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Comprehensive characterization of European house dust contaminants: Concentrations and profiles, geographical variability, and implications for chemical regulation and health risk

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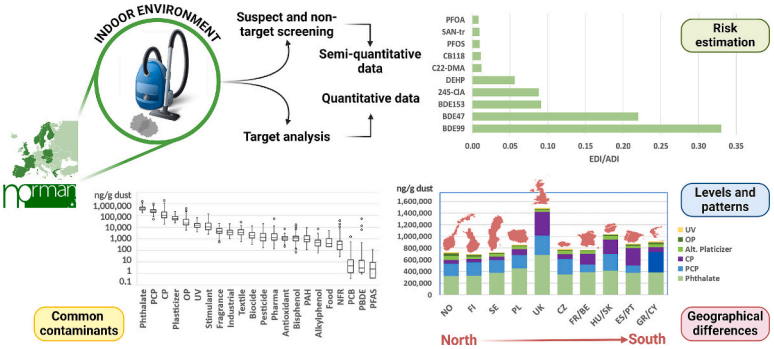
HIGHLIGHTS

- Over 1200 anthropogenic compounds were tentatively identified in European house dust
- In total, 262 house dust contaminants were commonly detected and quantified or semi-quantified
- Multivariate data analysis show geographical trends, with north-south gradients across Europe
- Relative abundance of regulated compounds and substitutes indicate limited effect of regulation
- The results call for proactive measures to prevent hazardous chemicals from entering the market

GRAPHICAL ABSTRACT



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ABSTRACT

This study investigated the concentration profiles and geographical variability of contaminants in house dust across Europe. A collaborative trial (CT) was organized by the NORMAN network using pooled dust and advanced chromatographic and mass spectrometric techniques combined with suspect screening and non-target screening (NTS). Over 1200 anthropogenic compounds were tentatively identified. Additionally, seventy-five individual samples were subjected to target analysis and NTS. The median concentrations of most contaminants varied <3-fold across Europe, and the contaminant profile of European dust was similar to that of North American dust, which was investigated in a previous CT. This similarity may be attributed to the use of similar consumer articles and building materials throughout the developed world. Multivariate data analysis revealed geographical trends in contaminant distribution, with north-south gradients across Europe. Geographical trends were more frequently found for compounds with rapid release (pharmaceuticals, personal care products, fragrances, pesticides, biocides) and smoke-related compounds. The concentrations of chlorinated paraffins, polycyclic aromatic hydrocarbons (PAHs), perfluorinated alkyl substances and stimulants generally increased from north to south, whereas the biocides levels decreased from north to south. Despite widespread presence of in-use contaminants in dusts, some of the highest risks come from compounds that have been restricted for decades or more. These include di(2-ethylhexyl) phthalate (DEHP), polychlorinated biphenyl (PCB) 118 and polybrominated diphenyl ethers 47, 99, and 153. DEHP remains the most abundant contaminant in European house dust, while the other compounds are classified as persistent organic pollutants (POPs). Moreover, there is a striking lack of reliable toxicity data, particularly for emerging compounds. For instance, although acceptable daily intakes (ADIs) were examined for 202 compounds, only 46 had consensus-based ADI values. The results highlight the need for proactive measures to prevent hazardous chemicals from entering the market and for careful selection of substitute chemicals, when such are needed, to avoid regrettable substitutions.

1. Introduction

According to the European Chemical Environmental Bureau, approximately 200,000 chemicals are used in Europe and global chemical sales more than doubled between 2000 and 2017 (EEB, 2024). The United Nations Environmental Programme (UNEP) projects these figures to double again by 2030 and points out that >60 % of the volume of chemicals registered for use in the European Union (EU) are classified as hazardous (UNEP, 2019). Many of these chemicals are used in European consumer products and building materials, from which they may be emitted to a greater or lesser extent. Once emitted, the chemicals will partition between various compartments of the indoor environment, including house dust. Considering that people typically spend at least half of their day at home (Nardi and Palladion, 2021; Mitra et al., 2022), exposure to residential house dust constitutes an important contributor to the human exposome.

It is therefore important to generate knowledge about contaminant concentrations within house dust and factors that might influence this.

A collaborative trial (CT) on suspect and non-target screening (NTS) of organic contaminants in a house dust sample from North America was organized by the NORMAN network (norman-network.net) in 2016 (Rostkowski et al., 2019). This trial revealed >2300 chemical constituents that could be identified (18 %) or tentatively identified (82 %) by 20 participants using gas chromatography (GC) or liquid chromatography (LC) coupled to mass spectrometry (MS), including both natural and anthropogenic compounds. However, these numbers likely constituted only a minor part of all house dust constituents. Hilton analyzed an extract of a reference material of house dust (NIST 2585) by comprehensive two-dimensional GC–MS (GC × GC–MS) and found over 10,000 peaks (Hilton et al., 2010). The cited study only covered the semi-volatile compounds (SVOCs) and, hence, the reference material contains many more chemicals beyond the investigated chemical domain.

In 2020, a new CT was organized by the NORMAN network with the aim to find and (tentatively) identify as many organic constituents as

possible in a sample of pooled European house dust using a wide range of MS-based suspect screening and NTS techniques. Subsequently, five of the 21 participant laboratories volunteered to perform analysis of the individual samples collected to obtain quantitative data on a selection of contaminants and semi-quantitative data on additional anthropogenic compounds identified using NTS. The aggregated data were evaluated with the primary aims to i) more comprehensively characterize organic constituents of European house dust, ii) generate knowledge about typical (median) concentrations and concentration ranges of contaminants, and iii) investigate geographical differences (within Europe and between Europe and North America). Furthermore, the contaminant concentrations were compared to available toxicity reference values to estimate the potential risk that residential dust contaminants pose to European citizens. An overview of the study design and methods used is shown in Fig. 1.

Consumer articles, household chemicals and building materials are traded on a global market leading to the hypothesis that residential house dust collected across Europe will contain similar contaminant concentrations and profiles. However, the climate, area use, chemical use policies, and consumer habits and preferences are expected to differ over the continent, leading to a second hypothesis – that there will be regional differences in the concentrations of a subset of the house dust contaminants. To obtain a wide coverage of anthropogenic contaminants, data from target analyses for well-known contaminants and contaminants of emerging concern (CECs) were combined with suspect screening and NTS using state-of-the-art GC–MS, GC × GC–MS and liquid chromatography – high-resolution mass spectrometry (LC-HRMS) techniques (Fig. 1). Various multi-variate statistical analysis tools were used to find and describe spatial trends in the distribution of house dust contaminants in Europe, a task that had previously not been undertaken. The results obtained are compared to literature data and are discussed in relation to the hypotheses formulated above and in relation to the contaminants' physicochemical properties, functional use categories and consumer habits and preferences. The challenges associated with risk assessment of contaminants in house dust are also discussed, including the lack of reliable toxicity data and potential additive effects. Finally, the relative abundance of regulated compounds and their substitutes are used as basis for a discussion on the effectiveness of past and current regulatory and voluntary efforts to reduce the emissions of and exposure to hazardous substances.

2. Material and methods

2.1. Sampling and samples

Individual house dust samples were collected in 2020 by members of the NORMAN network as part of a voluntary effort, using a standardized protocol (Abdallah et al., 2008). In each home, 1 m^2 of carpet was vacuumed for 2 min and in case of bare floors 4 m^2 was vacuumed for 4 min. Samples were collected using nylon sample socks (25 μm mesh size) that were mounted between the end of the telescopic tube and the foot of the vacuum cleaner. After sampling, the socks were closed with a twist tie, sealed in clean polyethylene zip-lock bags, and shipped to Umeå University. The sampling campaign resulted in 66 individual house dust samples. The samples were stored in the freezer (−18 °C) until the time all samples had been collected. They were then sieved (150–500 μm) to enhance sample homogeneity and comparability.

