Science of the Total Environment 806 (2022) 150494

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Emerging and legacy flame retardants in indoor air and dust samples of Tarragona Province (Catalonia, Spain)

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HIGHLIGHTS

- Indoor levels of novel FRs were higher than legacy FRs in Tarragona province.
- Levels of OPFRs were higher than HFRs in indoor air and dust.
- TCIPP was the most found FR in both indoor air and dust samples.
- Risk assessment from obtained levels were close to potential risk of FR exposure.

GRAPHICAL ABSTRACT

ARTICLE INFO

Article history:
Received 13 April 2021
Received in revised form 14 September 2021
Accepted 17 September 2021
Available online 24 September 2021

Editor: Damià Barceló

Keywords:
Flame retardants (FRs)
Indoor air
Dust
Organophosphate FRs (OPFRs)
Novel brominated FRs (NBFRs)
Tarragona Province (Catalonia Spain)

ABSTRACT

Flame retardants (FRs) are widely used in consumer products including furniture foam and electronic equipment such as computers, monitors and TVs. Over time, FRs can easily migrate into the surrounding environments. Since brominated FRs (BFRs) has been determined of high concern due to their environmental persistence, bioaccumulation and potential toxicity, novel FRs have emerged. The present study was aimed at identifying and quantifying the indoor levels of 41 legacy and novel FRs, which include 20 OPFRs and 21 HFRs (8 PBDEs, 3 HBCDDs, 5 NBFRs and 5 DECs) in Tarragona Province (Catalonia, Spain). The results have confirmed the presence of both legacy and novel FRs in air and dust of homes, schools and offices. To the best of our knowledge, this is the first European study measuring OPFRs at office environments and also confirming the presence of the following OPFRs: TEP, TCIPP, T2IPPP, TPPO, DCP, TMCP and B4IPPPP in indoor air, even some of them at high levels. OPFRs in general and TCIPP in particular showed high concentrations in air (94,599 pg/m^3 and 72,281 pg/m^3, respectively) and dust (32,084 ng/g and 13,496 ng/g, respectively) samples collected in indoor environments. HBCDDs were found at high levels in dust (32,185 ng/g), whereas the presence of PBDEs and DECs were low in both matrices (<160 pg/m^3 in air and <832 ng/g in dust). NBFRs showed higher levels than the two legacy FRs groups, which is supported by the current restrictions of these FRs (640 pg/m^3 in air and 1291 ng/g in dust). Samples of schools had significantly lower levels of NBFRs, but significantly higher concentrations of HFRs in air than in home samples, while dust levels of HFRs were significantly lower than those in samples of offices. Regarding human health risks, the current assessment suggests that those derived from exposure to FRs were lower -although close- to assumable risks, evidencing the potential of FRs for non-carcinogenic and carcinogenic risks, mainly due to the exposure to TCIPP, which was the main contributor together with ΣHBCDDs and also EHDPP.

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https://doi.org/10.1016/j.scitotenv.2021.150494

1. Introduction

Flame retardants (FRs) are a group of chemicals that are added to a wide variety of consumer products in order to prevent ignition and/or to reduce the spread of an already initiated fire (Björnsdotter et al., 2018). FRs are present in common items and goods such as foam and plastics used in textiles, electric and electronic equipment (i.e.: computers, monitors, printers and TVs), furniture and upholstery, construction and refrigeration products, as well as paints (de la Torre et al., 2020a). There are several families of FRs, both organic and inorganic compounds, based on bromine (e.g., high brominated aromatic and cycloaliphatic compounds), chlorine (e.g., chloroparaffins and dechhloranes (DECs) like dechlorane plus (DP)), phosphorus (e. g., phosphate-esters, phosphonates and phosphinates, ammonium phosphate), nitrogen (e. g., melamine and melamine derivatives), boron (e.g., sodium borate, borax and zinc borates), and metallic hydroxides. The choice of the appropriate FR, or their combinations, depends on the material compatibility, costs, desired performance for the final product, and the flammability standard (Cristale et al., 2016). Some FRs are used as additives, physically blended into rather than chemically bonded to the original materials. It facilitates the migration into the surrounding environment -via abrasion or volatilization- during products lifetime (Li et al., 2019). Due to their persistence and resistance to degradation, some FRs can accumulate in indoor dust, air, water, foods, and sediments (Domínguez-Morueco et al., 2018; Ekpe et al., 2020; Wu et al., 2020). Moreover, FRs can be biomagnified along the food chain, being present in human biosamples (Klinčić et al., 2020; Lee et al., 2020).

Representative brominated FRs (BFRs), also known as legacy halogenated FRs (HFRs), are hexabromocyclododecanes (HBCDDs) and polybrominated diphenyl ethers (PBDEs), which have been widely used primarily in three commercial mixtures: Penta-, Octa-, and Deca-BDEs. Their potential of neurotoxicity, carcinogenicity and/or thyroid endocrine disruption (Wang et al., 2019) led (European legislation) to the ban or restriction at concentrations lower than 0.1% by mass of Penta- and Octa-BDE in 2004, and of Deca-BDE in 2008 (Decision 2005/717/EC). In turn, Penta- and Octa-BDE were added to the POPs list under the Stockholm Convention in 2011, Deca-BDE in 2017 (UNEP, 2017), and HBCDD in 2013, being phased out in many countries in 2016, excepting China (Shi et al., 2018). Since then, and in response to the market demand and safety standards, the use of emerging FRs (EFRs) has increased. In recent years, novel BFRs (NBFRs), DECs, and organophosphate FRs (OPFRs) have even overtaken the use of PBDEs and HBCDD (Lee et al., 2020). Although EFRs are generally less persistent and bioaccumulative than HFRs, the safety of these compounds for environment and humans still needs to be elucidated (de la Torre et al., 2020a). Some NBFRs exhibit similar structure, lower vapor pressure and water solubility, as well as higher octanol-water coefficient than PBDEs. It suggests that these chemicals could have potential for bioaccumulation and toxicity (Ezechiáš et al., 2014). In fact, recent animal studies suggested that many NBFRs impaired endocrine system in rats (Guigueno and Fernie, 2017), and altered gene expression and transcriptional response, caused disruption of thyroid axis and decreased the fecundity in zebrafish (Wang et al., 2019).

With respect to OPFRs, non-halogenated phosphorus compounds have proven to be a better alternative according to costs and performance basis (Pantelaki and Voutsa, 2019). Regarding toxic potential, tris(2-chloroethyl) phosphate (TCEP) and tris(1,3-dichloropropyl) phosphate (TDCIPP), for example, were classified as carcinogens by the EU under Regulation (EC) N 1272/2008. On the other hand, triphenylphosphate (TPHP) and tri-n-butyl phosphate (TNBP) have been reported to be neurotoxic in zebrafish larvae and in rats (Behl et al., 2015; Shi et al., 2018). Furthermore, tris(2-butoxyethyl) phosphate (TBOEP) caused developmental toxicity in zebrafish (Han et al., 2014), while TDCIPP exhibited neurotoxicity in PC12 cells (Ospina et al., 2018). In addition, some OPFRs were suggested to be associated with the prevalence of asthma, atopic dermatitis and allergic rhinitis in humans (Araki et al., 2014).

Since FRs can enter into the body via inhalation, dermal contact, and/or unintentional ingestion, these chemicals -present in indoor environments- are a source of potential human health risks (Chen et al., 2020). Inhalation and ingestion are the main routes for human exposure (Rantakokko et al., 2019). Particularly vulnerable groups are children and toddlers, who are especially exposed to dust when crawling and playing on the floors, as result of hand-to-mouth behaviour, as well as due to their comparatively lower body weights (Cao et al., 2019). Some exposures to FRs have been also associated with developmental problems (Wei et al., 2015). Therefore, monitoring the presence of novel and legacy FRs families in air and dust of indoor environments is clearly necessary in order to establish the risks of human exposure, especially for sensitive population groups.

Recent investigations have confirmed the presence of both NBFRs and OPFRs -as well as PBDEs- in European indoor environments (i.e., homes, offices, schools and workplaces) (Kademoglou et al., 2017; Shoeib et al., 2019), even at higher concentrations than those found in sediments, sludge, soils and foods (Wang et al., 2019; Yadav et al., 2019). In Spain, recent studies have reported high levels of some OPFRs, such as TPHP in indoor dust (Björnsdotter et al., 2018; Velázquez-Gómez and Lacorte, 2020). In fact, higher levels of OPFRs -in comparison to the levels of PBDEs- have been reported (Cristale et al., 2016; de la Torre et al., 2020b). Reche et al. (2019) found levels of PBDEs in Spanish indoor air and dust lower than those reported in other European countries. Independently of these results, the presence of novel FRs remains still unclear in Spanish indoor environments. To the best of our knowledge, to date there is no data on the existence/levels of OPFRs in indoor air environments of Spain. Thus, this study was aimed at identifying and quantifying in Tarragona Province (Catalonia, Spain), the indoor levels of 41 FRs (both legacy and novel), which include 20 OPFRs and 21 HFRs (8 PBDEs, 3 HBCDDs, 5 NBFRs and 5 DECs). The differences in the levels of FRs in houses, offices and schools as well as source profiles were assessed. Finally, human exposure to these FRs and the associated health risks were also evaluated.

