic diseases.¹ Numerous epidemiological studies have reported that firefighters have elevated incidence and mortality rates for several types of cancer, including testicular, prostate, and non-Hodgkin's lymphoma, as well as increased risk of cardiovascular disease.²⁻⁴ These health disparities are thought to be driven, in part, by chronic exposure

to toxic substances encountered during routine firefighting activities.^{5,6}

Much of the scientific attention to date has focused on chemical exposures encountered at the fire scene. Combustion of building materials and consumer products can release a complex mixture of toxic compounds, including formaldehyde, hydrogen cyanide, polycyclic aromatic hydrocarbons (PAHs), flame retardants, and plasticizers.⁷ In recent years, per- and polyfluoroalkyl substances (PFAS) have emerged as a particular concern due to their widespread use, environmental persistence, and potential adverse health effects.⁸ PFAS have been linked to multiple adverse outcomes, including immunotoxicity, liver damage, reproductive harm, and cancer.^{9,10} Firefighters can be exposed to PFAS not only through firefighting foams (*e.g.*, aqueous film-forming foam, or AFFF), but also potentially through other occupational sources such as turnout gear. Turnout gear has historically been manufactured

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using PFAS-treated materials to provide a durable water- and oil-repellent finish to the outer shell of the firefighter turnout gear;¹¹ however, due to mounting pressure, several manufacturers have switched to non-PFAS treated textiles in firefighter turnout gear, although there are some associated tradeoffs with this transition.^{12,13}

While the focus on chemical exposures during fire suppression activities has been essential, less is known about exposures that may occur within the fire station environment itself, which was highlighted in a recent review on this topic.¹⁴ Fire stations serve as living quarters, training facilities, and vehicle storage areas, and firefighters often spend extended periods of time in these spaces. There is growing concern that contaminated turnout gear may act as a source of secondary exposure. Laboratory studies have demonstrated that PFAS-coated textiles, particularly those containing side-chain fluorinated polymers (SFPs) can off-gas volatile PFAS compounds, such as fluorotelomer alcohols (FTOHs) into the air^{15,16} and similar concerns have been raised for other semi-volatile organic compounds (SVOCs), such as flame retardants.^{17,18}

Despite these concerns, no published studies to date have systematically evaluated PFAS concentrations in fire station air. The potential for inhalation exposure to volatile and semi-volatile PFAS is particularly concerning given that PFAS have been detected in indoor air in locations with PFAS-treated products.^{19,20} Turnout gear may represent a significant and understudied source of indoor contamination. If so, airborne PFAS concentrations in fire stations could be elevated relative to typical residential or occupational environments.

In this study, we address this critical knowledge gap by measuring concentrations of volatile PFAS and other SVOCs in fire station air and assess differences based on turnout gear storage conditions. We specifically hypothesized that the turnout gear would be a source of volatile PFAS to the air and would therefore be higher in rooms where turnout gear was stored. We place particular emphasis on volatile PFAS, which may contribute significantly to internal exposure to more harmful PFAS such as perfluorooctanoic acid (PFOA), a compound with known toxicity and regulatory concern.²¹ Using inhalation intake estimates, we calculated the amount of time needed to reach a dose of PFOA equivalent to exposure that would occur if drinking water containing PFOA at the current US EPA Maximum Contaminant Level (MCL) drinking water standard of 4 ng L⁻¹. This work provides an important step in understanding the potential for chronic chemical exposure in fire station environments and informs future efforts to mitigate risks among this critical workforce.

Materials and methods

Sample collection

Eleven fire stations and one turnout gear cleaning and supply station were sampled across three different counties in North Carolina between May–September of 2024. Stations included in the study were identified *via* word of mouth and contacts with the research team. Fire Chiefs at each station provided approval before all air sampling equipment was deployed.

In each station, an active air sampler was deployed in the room where the turnout gear was stored, and a second sampler was simultaneously deployed in the day room. The day rooms in all stations were similar in design and were carpeted with multiple recliners, a few small tables and a television. Some stations stored their turnout gear in a separate closed room with a door that connected to the apparatus bay (identified in this study as “closed” rooms). Other stations had a more open design in which the turnout gear was stored either in lockers along the wall of the apparatus bay, or in an open hallway just off the apparatus bay (identified as “open” rooms in this study).