In addition, eight samples of pooled and sieved house dust were provided by EHESP, University of Rennes (n = 4, ca 100 μm), Umeå University (n = 3, 75–190 μm) and University of Antwerp (n = 1, 500

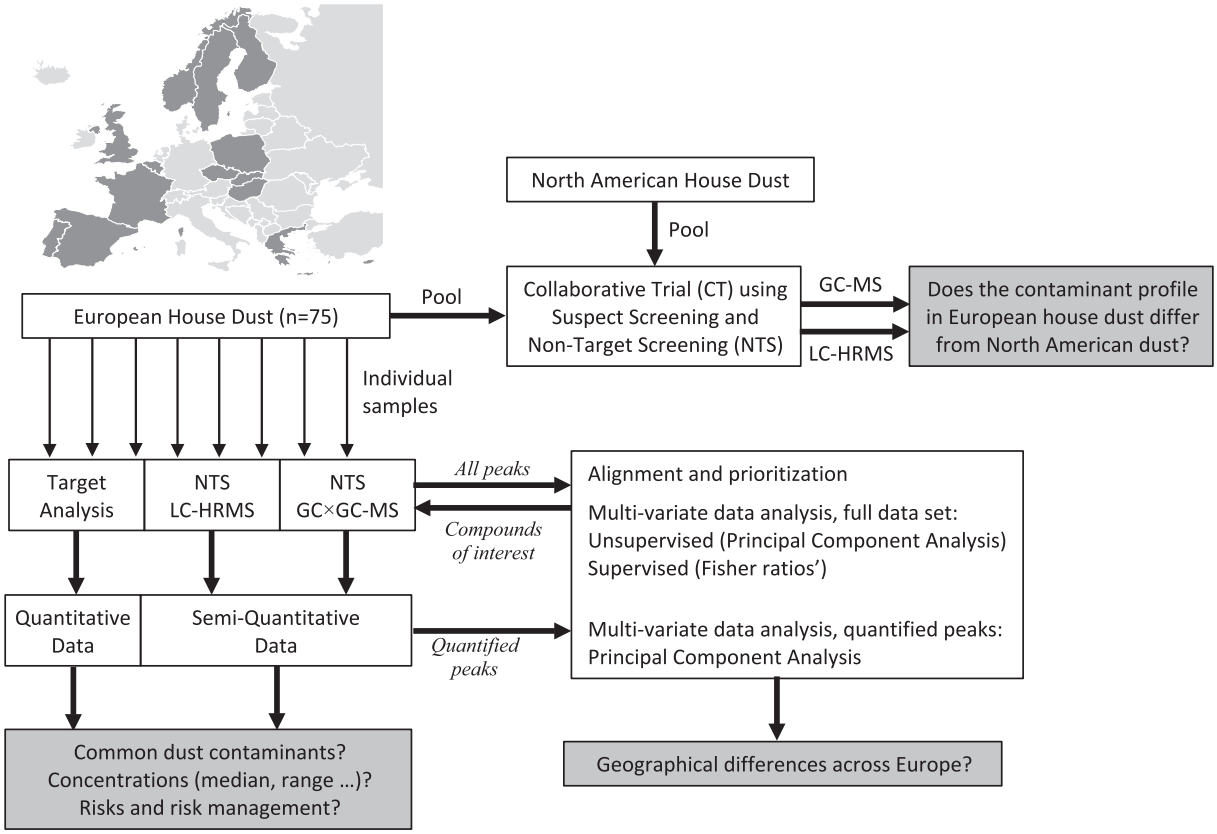

Fig. 1. Study layout for investigation of geographical differences in house dust contaminant profiles, typical concentrations of European dust contaminants, and associated potential risks. Sampled countries are marked in dark grey on the map. GC–MS: Gas chromatography – mass spectrometry; LC-HRMS: Liquid chromatography – high-resolution mass spectrometry; GC × GC–MS: Comprehensive two-dimensional GC–MS.

Table 1
Overview of the sampling and analyses performed within the NORMAN collaborative trial on European indoor dust using suspect and non-target screening (NTS) and house dust contaminant geographic distribution studies. Abbreviations: AP: alkylphenols, BP: bisphenols, CP: chlorinated paraffins, NFR: novel flame retardants, OP: organophosphate esters, PAH: polycyclic aromatic hydrocarbons, PFAS: perfluorinated alkyl substances, LC: liquid chromatography, GC: gas chromatography, MS: mass spectrometry. * Abbreviations explained in author affiliations.

Organization*	Dust sampling (country code, n)	Collaborative trial		Geographic distribution	
		LC-MS	GC–MS	Target analysis	NTS
Adam Mickiewicz University in Poznań	Poland (PL, 8)	–	–	–	–
AFIN-TS GmbH	–	Yes	–	–	–
Aristotle University of Thessaloniki	–	Yes	–	–	–
EHESP, Univ Rennes	France (FR, 4)	–	–	–	–
Environmental Institute	Hungary (HU, 2) Slovakia (SK, 5)	–	Yes	–	–
Finnish Institute for Health and Welfare	Finland (FI, 8)	–	–	–	–
Health Canada	–	Yes	–	–	–
IDAEA-CSIC	–	Yes	–	–	–
INEGI	Portugal (PT, 4)	–	–	–	–
INERIS	France (FR, 5)	Yes	–	–	–
IVL Swedish Environmental Research Institute	Sweden (SE, 3) Cyprus (CY, 1)	Yes	–	–	–
Nanjing University	–	Yes	–	–	–
NILU	Norway (NO, 7)	–	–	AP/BP/CP/NFR	–
NTNU	–	Yes	–	–	–
Örebro University	Sweden (SE, 5)	–	Yes	–	–
RECETOX, Masaryk University	Czech Republic (CZ, 5)	–	Yes	PFAS	–
Umeå University	Sweden (SE, 3)	Yes	Yes	PAH/Plasticizers	GCxGC–MS
US Environmental Protection Agency	–	Yes	–	–	–
University of Antwerp	Belgium (BE, 1)	Yes	Yes	OP	–
University of Athens	Greece (GR, 3)	Yes	Yes	–	LC-HRMS
University of Bordeaux	–	Yes	Yes	–	–
University of Birmingham	United Kingdom (UK, 3)	–	–	–	–
University of Cordoba	Spain (ES, 3)	Yes	–	–	–
University of Gdansk	Poland (PL, 5)	–	–	–	–
University of Luxembourg	–	Yes	–	–	–
University of Newfoundland	–	Yes	–	–	–
University of Queensland	–	Yes	Yes	–	–
Vito	–	Yes	Yes	–	–

μm). These samples did not contain dust from any of the individual dust samples. This resulted in 75 samples from 14 countries spread across Europe (Table 1 and Fig. 1).

A composite sample (10 g) was created by pooling aliquots, generally 20 % of each individual dust sample ($n = 67$) or dust pool sample ($n = 8$). However, the total amount of dust collected differed widely (31 mg to 7.1 g) and the sample contribution of very large samples was reduced (max 0.3 g) to avoid excessive dilution effects. The mass contribution for individual samples varied between 0.31 % and 3.0 %. The composite sample was used within the CT (Table 1) and it was split into two equal parts, one for analysis by the participant's regular method and one for preparation of an extract, which was also offered to the participants.