2. Materials and methods

2.1. Chemicals and materials

All chemicals and materials used in this study are listed in Supplementary Material- Section 1.

2.2. Sampling sites and matrices

To determine the levels of selected FRs in indoor environments of Tarragona Province, both air and dust samples were collected. Five offices, 5 schools and 10 homes were selected for sampling. A questionnaire on the characteristics of each room and the building in which sampling was conducted, as well as the room content, was filled (Table 1). Sampling was conducted between November 2019 and January 2020, taking into account that in winter, windows are closed and the air circulates through ventilation systems.

2.2.1. Indoor air sampling

Air samples were collected at 20 sampling sites. In each sampling site, two low volume active personal pumps (Leland Legacy, SKC Inc., Eighty Four, PA) were deployed for 24 h. Each sampler was connected to a sampling train with two cartridges. For HFRs sampling, polyurethane foam plugs (PUF) and glass fiber filter (GFF) cartridges (SKC, 22 × 110 mm, VERTEX Technics, Barcelona, Spain) were used, being adjusted at 12 L/min of total air flow rate (6 L/min per sampling train), which sampled a total volume of approximately 17.3 m^3. Regarding OPFRs, SPE cartridges (Isolute ENV+, 200 mg, 6 mL, Biotage, Uppsala, Sweden) were used, with an air flow rate of 5 L/min (2.5 L/min per

Table 1
Characteristics of indoor environments.

Sample	ID	Type	Location	Floor	Room area (m^2)	Electronic devices[a] (n items)	Wood furniture[b] (n items)	Plastic elements[c] (n items)	Synthetic textile elements[d] (n items)	Sampling date
House 1	H1	House	Santa Coloma de Queralt	2	20	3	11	0	6	31/10/19
House 2	H2	House	Santa Coloma de Queralt	2	14.5	3	6	0	0	24/11/19
House 3	H3	House	Santa Coloma de Queralt	0	12	1	10	0	8	23/11/19
House 4	H4	House	Sant Ramon	0	20	3	13	0	7	08/01/20
House 5	H5	House	Tarragona	1	15	2	3	1	1	26/11/19
House 6	H6	House	Cambrils	0	20	5	8	0	0	07/11/19
House 7	H7	House	Santa Coloma de Queralt	2	35	1	14	1	2	18/01/20
House 8	H8	House	Cambrils	1	25	5	9	0	1	07/12/19
House 9	H9	House	Santa Coloma de Queralt	3	42	7	10	0	7	25/12/19
House 10	H10	House	Santa Coloma de Queralt	2	30	6	11	0	6	11/01/20
Office 1	O1	Office	Tarragona	3	30	2	7	5	5	03/12/19
Office 2	O2	Office	Santa Coloma de Queralt	0	32	3	5	3	3	02/12/19
Office 3	O3	Office	Santa Coloma de Queralt	0	65	13	9	14	14	13/01/20
Office 4	O4	Office	Reus	3	28	11	9	6	6	23/12/19
Office 5	O5	Office	Tarragona	3	45	13	29	9	9	10/12/19
School 1	S1	School	El Morell	0	25	27	52	0	0	06/11/19
School 2	S2	School	Perafort	0	45	29	40	2	0	07/11/19
School 3	S3	School	Constantí	1	60	4	49	5	0	13/11/19
School 4	S4	School	Reus	1	18	1	19	0	0	14/11/19
School 5	S5	School	La Pobla de Mafumet	0	45	22	43	2	0	12/11/19

[a] Electronic devices include TVs, computers, printers ….
[b] Wood furniture include chairs, tables, shelfs … made of wood.
[c] Plastic elements include chairs, tables, furniture … made of plastic.
[d] Synthetic elements include pillows, matrices … made of plastic-derived textiles.

sampling train) and a volume of approximately 7.2 m^3. All samples were stored at −20 °C and protected from light until subsequent analyses.

In turn, $PM_{2.5}$ and PM_{10} were measured employing Dylos DC1100 Pro Air Quality Monitor (Particle Counter) in order to estimate both gas and particulate phases of air. Table S1 summarizes the ranges of concentrations >0.5 μm and >2.5 μm recorded at the selected environments and the estimated gas and particulate phases.

2.2.2. Deposited dust sampling

Dust samples in the different indoor environments (Table S2) were collected from surfaces that were elevated more than 0.5 m from the ground floor (i.e., tables, desks, shelves, furniture) using a natural fiber broom and metal dustpan. In order to avoid cross contamination of FRs from collection devices, these were cleaned up with water and dried in an oven at 60 °C between samplings. Samples were then transferred to clean glass vials and stored at −20 °C (protected from light) until extraction. A total of 9 out 20 dust samples were collected at sufficient quantities to perform analytical determinations: 2 offices, 3 schools and 4 homes.

2.3. Chemical characterization of FRs

The procedures used for the chemical analysis of FRs was based on previously reported methods (Barón et al., 2012, 2014; Giulivo et al., 2016).

2.3.1. Sample pre-treatment, extraction and clean up

Surrogate standards were isotopically labelled analogues of selected analytes. In total, 19 different isotopically labelled standards were used for analyzing 41 FRs (Table S3).

For HFRs extraction in air samples, PUFs and GFFs of the same cartridges were extracted together in closed glass tubes and spiked with 3 ng of 13C-PBDEs mixture and 3.2 ng of 13C-*syn*-DP mixture. After being kept at 4 °C overnight, extraction was done with 20 mL of n-hexane/acetone (1:1, v/v) in an ultrasonic bath for 20 min. This procedure was performed twice for each sample, being then both extracts combined. Subsequently, samples were dried under a gentle stream of nitrogen before the reconstitution with 5 mL of *n*-hexane. Samples were purified by solid phase extraction (SPE) using Al—N cartridges (Biotage, Uppsala, Sweden) (5 g) and eluted with 20 mL of hexane: DCM (1:2 v/v). Finally, samples were dried with a stream of nitrogen and reconstituted with 40 μL of toluene.

OPFRs extraction of air samples was carried out according to Tao et al. (2019). Samples of cartridges were spiked with 15 ng of labelled OPFR mixture and kept at 4 °C overnight in order to equilibrate before extraction. Samples were eluted with 10 mL of acetone. At each sampling point, the two SPE cartridges were collected, being their corresponding extracts combined. After drying under a gentle stream of nitrogen, samples were reconstituted with 200 μL of methanol.

Concerning to dust samples, about 1 and 0.5 g of dust were spiked with 3 ng of ^{13}C-PBDEs mixture and 3.2 ng of ^{13}C-*syn*-DP, as well as with 40 ng of labelled OPFR mixture, respectively. Samples were kept at 4 °C overnight to equilibrate until extraction.

For the extraction of HFRs from dust samples, these were ground with copper (1:2) and neutral alumina (1:2), and loaded into a 22 mL extraction cell containing 6 g of neutral alumina. Dead volume was filled with hydromatrix. Samples were simultaneously extracted and cleaned through pressurized liquid extraction (PLE), which was carried out using an Accelerated Solvent Extraction (ASE) 350 system (Dionex, Sunnywale, CA, USA), with hexane:DCM (1:1) as solvent at a pressure of 1500 psi and 100 °C. Extracts were concentrated to incipient dryness under a gentle stream of nitrogen and re-dissolved with toluene for a final volume of 40 μL.

Regarding OPFRs extraction in dust, sample was ground with copper (1:1) and loaded into a 22 mL extraction cell. Dead volume was filled with hydromatrix. Samples were extracted with hexane:acetone (1:1) at a pressure of 1500 psi and 100 °C. Extracts were then dried under a gentle stream of nitrogen, being samples reconstituted with 500 μL of methanol.

2.3.2. Instrumental analysis

PBDEs and NBFRs (HBB, DBDPE, PBEB, EH-TBB and BEH-TEBP) analyses were performed on a GC Agilent 7890A equipped with a DB-5 ms column and coupled to an Agilent 7000B triple quadrupole mass spectrometer, using electronic ionization (EI). The instrumental conditions and the spectrometric determination were previously described (Barón et al., 2014; Eljarrat et al., 2007). Due to their low sensitivity with GC-EI-MS-MS, BDE-209 and DBDPE were analyzed by GC–MS

with the same chromatographic conditions in an Agilent 5975A mass spectrometer, using negative ion chemical ionization (NICI) (Eljarrat and Barceló, 2004). The analysis of DECs was performed by GC-NICI-MS-MS as previously reported (Barón et al., 2012). After analysis of the previous HFRs, extracts were spiked with 1 ng of labelled HBCDDs and re-dissolved in 40 μL of methanol to analyze HBCDDs, using a Thermo Scientific LC-MS/MS system with instrumental conditions previously reported (Guerra et al., 2010).