Pumps and samplers were typically placed in an elevated location near a wall to ensure they would not disrupt operations in the station (see SI Fig. S1 for photos of air samplers deployed in a closed and open station design). To support the air sampling, we used an AirCheck Connect pump (SKC Inc. Model # 220-4000S) attached to a low volume polyurethane foam (PUF/XAD-2/PUF) sorbent tube (SKC Inc. Model #226-143) operated at 2.0 L min⁻¹. Pumps were purchased new from SKC Inc. and were calibrated by the company prior to use. Air sampling occurred at each location for 2–3 consecutive days. The following information was collected with each sample: the date and time the pumps were turned on and off, the volume of air sampled, and average temperature in the room during sampling. Temperatures in all indoor rooms ranged from 21–26 °C while temperatures in the gear rooms that were open to the apparatus bay ranged from 26–31 °C. In the latter settings, it was noted that the apparatus bay doors were open during both set up and taken down of the air samples. While setting up the air sampling equipment, the study team investigated several sets of turnout gear in each station to review information on the NFPA labels. A review of the labels indicated that the turnout gear was sourced from different manufacturers and over different years, with a majority being manufactured prior to 2024. Based on this information, and information on PFAS treatments in turnout gear available in the published literature,¹³ it seems likely that a majority of the turnout gear in these rooms likely had intentional PFAS treatments to the garments.

PUF extraction

PUF samples were extracted *via* Soxhlets using ethyl acetate for ~20 hours. A suite of isotopically labeled internal standards (see Tables S1–S3) was spiked into each PUF sample prior to extraction and used for quantification of target analytes. Extracts were then evaporated to ~1 mL using a Turbo Vap nitrogen evaporation system plumbed with ultra-high purity nitrogen gas. Prior to instrument analysis, a suite of isotopically labeled recovery standards was added to each extract and then the extract was filtered using a nylon syringe filter (0.45 microns). Recovery standards were used to quantify recovery of the internal standards. Five field blanks (PUF cartridges taken into the field) were extracted alongside samples.

Instrumental analysis

Extracts were first analyzed for 14 volatile PFAS using an Exploris GC-Orbitrap (Thermo Scientific™). Extracts were then



separately analyzed for SVOCs and BFRs using a Q-Exactive GC-Orbitrap (Thermo Scientific™) operated in electron impact mode, and an Exactive GC-Orbitrap (Thermo Scientific™) operated in negative chemical ionization mode, respectively. Both systems have a maximum scan range of 30 to 3000 m/z , are capable of mass resolution up to 100 000 (m/z 272), and mass accuracy below 1 ppm (internal calibration). Instrument settings for the Exploris GC-Orbitrap method can be found in (ref. 22) and the Q-Exactive method for SVOCs can be found in (ref. 23). The Exactive-GC was equipped with an Agilent™ DB-5MS GC Column capillary column (10 m × 0.25 mm ID, 0.1 μm film thicknesses) with helium as the carrier gas flowing at 1.3 mL min⁻¹. The Programmable Temperature Vaporizer (PTV) Inlet is operated in splitless injection mode with a 1 μL injection. The GC oven temperature program was 80 °C for 2 min, 80 to 250 °C at 20 °C min⁻¹, 250 to 260 °C at 1.5 °C min⁻¹, 260 to 300 at 25 °C min⁻¹ with a final hold of 9 min hold at 300 °C (total run time 28 min). The transfer line and ion source were held at 300 °C for the duration. Methane gas was used as the reagent gas with a flow of 1.5 mL min⁻¹. For this analysis, the electron energy was set to 70 eV and the QE-GC was operated in full scan negative chemical ionization (NCI) mode with an automatic gain control (AGC) of 1 × 10⁶, a maximum IT of 200 ms, and a mass resolution of 60 000 (at m/z 200). Samples were run with a scan range of 70–800 m/z and quantified using the Tracefinder software.

QAQC

Field blanks (non-deployed air cartridges) were analyzed in tandem with samples to assess the levels of contamination and were used to estimate the method detect limit (MDL). The MDL was calculated as the average plus three times the standard deviation of analytes in field blanks and normalized to the volume of air sampled. If sample values exceeded the MDL, they were then blank subtracted with the average blank value to produce blank subtracted values. MDL values for all target analytes are listed in Tables S1–S3.

Exposure estimates

Inhalation exposure to 8:2 FTOH and conversion to perfluorooctanoic acid (PFOA) in blood were assessed with data generated in this study. Parameters used to support the exposure estimate can be found in Table S7. We estimated the amount of time (T ; in hours) needed to reach an exposure equivalent (EE) to the acceptable daily dose (ADD) for PFOA when breathing in air at the levels measured in this study. The ADD used in this study was 4 ng day⁻¹ and was calculated using the current U.S. Drinking Water Maximum Contaminant Level (MCLs; 4 ng L⁻¹) and the drinking water ingestion rate of 1.04 L day⁻¹ (EPA Exposure Factors Handbook, 2011; Table 3-1). The EE was therefore 4 ng, and the time needed to reach this equivalent was estimated using eqn (1):

$$T(\text{hours}) = \text{EE} / (C_A^* \ln hR \times M_R) \quad (1)$$

where C_A indicates the measured 8:2 FTOH concentration (ng m⁻³) measured in this study, $\ln hR$ indicates the average