Moreover, a house dust extract was prepared by sequential extraction of 5 g of pooled dust with ultrasonication, in 50 mL glass centrifuge tubes, using two 25 mL portions of hexane:acetone (3:1, *v*/v), and two 25 mL portions of 100 % acetone. This procedure has previously been shown to efficiently extract a wide range of dust contaminants (Moschet et al., 2018). Between each extraction, the sample was centrifuged, and the supernatant was removed. Following suitable pooling and evaporation (until near dryness) the final extracts were prepared in 25 mL hexane:acetone (1:1, v/v). The final extract contained an equivalent of 200 mg of dust per mL of solvent. A blank extract was prepared in the same way and was shipped together with the samples.

Participants in the CT were provided with one 200 mg aliquot of house dust and one 1 mL aliquot of dust extract (originating from 200 mg of dust). The type of analysis performed by each laboratory is given in Table 1. LC-HRMS data generated using electrospray ionization (ESI) and GC–MS data generated using electron ionization (EI) were used in the current publication.

The remaining portions of house dust (80 % *w*/w in most cases) were used to investigate the spatial variability of dust contaminants across Europe (Fig. 1). However, if the remaining dust weight exceeded 1 g, only 1 g were used, to prevent a few large samples from skewing multivariate statistical models created using NTS data (c.f. section 2.4). The individual samples were sequentially extracted with two 5 mL portions of methanol, two 5 mL portions of hexane:acetone (3:1, *v*/v) and two 5 mL portions of hexane. Four procedural blank samples were prepared in the same manner as the samples. The extracts were split into five equal parts (1 mL each).

Two of the 1 mL methanol aliquots (first and second extraction) were pooled, transferred to a HPLC vial and shipped for analysis of perfluorinated alkyl substances (PFAS). In addition, four composite samples were created by pooling 1 mL portions from each of the six sequential extracts. The extract volumes were reduced to 3 mL and each composite was transferred to a 4 mL vial and shipped for target analysis.

2.2. Target analysis and non-target screening

The following compounds were targeted: PFAS (at RECETOX), alkylphenols (APs), bisphenols (BPs), short-, medium- and long chain chlorinated paraffins (CPs), and novel flame retardants (NFRs) (at NILU), organophosphate esters (OPs; University of Antwerp) and polycyclic aromatic hydrocarbons (PAHs) and plasticizers (Umeå University). The measurement accuracy at the limit of quantification for the detected compounds were generally within ±30 %, but were higher for branched alkylphenols, 2,4-substituted bisphenols and decabromodiphenyl ethane (DBDPE; ± 50–60 %), and chlorinated paraffins (± 100 %). The methods and the quality assurance/quality control (QA/QC) measures are described in the Supplementary material.

2.3. Non-target screening

NTS was performed for both polar contaminants (University of Athens) and non-polar contaminants (Umeå University) using LC-

HRMS/MS and GC × GC–MS, respectively. NTS for LC amenable compounds was performed using a Bruker Maxis Impact LC quadrupole time-of-flight (QTOF) MS/MS instrument (Bruker Daltonics, Bremen, Germany) operated at a resolution of $R = 40{,}000$ and equipped with a C18 column. NTS for GC amenable compounds were performed using a Leco BT4D GC × GC unit resolution MS instrument (St Joseph, MI, USA) equipped with a non-polar/polar column set. Details on instrumental analysis are given in Supplementary material.

2.3.1. Generation, processing and QA/QC of LC-HRMS/MS non-target screening data

Data was collected in both positive and negative electrospray ionization (ESI) mode. Each batch of samples was accompanied by a procedural blank to capture any potential contamination originating from solvents, consumables, and laboratory conditions. For retention time normalization and quality assurance/quality control, the "UoA RTI" mixture was analyzed periodically throughout the sequence (Aalizadeh et al., 2021). Instrument performance was continuously monitored through the peak area and retention time of the RTI standard mixture, injected every 10 samples. No significant sensitivity loss or notable retention time drift was observed, with retention time stability maintained within 0.1 min.

Data was converted to mzML format using XCPWrapper 3.0.9.2 by Bruker Daltonics (Bremen, Germany) within Bruker DataAnalysis v4.3. The mzML files, along with sample metadata, were uploaded to the NORMAN Digital Sample Freezing Platform (DSFP), with the collection assigned a persistent DOI doi:10.60930/dsfp.xa5r-j469) for global accessibility. All data is openly available on the DSFP platform.

Data was processed using a standardized workflow (Alygizakis et al., 2023). Briefly, each sample was processed using centWave algorithm (through XCMS R-package v. 4.2.2) for peak picking (Tautenhahn et al., 2008) with previously optimized ppm and peak width parameters (Libiseller et al., 2015). The peak lists were processed for componentization, which is a procedure for grouping peaks coming from the same compound (adducts, isotopic peaks, in-source fragments) using the nontarget R-package (v. 1.9) (Loos and Singer, 2017). The component lists were stored in a standardized way that allowed them to be screened retrospectively for the SusDat compounds amenable to LC-MS. The substances that were detected passed a number of criteria: mass accuracy <2 mDa, plausible retention time, plausible isotopic fit, presence of qualifier fragment ions. The credibility of the identifications was evaluated based on the collected evidence from the HRMS data and only tentatively identified compounds with an identification point (IP) score >0.50 (Alygizakis et al., 2023) were semi-quantified using QSPR ionization efficiency prediction models (Aalizadeh et al., 2022).

2.3.2. Generation, processing and QA/QC of GC × GC–MS non-target screening data

Prior to analysis, the samples were fortified with 1 mL of toluene and evaporated to 200 μL. An n-alkane mixture (C8-C40 in toluene) was injected with each batch of samples and was used to establish linear retention index (RI) values for the dust contaminants (van den Dool and Kratz, 1963). Leco ChromaTOF software was used for instrument control and part of the data evaluation. It was complemented by Analyzer Pro XD software from SpectralWorks Ltd. (Runcorn, UK), which was primarily used for data alignment. In the first step of data processing, peak picking was performed with Analyzer Pro XD using a minimum signal-to-noise ratio (S/N) threshold of 10. All detected peaks were compared and aligned with regard to the first-dimension retention time (±20 RI units), second-dimension retention time (±0.2 s), and EI spectra similarities (600, forward matching) using the Matrix Analyzer module of the software. This resulted in a matrix table with compounds/features as rows, samples as columns, and total peak areas as cell values. In addition, the consensus spectrum for each component was automatically searched against the NIST 2020 mass spectral library (NIST; Gaithersburg, MD, USA). Compounds that match an entry in the database were assigned a tentative name and the remaining components were assigned sequential numbers.

The matrix with tentatively identified compounds and unknown features was exported to MS Excel and was processed to remove column bleed features, biogenic compounds, and features that were abundant in blank samples (blank average area/sample average area >0.2). In addition, branched alkanes and cycloalkanes were removed, as these compounds were poorly resolved and could not be reliably aligned. The remaining compounds and features were further filtered to remove those that were not detected in at least 10 % of all samples or at least one third of the samples from one country.

The compounds and features in the filtered data set were manually scrutinized to find and tentatively identify as many anthropogenic compounds as possible. The spectral quality, NIST library spectra similarity and probability, and agreement of experimental and NIST database RI values were considered in the data curation process. The peak positions on the GC × GC plane were used to support the tentative identification of members of homologous series that were missing in NIST 2020 (Rostkowski et al., 2019). Based on the spectral quality, the degree of agreement with NIST data, and available metadata (previously reported in house dust, consumer products, number of patents or PubChem references, etcetera) an identification confidence level was assigned according to the scale of Schymanski et al. (2015), modified to fit GC–MS data by Röhler et al. (2021).

