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Investigation of seasonal changes in flame retardant concentrations in car interiors

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We investigated how temperature influences flame retardant levels in vehicle air and dust, and analyzed vehicle interior materials to identify major sources. Airborne concentrations of flame retardants varied dramatically, with some compounds increasing over 50 000-fold at elevated summer temperatures compared to winter temperatures. In contrast, dust levels were more stable but consistently high, reflecting substantial flame retardant content in interior materials. All sampled vehicle materials contained more than 1 μg g^{-1} of three chlorinated organophosphate flame retardants. The combination of high material concentrations and temperature-driven emissions suggests that individuals who spend an extended time in vehicles may face elevated exposure to these chemicals, particularly during warmer conditions.

Environmental significance

Vehicle interiors consistently contain higher levels of flame retardants than many other indoor environments, yet they remain underexplored in chemical exposure assessments. Unlike most indoor environments, vehicles are compact, enclosed spaces that experience frequent and extreme temperature fluctuations – conditions that may amplify chemical emissions. This study shows that temperature markedly increases airborne concentrations of flame retardants in vehicle interiors, while dust-associated levels remain relatively stable. These findings underscore the importance of evaluating both air and dust compartments when assessing human exposure in vehicles and provide critical insight for understanding real-world chemical emissions in transportation environments.

1 Introduction

Flame retardants (FRs) such as BDE 209 (decabromodiphenyl ether) and organophosphate esters (OPEs) are an exposure concern in vehicles, given that high levels of FRs are consistently detected in vehicle dust.[1–5] A wide range of human health effects are associated with exposure to these compounds,[6–10] and many jurisdictions have undertaken actions to regulate their use.[11–14] There is evidence of exposure from cars contributing to overall human exposure to FRs,[15] and one study estimated that up to 60% of daily exposure to some FRs can come from personal transport.[16] However, our understanding of factors in the vehicle environment that drive emissions and exposures is incomplete, particularly concerning which vehicle parts are most relevant to FR exposure, and how environmental conditions in the car can enhance exposures.

While it is known that vehicles are unique indoor environments due to their extreme temperature ranges and greater influence of solar radiation, relatively few studies have focused on the impact of seasonal temperature variations in the release of plastic additives from vehicle materials, although car internal temperatures can exceed outdoor temperatures by more than 20 °C.[17–19] To our knowledge, one study on PBDEs[20] and two on OPEs[21,22] have investigated temperature-driven changes in a real vehicle environment and several studies have been conducted in car test chambers.[21,23,24]

Our study examined temperature-driven changes in levels and distributions of FRs in car interior environments by comparing vehicle dust and air from the same car in winter and summer, supported by direct measurements of FRs in car interior materials to contribute to our understanding of seasonal differences in FR emission profiles and potential exposures.

2 Methods

2.1 Study design

A 2015 Hyundai i20 was investigated in two temperature conditions. Winter sampling occurred at temperatures below 10 °C, in the shade for 10 days, to minimize heating by the sun. Summer sampling occurred at daytime temperatures around 30 °C, in full sun for 5 days, with a southern exposure, to maximize the impact of temperature (Fig. S1). Weather conditions prevented identical sampling durations in winter and summer. The winter sampling took place in January/February 2024 in Špindlerův Mlýn, Czechia and the summer sampling

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in September 2024 in Brno, Czechia. No new car interior elements were introduced between the winter and summer sampling periods and the regular use pattern of the car was not altered between the sampling periods.

2.2 Sampling methods

During each temperature experiment, car indoor air was sampled using a passive air sampler consisting of a precleaned polyurethane foam (PUF) disk (Fig. S2), suspended without a housing from the rearview mirror. Dust from the car dashboard and front seats was collected at the start and end of each sampling period, using a vacuum cleaner and a forensic filter sampling head with a pre-weighed quartz microfibre filter (Fig. S3 and S4). Composite samples were collected to maximize the representativeness of the whole car interior.[25] A summary of the sample collection times and conditions is given in Table S1. After sampling, dust and air samples were wrapped in aluminum foil in separate zip bags, transported at temperatures of 5 °C to the analytical laboratories and stored at −18 °C until processing. Small amounts of car interior materials were also collected, covering four material types: foam from the seats (n = 4), seat textile fibres, soft plastic from the top layer of the dashboard and hard plastic from the center console of the car, doors, and front part of the dashboard (Fig. S5 and S6). Solid samples were collected in glass vials and stored at 4 °C until processing.