inhalation rate of an adult male (0.6 m³ hour; EPA Exposure Factors Handbook, 2011; Table 6-1), and M_R indicates the estimated metabolic conversion (%) of 8:2 FTOH to PFOA in the blood. The metabolic conversion of 8:2 FTOH used here was 5% and was based on data reported in Himmelstein *et al.*²⁴ In this study the authors exposed adult Sprague-Dawley rats to 8:2 FTOH *via* inhalation and assessed its toxicokinetics and levels of metabolites in blood plasma. Notably, this study conducted a route-to-route comparison of exposure for oral exposure *vs.* inhalation and the authors noted a similar kinetic yield of PFOA after oral dosing in male rats relative to inhalation exposure. Fromme *et al.*²⁵ and Schlummer *et al.*²⁶ also used a 5% conversion rate in their assessment of PFOA exposure based on inhalation of 8:2 FTOH.

Statistical analyses

Non-parametric approaches were generally employed to assess significant differences among analytes based on room sampled with the exception of a few chemicals in which the data distribution was found to be normal using a Shapiro Wilks test (Dibenzofuran, Fluoranthene, Pyrene, PCB11, NP, DMP). Specifically, we used either a Kruskal-Wallis or an ANOVA test (depending on normality of the data) with a post-hoc Mann-Whitney test to determine if levels were significantly different among the rooms sampled. We also employed Spearman correlation analyses to assess correlations among analytes and among rooms in the same station. A p -value of 0.05 or less was considered statistically significant.

Results and discussion

Air sampling was conducted at 11 different fire stations between May–September of 2024. Most buildings were constructed in the 1980s–1990s; however, a few were constructed in the late 2010s. In one station an air pump malfunctioned, and no sample was collected for the corresponding day room. One station had two different rooms where turnout gear was stored (both closed rooms) and both were sampled. The majority of samples were collected at fire stations; however, a fire service department turnout gear supply and storage unit (henceforth identified as “supply”) was also sampled. The supply site contained a turnout gear washing machine extractor used to clean gear for multiple stations. It was sampled at three different locations, including adjacent to the extractor and in a supply room. Ultimately, we collected air samples from 10 days rooms and 12 turnout gear storage rooms among the fire stations, 7 of which were a closed design (*i.e.*, in a room with a closed door), and 5 of which were in an open design. The latter describes cases where gear was stored in lockers hanging either along the wall or off an open hallway connected to the apparatus bay.

Levels of volatile PFAS measured in the air of the fire stations and the supply unit are provided in Table 1. Seven of the 14 targeted PFAS were detected frequently (>50% of samples) among the fire stations. The most abundant PFAS measured, based on median levels, were the fluorotelomer alcohols. 6:2 FTOH had the highest median levels at 15.2 ng m⁻³, whereas



Table 1 Descriptive statistics for the volatile PFAS measured in fire station indoor air samples ($n = 25$)

Chemical	Acronym	Overall DF (%)	Detection limit (ng m ⁻³)	25th Percentile (ng m ⁻³)	50th Percentile (ng m ⁻³)	75th Percentile (ng m ⁻³)	Maximum (ng m ⁻³)
2-(Perfluorohexyl)ethanol	6 : 2 FTOH	100	0.2	6.38	15.23	67.89	657.5
2-(Perfluorooctyl)ethanol	8 : 2 FTOH	100	0.03	3.10	7.32	21.69	1067
2-(Perfluorodecyl)ethanol	10:FTOH	100	0.03	2.48	9.33	25.82	133.2
2-Perfluorohexyl ethyl acrylate	6 : 2 FTAC	12	0.01	NA	NA	NA	0.48
2-Perfluorooctyl ethyl acrylate	8 : 2 FTAC	68	0.01	NA	0.02	0.101	0.708
2-Perfluorodecyl ethyl acrylate	10 : 2 FTAC	36	0.01	NA	NA	0.016	0.526
2-Perfluorohexyl ethyl methacrylate	6 : 2 FTMAC	100	0.01	0.087	0.157	0.197	11.47
2-Perfluorooctyl ethyl methacrylate	8 : 2 FTMAC	0	0.02	NA	NA	NA	NA
2-(<i>N</i> -methylperfluoro-1-butanesulfonamido)ethanol	MeFBSE	100	0.02	3.87	7.61	37.57	147.9
<i>N</i> -Methyl- <i>N</i> -(2-hydroxyethyl)perfluorooctanesulfonamide	MeFOSE	40	0.01	NA	NA	0.069	0.433
Ethyl- <i>N</i> -(2-hydroxyethyl)perfluorooctane sulfonamide	EtFOSE	0	0.82	NA	NA	NA	NA
<i>N</i> -Methylperfluorooctanesulfonamide	MeFOSA	0	0.02	NA	NA	NA	NA
<i>N</i> -Ethylperfluorooctane sulfonamide	EtFOSA	0	0.02	NA	NA	NA	NA
Parachlorobenzotrifluoride	PCBTF	96	0.16	0.98	2.93	5.98	8.01