For OPFR analyses, online sample purification and analysis were performed with a Thermo Scientific TurboFlow™system, consisting of a triple quadrupole (QqQ) MS with a heated-electrospray ionization source (H-ESI), two LC quaternary pumps and three LC columns, two of them for purification and another one for separation. The TurboFlow™ purification columns here used were the following: Cyclone™-P (0.5x50mm) and C18-XL (0.5 × 50 mm). Chromatographic separation was subsequently achieved using an analytical column Purosphere Star RP-18 (125 mm × 0.2 mm), with a particle size of 5 μm (Giulivo et al., 2016). Selective reaction monitoring (SRM) details are summarized in Supplementary Material (Tables S4–S8).

2.4. Quality assurance/quality control

Quality Assurance/Quality Control (QC/QA) procedures were followed throughout sampling and analysis, including analytical instrumentation calibration and use of certified patterns. Instrumental parameters such as recoveries, relative standard deviations (RSDs), limits of detection (LODs) and limits of quantification (LOQs) are summarized in Tables S9 and S10, for air and dust samples, respectively. Analytical parameters for dust were previously reported (Barón et al., 2012; Barón et al., 2014; Giulivo et al., 2016; Guerra et al., 2010; Olivero-Verbel et al., 2021), while analytical parameters for air samples were defined in the current study. Recovery tests were conducted with spiked samples, being tests done by triplicate. Recoveries ranged between 41 and 119% for all FRs, always being within the range of acceptability (40–120%) for analytical methods, based on quantification by isotopic dilution. For each batch of samples, a blank was added. These blanks included the PUF + GFF cartridges for HFR analyses, and the SPE cartridges for OPFR analyses. Thus, the potential contamination due to the sampling system was assessed. Blank levels were subtracted from those of the corresponding samples.

2.5. Risk assessment

For human health risk assessment of FRs, exposure through air inhalation (Inh) and dust ingestion (Ing) was calculated for adult and children (Eqs. (1) and (2)):

$$Inh = \frac{\sum (Cair_i \times Tf_i)}{BW} \quad (1)$$

$$Ing = \frac{\sum (Cdust_i \times Tf_i)}{BW} \quad (2)$$

where $Cair_i$ and $Cdust_i$ are the concentrations of FRs in air and dust in environment i; Tf_i is the time fraction of day expended in each environment i; and BW is the body weight. Exposure was calculated for adults (>18 years old) and children (6–12 years old). For adults, a time distribution of 8 h in offices and 16 h at home was assumed, while for children, a distribution of 6 and 18 h was considered for schools and homes, respectively. Calculations were done by means of a Montecarlo simulation (100,000 iterations) using Oracle Crystalball software (version 11.1.2.4.850), based on the propagation of parameters variability and uncertainty given by probability function. Parameters and their probabilistic distributions are shown in Table S11.

Non-carcinogenic and carcinogenic risks were assessed considering the total EDI (sum of inhalation and dust ingestion exposure) of adults and children, according to Eqs. (3) and (4).

$$Non\ carcinogenic = \frac{EDI}{RfD} \times \frac{ED}{AT} \quad (3)$$

$$Carcinogenic\ risk = EDI \times SFO \times \frac{ED}{AT} \quad (4)$$

where ED is the exposure duration, 30 and 6 years for adults and children, respectively; AT is the average duration: 30 and 6 years for adults and children, respectively, for non-carcinogenic (non-CR), and 70 years for carcinogenic risks (CR) (U.S. EPA, 1989; U.S. EPA, 2008). In turn, RfD is the oral reference dose and SFO is the oral slope factor. Only FRs with defined RfD and SFO, according to the U.S. Environmental Protection Agency (USEPA), as well as data from the literature (Brommer et al., 2012; Chen et al., 2020; He et al., 2018; Li et al., 2018; Van den Eede et al., 2011) were included in the current assessment (Table S12). In fact, since RfD is established, but there is no SFO for TEP, TPHP, EHDPP, ΣHBCDDs, and DBDPE, only non-CRs were calculated for these FRs.

2.6. Statistical analysis

The statistical analysis was conducted using IBM SPSS Statistics 22 (Chicago, IL). Values below the limit of detection (LOD) were considered as zero. In order to get an overview of the presence of FRs in indoor environments of Tarragona Province, outliers were included in the statistical analysis leading to avoid bias when mean values are compared. Normality in levels of FRs was tested by using the Kolmogorov-Smirnov test. Differences among levels of FRs were assessed using *t*-test or ANOVA test, followed by a subsequent post-hoc test (Bonferroni and T3 Dunnet tests, for normal and non-normal values, respectively). Finally, two-tailed bilateral correlations were used considering Pearson and Spearman coefficients for normal and non-normal samples, respectively. A p-value <0.05 was considered as statistically significant.

3. Results and discussion

3.1. General information

In this study, 41 FRs were measured in 20 indoor environments of Tarragona Province. The levels of FRs were individually assessed and also grouped according to their respective families: ΣOPFRs and ΣHFRs. The last family included ΣPBDEs, ΣHBCDDs, ΣNBFRs and ΣDECs. The restrictions worldwide on legacy FRs, which have been implemented in recent years, have led to OPFRs use gradually being overtaken (Chupeau et al., 2020). This has also been reflected in the current results. Interestingly, ΣOPFRs levels were significantly ($p < 0.001$) higher -more than 100-times- than the concentrations of ΣHFRs in air samples (94,599 vs. 806 pg/m^3) (Fig. 1a). Individual concentrations of OPFRs were high, which agrees with their volatility properties. In spite of the considerable list of measured OPFRs and their high concentrations, similar levels of ΣOPFRs and ΣHFRs were found in dust samples (32,084 and 34,306 ng/g, respectively) (Fig. 1b). It is due to the very high levels of ΣHBCDDs found in dust samples, which were not observed in air. HBCDD are FRs with low to moderate potential to volatilize are banned since 2016 in Europe, but not in China (Shi et al., 2018). The purchasing products from countries like China, and mainly the presence of these chemicals in furniture and devices before the restrictions, together with the persistence of HFR in indoor environments would explain the presence of these compounds in the current dust samples.

Differences between physic-chemical properties of FRs may lead to variability in the presences of FRs in air (Table 2) and dust (Table 3). As expected, all sampling environments exhibited differences in FRs profiles because of variability in the features, year of construction, and

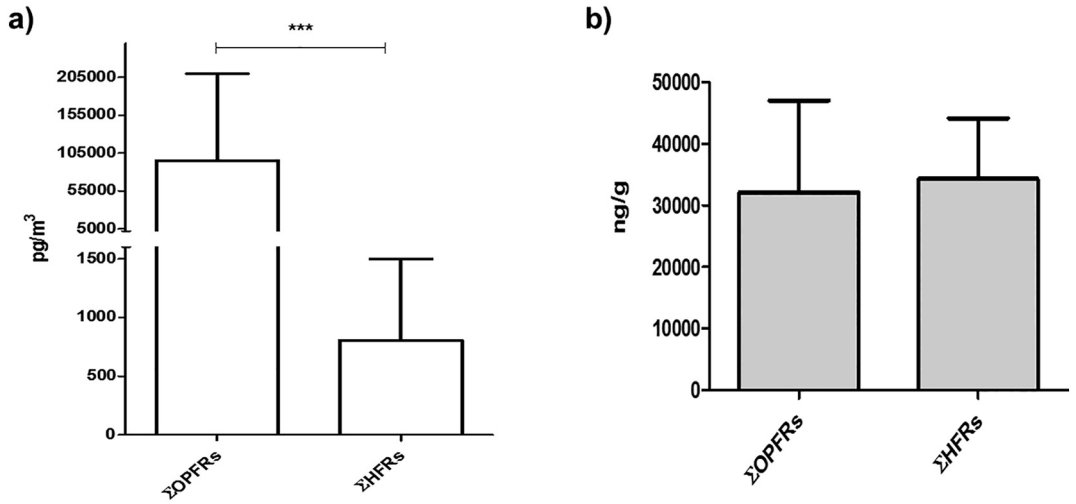

Fig. 1. Levels of ΣOPFRs and ΣHFRs in (a) air (pg/m^3) and (b) dust (ng/g). ***Significant differences between groups existed at $p < 0.001$.

construction materials (for example) (Tables S13 and S14). Among all FR analyzed, 4 (TPP, PBEB, Dec-602 and Dec-603) were not detected in any sample, either air or dust.

3.2. Levels of FRs in indoor air

Regarding OPFRs (Table 2), TPP, TBOEP, 2IPPDPP, RDP, 4IPPDPP, IDPP and THP were below the LOD in all samples (n = 20). On the other hand, although the detection frequency of some OPFRs (TDCIPP, TPHP and EHDPP) in air was below 25%, they presented high levels. In contrast, TMCP and B4IPPPP, also found in a few samples, showed the lowest concentrations (9.2 pg/m^3 and 9.4 pg/m^3, respectively). TCIPP showed the highest mean levels (72,281 pg/m^3), followed by TEHP (15,214 pg/m^3) and T2IPPP (2601 pg/m^3). These three chemicals presented significant ($p < 0.05$) higher levels than the rest of OPFRs, excepting TNBP (1388 pg/m^3). Nowadays, TCIPP is employed in many consumer products due to its lower cost in comparison to other OPFRs such as TDCIPP, which is used when a special high degree of flame retardancy is required (Zuiderveen et al., 2020). Other significant differences existed between the other OPFRs, ranging 50 and 1400 pg/m^3 (Fig. S1).