All tentatively identified compounds were added to a compounds-of-interest list (Fig. 1), which was reprocessed using a ChromaTOF target analyte finding method. At least two abundant ions (of which one was used as quantifier ion, QI) were included in the method and a low signal threshold (S/N = 10) was used to enhance the detection frequency. The integrations were checked manually and adjusted when required. All compounds were semi-quantified using the most closely eluting $^{13}C_{12}$-PCB (added together with internal standard (IS) for PAHs and phthalates, see Supplementary material). The total component area of the analyte was obtained by dividing the QI area by the relative contribution of the QI to the total ion abundance of all ions in the analytes mass spectrum (extracted from the most abundant sample). The total component area of the $^{13}C_{12}$-PCB was calculated in the same way and the analyte concentration was obtained using the isotope dilution method.

2.4. Multivariate statistical evaluation of data

2.4.1. Unsupervised PCA

Principal component analysis (PCA) was used to create an overview of major trends and groups among objects (samples) and variables (compounds/features). PCA was performed using SIMCA 16 software (Umetrics suite; Sartorius, Umeå, Sweden). Two sets of data were evaluated: i) all compounds and features that were found in at least 10 % of all samples or at least one third of the samples from one country and ii) all compounds that were quantified using target analysis or semi-quantified using NTS methodologies. The data was log-transformed to bring the values closer to normal distribution. It was also scaled to unit variance and mean centered to reduce the influence of contaminant concentrations, thereby emphasizing profile differences. Cross validation was used to determine the number of significant principal components (PCs).

2.4.2. Supervised PCA

The unsupervised PCA was complemented by a supervised data evaluation of GC × GC–MS data based on Fisher ratios (F-ratios), which is a measure of the within group variability versus the between group variability. A tile-based F-ratio software (ChromaTOF tiles) was used to reduce the influence of retention time variations and among samples (Parsons et al., 2015). The raw data from ChromaTOF was directly imported into the software and the data was then normalized to the IS area and sample weight. Several groupings were tested to find the one with

the best differentiation between groups. An F-ratio cut-off value of 5 was used to extract the compounds that differed the most between groups. The features that exceeded that threshold were compiled in Excel and were subjected to PCA, as described above.

2.5. Calculations of estimated daily intakes (EDIs)

Estimated daily intakes (EDIs) of individual contaminants in residential dust via non-dietary ingestion were estimated based on the average concentrations in European house dust. Exposure was estimated for a child less than six years old, based on exposure factors from the US Environmental Protection Agency (US-EPA) exposure factors handbook (US-EPA, 2011), as children typically have the highest dust ingestion rates and consequently higher exposure than adults. EDIs were calculated assuming a dust ingestion rate of 40 mg/day and a child body weight of 18.6 kg, and using European medians (normal scenario) and 90th-percentiles (high-risk scenario) for concentrations in dust.

ADIs include indicators of acceptable intake thresholds using different terminology from different regulatory agencies or studies (Demirtepe et al., 2019), e.g., acceptable daily intakes (ADIs), tolerable daily intakes (TDIs), minimal risk levels (MRL) or reference doses (RfD). Although different agencies may use different underlying assumptions when calculating such thresholds, published agency values typically represent the consensus of multiple studies evaluated based on expert knowledge, and were taken without adjustment. When consensus values were not available, NOAELs were used with a safety factor of 100 (European Commission, 2003). NOAELs were selected according to the following criteria: published in 2004 or later, reflecting oral intake, and excluding acute/short-term/sub-chronic exposures. When NOAELs for multiple endpoints were available, the most sensitive endpoint was selected. Cancer risks were not considered. ADIs used for comparison with EDIs were based on toxicology data available in the US-EPA CompTox Chemistry Dashboard as of March 2024.

3. Results and discussion

3.1. Performance characteristics of the target, suspect and non-target analyses

It is difficult to estimate the measurement accuracy for semi-quantitative analyses (Pu et al., 2024). However, a comparison was made between concentrations obtained using target analysis and NTS semi-quantification by GCxGC–MS for five OPs (tris(2-chloroethyl) phosphate, tris(2-chloroisopropyl) phosphate, tris(1,3-dichloro-2-propyl) phosphate, triphenyl phosphate, and 2-ethylhexyl diphenyl phosphate) and six PAHs (fluorene, phenanthrene, fluoranthene, pyrene, chrysene, and benzo[*a*]pyrene). The average deviation between the target quantification and NTS semi-quantification results ranged between – 10 % and – 40 % for the OPs and between +20 % and + 60 % for the PAHs. Thus, the reported GC × GC–MS semi-quantitative data are assumed to be within a factor 2 around the actual value (50 % to 200 %). The evaluation of semi quantification results for LC-MS/MS indicate comparable mean errors, but larger deviations, up to an order of magnitude (Aalizadeh et al., 2022). The wider range was attributed to difficulties in accurately predicting ESI ionization efficiencies.

In total, 262 house dust contaminants were commonly detected (detection frequency > 10 %) and quantified (77) by target analysis or semi-quantified (185) using NTS methodologies (Section 2.3). More than half of the target compounds (60 %) were also detected using the NTS methodologies, mainly plasticizers, OPs and PAHs. The compounds that were not detected were generally present at low concentrations. The full results of the target analysis are presented in Table S2. The concentrations of individual dust contaminants generally spanned a range of two orders of magnitude, which is considerably more than the measurement uncertainty of both the target analysis and NTS semi-quantification methods. No differentiation has therefore been made between quantitative and semi-quantitative results in Section 3.3 and Section 3.4, which summarize the results of the analysis of the individual dust samples.

In addition, results from the analysis of the house dust composite sample (Fig. 1) were reported by 21 laboratories. Many of the participating organizations had previously contributed data for the first NORMAN Collaborative Trial on non-target analysis of house dust, which was conducted using house dust from North America (Rostkowski et al., 2019). A list of chemicals reported in that exercise can be found in the NORMAN Suspect List Exchange (NORMAN-SLE; Taha et al., 2022) under the name “S35 - INDOORCT16” and on Zenodo (DOI: https://doi.org/10.5281/zenodo.2653206). Similar methodologies were used in the two CTs, the reported data were curated in a similar way, and the results could therefore be used to compare the contaminant profiles of North American and European house dust. A comparison of the contaminant profiles of house dust from Europe and North America is made in Section 3.4.4. The results of this collaborative trial (Excel file “Supplement_Compound list” in Supplementary material) have also been uploaded to the NORMAN-SLE as list “S120 - DUSTCT24” (DOI: https://doi.org/10.5281/zenodo.13835255).

The results of the CT indicate that there is still some way to go until we are able to generate comparable suspect screening and NTS results across multiple laboratories. The number of compounds reported by the participating laboratories varied widely, ranging from 15 to 581, which accounts for 1 % to 48 % of the 1210 compounds reported. Only one contaminant (triphenyl phosphate) was detected by 75 % of the participants, 10 were detected by at least 50 % of the participants, and 53 were detected by at least 25 % of the participants (c.f. Excel file “Supplement_Compound list”; compounds with high detection frequency are color coded in green). Moreover, most contaminants (70 %) were reported by only one of the participants. Similar percentages of unique contaminants, 66 % for LC-HRMS and 64 % for GC–MS, were obtained in a previous NORMAN collaborative trial (Rostkowski et al., 2019). However, an improvement was noted for some groups of contaminants, e.g. OPs, abundant bisphenols and abundant PFAS. These were reported by a greater share of laboratories in the current study as compared to the previous. In order to improve the analytical practices, the NORMAN network has recently published an extensive guidance document for suspect screening and NTS (Hollender et al., 2023).