the highest measurement overall was 8 : 2 FTOH at 1067 ng m⁻³. FTOHs are commonly detected in indoor air given their known use in materials common to most buildings, such as paint.²⁷ FTOH levels measured here are similar to levels reported in occupational settings and outdoor shops reported by Schlummer *et al.*²⁶ and Fraser *et al.*,²⁸ but generally higher than residential settings reported by Winkens *et al.*²⁹ and Eichler *et al.*³⁰ MeFBSE was the second most abundant PFAS measured after the FTOHs with a median of 7.61 and a maximum of 147.9 ng m⁻³. The other frequently detected PFAS include 6 : 2 FTMAC and PCBTF. FTOHs, MeFBSE and 6 : 2 FTMAC are among the most frequently detected PFAS measured in firefighter turnout gear in prior studies and suggest they may be a primary source of these compounds in the air measured.³¹ We hypothesize that these PFAS are released due to weathering of the textiles that could release some of the SFPs applied to the textiles, or unreacted monomers, and lead to volatilization of these

smaller, neutral PFAS as shown in SI Fig. S2. In contrast, PCBTF, while considered a PFAS due to the presence of the fully fluorinated carbon atom, is not typically used as a water and oil repellent. PCBTF is typically used as a solvent in coatings, caulks, adhesives and cleaning products and can be found in paints and sealants.³² However, it is listed on California's Proposition 65 list as a potential carcinogen.³³ Only a few occupational studies have measured PCBTF in air where they were detected at levels much higher ($\mu\text{g m}^{-3}$ to mg m^{-3}) than levels reported in the current study.³⁴ California's Office of Environmental Health (OEHHA) reports a No Significant Risk Level (NSRL) of 23 μg per day.³⁵ Median levels measured in this study for PCBTF were 2.93 ng m⁻³ with a maximum of 8.01 ng m⁻³, far below levels of concern based on this NSRL.

We compared PFAS levels by room to assess differences (Fig. 1). All three FTOHs, along with 6 : 2 FTMAC and 8 : 2 FTAC were significantly higher in the closed vs. open turnout gear

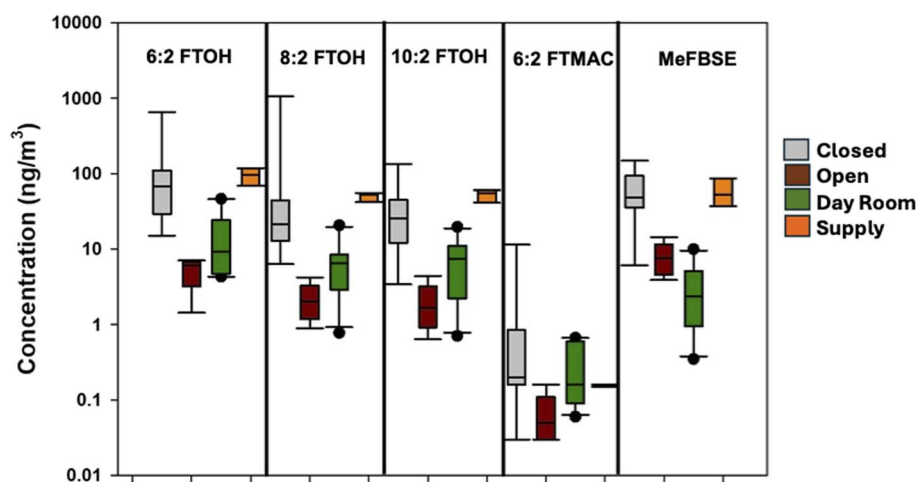


Fig. 1 Volatile PFAS levels measured in indoor air of North Carolina fire stations. Grey = gear room closed ($n = 7$); dark red = gear room open ($n = 5$); green = day rooms ($n = 10$); orange = supply ($n = 3$).



storage rooms (Table S5). Median MeFBSE levels in the closed turnout gear storage rooms were higher than the open rooms but did not reach statistical significance ($p = 0.47$); however, levels in the closed storage rooms were significantly higher than levels measured in the day rooms by a factor of 20 ($p < 0.05$). While we only sampled five stations with an open design for turnout gear storage, these data suggest that the open design leads to greater dilution and air change rates that contribute to the lower PFAS levels. Taken together, these results also reinforce our hypothesis that the primary source of these PFAS is off gassing from the PPE (turnout gear) stored in these rooms as no other materials or products were present in these rooms. While it seems likely that the turnout gear itself is the source, it's more challenging to discern whether the PFAS originated from contamination brought into the station (on the PPE) through firefighting activities (*i.e.* exposure to soot), off-gassing or leaching from the PFAS treatments intentionally applied to the turnout gear, or from secondary sources such as cross-contamination while laundering. More research is needed to differentiate the specific PFAS source.