In relation to HFRs, BDE-154, BDE-153, BDE-183, β-HBCDD, PBEB, Dec-602, and Dec-603 were not detected in any air sample (n = 20), while HBB, α-HBCDD and γ-HBCDD were detected only in one school at low concentrations (<2 pg/m^3). Furthermore, detection frequencies of Dec-604, *syn*-DP and *anti*-DP were 15%, 5% and 10%, respectively, with concentrations below 5 pg/m^3. PBDEs exhibited lower levels -below 71 pg/m^3- than almost all OPFRs. The PBDE with the highest mean value was BDE-47 (70.3 pg/m^3), followed by BDE-209 (45.8 pg/m^3). BDE-28 showed the lowest concentrations (3.32 pg/m^3), being significantly ($p < 0.05$) lower than those of the remaining PBDEs, with the exception of BDE-100 (8.6 pg/m^3). Similarly, in a recent study also conducted in Spain, BDE-209 was the most detected PBDE in air, contributing 34% to ΣPBDEs (de la Torre et al., 2020a).

NBFRs showed lower detection frequencies than PBDEs, but at higher concentrations -from 100 to 343 ng/m^3- for all compounds, excepting HBB (1.72 ng/m^3). Significant ($p < 0.05$) differences were noticed between HBB vs. EH-TBB and DBDPE, as well as between EH-TBB vs. BEH-TBP. Gas/particle partition of each FR in indoor air was calculated according to Sühring et al. (2016). Results are shown in Table S1. The used equations considered the air total suspended particles (TSP) (Table S1) concentration and particle-gas partition coefficient of each flame retardant. It, in turn, considers octanol/air partition coefficient of each flame retardant taken from Chemspider database (http://www.chemspider.com/) and a fraction of organic matter in TSP of 0.40 (Sánchez-Soberón et al., 2019; Sühring et al., 2016). Equation and parameters are shown in Supplementary Material- Section 2. Lighter FRs such as TEP, TECP TCIPP, and TNBP were mainly present (>98%) in gas phase, while heavier FRs such as T2IPPP, TEHP, BDE-209, as well as NBFRs (EH-TBB, BEH-TBP, DBDPE) and *syn*-DP, were present mainly (>92%) in the particle phase. It should be highlighted that smokers' homes (H2 and H10) showed levels of suspended particles above 0.5 μm, which were 20 times higher than those found in non-smokers indoor environments. This influences the fraction of FRs in air particulate, being higher in the smokers' homes, but not showing the highest concentration of the total FRs levels in air.

Grouping FRs by families, ΣOPFRs registered significant ($p < 0.01$) higher levels (94,599 pg/m^3) than other families like ΣPBDEs (159 pg/m^3), ΣHBCDDs (1 pg/m^3), ΣNBFRs (640 pg/m^3) and ΣDECs (5 pg/m^3) (Fig. 2a). In general terms, the current results are in good accordance with the replacement of PBDEs and HBCDDs by OPFRs and NBFRs, as result of policy intervention (Drage et al., 2020) in recent years.

3.3. Levels of FRs in dust of indoor environments

In general, some differences were observed in the concentration profile of FRs in dust samples (Table 3). We detected more FRs in dust than in air, including TBOEP, 2IPPDPP, RDP, 4IPPDPP, IDPP, BDE-153, BDE-183, BDE-154 and β-HBCDD. In contrast, Dec-604, which was detected in air samples, showed levels below LOD in all dust samples. With respect to OPFRs, TEP (n = 2), B4IPPPP (n = 3), TBOEP (n = 2) and IDPP (n = 1) were only detected at levels below 30.8 ng/g. Similarly to the results obtained in air, TCIPP, which has a lower cost than other OPFRs, was the most found OPFRs in dust (13,496 ng/g), followed by T2IPPP (5055 ng/g). These two OPFRs presented significantly ($p < 0.01$) higher levels than the rest of OPFRs (B4IPPPP, TBOEP, 4IPPDPP, TEP, TNBP, and IDPP). TDCIPP, TCEP, TPHP and EHDPP levels ranged from 2263 to 3447 ng/g. The differences between OPFRs levels are depicted in Fig. S2.

With respect to HFRs, HBB and *syn*-DP were found in three (1 office and 2 schools) and two (2 offices) samples, respectively, at values below 7.4 ng/g. In relation to PBDEs, BDE-209 exhibited significant ($p < 0.05$) higher levels (774 ng/g) than BDE-28, BDE-100, BDE-154, BDE-153 and BDE-183, all of them with levels below 6.3 ng/g. This, together with the results from air, suggests that BDE-209 was the most used PBDEs in furniture and devices among the environments here investigated.

The three measured HBCDDs in dust samples (α-HBCDD, β-HBCDD and γ-HBCDD) had levels ranging from 3801 to 18,818 ng/g, with β-HBCDD and γ-HBCDD showing significant differences ($p < 0.01$). In

Table 2
Air concentrations (pg/m^3) of FRs in indoor environments (n = 20).

	Detection frequency (%)	Mean	Standard deviation	Minimum	Median	P95	Maximum
TEP	55	1026	2023	<LOD	76.0	7472	7570
TCEP	90	1251	1532	<LOD	539	6199	6342
TPPO	60	98.0	164	<LOD	34.5	648	667
TCIPP	85	72,281	111,033	<LOD	35,541	390,435	390,863
TPP	0	<LOD	–	–	–	–	–
TDCIPP	15	165	499	<LOD	<LOD	1729	1741
TPHP	25	146	516	<LOD	<LOD	2211	2315
TNBP	90	1388	1009	<LOD	1155	4461	4562
DCP	50	52.1	68	<LOD	15.9	212	213
TBOEP	0	<LOD	–	–	–	–	–
2IPPDPP	0	<LOD	–	–	–	–	–
RDP	0	<LOD	–	–	–	–	–
4IPPDPP	0	<LOD	–	–	–	–	–
TMCP	25	9.4	21.5	<LOD	<LOD	71.5	71.8
EHDPP	10	356	1559	<LOD	<LOD	6634	6976
B4IPPPP	20	9.2	23.7	<LOD	<LOD	90.1	91.8
IDPP	0	<LOD	–	–	–	–	–
T2IPPP	85	2601	2545	<LOD	1762	10,674	10,976
THP	0	<LOD	–	–	–	–	–
TEHP	85	15,214	16,739	<LOD	8228	61,986	63,265
∑OPFRs	95	94,599	115,117	<LOD	58,330	419,252	419,809
BDE-28	85	3.3	2.2	<LOD	3.0	9.3	9.5
BDE-47	95	70.3	47.0	<LOD	58.1	242	249
BDE-100	95	8.6	4.8	<LOD	7.5	23.3	23.6
BDE-99	100	31.3	15.9	13.6	27.0	78.8	79.8
BDE-154	0	<LOD	–	–	–	–	–
BDE-153	0	<LOD	–	–	–	–	–
BDE-183	0	<LOD	–	–	–	–	–
BDE-209	60	45.8	53.1	<LOD	43.0	198	202
∑PBDEs	100	159	88.2	59.1	139	409	413
α-HBCDD	5	0.2	0.9	<LOD	<LOD	3.9	4.1
β-HBCDD	0	<LOD	–	–	–	–	–
γ-HBCDD	5	0.8	3.6	<LOD	<LOD	ND	16.2
∑HBCDDs	5	1.01	4.50	<LOD	<LOD	19.3	20.3
PBEB	0	<LOD	–	–	–	–	–
HBB	5	1.7	7.7	<LOD	<LOD	32.7	34.4
EH-TBB	85	343	482	<LOD	55.6	1564	1591
BEHTBP	35	101	152	<LOD	<LOD	403	403
DBDPE	60	195	200	<LOD	203	653	661
∑NBFRs	100	641	684	7.49	278	2005	2014
Dec-602	0	<LOD	–	–	–	–	–
Dec-603	0	<LOD	–	–	–	–	–
Dec-604	15	5.1	12.7	<LOD	<LOD	41.3	41.7
syn-DP	5	1.1	4.7	<LOD	<LOD	19.9	20.9
anti-DP	10	1.1	3.6	<LOD	<LOD	14.0	14.3
∑DECs	15	5.1	12.7	<LOD	<LOD	41.3	41.7
∑HFRs	100	806	691	139	564	2187	2192

<LOD: below limit of detection; P95: percentile 95th.

comparison to HBCDDs, NBFRs showed lower levels. The levels of both DBDPE and BEH-TBP (753 and 508 ng/g, respectively) were significantly higher than those of EH-TBB and HBB. When Deca-BDE mixture was banned, DBDPE was introduced as an alternative to be used in materials used in homes and workplaces (McGrath et al., 2018). In turn, BEH-TBP and EH-TBB were introduced as alternatives for Penta-BDE mixture (Tao et al., 2016). Although no specific data on the production and use of these NBFRs are currently available, substantial production is likely, which would agree with the present results.