3.2. Results overview

The NTS data generated by GCxGC–MS, including both tentatively identified compounds and unknown features, were evaluated using multivariate data analysis (MVDA) of the full data, including both identified compounds and unknown features, to provide a first overview of the spatial variability of house dust contaminants across Europe. A weak PCA model was obtained with an explained variance of <30 % and no apparent grouping of samples, which suggests that the within-country variability was comparable to, or greater than, the variability between countries. This outcome supports the primary hypothesis of the study, i.e. that house dust contaminants are ubiquitous and that their distribution across Europe is mainly uniform. The results of the second NORMAN collaborative trial on house dust offer insights into the extent of the contamination, revealing 1210 anthropogenic compounds in European house dust at a (self-reported) identification confidence level of 3 or better. A list of these (tentatively) identified contaminants, along with information on the detection technique(s) and frequencies, is provided in the Excel file “Supplement_Compound list”.

However, using F-ratio based supervised data evaluation (Section 2.4.2) it was possible to observe spatial trends among the samples collected across Europe. Several groupings were tested, with the best separation of groups obtained when dividing the samples into three regions: North Europe (Finland, Norway, and Sweden), Middle Europe (Belgium, Czech Republic, France, Hungary, Poland, Slovakia, and UK), and South Europe (Cyprus, Greece, Portugal, and Spain). In addition, the

analysis revealed a North-South trend in the contaminant concentrations and distribution. This finding supports the second hypothesis of our study, i.e. that differences in consumer preferences, habits and use patterns across Europe are reflected in the contaminant profiles and levels found in house dust.

A PCA was also conducted using all quantitative and semi-quantitative data to obtain a visual representation of how the house dust contamination varies across Europe. In cases when both were available, priority was given to quantitative data over semi-quantitative data and to GC × GC–MS NTS data over LC-ESI-HRMS NTS data. The 3D graph of the results (Fig. 2) clearly shows that there is a spatial trend in the data, with samples from South Europe found at the top of the graph, samples from North Europe at the bottom, and samples from the middle latitudes of Europe (Mid Europe) in between. There is a continuous spread within each group with samples from the same country/region clustering together. Moving from bottom-front to top-back of the graph, we find samples originating from Sweden, Finland/Norway, Poland, Czech Republic/Slovakia, Belgium/France, Cyprus/Greece/Hungary/UK, Spain and Portugal. The three principal components (PCs) were found to capture variations related to contaminant concentration (PC1; 88.5 % of variance), sample type (individual/pooled; PC2; 1.0 %) and latitude of sampling location (north-south; PC3; 0.9 %). PC1 has negative values in the forward part of the box and positive values in the rear. Thus, given the location of the samples in the graph (Fig. 2), the dust samples from the south of Europe appear to contain higher concentrations of contaminants than samples from the north.

In the following sub-section (3.3) the median total concentrations of contaminants and contaminant classes in European house dust are presented and also compared with data from similar studies. In section 3.4, we discuss the geographical differences (north to south) in: total concentrations and compound class distribution (section3.4.1), contaminant class concentrations and profiles (section 3.4.2), and concentrations of individual contaminants (section 3.4.3). Thereafter, we compare the dust contaminant profile of European house dust with that of North American house dust (section 3.4.4). The concentrations of European dust contaminants and the geographic distribution are discussed in relation to their physicochemical properties, functional use categories and consumer preferences and habits (section 3.5). In section 4, the individual contaminant concentrations are used to calculate EDIs, which are compared to ADIs and similar values to estimate the margin of safety for human exposure to house dust.

3.3. European house dust contaminant concentrations

The total concentrations of the quantified or semi-quantified house dust contaminants are presented as box and whisker plots in Fig. 3. The contaminant classes were found to span more than five orders of magnitude, ranging from PFAS at a few ng/g to phthalates at almost 1 mg/g. Plastics additives (incl. Phthalates, other plasticizers, OPs, and CPs) and personal care product (PCP) chemicals had the highest concentrations, while halogenated persistent organic pollutants (POPs) were present at the lowest concentrations. The median concentrations of UV absorbers, stimulants and fragrances were relatively high (ca 10 μg/g). Compounds related to textile industry, compounds with broad industrial use (including phenols), biocides, pesticides, pharmaceuticals, and PAHs were present at moderate median concentrations (around 1 μg/g). The high-level dust contaminants generally span one to two orders of magnitude in concentration; medium level contaminants span two orders of magnitude, and the low-level contaminants span more than two orders of magnitude. The latter is partially explained by a few samples with highly elevated levels of POPs (above 1 μg/g).

The median concentrations of individual house dust contaminants (ng/g dust) are presented in Fig. 4. The compounds have been grouped by compound class and arranged in decreasing order of concentration. Detailed information on the concentrations, detection frequencies, identification confidence levels, summary statistics and abbreviations of the contaminants are provided in Table 2 and Table S3. Major contaminants in each group are highlighted and discussed below, starting with the most abundant contaminant groups, plasticizers and PCPs.

Phthalates, specifically di(2-ethylhexyl) phthalate (DEHP), was the contaminant class (light green in Fig. 4) and individual contaminant (250 μg/g), respectively, found at highest median concentration.

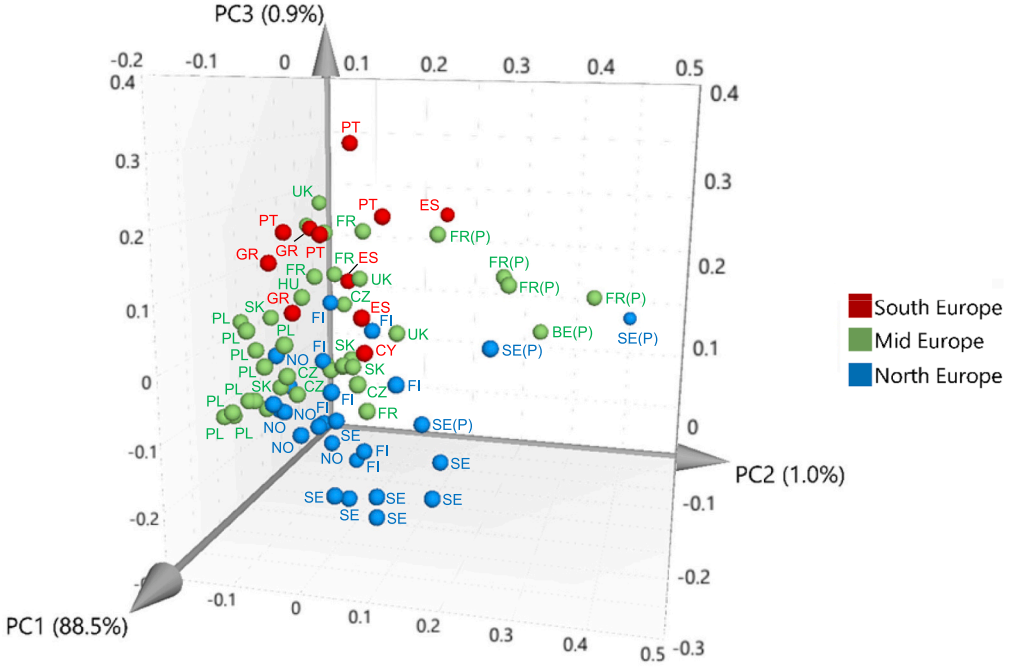

Fig. 2. 3D-plot from principal component analysis of samples of European house dust, with objects color-coded according to region. The country of origin of the samples are indicated by their 2-letter country code (see Table 1) and pooled samples are indicated by "(P)". The explained variance of each principal component (PC) is given in parentheses.