In addition to the volatile PFAS, we also measured air concentrations for an additional 120 SVOCs that included organophosphate esters (OPEs), polycyclic aromatic hydrocarbons (PAHs), pesticides, polychlorinated biphenyls (PCBs), phthalates and phthalate alternatives and brominated flame retardants (BFRs). Measured levels for these SVOCs can be found in Table S4. In general, PAHs, dibenzofuran (a combustion byproduct) and phthalates were the most abundant SVOCs detected based on median levels, although there were several OPEs, pesticides and other chemicals (*i.e.* industrial or personal care product chemicals) that were commonly detected and relatively abundant. The latter includes TEP, TCPP, DEET, chlordane, 4-tert octylphenol, nonylphenol and lillial. BFRs and some of the higher molecular weight PAHs and phthalates were not commonly detected in these samples which likely reflect the sampling design. We employed PUF sorbent tubes for air sampling which are designed to capture vapor phase chemicals and may not be efficient at sampling chemicals with higher molecular weights and lower vapor pressures that typically sorb to particles/aerosols at room temperature.³⁶

Naphthalene was found to have the highest median and maximum levels of these SVOCs, followed by dibutyl phthalate (DBP). Similar to the PFAS, several SVOCs were found to be present at significantly higher levels in the closed rooms compared to the open rooms (Table S5), including TiBP, dibenzofuran, acenaphthene, phenanthrene, anthracene, DMP, and DEP. In addition, TEP, Fluoranthene, and DMP were significantly higher in closed rooms compared to the day rooms. A majority of these are PAHs, which accumulate on turnout gear due to exposure to soot and smoke⁷ and may also off-gas from the gear during storage. Interestingly, lillial and TCPP were found to have higher, although not statistically significant, levels in the day room compared to other sampling locations. Lillial is a common synthetic fragrance used in personal care products and cleaning products and thus higher levels may stem from the greater concentration of people in the day room. TCPP is a common flame retardant used in both

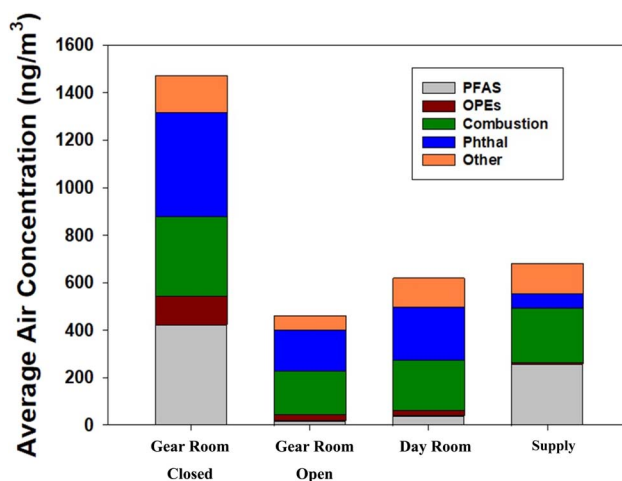


Fig. 2 Average levels of SVOC classes measured in indoor air of North Carolina fire stations. Gear room closed ($n = 7$); gear room open ($n = 5$); day rooms ($n = 10$); supply ($n = 3$). PFAS includes the sum of 7 analytes, OPEs includes the sum of 7 analytes, Combustion includes the sum of 10 combustion byproducts, Phthal includes the sum of 4 phthalates, and Other includes the sum of PCB 11, PCB28, DEET, chlordane, 4tOP and NP.

furniture and insulation and may reflect a greater source from furniture in the day room.^{37,38}

Combining all results, we calculated the average air concentrations for each frequently detected (>50%) chemical among the four different locations sampled. As evident in Fig. 2, the sum concentration of all chemicals measured was highest in the closed turnout gear storage rooms and was approximately two times higher than the other rooms. The contribution from volatile PFAS was also highest in the closed turnout gear storage rooms, and supply, relative to the day rooms and open turnout gear storage locations. Phthalates and OPEs also contributed more to the overall chemical burden in the closed turnout gear storage rooms compared to all other sites sampled. Taken together, these results suggest that the air concentration is more contaminated with volatile PFAS and SVOCs in general, in these closed rooms used for turnout gear storage.