Interior car temperatures during the experiment periods were monitored using a Testo 175T1 temperature logger placed in the middle of the car (near the gear shift lever, always in the shade), recording at 15 min intervals, and outside temperature was obtained from the nearest weather station at 30 min intervals (Fig. S1).

2.3 Extraction and analysis

Complete information on dust sample extraction, clean-up and analysis has been published previously[25] and is given in detail in the SI in Texts S1 and S2. Dust samples with filters were ground using a ball mill and weighed. Dust and car material samples (plastic and foam) were divided into two aliquots for separate extraction of OPEs and BDE 209. For OPE analysis, samples were extracted by ultrasonic extraction in methanol and analysed by LC-MS. For BDE-209 analysis, samples were extracted by ultrasonic extraction in 1 : 1 hexane:acetone and analyzed by GC-HRMS. PUFs were cut into two fractions and extracted by Soxhlet extraction, using methanol for OPE analysis, and dichloromethane for BDE 209 analysis.

BDE 209 and 14 OPEs (TCEP, TCIPP, TDCIPP, TBOEP, TPhP, oTMPP, CDP, TDBPP, TnPP, ip-TPP, TEHP, TEP, TBP and EHDPP) were analysed. Full compound names and identifiers are given in Table S2. Standards are given in Table S3.

2.4 QA/QC and data analysis

Field blanks for air, consisting of a precleaned PUF, were collected in each sampling period. For dust, three clean filters were used as field blanks. Details on blanks are given in Table S4. Method detection limits (MDLs) were calculated based on the average of the field blank + 3*standard deviation, and all sample masses above the MDL were corrected based on the average of the field blanks. FR masses accumulated on PUF disks were converted to concentrations following the GAPS template[26] based on a baseline sampling rate for indoor PUF without housing from Vojta *et al.*,[27] the average indoor temperature of the car during each period, and the specific duration of each sampling period, to ensure comparability across seasons. Air concentrations were assumed to represent bulk air. More information is in Text S3.

2.5 Modelling of vehicle conditions

To evaluate whether our measured concentrations reflected equilibrium conditions in vehicles, we used two-compartment partitioning models to calculate dust concentrations based on measured air concentrations. The gas-phase concentration of FRs was estimated from the temperature-adjusted octanol–air partitioning coefficient (K_{OA}), particle organic matter content, and total suspended particles in vehicle air.[28] Subsequently, these estimated gas-phase air concentrations were used to estimate a concentration associated with settled dust under equilibrium conditions, based on the K_{OA}, density of dust particles and organic matter associated with settled dust.[29] A detailed description can be found in Text S4, Fig. S7 and S8, Table S5. We also performed a sensitivity analysis on the modelled distributions, also detailed in Text S4.

3 Results

Of the 15 FRs examined, BDE 209 and 12 OPEs were detected in at least one matrix: TCEP, TCIPP, TDCIPP, TBOEP, TPhP, oTMPP, CDP, TEP, EHDPP, TDBPP, TnPP and TBP. ip-TPP and TEHP were not detected in any matrix. EHDPP, TDBPP and TnPP were detected in only 1 or 2 samples at low levels and are not considered further in the analysis.

3.1 Concentrations in car air

Four FRs (TCEP, TDCIPP, TEP and BDE 209) were detected in vehicle air in winter (Fig. 1a), with the highest concentration for TCEP at 23 ng m^{-3}. In summer, nine OPEs were detected in vehicle air (TCEP, TCIPP, TDCIPP, TBOEP, TPhP, oTMPP, CDP, TEP and TBP). TCEP and TCIPP were detected at the highest concentrations (16 000 ng m^{-3} and 4200 ng m^{-3}, respectively). There was a clear influence of seasonal temperature differences on vehicle air concentrations of OPEs; in summer, OPEs were 7–97000 times higher than in winter vehicle air (Fig. 1a and Table S6). These changes were most pronounced for TCEP (23 ng m^{-3} in winter *vs.* 16 000 ng m^{-3} in summer) and for TCIPP (<32 ng m^{-3} in winter *vs.* 4200 ng m^{-3} in summer). TEP, the most volatile of the measured compounds, had the smallest winter-summer difference: 8.8 ng m^{-3} in winter *vs.* 62 ng m^{-3} in summer.