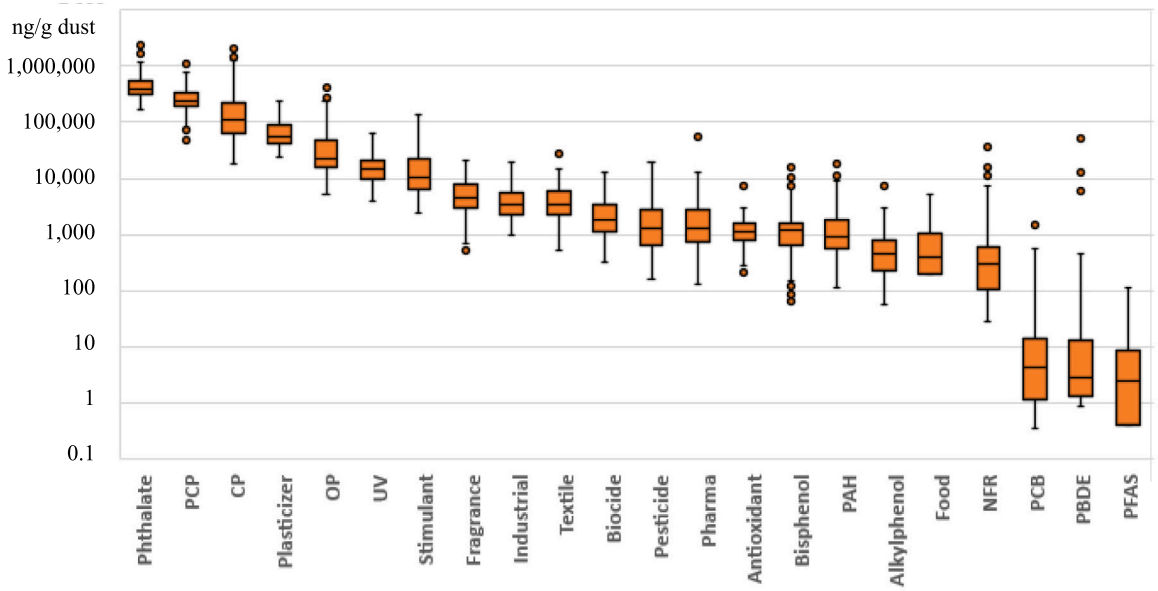

Fig. 3. Box and whisker plot of total concentrations of dust contaminant classes. Circles indicate outlier values above and below the quartiles. CP: Chlorinated paraffin; Plasticizer: Non-phthalate plasticizer; OP: Organophosphate ester; UV: UV screen; Textile: Textile related compound; PAH: Polycyclic aromatic hydrocarbon; Food: Food related compound; NFR: Novel flame retardant; PBDE: Polybrominated diphenyl ether; PFAS: Polyfluoroalkyl substances.

Despite considerable voluntary and legislative efforts to reduce their use in consumer products and building materials, this study demonstrates that phthalates persist as a prominent class of indoor contaminants. The DEHP substitutes di-isononyl phthalate (DINP; 40 μg/g), di-isodecyl phthalate (DIDP; 16 μg/g) and benzyl 2-ethylhexyl phthalate (BzEHP; 6 μg/g) were found at approximately 1/10th of the concentration of DEHP (250 μg/g). The low molecular weight phthalates were detected at similar or lower levels than BzEHP. The alternative plasticizers (bright green in Fig. 4) were found within a similar concentration range (0.35–22 μg/g). Tributyl acetylcitrate (ATBC), di-isononyl cyclohexane (DINCH), dipropylene glycol dibenzoate (DPGDB), diethylene glycol dibenzoate (DEGDB), and di-2-ethylhexyl adipate (DEHA) were the individual alternative plasticizers found at highest concentrations. Chlorinated paraffins (CPs) and OPs, which are used as both plasticizers and flame retardants, were detected at concentrations comparable to those obtained for phthalates and alternative plasticizers, respectively. Long chain chlorinated paraffins (LCCPs; 87 μg/g) and tris(2-chloroisopropyl) phosphate (TClPP; 6 μg/g) and tris(2-butoxyethyl) phosphate (TBOEP; 3 μg/g), respectively, were the most abundant contaminants in the two compounds classes.

The personal care product (PCP) class was divided into four sub-groups: glycols and polyethylene glycols (PEGs), wax esters (WE), alkyl benzoates (AB) and mono- and dimethyl alkyl amines (DMAs) (blue in Fig. 4, left column). The concentrations of the "glycols" correlated to the molecular size and decreased from glycerol (GLYC; 85 μg/g), through diethylene glycol (EG; 5 μg/g) and dipropylene glycol (DPG; 4 μg/g) to PEGs of various chain lengths (1–3 μg/g). Wax esters were the second most abundant sub-group, with stearyl palmitate (16:18-WE; 47 μg/g) and cetyl palmitate (16:16-WE; 23 μg/g) detected at the highest levels. Alkyl benzoates with chain lengths ranging from C12 to C15 (C1X-AB) were the third most abundant sub-group, with concentrations ranging from 3 μg/g to 8 μg/g. The final sub-group, amines/amides, mainly contain DMAs with chain lengths ranging from C10 to C20 (0.2–4 μg/g), showing a homologue distribution similar to that reported for dimethyl lauryl amine (DIMLA) technical formulations (Eastman, 2024). According to the EU cosmetics ingredient database (CosIng, 2024), the NORMAN SLE suspect list "S13 - EUCOSMETICS" (Taha et al., 2022), the US-EPA chemical and products database (CPDat) (Dionisio et al., 2018) and the US-EPA functional use database (FUse), the most prevalent PCPs are used in a variety of consumer products including cosmetics and hair and skin conditioning products, with a major use as emollients in skin conditioners (US-EPA, 2024a, 2024b).

Fragrances have been assigned as a separate class. Although many of these compounds are used in PCPs, there are also other uses, such as air fresheners and masking agents in detergents and household cleaners. Four fragrances: benzyl salicylate (BS), veramoss (Vera), benzyl benzoate (BB) and sclareolide (Sclar), typically used as masking compounds, fixatives or base notes were found at the highest concentrations (0.4–1 μg/g). In addition, two polycyclic musk fragrances: galaxolide (HHCB, 0.3 μg/g) and tonalide (AHTN, 0.1 μg/g), were detected at almost as high concentrations.

UV absorbers have an even broader range of applications and, consequently, form a rather large group (15 compounds; yellow). These chemicals are used in both PCPs and plastic products as UV screens and light stabilizers, respectively. The most abundant UV absorbers were octocrylene (OC; 6 μg/g), trans- and cis-2-ethylhexyl-4-methoxycinnamat (EHMC; 2 μg/g and 0.6 μg/g), benzophenone (BP, 1 μg/g), and 2-ethylhexyl salicylate (2EHS; 0.7 μg/g).