SI Fig. S3 presents Spearman correlation coefficients for all chemicals measured. Not surprisingly, many chemicals within the same class were significantly correlated with each other. Volatile PFAS were all moderately to highly correlated ($r_s = 0.47-0.92$), except for PCBTF which was not correlated with any of the other PFAS and suggests that PCBTF is coming from another source, possibly paints, sealants or consumer products. Table S6 presents correlations between paired day rooms and turnout gear storage rooms by station. Despite the small number of stations sampled, several chemicals were strongly and significantly correlated in paired rooms from the same station, including 8:2 FTAC, MeFBSE, TCPP and DBP ($R_s = 0.68-0.78$). TCPP and DBP, which displayed the highest correlations, are used as flame retardants and plasticizers, the former frequently found in furniture and insulation, and the latter often used as a plasticizer in PVC plastics. Primary uses of MeFBSE and 8:2 FTAC are not well known, although, both may



be residual intermediates, or weathering products, from SFP treatments in firefighter turnout gear. Both PFAS, and others, have been identified in firefighter turnout gear manufactured to meet NFPA standards, and they may be added to other textiles to provide water repellency.³¹ Thus, the significant correlation, along with the higher levels found in turnout gear storage areas, suggests that emissions from the gear are influencing air levels in the day rooms; however, more research is needed to verify this hypothesis.

Overall, these data demonstrate that volatile PFAS are a significant contributor to the SVOC burden, particularly in closed turnout gear storage areas. The maximum concentration recorded was for 8 : 2 FTOH at a concentration of 1067 ng m⁻³. 8 : 2 FTOH has been shown to metabolize to PFOA following inhalation in a prior rodent study.²⁴ Given PFOA is the only PFAS currently labeled as a carcinogen,²¹ we further investigated potential exposure to PFOA *via* inhalation and transformation of 8 : 2 FTOH. Using eqn (1) and the parameters provided in Table S7, we calculated how long an individual could inhale the air at the measured concentrations in our study before reaching a daily exposure equivalent to the current drinking water MCL for PFOA, and determining if inhalation could lead to exposures above the acceptable daily dose (ADD). In the closed turnout gear storage room with the highest concentrations of 8 : 2 FTOH, we estimated it would take less than 10 minutes to reach a dose equivalent to the ADD (Table S8). For the remaining closed turnout gear storage rooms (including supply), it would take anywhere from 3–24 hours to reach this threshold, with an average of 7.4 hours. Estimates over 24 hours would be considered below the ADD as it would take more than 24 hours to reach this exposure. Most firefighters do not spend long periods of time in the turnout gear storage room, so the risk associated with 8 : 2 FTOH in most rooms should be minimal. However, it is important to note that we did not include a 10 fold uncertainty factor in our estimates for the cross-species comparison. If we did include an uncertainty factor, then the estimated risk would be higher. When including the 10 fold uncertainty factor, the average time needed to reach the ADD was reduced to 45 minutes in the closed turnout gear storage areas. Given that the closed turnout gear storage rooms had higher abundances of all measured SVOCs, there may be a need to consider additional ventilation and/or air filtration of these rooms in the future to minimize exposure risks associated with other chemicals among firefighters. Firefighters already face elevated exposures to a number of hazardous chemicals, and the occupation of firefighting itself is classified as a carcinogen.¹ Therefore, any and all exposure mitigation strategies could provide a benefit to fire service members.

Our research team communicated these findings to the fire departments. To address exposure concerns in the room with the highest 8 : 2 FTOH levels, we recommended and installed an air filter/purifier containing activated carbon. Further monitoring of the room levels after installation of the air filter confirmed that the levels of volatile PFAS were reduced, and specifically 8 : 2 FTOH was reduced by 80% (data not shown).

Overall, this study demonstrates that volatile PFAS and a suite of other SVOCs are detectable in the air of fire stations.

While most levels measured here are comparable to other residential settings, levels of the volatile PFAS in the closed turnout gear storage areas and the turnout gear supply depot were elevated compared to residential settings and may pose some risk for exposure to chemicals like PFOA. Limitations of our study include a small number of stations sampled in one state. We do not have any information on the ventilation rates of these stations, or these rooms, both of which may affect air levels. In addition, this study was conducted during the warmest months of the year in North Carolina, where outdoor temperatures can reach over 100 °F. Importantly, some stations did have the apparatus bay doors open during parts of the air sampling, which may further influence air measurements, particularly in the apparatus bay. Despite these limitations, our measurements do align with prior research on air levels of SVOCs. Additional and more robust studies are needed to understand primary sources of these SVOCs and to determine variability over time, particularly over seasons with changes in ambient temperature and ventilation rates.

Conflicts of interest

The authors declare no conflicts of interest.

Data availability

Data for this article, including all individual air measurements, are available at the Duke Research Data Repository at <https://doi.org/10.7924/r4xs60d1h>.