3.2 Concentrations in car dust

Two dust samples per experimental period were analyzed, dust from the start and end of the sampling periods (Table S1 and

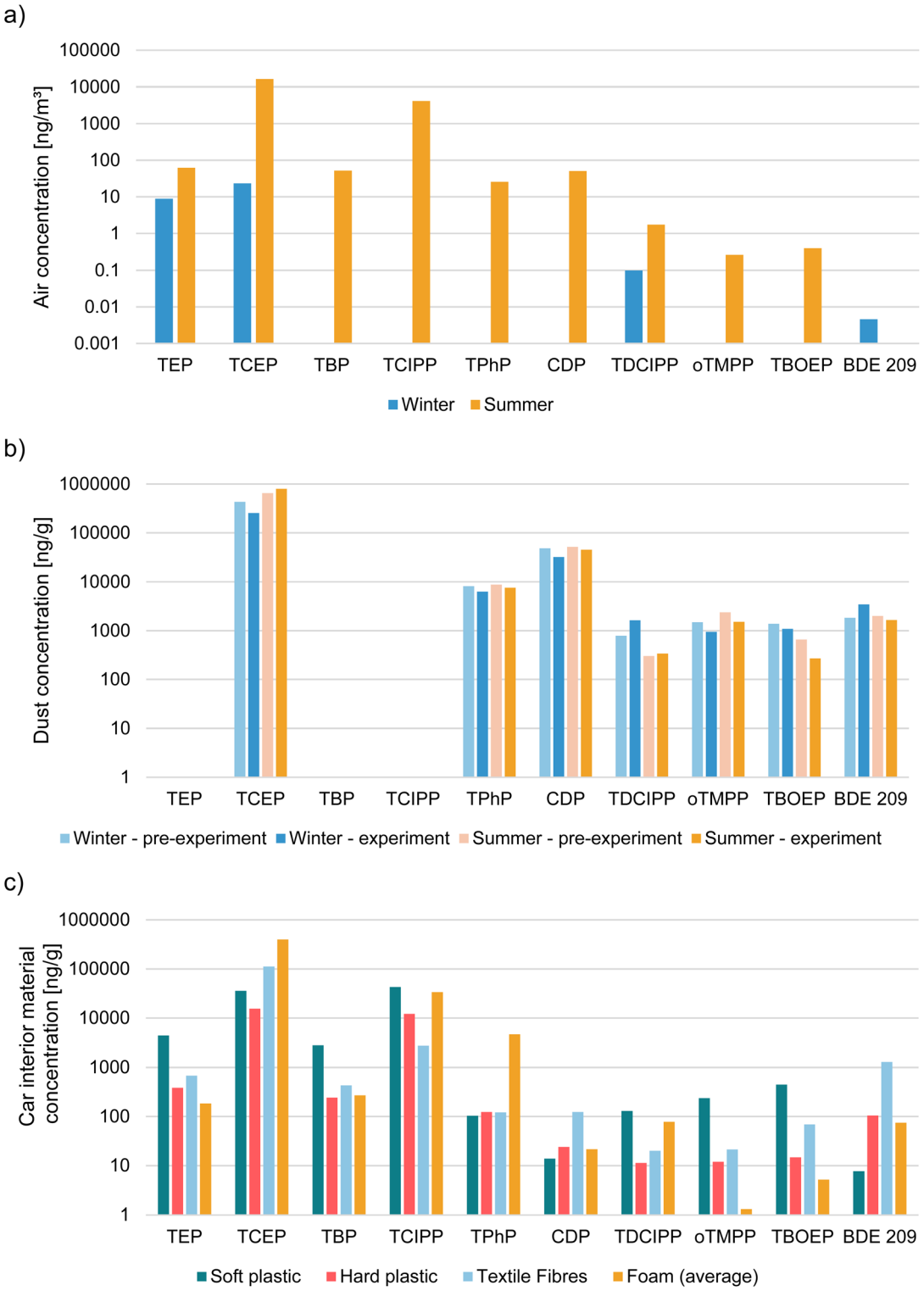

Fig. 1 Concentrations of OPEs and BDE 209 in: (a) car air in winter and summer, (b) car dust in winter and summer, (c) car interior materials. Compounds ordered from lowest K_{OA} to highest. In panel (b) the "pre-experiment" sample is dust collected at start of experimental period, and "experiment" sample is dust collected at end of experimental period.