The remaining industry related chemicals have been sub-divided into the following categories (in decreasing concentration order): chemicals with multiple uses (Industrial), textile industry related chemicals (Textile), bisphenols, and alkylphenols (Phenol). The Industrial category includes one monomer (2,4-Toluene diisocyanate; 2,4-TDI), one oligomer (styrene-acrylonitrile trimer, SAN-tr), one solvent (2,4,7,9-Tetramethyl-5-decyne-4,7-diol; TMDD) and one transformation product (benzothiazolone; BTON) with concentrations ranging between 0.3 μg/g and 1 μg/g. The textile dye indigo (0.8 μg/g) and the bleach activator tetraacetylethylenediamine (TAED; 0.4 μg/g) were the two most abundant chemicals in the Textile category. The remaining members are mainly chlorinated amines (ClA) and chlorinated nitroamines (Cl-NA), which are either used in azo dye production or formed by degradation of such (Pinheiro et al., 2004). Many of these compounds are either toxic or carcinogenic (Pinheiro et al., 2004). The most abundant of the bisphenols was bisphenol A (BPA; 0.8 μg/g), despite voluntary and legislative (e.g. EU, 2018) efforts to phase out this chemical. The BPA substitutes bisphenol S (PBS; 0.1 μg/g) and bisphenol F (BPF; 0.03 μg/g) were

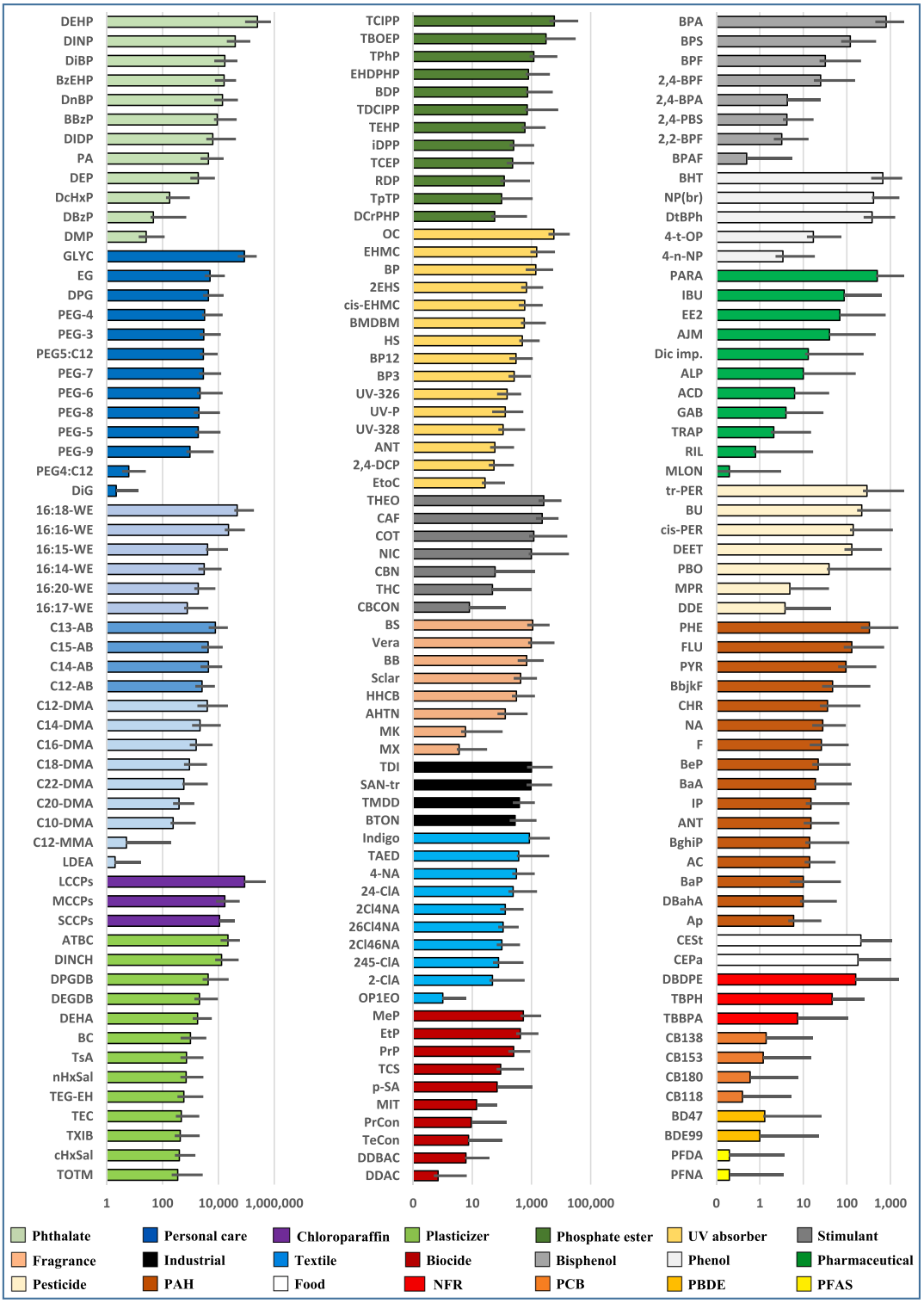

Fig. 4. Median concentrations of individual house dust contaminants (ng/g dust). Error bars indicate the 10th and 90th percentile. The compounds have been grouped by compound class and arranged in decreasing concentration order. The personal care product class has been sub-divided into glycols and polyethylene glycols (PEG), wax esters (WE), alkyl benzoates (AB) and mono- and dimethyl alkyl amines (DMA). Compound abbreviations are explained in Table 2 and Table S3 (Supplementary material). Plasticizer: non-phthalate plasticizer; Textile: textile related compound; PAH: polycyclic aromatic hydrocarbon; Food: food related compound; NFR: novel flame retardant; PCB: polychlorinated biphenyl; PBDE: polybrominated diphenyl ether; PFAS: polyfluorinated alkyl substance.

detected at approximately one and two orders of magnitude lower concentrations, respectively. In the final sub-group, "Phenol", the highest concentrations were observed for two antioxidants: butylated hydroxytoluene (BHT; 0.7 μg/g) and 2,4-di-tert-butylphenol (DtBPh; 0.4 μg/g), along with branched 4-nonylphenols (NP(br)).

The European house dust also contained four substance classes with inherent biological activity: biocides, pesticides, pharmaceuticals, and stimulants. The stimulants were found at highest concentrations, dominated by theobromine (THEO), caffeine (CAF), nicotine (NIC), and cotinine (COT) at concentrations ranging between 1 μg/g and 3 μg/g.

Table 2
Summary of concentrations (ng/g) of commonly studied and frequently detected (>20 %) chemicals in EU house dust samples with literature reference values from similar (mainly European, 2009–2024) studies. The compounds are sorted according to their highest median concentrations and the frequency of detection (Freq.). CL, identification confidence level according to Schymanski et al. (2015); "Mean" is the geometrical mean concentration; *n* is the number of studies. For an extensive list of contaminants, incl. Compounds with detection frequency 10 % to 20 %, and literature references, see Table S4 (Supplementary material).