Finally, in the DEC group, *syn*-DP and *anti*-DP were found at very low concentrations (2.2 and 6.5 ng/g respectively). DP, currently an alternative for Deca-BDE mixture, has been used as a FR since the 1960s in a wide range of applications such as electronic wiring and cables, plastic roofing materials, and hard plastic connectors in televisions and computer monitors, and furniture (Sharkey et al., 2020). DEC levels below the detection limits can be explained either due to the degradation during the use of the products, or when DP is released into the environment, because of the exposure to air, light, and other coexisting agents (Zhu et al., 2007). However, uncertainties remain when interpreting their transport and fate. Concentration profiles vary across environments due to changes in mass, number, and emission characteristics of the possible FRs sources used in building materials and consumer products (Fromme et al., 2014b). According to FRs families, ΣHBCDDs and ΣOPFRs registered similar levels (32,185 and 32,084 ng/g, respectively), being significantly ($p < 0.05$) higher than those of DECs (8.8 ng/g) and PBDEs (831 ng/g), but not than those of NBFRs (1291 ng/g) (Fig. 2b).

3.4. Correlations and associations between FRs levels and indoor environmental parameters

Some correlations between FRs of the same family in air (Fig. S3) and dust (Figs. S4 and S5) were observed due to the co-occurrence and complementary use of these compounds in same furniture and devices, as well as because of the similarities in the purchase time of the consumer products (Dodson et al., 2012). Other correlations were found, but, due to the lack of data on the use of particular FRs and mixtures in furniture, textiles or devices, it was difficult to establish the reason for these relationships. BDE concentrations were correlated between them in both analyzed matrices, being BDE-100, BDE-47 and BDE-99 significantly and positively correlated ($r > 0.8$, $p < 0.01$), while BDE-28 and BDE-209 correlated significantly and negatively in air ($r < -0.5$, $p < 0.05$).

Table 3
Dust concentrations (ng/g) of FRs in indoor environments (n = 9).

	Detection frequency (%)	Mean	Standard deviation	Minimum	Median	P75	Maximum
TEP	33	12.7	21.6	<LOD	<LOD	34.5	55.9
TCEP	100	3255	7839	79.3	475	1478	24,104
TPPO	100	415	556	19.0	116	925	1550
TCIPP	89	13,497	27,933	415	3261	13,366	86,754
TPP	0	<LOD	–	–	–	–	–
TDCIPP	89	3447	7066	137	519	4680	21,536
TPHP	89	2566	3829	343	1518	2225	12,615
TNBP	89	20.3	16.0	<LOD	20.0	31.9	53.0
DCP	100	933	775	196	772	1096	2846
TBOEP	22	11.8	34.6	<LOD	<LOD	1.8	104
2IPPDPP	78	48.5	71.2	<LOD	31	64.2	228
RDP	78	89.5	103	<LOD	35	184	292
4IPPDPP	78	12.2	14.3	<LOD	9.0	19.1	45.7
TMCP	100	173	162	41.4	107	283	539
EHDPP	56	2263	6544	<LOD	14.0	310	19,711
B4IPPPP	33	3.3	5.2	<LOD	<LOD	8.3	13.6
IDPP	11	30.8	92.3	<LOD	<LOD	<LOD	277
T2IPPP	100	5055	2788	1319	5352	7844	9304
THP	0	<LOD	–	–	–	–	–
TEHP	67	252	309	<LOD	150	575	837
∑OPFRs	100	32,084	44,874	5822	17,231	35,887	148,215
BDE-28	100	0.5	0.5	0.1	0.2	0.6	1.7
BDE-47	78	12.1	11.4	<LOD	6.8	24.6	31.2
BDE-100	100	4.5	5.0	0.4	2.0	9.7	14.0
BDE-99	100	22.8	28.6	2.4	11.7	30.8	94.2
BDE-154	100	4.4	7.0	0.4	1.7	5.1	22.3
BDE-153	100	6.2	8.6	0.9	3.4	7.6	28.1
BDE-183	100	6.2	6.4	1.2	4.7	7.5	22.2
BDE-209	100	774	651	142	660	919	2368
∑PBDEs	100	831	664	183	668	1009	2444
α-HBCDD	100	9566	9081	267	5944	20,133	24,717
β-HBCDD	78	3801	4727	<LOD	942	9089	11,745
γ-HBCDD	100	18,818	15,384	1085	13,524	37,600	40,623
∑HBCDDs	100	32,185	28,716	1352	18,438	69,057	71,365
PBEB	0	<LOD	–	–	–	–	–
HBB	33	7.40	19.7	<LOD	<LOD	4.2	59.6
EHTBB	100	22.7	16.5	3.30	20.4	37.4	49.4
BEHTBP	100	753	554	137	861	1291	1540
DBDPE	100	508	568	36.7	375	844	1793
∑NBFRs	100	1291	807	190	1523	2060	2282
Dec-602	0	<LOD	–	–	–	–	–
Dec-603	0	<LOD	–	–	–	–	–
Dec-604	0	<LOD	–	–	–	–	–
syn-DP	22	2.2	5.5	<LOD	<LOD	2.6	16.5
anti-DP	67	6.5	9.5	<LOD	2.8	15.3	26.1
∑DECs	100	8.8	807	190	<LOD	<LOD	2282
∑HFRs	100	34,307	29,633	2644	21,169	72,569	74,799

<LOD: <limit of detection; P75: percentile 75th.

Interestingly, DCP was negatively correlated with DBDPE and TNBP in air samples ($r = -0.5$, $p < 0.01$) (Fig. S3).

On the other hand, regarding dust samples, correlations ($r > 0.678$, $p < 0.05$) between FRs of the same family were found for OPFRs (2IPPDPP- 4IPPDPP, 2IPPDPP- IPP, 4IPPDPP-IPP), PBDEs (BDE-99-BDE-154, BDE-99-BDE-153, BDE-154-BDE-153, BDE-100-BDE-154), and NBFRs (EH-TBB, BEH-TBP), as well as HBCDDs (α-HBCDD-β-HBCDD, α-HBCDD-γ-HBCDD, β-HBCDD-γ-HBCDD) (Figs. S4 and S5). Moreover, ΣNBFRs significantly correlated with ΣHFRs in air samples ($r = 0.95$, $p < 0.01$), while significant correlations existed in dust samples between ΣPBDEs and ΣOPFRs, ΣNBFRs and ΣHFRs ($r > 0.70$, $p < 0.01$), as well as between ΣHBCDDs and ΣHFRs ($r = 0.97$, $p < 0.001$) (Figs. S4 and S5). Finally, the levels in air were significantly and positively correlated to the concentrations in dust only for BDE-28 and BDE-47 ($r > 0.60$, $p < 0.05$).

In general terms, the area of the room where FRs were collected was negatively correlated with FR levels. Notwithstanding, statistical significance was only reached for the levels of TCPP, α-HBCDD and γ-HBCDD in dust. In contrast, ΣDECs levels in air showed a significant -but weak- positive correlation with the room area ($p = 0.46$; $p < 0.05$). This indicates that high levels of FRs were present in small rooms and vice-versa, which might be related with less ventilation in small rooms, leading to accumulation of indoor chemicals.

The presence of electronic devices, wood furniture, plastic furniture and synthetic textiles in the environments was positively associated only with some FRs levels in dust. The number of electronic devices significantly and positively correlated with RDP, β-HBCDD and BDE-47; wood furniture with RDP, BDE-100 and BDE-154; plastic furniture with α-HBCDD and DBDPE; while the number of synthetic textiles correlated with TDCIPP and DBDPE ($r > 0.68$, $p < 0.05$). A similar tendency was also observed when considering FRs families in dust samples. Thus, the number of electronic devices positively correlated with ΣOPFRs, ΣHBCDDs and ΣHFRs ($r > 0.670$, $p < 0.05$); while the number of plastic elements was positively correlated with ΣNBFRs and ΣHFRs ($r > 0.670$, $p < 0.05$) (Figs. S4 and S5). The variability in the correlations might be associated with emission characteristics of these FRs as sources.