Supplementary information (SI): photos of air sampling, SVOC air correlation analyses, full chemical details for targeted chemicals, QAQC information, descriptive statistics for individual SVOCs, and full details for the inhalation exposure estimates. In the fire stations, a correlation heat map displaying correlations of SVOCs in air, and full chemical information on the target analy. See DOI: <https://doi.org/10.1039/d6em00069j>.

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References

- 1 P. A. Demers, *et al.*, Carcinogenicity of occupational exposure as a firefighter, *Lancet Oncol.*, 2022, 23(8), 985–986.
- 2 H. Jalilian, *et al.*, Cancer incidence and mortality among firefighters, *Int. J. Cancer*, 2019, 145(10), 2639–2646.
- 3 D. J. Lee, *et al.*, Cancer risk among career male and female Florida firefighters: Evidence from the Florida Firefighter Cancer Registry (1981–2014), *Am. J. Ind. Med.*, 2020, 63(4), 285–299.



- 4 R. D. Daniels, *et al.*, Mortality and cancer incidence in a pooled cohort of US firefighters from San Francisco, Chicago and Philadelphia (1950-2009), *Occup. Environ. Med.*, 2014, **71**(6), 388–397.
- 5 M. M. Dahm, *et al.*, Creation of a retrospective job-exposure matrix using surrogate measures of exposure for a cohort of US career firefighters from San Francisco, Chicago and Philadelphia, *Occup. Environ. Med.*, 2015, **72**(9), 670–677.
- 6 N. L. DeBono, *et al.*, Firefighting and Cancer: A Meta-analysis of Cohort Studies in the Context of Cancer Hazard Identification, *Saf. Health Work*, 2023, **14**(2), 141–152.
- 7 K. W. Fent, *et al.*, Flame retardants, dioxins, and furans in air and on firefighters' protective ensembles during controlled residential firefighting, *Environ. Int.*, 2020, 140.
- 8 N. U. S. Mazumder, *et al.*, Firefighters' exposure to per-and polyfluoroalkyl substances (PFAS) as an occupational hazard: A review, *Front. Mater.*, 2023, **10**, 1143411.
- 9 E. M. Sunderland, *et al.*, A review of the pathways of human exposure to poly- and perfluoroalkyl substances (PFASs) and present understanding of health effects, *J. Expo. Sci. Environ. Epidemiol.*, 2019, **29**(2), 131–147.
- 10 NASEM, Guidance on PFAS Exposure, Testing, and Clinical Follow-Up, National Academies of Science, Engineering and Math, 2022, <https://nap.nationalacademies.org/catalog/26156/guidance-on-pfas-exposure-testing-and-clinical-follow-up>.
- 11 G. F. Peaslee, *et al.*, Another Pathway for Firefighter Exposure to Per- and Polyfluoroalkyl Substances: Firefighter Textiles, *Environ. Sci. Technol. Lett.*, 2020, **7**(8), 594–599.
- 12 N. U. S. Mazumder, *et al.*, Toward the future of firefighter gear: Assessing fluorinated and non-fluorinated outer shells following simulated on-the-job exposures, *J. Ind. Text.*, 2023, **53**, DOI: [10.1177/15280837231217401](https://doi.org/10.1177/15280837231217401).
- 13 N. J. Herkert, *et al.*, Per- and Polyfluoroalkyl Substances (PFAS) and Brominated Flame Retardants (BFRs) in Firefighter Turnout Gear: Two Chemical Classes of Concern to Consider, *Environ. Sci. Technol. Lett.*, 2025, **13**, 28–33.
- 14 K. Bralewska, Air pollution inside fire stations: State-of-the-art and future challenges, *Int. J. Hyg Environ. Health*, 2024, 255.
- 15 R. Aranda-Rodriguez, *et al.*, PFAS emissions from functional textiles using micro-chamber and thermal desorption coupled to two-dimensional gas chromatography-time of flight mass spectrometry (TD-GCxGC-TOF MS), *J. Chromatogr. A*, 2024, 1733.
- 16 S. Schellenberger, *et al.*, An Outdoor Aging Study to Investigate the Release of Per- And Polyfluoroalkyl Substances (PFAS) from Functional Textiles, *Environ. Sci. Technol.*, 2022, **56**(6), 3471–3479.
- 17 J. Rovira and J. L. Domingo, Human health risks due to exposure to inorganic and organic chemicals from textiles: A review, *Environ. Res.*, 2019, **168**, 62–69.