Fig. 1b). However, all calculations of seasonal differences and partition modelling were based on the second samples (*i.e.* Winter – sample 2 and Summer – sample 2), which coincide with the experiment duration.

Seven FRs were detected in car dust: TCEP, TDCIPP, TBOEP, TPhP, oTMPP, CDP and BDE 209. The highest concentration was found for TCEP, with concentrations of 260 000 ng g^{-1} in winter and 790 000 ng g^{-1} in summer, which was also the

largest summer increase (Fig. 1b). Concentrations in dust had smaller variations between winter and summer than air concentrations; concentrations did not substantially vary for CDP, TPhP and oTMPP (Fig. 1b and Table S7); BDE 209 was 2× lower in summer than in winter, TDCIPP 4.8× lower and for TBOEP 4.1× lower.

3.3 Concentrations in car materials

Eight FRs were detected in car interior materials (TCEP, TCIPP, TDCIPP, TPhP, CDP, TEP, TBP and BDE 209, Fig. 1c). Of these, TCEP, TCIPP, TPhP, TEP, TBP and BDE 209 were quantified at levels above >1 μg g^{-1} in at least one of the sampled materials. The highest levels of FRs were found in foam, with maxima of 640 000 ng g^{-1} TCEP, 76 000 ng g^{-1} TCIPP and 5200 ng g^{-1} TPhP; followed by fibres, with 110 000 ng g^{-1} TCEP, 2700 ng g^{-1} TCIPP and 1300 ng g^{-1} BDE 209. Soft plastic dashboard pieces contained 43 000 ng g^{-1} TCIPP and 36 000 TCEP, and hard plastic contained 15 000 ng g^{-1} TCEP and 12 000 ng g^{-1} TCIPP.

TCEP and TCIPP were detected in concentrations >1 μg g^{-1} in all parts of the car examined, while TPhP only in foam (median 4600 ng g^{-1}) and BDE 209 only in fibres (1300 ng g^{-1}), suggesting different specific uses in the vehicle. The concentrations of FRs also varied in individual foam samples; for example, TCEP had higher concentrations in the rear of the front seats and TCIPP had 10× higher concentrations in the rear seat cushions (Table S8), indicating heterogeneity even within the same material types from the same car.

While many products contained high FR levels, the FR content needed to impart flammability is typically higher than the levels we observed; additive FR concentrations to impart flammability are typically in the range of 5–25%,[30] but individually the FR concentrations in our vehicle materials only reached a maximum of 0.06%. However, individual FRs may be used in combination; for example, the plastics analysed had similar amounts of both TCEP and TCIPP, suggesting intentional combined use, and the flammability rating of products may have been achieved by the inclusion of other FR compounds that were not a part of our target methods.

4 Discussion

4.1 Air-dust distributions

For OPEs in summer, measured dust concentrations of TDCIPP, TBOEP, TPhP, oTMPP, and CDP were substantially lower than those predicted based on air concentrations at the average summer vehicle temperature of 33.22 °C. Only TCEP had a higher measured dust compared to the predicted dust concentration (17× higher). In winter, measured dust concentrations were consistently higher than predicted. The predicted dust concentrations had strong variations based on temperature changes in the car, which was not observed in the measured dust, which exhibited more consistency across seasons/temperature conditions. This suggests that vehicle dust does not immediately equilibrate with air concentrations, but rather reflects longer-term vehicle conditions, combining the influence of air-dust partitioning, abrasion and direct partitioning.[31,32] TCEP alone was consistently higher in measured dust than predicted values, suggesting a particular source to dust that may not be freely exchangeable with air, *e.g.*, abrasion, which is plausible given that TCEP was the highest concentration FR in both textiles and foam (Fig. 1c). Predicted dust concentrations of BDE-209 suggested similar emission pathways: in winter BDE-209 was predicted to be <100 ng g^{-1} in dust, yet the measured concentration was 3400 ng g^{-1}. This, along with the high concentrations of BDE-209 detected in textiles, suggested abrasion as a plausible source of BDE-209 to dust.