Few significant differences in the levels of FRs depending on the type of indoor environment (homes, offices and schools) were found. In air samples, offices showed significant ($p < 0.05$) lower levels of TNBP and TEHP than those detected in homes (586 vs 1954 pg/m^3, respectively) and schools (4754 vs 16,506 pg/m^3, respectively) (Table 4). In contrast, all the three measured HBCDDs in dust samples were higher

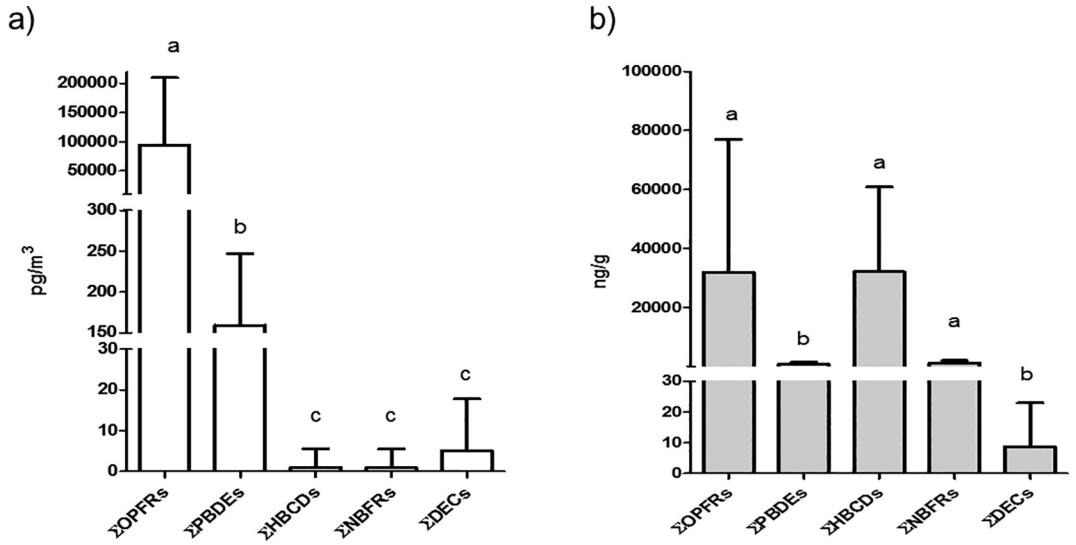

Fig. 2. Levels of FRs families (ΣOPFRs, ΣPBDEs, ΣHBCDs, ΣNBFRs and ΣDECs) in a) air (pg/m^3) and dust (ng/g) samples. Data not showing a common superscript (a, b, c) indicate significant differences between them.

in offices than in homes, being the differences significant for α-HBCDD (20,803 vs 2455 ng/g, $p < 0.05$) and γ-HBCDD (38,523 vs 7830 ng/g, $p < 0.01$) (Table 5). The comparatively low levels of HBCDDs found at homes may be justified by the less content of electronic devices than that observed in other environments. However, it would not explain the higher concentrations of other FRs. With respect to FRs families, the levels of ΣNBFRs in air were significantly higher at homes than in schools (950 vs 231 pg/m^3). In addition, ΣHFRs differed significantly between sampling environments in a different way. Thus, air levels at homes were significantly ($p < 0.05$) higher than those in schools (1145 pg/m^3 vs 379 pg/m^3), while dust levels at homes were lower ($p < 0.001$) than those found in offices (12,009 ng/g vs 72,128 ng/g).

3.5. Comparison of FR levels in the present study with European levels in air and dust samples

Tables 4 and 5 summarize FRs levels in indoor air and dust, which have been reported for various European countries for the last 5 years. The current levels have been compared with countries from Europe since the FR legislation, building materials employed, furniture and devices are more similar in comparison to the non-European countries. Table 4 shows levels of FRs measured at indoor air in Spain (this study; Reche et al., 2019), Norway (Cequier et al., 2014; Tay et al., 2017), Ireland (Wemken et al., 2019), UK (Tao et al., 2016) and Germany (Fromme et al., 2014a), at homes, offices, schools, and/or day-care centers. To the best of our knowledge, the present investigation is the first one in Europe confirming the presence of a number of OPFRs in air: TEP, TCIPP, and T2IPPP -even at high levels- as well as TPPO, DCP, TMCP and B4IPPPP at low levels (about 100 pg/m^3 or below). Moreover, this has been the first study measuring OPFRs in air at offices in Europe. A detailed comparison between concentrations is given in Supplementary Material-Section 3. The differences between countries can be due to factors such as sampling procedure variation in furniture profiles, electronic devices, ventilation, or the year of construction, among others.

Table 5 summarizes the presence of FRs in indoor dust samples in various European countries (Spain, Italy, Finland, Ireland, Germany, Netherlands, Norway, Portugal and UK) (Simonetti et al., 2020; Rantakokko et al., 2019; Reche et al., 2019; Wemken et al., 2019; Larsson et al., 2018; Sugeng et al., 2017; Tay et al., 2017; Coelho et al., 2016; Cristale et al., 2016; Tao et al., 2016; Xu et al., 2016; Cequier et al., 2014). It is important to notice that methodology of dust sampling differs from the other authors which needs to take it into account in detailed comparison of levels in Table 5. In the present study, it was done by using natural fibres broom and metal dustpan from superficies instead of vacuum cleaner due to technical reasons in the collection of high amounts needed for analysis. Sweeping has been described as an alternative for house dust collection in areas where vacuum collection is not feasible for the analysis of chemicals as pesticides (Birn et al., 2012) and methodology in our study allowed the detected 37 out 41 measured FRs. In relation to OPFR and for the very first time in Europe, we detected TEP, 4IPPDPP and B4IPPPP (at levels <25 ng/g), as well as TPPO, DCP, RDP, TMCP, 2IPPDPP, IDPP and T2IPPP, the last one above 3000 ng/g in indoor dust.

3.6. Human exposure and risk assessment

Due to the wide use of FRs, a comprehensive analysis on human exposure and risk assessment is clearly needed. In addition to indoor-inhaled FRs, these compounds can be unintentionally ingested through dust in different indoor environments (Shi et al., 2018). Total (inhaled and ingested) daily intakes (EDI) of FRs for adults (men and women) ≥18 years old, and children (boys and girls) aged between 6 and 12 years old, were estimated. Spending time at home leads to a higher air exposure for adults and children for all FRs (data not shown). Nevertheless, dust exposure of almost FRs was higher at offices and schools in comparison to homes, for adults and children, respectively. When total exposure was considered, spending time at home affected mainly the OPFRs: TEP, TNBP, DCP, 2IPPDPP, 4IPPDPP, TMCP, B4IPPPP, T2IPPP, TEHP, as well as the PBDEs: BDE-28, BDE-47, BDE-99, BDE-100, BDE-183, and EH-TBB, BEH-TBP and Dec-604, in both adults and children.

In general, exposure levels were higher in children than in adults (Table 6). Despite the lower inhalation rate for children with respect to adults, the high rates of dust ingestion and time fraction at home, and the lower body weight, lead to higher -approximately twice- EDIs in children. According to the above results, TCIPP showed the highest mean EDI than any other FR, with values of 19.5 and 29.3 ng/kg/day for adults and children, respectively, and with a contribution of air inhalation above 75%. In fact, TCIPP values corresponded to approximately one-half of the total estimation for ΣOPFRs. Taking into account the FRs families, ΣOPFRs showed the highest exposure (30.4 and 59.6 ng/kg/day for adults and children, respectively), followed by ΣHBCDDs (9.10 and 23.2 ng/kg/day for adults and children, respectively). The other FRs families presented lower levels, but were similar between them: ΣNBFRs: 0.60 and 1.68 ng/kg/day for adults and children, respectively, ΣPBDEs: 0.31 and 0.88 ng/kg/day for adults and

Table 4
Levels (pg/m^3) of FRs in indoor air in present study and in studies performed in Europe.

	Present study			Reche et al. (2019)	Wemken et al. (2019)			Tay et al. (2017)		Tao et al. (2016)		Xu et al. (2016)		Fromme et al. (2014a)	Cequier et al. (2014)	
Year	2020			2019	2019			2017		2016		2016		2014	2014	
Country	Spain			Spain	Ireland			Norway		UK		Norway		Germany	Norway	
Location	Homes	Offices	Schools	Home	Homes	Offices	Schools	Stationary	Personal	Homes	Offices	Stationary	Personal	Daycare centres	Homes	Schools
n	10	5	5	7	28	28	31	60	13	15	20	58	31	63	48	6
Levels	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Median	Median	Mean	Mean	Mean
TEP	1948	34	176													
TCEP	856	1603	1692									3000	3000	<2200	3.2	7.0
TPPO	76.1	107	133													
TCIPP	28,760	114,724	116,882													
TDCIPP	174	313	<LOD									<LOD	<LOD		0.4	0.1
TPHP	44.3	497.0	<LOD									1000	<LOD	<LOD	0.4	0.1
TNBP	1954	586	1056									<LOD	<LOD	4000	9.7	3.1
DCP	41.9	73.3	52.5													
TBOEP	<LOD	<LOD	<LOD									<LOD	<LOD		1.5	18.6
TMCP	7.2	19.1	4.2													
EHDPP	13.9	1395	<LOD												0.5	1.0
B4IPPPP	9.2	11.5	7.1													
T2IPPP	3329	2926	819													
TEHP	16,507	4754	23,091									<LOD	<LOD			
BDE-28	3.7	2.9	2.9					<1.5	<LOD	22	43,894				9.53	7.99
BDE-47	79.1	58.2	64.6	1.94	44,078	160	43,866	44,046	43,924	120	44				178	178
BDE-100	10.0	6.6	7.8	0.6				<0.11	<LOD	44	3				10.9	9.1
BDE-99	37.3	22.9	27.8	1.	37	48	43,960	<0.25	43,959	130	43,870				40.5	26.3
BDE-154	<LOD	<LOD	<LOD	0.3				<0.11	<LOD	14	0.8				2.3	0.6
BDE-153	<LOD	<LOD	<LOD	0.3				<0.20	<LOD	24	0.6				7.6	0.7
BDE-183	<LOD	<LOD	<LOD	0.2	0.8	0.5	0.5	<0.95	<LOD	44,045	<1.0				11.4	<LOD
BDE-209	54.3	34.2	40.4	3.6	880	420	1600	43,953	64	660	74				323	22.7
α-HBCDD	<LOD	<LOD	1		10	86	33	<0.16	43,927	43	44,051					
β-HBCDD	<LOD	<LOD	<LOD		43,954	42	160	<0.064	15	17	43,866					
γ-HBCDD	<LOD	<LOD	3		100	90	96	<0.18	43,869	270	55					
PBEB	<LOD	<LOD	<LOD					<LOD	<LOD	43,983	43,865				1.3	<LOD
HBB	<LOD	<LOD	6.9					<0.6	<LOD	11	19				12.4	4.2
EH-TBB	592	172	16.7					150	120	44,047	22					
BEHTBP	177	48.1	<LOD					44,046	20	10	43,863					
DBDPE	181	211	207	13.7	390	240	460	43,862	43,868	26	43,958					
Dec-602	<LOD	<LOD	<LOD	0.4												
Dec-603	<LOD	<LOD	<LOD	0.7												
Dec-604	10.2	<LOD	<LOD													
syn-DP	2.1	<LOD	<LOD	10.4												
anti-DP	1.4	<LOD	1.6	1.2												