- 18 V. N. Mokoana, J. K. O. Asante and O. J. Okonkwo, A review on volatilization of flame retarding compounds from polymeric textile materials used in firefighter protective garment, *J. Fire Sci.*, 2023, **41**(4), 107–121.
- 19 H. Fromme, *et al.*, Neutral polyfluorinated compounds in indoor air in Germany - The LUPE 4 study, *Chemosphere*, 2015, **139**, 572–578.
- 20 B. I. Freberg, *et al.*, Occupational Exposure to Airborne Perfluorinated Compounds during Professional Ski Waxing, *Environ. Sci. Technol.*, 2010, **44**(19), 7723–7728.
- 21 S. Zahm, *et al.*, Carcinogenicity of perfluorooctanoic acid and perfluorooctanesulfonic acid, *Lancet Oncol.*, 2024, **25**(1), 16–17.
- 22 T. Hoxie, *et al.*, Silicone Wristbands as a Personal Passive Sampler to Evaluate Indoor Exposure to Volatile and Non-volatile PFASs, *Environ. Sci. Technol.*, 2024, **58**(37), 16316–16326.
- 23 N. J. Herkert, *et al.*, Wristband Personal Passive Samplers and Suspect Screening Methods Highlight Gender Disparities in Chemical Exposures, *Environ. Sci. Technol.*, 2024, **58**(35), 15497–15510.
- 24 M. W. Himmelstein, *et al.*, 8:2 fluorotelomer alcohol: A one-day nose-only inhalation toxicokinetic study in the Sprague-Dawley rat with application to risk assessment, *Toxicology*, 2012, **291**(1–3), 122–132.
- 25 H. Fromme, *et al.*, Perfluorinated compounds – Exposure assessment for the general population in western countries, *Int. J. Hyg Environ. Health*, 2009, **212**(3), 239–270.
- 26 M. Schlummer, *et al.*, Detection of fluorotelomer alcohols in indoor environments and their relevance for human exposure, *Environ. Int.*, 2013, **57–58**, 42–49.
- 27 L. Cahuas, *et al.*, Paints: A Source of Volatile PFAS in Air? Potential Implications for Inhalation Exposure, *Environ. Sci. Technol.*, 2022, **56**, 17070–17079.
- 28 A. J. Fraser, *et al.*, Polyfluorinated Compounds in Serum Linked to Indoor Air in Office Environments, *Environ. Sci. Technol.*, 2012, **46**(2), 1209–1215.
- 29 K. Winkens, *et al.*, Perfluoroalkyl acids and their precursors in indoor air sampled in children's bedrooms, *Environ. Pollut.*, 2017, **222**, 423–432.
- 30 C. M. A. Eichler, *et al.*, Cloth-Air Partitioning of Neutral Per- and Polyfluoroalkyl Substances (PFAS) in North Carolina Homes during the Indoor PFAS Assessment (IPA) Campaign, *Environ. Sci. Technol.*, 2023, **57**(40), 15173–15183.
- 31 D. J. Muensterman, *et al.*, Disposition of Fluorine on New Firefighter Turnout Gear, *Environ. Sci. Technol.*, 2022, **56**(2), 974–983.
- 32 M. Liu, *et al.*, Nonpolymeric Per- and Polyfluoroalkyl Substances (PFAS) and Side-Chain Fluorinated Polymers in Canadian Building Products, *Environ. Sci. Technol.*, 2025, **59**(38), 20642–20652.
- 33 M. L. Kim-Fu, *et al.*, Fluorinated aromatic PBCTF and 6:2 diPAP in bridge and traffic paints, *Environ. Sci. Process. Impacts*, 2024, **26**(12), 2158–2165.
- 34 E. G. Lee, *et al.*, Assessing Exposures to 1-chloro-4-(trifluoromethyl) Benzene (PCBTF) in U.S. Workplaces, *J. Occup. Environ. Hyg.*, 2015, **12**(7), D123–D130.
- 35 (OEHTA), C.O.o.E.H.H.A., Final Statement Of Reasons Title 27, California Code Of Regulations Section 25705(B) Specific Regulatory Levels Posing No Significant Risk No Significant Risk Level: P-Chloro-A,A,A-Trifluorotoluene, Oehha, 2020,



- pp. 1–13, <https://oehha.ca.gov/proposition-65/crn/ amendment-section-25705-specific-regulatory-levels-posing-no-significant-risk-p-chloro-aaa>.
- 36 T. Harner and M. Shoeib, Measurements of octanol-air partition coefficients (K-OA) for polybrominated diphenyl ethers (PBDEs): Predicting partitioning in the environment, *J. Chem. Eng. Data*, 2002, **47**(2), 228–232.
- 37 E. D. Schreder, N. Uding and M. J. La Guardia, Inhalation a significant exposure route for chlorinated organophosphate flame retardants, *Chemosphere*, 2016, **150**, 499–504.
- 38 P. C. Hartmann, D. Bürgi and W. Giger, Organophosphate flame retardants and plasticizers in indoor air, *Chemosphere*, 2004, **57**(8), 781–787.