4.2 Seasonal influences

Our study showed a clear seasonal difference in air and dust concentrations of most FRs in vehicle environments, particularly in vehicle air.

In air, this effect has also been noted by Hoehn *et al.*[22] during real vehicle use at ambient temperatures between −5 and 30 °C, finding 2–5 times higher air concentrations of TEP, TiBP, TNBP, and TCIPP in summer *vs.* winter. Hoehn *et al.*[22] calculated that the concentration of TCIPP in air increased by an average of 12% for each 1 °C increase in ambient temperature, and attributed this to the emission of TCIPP from vehicle foam, as median TCIPP concentrations were higher in cars where TCIPP was detected in the foam, especially in summer. A further contribution to the enhanced air concentrations of OPEs in vehicle air could be photo-oxidation of the polymer matrix of car interior surfaces, particularly those with substantial exposure to solar radiation in summer, *e.g.*, in dashboard plastics. BDE-209 showed an opposite effect, with higher concentrations in air during colder sampling;[22] we observed a similar pattern, as BDE-209 was only detected in air in winter. This may be due to debromination of BDE-209 in summer.[20,33,34] Temperature influences in vehicle air have also been supported by chamber tests: Lexen *et al.*[24] reported a 700-fold increase of TDCIPP in the car air during a whole car chamber test with temperature increase from 25 to 80 °C, and Wensing *et al.*[21] similarly reported increasing concentrations of TDCIPP and TCIPP in vehicle air with increasing temperatures.

The elevated air concentrations in vehicles, particularly at summer temperatures, suggest the potential importance of inhalation exposure to FRs in such environments. Inhalation exposure is often discounted in FR exposure estimates, despite evidence that it can be an important pathway for selected OPEs.[14,35,36] Inhalation of gaseous and particle FRs may make a more substantial contribution to overall exposure for those exposed to summer vehicle environments.

In dust, TCEP had higher dust concentrations in summer *vs.* winter, suggesting increasing emission of a source to dust at higher temperatures. In contrast, TDCIPP and TBOEP had summer concentrations in dust 4× lower than winter, suggesting shifting partitioning from dust to air. BDE 209 had 2× higher concentration in dust in winter than in summer, similar to the pattern observed in air. This aligns with the higher winter concentrations of BDE-209 in dust reported by Dubowski *et al.*,[20] attributed to debromination of BDE-209 in dust during summer.

TDCIPP, TBOEP, TPhP, oTMPP and CDP had relatively low concentrations in dust in summer compared to the prediction for the average exposure temperature (33.22 °C). This was likely because the air responded rapidly to temperature increases in the vehicle, and more volatile compounds (*e.g.*, TEP) equilibrated more quickly within the vehicle, while less volatile compounds (*e.g.*, BDE-209) had limited response to temperature changes. In contrast, the equilibration between air and dust for many of these intermediate volatility chemicals (*e.g.*, TDCIPP, TBOEP, TPhP, oTMPP and CDP) took longer, as has been noted in other indoor environments.[28,37] Thus, the high dust concentrations predicted based on air concentrations were not observed in real vehicle conditions.

4.3 Sources of FRs

The high concentrations of TCEP in car parts suggested intentional use of TCEP as an FR in vehicle parts, which, through temperature-driven emission from vehicle parts, translated into elevated concentrations in car air and dust. TCIPP was also present in all tested vehicle materials at high concentrations, and had a clear increase in summer air concentration (<32 ng m^{-3} in winter and 4200 ng m^{-3} in summer), suggesting a source with strong seasonally-impacted temperature differences triggering emission to air in summer, *e.g.*, the soft plastic dashboard covering, which contained 43 000 ng g^{-1} and was exposed to direct sunlight with potential for photo-oxidation of the polymer matrix itself, as well as some of the highest temperatures inside the vehicle. BDE 209 was primarily found in dust, and we presume that an important source in the vehicle was textile fibres (1300 ng g^{-1}), particularly given that the lower variations in concentrations between winter and summer suggested abrasion as an emission pathway. The primary source of TPhP in the vehicle was likely foam (4400–5200 ng g^{-1}), which influenced both dust concentrations and the increase in air concentration in summer. CDP had very low concentration in car materials, but concentrations in the dust up to 52 000 ng g^{-1}, suggesting an interior source that we did not sample, such as rubber floor mats or carpeted wall sections, was contributing to dust concentrations.