children, respectively, and ΣDECs: 0.01 ng/kg/day for both adults and children. Inhalation was a predominant route only for ΣOPFR, with around 65% of contribution, while dust ingestion was the dominant pathway for ΣPBDEs (87%), ΣHBCDD (100%), ΣNBFRs (67%) and ΣDECs (60%). Previous studies reported that the EDI of PBDEs from dust was lower than the RfD established by the USEPA (Cequier et al., 2014). In the present study, we observed low EDI of PBDEs, probably related to the decrease in their use in recent years.

The results of the current investigation show that, individually, all FRs having defined toxicological values, showed acceptable human health risks based on the non-CR and CR (Table 6). TCIPP and EHDPP had the highest individual non-CR, both below 0.01.The mean non-CR values for ΣHBCDDs for adults, and especially for children (0.05 and 0.12, respectively), were close, but still below the limit value. Moreover, 95th percentile non-CR due to exposure to ΣHBCDDs was set at 0.14 and 0.35 for adults and children, respectively. Regarding CR in children, both ΣOPFRs and ΣTOTAL exhibited the highest levels, being $1.1 \cdot 10^{-7}$ and $3.3 \cdot 10^{-7}$ for mean and 95th percentile, respectively, which are below the levels considered as acceptable (10^{-6}) according to national and international standards.

The present risk assessment suggests that human exposure to FRs is of concern. The non-CR -considering all FRs exposure- showed values below, but close to the limit (HQ = 1), especially in children. As above indicated, TCIPP was the main contributor, together with ΣHBCDDs and also EHDPP. In turn, total carcinogenic risk was below the 10^{-6} threshold, for both ΣOPFRs and ΣHFR, being 100 times lower for the last one. The current assessment only accounts for the potential of FRs at homes, offices and schools. Other indoor environments, like cars (for example), contain high amounts of FRs, and therefore, are also potential important sources of these pollutants. Thus, our results can be considered as not negligible since they support the possible potential of FRs for both non-CR and CR due to indoor exposure. On the other hand, human exposure through diet means another important source of FRs intake additional to inhalation and dust ingestion, mainly for HFRs (Gbadamosi et al., 2021). If in the current results, dietary exposure is also included, health risks can increase, being closer to the limit values. Furthermore, the fact that FRs could exhibit combined effects in mixture is not taken into account.

4. Conclusions

The results of the present study have confirmed the presence of legacy and novel FRs in both air and dust of homes, schools and offices of Tarragona Province (Catalonia, Spain). To the best of our knowledge, this is the first study in Europe focused on measuring OPFRs in air at offices, and also in having detected OPFRs in indoor air (TEP, TCIPP, T2IPPP, TPPO, DCP, TMCP and B4IPPPP), even some of them at high levels. OPFRs in general and TCIPP in particular, showed high concentrations in both air and dust of indoor environments. High levels of HBCDD were found only in dust, while the presence of PBDEs and DECs was low in both matrices. NBFRs showed higher levels than these two legacy FRs groups, which is in agreement with the current restrictions of these FRs.

Table 5
Levels (ng/g) of FRs in indoor dust in present study and in studies performed across Europe.

	Present study			Simonetti et al. (2020)		Rantakokko et al. (2019)	Reche et al. (2019)	Wemken et al. (2019)			Larsson et al. (2018)	Sugeng et al. (2017)	Tay et al. (2017)	Coelho et al. (2016)	Cristale et al. (2016)			Tao et al. (2016)		Xu et al. (2016)	Cequier et al. (2014)	
Year	2020			2020		2019	2019	2019			2018	2017	2017	2016	2016			2016		2016	2014	
Country	Spain			Italy		Finland	Spain	Ireland			Germany	Netherlands	Norway	Portugal	Spain			UK		Norway	Norway	
Location	Homes	Offices	Schools	Homes	Workplace	Homes	Homes	Homes	Offices	Schools	Preschools	Homes	Homes	Homes	Homes	Schools	Theatres	Homes	Offices	Homes	Homes	Schools
n	4	2	3	7	5	40	7	29	31	32	100	14	60	28	5	4	3	45	47	61	48	6
Sampling	BD			VC		VC	QF-PM10	VC			VC	VC	VC	VC	VC			VC and NS		VC	VC	
Levels	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Mean	Median	Median	Mean	Mean	Mean	Mean	Mean	Mean	Median	Mean	Mean
TEP	18.0	<LOD	14.1																			
TCEP	249	695	8969			1110					<LOD	205		58						455	796	2080
TPPO	171	636	594																			
TClPP	1785	45,136	8019			3530					<LOD	3641									5740	1890
TPP	<LOD	<LOD	<LOD											3.0								
TDClPP	1920	11,076	398			1350					43,832	3752		60						1130	797	2730
TPHP	1374	6479	1546			1060					43,862	357		1000						1228	1240	2400
TNBP	25.1	14.7	17.5			62.6														<LOD	119	42.7
DCP	745	616	1396																			
TBOEP	<LOD	52.1	0.8			16,700					64									6796	18,700	89,800
2IPPDPP	73.8	7.9	41.7																			
RDP	12.9	114	175																			
4IPPDPP	17.5	4.6	10.3																			
TMCP	164	66.1	255																			
EHDPP	135	9856	37.7			953																
B4IPPPP	7.0	<LOD	<LOD																			
IDPP	69.2	<LOD	<LOD																			
T2IPPP	6282	3249	4624																			
TEHP	494	364	183			434														710		
BDE-28	0.4	0.1	0.7	<LOD		1.92							0.5	0.35	<LOD	<LOD	<LOD	44,075	44,077		1.0	0.4
BDE-47	7.4	7.0	21.7	44,001	44,073	75.9	32.1	26	16	9	43,991		22	12	43,897	<LOD	43,870	14	83		199	61.6
BDE-100	1.4	3.4	9.4	34.2	43,890	37.8	19.8				43,863		0.16	43,863	<LOD	<LOD	44,084	43,865	18		61.1	8.9
BDE-99	11.1	19.1	40.7	37.4	46.3	217	27.2	45	26	20	11		27	12	44,079	<LOD	43,937	31	100		348	45.8
BDE-154	1.0	3.1	9.7	43,927	43,956	18.2							44,044		<LOD	<LOD	<LOD	2	44,019		24.7	7.4
BDE-153	3.0	5.0	11.2	<LOD		52.6					<LOD		43,953	43,891	44,081	<LOD	44,017	44,047	28		39.7	11.9
BDE-183	8.7	4.2	4.4	43,933	73.7			43,834	11	2			43,985	43,866	33.4	<LOD	<LOD	43,928	29		18.2	7.3
BDE-209	514	1514	628	<LOD	236.80	674	942	58,000	4200	17,000	270	492	940	780	3526	710	4754	34,000	9100		6840	1120
α-HBCDD	2455	20,813	11,550					1500	520	680	57		94	200				2300	1100			
β-HBCDD	295	9044	4980					680	160	350	15		23	43				1100	300			
γ-HBCDD	7830	38,523	20,334					670	170	620	34		43	130				4900	630			
PBEB	<LOD	<LOD	<LOD			<LOD							0.51					43,892	2		0.3	0.02
HBB	<LOD	0.6	21.7	<LOD	63.6	609							0.43					44,044	14		1.5	1.6
EH-TBB	15.8	27.2	28.9			8.9					<LOD		25					21	120		16.1	2.7
BEHTBPE	634	756	911			243					80		300	13							132	99.9
DBDPE	233	1409	272			208	572	39,000	12,000	48,000	120		70	60				240	1600		8.73	13.4
syn-DP	<LOD	10.1	<LOD				169															
anti-DP	6.6	16.1	<LOD				892															

VC: Vacuum cleaner; QF-PM10vacuum pump connected to a 37 mm quartz fiber filter with a PM_{10} cutpoint; NS: nylon sample socks (25 μm pore size).BD: natural fibber broom and metal dustpan.

Table 6
Total (air inhalation plus dust ingestion) estimated daily intake (EDI) in adults and children and contribution of inhalation pathway.

	EDI- adults (ng/kg/day)								EDI- children (ng/kg/day)							
	Mean	SD	Min	P50	P75	P95	Max	% Inh[a]	Mean	SD	Min	P50	P75	P95	Max	% Inh[a]
TEP	0.33	0.46	0.00	0.19	0.39	1.08	14.0	99	0.76	1.04	0.01	0.45	0.88	2.40	32.5	97
TCEP	0.40	0.29	0.03	0.33	0.49	0.91	12.3	70	3.78	6.45	0.07	2.07	3.99	12.0	419	14
TPPO	0.12	0.18	0.00	0.07	0.14	0.37	12.6	18	0.42	0.52	0.01	0.27	0.49	1.23	27.3	11
TCIPP	19.5	18.7	0.85	14.4	23.4	49.8	740	75	29.3	24.2	1.70	22.9	35.6	71.0	956	85
TDCIPP	1.58	3.19	0.01	0.74	1.64	5.40	164	4	2.14	4.20	0.01	0.97	2.20	7.55	193	3
TPHP	0.99	1.79	0.01	0.51	1.06	3.32	91.4	5	1.93	2.03	0.05	1.33	2.36	5.45	58.5	1
TNBP	0.39	0.23	0.03	0.33	0.48	0.81	3.69	98	0.88	0.53	0.09	0.75	1.08	1.87	8.18	96
DCP	0.23	0.29	0.00	0.14	0.27	0.70	7.90	6	1.25	1.31	0.04	0.86	1.53	3.51	42.6	2
TBOEP	0.01	0.02	0.00	0.01	0.01	0.04	1.43	0	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0
2IPPDPP	0.02	0.04	0.00	0.01	0.02	0.06	3.90	0	0.09	0.18	0.00	0.04	0.10	0.30	24.0	0
RDP	0.01	0.03	0.00	0.01	0.02	0.05	0.97	0	0.07	0.08	0.00	0.05	0.09	0.21	2.10	0
4IPPDPP	0.00	0.01	0.00	0.00	0.00	0.01	0.46	0	0.02	0.03	0.00	0.01	0.02	0.07	2.03	0
TMCP	0.04	0.06	0.00	0.02	0.05	0.14	3.07	7	0.26	0.30	0.00	0.17	0.31	0.76	6.69	1
EHDPP	1.15	2.60	0.01	0.51	1.15	4.00	176	10	0.15	0.27	0.00	0.08	0.17	0.52	15.2	3
B4IPPPP	0.00	0.01	0.00	0.00	0.00	0.01	0.53	63	0.01	0.01	0.00	0.01	0.01	0.03	0.96	36
IDPP	0.01	0.05	0.00	0.00	0.01	0.05	5.85	0	0.07	0.18	0.00	0.02	0.06	0.27	11.3	0
T2IPPP	2.42	2.31	0.17	1.80	2.85	6.14	108	33	9.25	8.47	0.45	6.85	11.3	24.1	275	14
TEHP	3.28	3.99	0.11	2.14	3.83	9.57	127	97	9.17	9.06	0.66	6.68	10.6	23.6	305	97
ΣOPFRs	30.4	25.4	1.78	23.9	36.5	72.2	632	65	59.6	37.0	5.3	50.7	73.4	127	638	63
BDE-28	0.00	0.00	0.00	0.00	0.00	0.00	0.01	90	0.00	0.00	0.00	0.00	0.00	0.00	0.02	72
BDE-47	0.02	0.01	0.00	0.02	0.02	0.04	0.30	89	0.05	0.03	0.00	0.04	0.06	0.11	0.68	71
BDE-100	0.00	0.00	0.00	0.00	0.00	0.01	0.05	78	0.01	0.01	0.00	0.01	0.01	0.02	0.13	50
BDE-99	0.01	0.01	0.00	0.01	0.01	0.03	0.45	66	0.04	0.04	0.00	0.03	0.05	0.10	1.77	40
BDE-154	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0	0.00	0.01	0.00	0.00	0.00	0.01	0.50	0
BDE-153	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0	0.01	0.01	0.00	0.00	0.01	0.02	0.63	0
BDE-183	0.00	0.00	0.00	0.00	0.00	0.01	0.30	0	0.01	0.02	0.00	0.01	0.01	0.03	0.63	0
BDE-209	0.27	0.38	0.01	0.16	0.31	0.86	17.1	4	0.76	0.88	0.03	0.51	0.91	2.16	96.7	3
ΣPBDEs	0.31	0.40	0.02	0.19	0.35	0.92	16.3	13	0.88	0.87	0.05	0.63	1.06	2.37	24.4	10
α-HBCDD	2.61	3.38	0.03	1.61	3.12	8.10	117	0	6.35	7.30	0.10	4.17	7.71	18.9	209	0
β-HBCDD	0.98	1.19	0.01	0.62	1.18	2.99	31.3	0	1.96	3.33	0.01	1.05	2.20	6.50	164	0
γ-HBCDD	5.50	6.94	0.08	3.41	6.59	17.0	171	0	14.8	17.3	0.18	9.59	17.9	44.1	561	0
ΣHBCDDs	9.10	11.4	0.12	5.70	10.9	27.8	312	0	23.2	28.0	0.33	15.0	28.0	69.1	937	0
HBB	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0	0.01	0.01	0.00	0.00	0.01	0.03	0.66	0
EHTBB	0.12	0.11	0.01	0.09	0.15	0.31	2.94	95	0.25	0.24	0.01	0.18	0.30	0.66	9.94	90
BEHTBP	0.24	0.32	0.00	0.15	0.27	0.73	11.8	14	1.01	1.18	0.02	0.66	1.20	2.94	47.4	6
DBDPE	0.24	0.26	0.01	0.17	0.28	0.66	7.54	20	0.42	0.41	0.02	0.30	0.50	1.10	16.0	22
ΣNBFRs	0.60	0.56	0.04	0.45	0.71	1.50	26.1	33	1.68	1.54	0.09	1.25	2.02	4.26	42.0	22
Dec-604	0.00	0.00	0.00	0.00	0.00	0.01	0.26	100	0.00	0.01	0.00	0.00	0.00	0.01	0.66	100
syn-DP	0.00	0.00	0.00	0.00	0.00	0.00	0.08	26	0.00	0.00	0.00	0.00	0.00	0.00	0.16	100
anti-DP	0.00	0.01	0.00	0.00	0.00	0.01	0.21	8	0.01	0.01	0.00	0.00	0.01	0.02	0.68	10
ΣDECs	0.01	0.01	0.00	0.00	0.01	0.02	0.32	36	0.01	0.02	0.00	0.01	0.01	0.03	0.92	43
ΣHFRs	10.0	12.2	0.18	6.37	11.2	30.2	354	2	25.8	30.1	0.54	17.0	31.2	75.4	1325	2

P50, P75 and P95 are 50th, 75th and 95th Percentile, respectively.
THP, TPP, Dec-602, Dec-603, PBEB were not calculated due air and dust samples were <LOD in all samples.
[a] % Inh: Mean contribution of indoor air inhalation pathway to total exposure (air inhalation plus dust ingestion).

Factors influencing environmental profiles lead to variability in correlations between FRs and the number of electronic devices, and wood and plastic furniture, as well as in the comparison among homes, schools and offices. This agrees with the differences in levels found in other European countries. Further research should be focused on evaluating levels in other indoor environments in order to clarify the potential risk of novel FRs. The current risk assessment suggests that exposure to FRs was below the assumable health risks (hazardous quotient and cancer risk below 1 and 10^{-6}, respectively). However, there is a lack of knowledge regarding toxicity of novel FRs and the interaction in FR mixture exposure. Therefore, both widely use and high concentrations in indoor environments, clearly indicate the need of assessing the potential toxicity, persistence and bioaccumulation of these replaced FRs.

Funding

This work was supported by the Spanish Ministry of Science, Innovation and Universities (MCIU), the State Research Agency (AEI) and the European Regional Development Fund (ERDF) [FLAMERISK project, grant number RTI2018-095466-B-I00]. J. Rovira received a postdoctoral fellowship from "Juan de la Cierva-incorporación" program of the MCIU [grant number IJC2018-035126-I]. IDAEA-CSIC is a Centre of Excellence Severo Ochoa (MCIU) [grant number CEX2018-000794-S].

CRediT authorship contribution statement

Roser Esplugas: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Resources, Data curation, Writing – original draft, Visualization. **Joaquim Rovira:** Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Resources, Data curation, Writing – review & editing, Visualization, Supervision, Project administration. **Montse Mari:** Conceptualization, Validation, Formal analysis, Investigation, Resources, Writing – review & editing, Supervision, Project administration. **Julio Fernández-Arribas:** Methodology, Software, Formal analysis, Resources, Data curation, Writing – review & editing, Visualization. **Ethel Eljarrat:** Conceptualization, Validation, Formal analysis, Investigation, Resources, Data curation, Writing – review & editing, Visualization, Supervision. **José L. Domingo:** Investigation, Writing – review & editing, Supervision, Project administration, Funding acquisition. **Marta Schuhmacher:** Conceptualization, Validation, Investigation, Writing – review & editing, Visualization, Supervision, Project administration, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

Biotage is acknowledged for providing SPE cartridges.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.150494.

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