OPEs appear to be the dominant FRs used in vehicles since the early 2000s.[2,25,38–45] TCIPP is one of the most frequently reported OPEs, with medians of 27 000 to 630 000 ng g^{-1} in fabrics, foams and shredded materials.[46,47] In most cases, foam contained higher concentrations of OPEs than fabrics.[47] Brominated FRs have also been consistently detected in car parts and end-of-life vehicles.[48–51] As in our study, car textiles have been typically associated with the highest concentrations of BDE 209,[49,51,52] with median BDE 209 concentrations in seat fabrics of 3600–48 033 000 ng g^{-1}.[34,49,51–54]

Overall, studies identify a lack of consistency in the composition and levels of FRs in car materials. Large ranges exist in FR concentrations in vehicle components, and compounds detected in one study are below detection limits in another. For example, Vojta *et al.*[55] did not detect BDE 209 in any vehicle plastic, textiles or foam, while many other studies, the current study included, have consistently reported BDE 209 at levels above >1 μg g^{-1} in car textiles, foam and coating. Hoehn *et al.*[22] confirmed the presence of BDE 209 in only one foam sample out of 52 cars, while Shin and Baek[54] reported concentrations of BDE 209 in vehicle foam and fabrics in all samples ($n = 5$). This makes generalizations of sources and emission profiles in vehicles difficult, and consequently limits our ability to estimate human exposure to FRs from vehicle materials and indoor environments.

4.4 Limitations

This experiment covered only one car, and thus, the concentrations of individual FRs and seasonal patterns may differ in other vehicles, although trends are in line with what has been identified in other studies to date. The age of the car (2015 model) may also affect the mixture of FRs present, although vehicle age has not been found to be a strong determinant of FR levels in vehicles.[25] While temperature was constantly monitored during sampling periods, UV radiation, also expected to differ between winter and summer, was not quantitatively assessed within the car interiors. Samples of car materials, particularly the soft and hard plastics, were small, which can introduce some higher uncertainties when converting to concentrations by mass. This experiment represents a stationary car; ventilation is expected to change air concentrations after starting and while driving. Lexén *et al.*[24] found that use of air conditioning decreased particulate phase chemicals by 40–80% and gas phase chemicals by over 90%. In contrast, Staaf and Östman[16] found the opposite; no decrease in TCIPP and TEP in a car with "normal" ventilation.

5 Conclusion

This study investigated the seasonal effects on dust and air concentrations of selected currently used FRs (BDE 209 and OPEs) in one car, in the context of the FR concentrations in car interior materials. We found high concentrations of BDE-209 and chlorinated OPEs in car air, dust and materials. All quantified OPEs had an increase in air concentrations in summer, while dust concentrations were less affected. TCEP was found in the highest concentrations in all matrices, and in air had very high concentrations with a strong influence of temperature.

The indoor temperature of a vehicle has the potential to significantly impact exposure to FRs. High exposures can be expected in the first moments of using a car, especially before ventilation/air conditioning has decreased interior concentrations. Due to the high concentrations in dust and material, dermal contact is also a likely exposure route, especially for TCEP, TCIPP and BDE 209. These high vehicle exposures are expected to most affect people who spend a lot of time in their cars, such as taxi and delivery drivers, and who use their cars regularly, for example, those with long daily commutes who also park in open, uncovered parking lots during the day. Proper ventilation before and during driving could help to reduce exposures, especially during warm months, as could regular vacuuming and use of reflective/thermal insulating screens to reduce the temperature inside parked cars.

Author contributions

PS: conceptualization, investigation, formal analysis, writing – original draft, writing – review and editing, SRJ: methodology, investigation, supervision, writing – review and editing, JK: methodology, investigation, OA: methodology, investigation, LM: conceptualization, supervision, resources, formal analysis, writing – review and editing.

Conflicts of interest

There are no conflicts to declare.

Data availability

The data supporting this article have been included as part of the supplementary information (SI). Supplementary information is available. See DOI: https://doi.org/10.1039/d5va00228a.

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