Feature	Abbr.	CL	Freq. %	Range (min – max)	Mean	Median	Literature median*	Lit. *n*	Literature range of medians
Phthalate (target analysis; CL = 2 semi-quantified)									
Di(2-ethylhexyl) phthalate	DEHP	1	100	110,000 - 1200,000	300,000	250,000	250,000	13	14,000 - 1600,000
Di-isononyl phthalate	DINP	1	100	8800–590,000	61,000	40,000	110,000	7	26,000–210,000
Di-isobutyl phthalate	DiBP	1	100	5400–280,000	23,000	17,000	9400	8	3000–104,000
Benzyl-2-ethylhexyl phthalate	BzEHP	2	100	5200–190,000	16,000	19,000	3100*	1	
Di-n-butyl phthalate	DnBP	1	100	3300–470,000	30,000	14,000	15,000	10	3200 - 800,000
Butylbenzyl phthalate	BBzP	1	100	260–240,000	17,000	9100	8500	9	2700 - 99,000
Di-isodecyl phthalate	DIDP	1	100	400–280,000	18,000	6300	26,000	7	10,000 - 56,000
Phthalic anhydride	PA	2	100	850–63,000	7000	4400	370	1	
Diethyl phthalate	DEP	1	100	430–11,000	2700	1900	2000	10	230–31,000
Dicyclohexyl phthalate	DcHxP	1	100	27–5200	180	410	210*	1	
Dimethyl phthalate	DMP	1	100	2–860	59	26	200	10	100–15,000
Personal care product (PCP, semi-quantified)									
Glycerol	GLYC	2	100	21,000–400,000	85,000	92,000	–	–	
Stearyl palmitate	16/18-WE	2	100	570–210,000	59,000	47,000	up to 20 μg/g	1	
Cetyl palmitate	16/16-WE	2	100	1200–100,000	29,000	23,000	20-200 μg/g	1	
Benzoic acid, tridecyl ester	C13-AB	2	100	1800–67,000	7800	9800	–	–	
N,*N*-Dimethyl dodecylamine	C12-DMA	2	100	580–71,000	4000	7900	–	–	
Dipropylene glycol	DPG	2	100	250–24,000	5600	4400	up to 20 μg/g	1	
Tetraethylene glycol	PEG-4	2	100	360–34,000	4500	3200	up to 20 μg/g	1	
Chloroparaffin (CP; target analysis)									
Long-chain chloroparaffins	LCCPs	1	100	3900 - 1900,000	190,000	87,000	20,000	2	5000–19,000
Medium-chain chloroparaffins	MCCPs	1	100	4900–100,000	23,000	17,000	100,000	2	31,000–100,000
Short-chain chloroparaffins#	SCCPs	1	100	2400 - 100,000	14,000	11,000	12,000	1	
Alternative plasticizer (semi-quantified)									
Tributyl acetylcitrate	ATBC	2	100	4700–76,000	23,000	22,000	5600	5	1900 - 30,000
Di-isononyl cyclohexane	DINCH	1	100	2000–130,000	20,000	13,000	13,000	4	2200 - 74,000
Dipropylene glycol dibenzoates	DPGDB	2	100	840–140,000	10,000	4300	5400	2	90–6200
Diethylene glycol dibenzoate	DEGDB	2	100	380–82,000	4600	2100	300*	1	
2-Ethylhexyl adipate	DEHA	1	100	260–15,300	2400	1800	3100	4	520–5400
Tributyl citrate	TBC	2	100	120–13,000	1600	1000	360*	1	
Triethylene glycol di(2-ethylhexoate)	TEG-EH	2	99	<50–9900	1100	580	1700	1	
Texanol #	TXIB	1	100	20–7400	820	430	1300*	1	
Tri(2-ethylhexyl) trimellitate	TOTM	2	100	79–47,000	1500	350	2300	2	1900 - 5200
Organophosphate ester (OP; target analysis)									
Tris(2-chloroisopropyl)phosphate	TClPP	1	100	320–270,000	16,000	5900	2000	17	730–65,000
Tris(2-butoxyethyl) phosphate	TBOEP	1	99	<20–410,000	17,000	3100	3800	12	830–22,000
Triphenyl phosphate	TPhP	1	100	220–110,000	3900	1200	890	19	270–4300
2-Ethylhexyl diphenyl phosphate	EHDPHP	1	99	<20–12,000	1600	790	620	9	200–2700
Bisphenol A bis(diphenyl phosphate)	BDP	1	97	<20–54,000	2300	740	140	3	35–21,000
Tris(1,3-dichloro-2-propyl) phosphate	TDCIPP	1	99	<20–200,000	5400	720	530	16	22–10,000
Tris(2-ethylhexyl) phosphate	TEHP	1	100	67–26,000	1200	600	400	5	160–1700
Tris(2-chloroethyl) phosphate	TCEP	1	100	37–9800	540	230	450	17	17–6900
Resorcinol bis(diphenyl phosphate)	RDP	1	91	<20–150,000	2700	120	39	3	2–1500
Tris(4-methylphenyl) phosphate	TpTP	1	89	<20–31,000	1200	98	190	12	20–2700
Triisobutyl phosphate	TiBP	1	54	<20–4800	200	22	500	7	46–5300
Triethyl phosphate	TEP	1	100	<20–920	90	<20	200	1	
Tributyl phosphate	TnBP	1	45	<20–2000	140	<20	110	16	28–5600
Antiblaze V6 #	V6	1	26	<20–21,000	490	<20	17	2	4–70
UV absorber (semi-quantified)									
Octocrylene	OC	1	100	1300–36,000	7000	5700	11,000	2	4400 - 18,000
2-Ethylhexyl-4-methoxycinnamat	tr-EHMC	1	100	160–28,000	2600	1500	1700	2	1500–1900
Benzophenone	BP	2	100	480–49,000	2600	1400	780*	2	750–820
2-Ethylhexyl salicylate	2EHS	2	100	44–3500	920	680	110	1	
Avobenzone	BMDBM	2	100	35–6000	1000	570	2600	1	
Homosalate	HMS	2	100	22–2900	670	490	2400*	2	83–4600
Octabenzone (Benzophenone 12)	BP12	2	100	36–5100	440	300	500	1	
Oxybenzone (Benzophenone 3)	BP3	2	100	45–2800	390	260	1900	1	
Tinuvin 326	UV-326	2	100	55–1000	180	150	140	2	73–200

(*continued on next page*)

Table 2 (*continued*)

Feature	Abbr.	CL	Freq. %	Range (min – max)	Mean	Median	Literature median*	Lit. *n*	Literature range of medians
Tinuvin P	UV-P	2	100	51–1900	210	130	220	2	94–340
Tinuvin 328	UV-328	2	100	24–9000	340	110	100	1	
Stimulant (semi-quantified)									
Caffeine	CAF	2	100	580–9800	3000	2300	2600	1	
Cotinine	COT	2	100	77–37,000	4800	1200	300	1	
Nicotine	NIC	2	100	67–290,000	12,300	990	1000	1	
Delta-9-THC (Dronabinol)	THC	2	60	<5–15,000	690	48	1300	1	
Fragrance (semi-quantified)									
Veramoss	Vera	3	100	27–15,000	980	2000	–	–	
Benzyl salicylate	BS	2	100	110–6100	1100	1400	up to 20 μg/g	1	
Galaxolide	HHCB	1	100	19–2300	460	310	900	3	750–980
Tonalide	AHTN	1	100	34–1700	250	130	410	3	320–520
Musk ketone	MK	1	99	<1–300	5.9	30	45	1	
Musk xylene	MX	1	92	<0.5–270	3.6	13	41	1	
Pharmaceutical (semi-quantified)									
Paracetamol	APAP	2	100	8–12,000	1400	500	300*	1	
Ibuprofen	IBU	2	72	<10–53,000	950	87	750	1	
Industrial chemical (semi-quantified)									
2,4-Toluene diisocyanate	TDI	2	100	140–16,000	1900	1000	2900	1	
2,4,7,9-Tetramethyl-5-decyne-4,7-diol	TMDD	2	100	63–7400	570	390	–	–	
Benzothiazolone	BTON	2	100	16–14,000	750	280	3200	1	
Textile related (semi-quantified)									
Indigo	Indigo	2	100	3–31,000	1900	850	9300	1	
N,*N*,*N*′,*N*′-Tetraacetylethylenediamine	TAED	2	100	17–9700	1200	370	up to 20 μg/g	1	
2,4-Dichloroaniline	24-ClA	2	100	34–6300	570	240	–	–	
4-Nitroaniline	4-NA	2	100	16–3700	480	310	–	–	
2-Chloro-4-nitroaniline	ClNA	2	100	14–2000	200	130	–	–	
2-Chloro-4,6-dinitroaniline	ClDNA	2	100	11–1100	160	100	CL 1*	1	
2,6-Dichloro-4-nitroaniline	DClNA	2	100	10–820	140	110	CL 2*	1	
2,4,5-trichloroaniline	245-ClA	2	100	13–1200	170	77	100	1	
Biocide (semi-quantified)									
Methylparaben	MeP	2	100	36–6100	750	530	1400	2	910–1800
Ethyl paraben	EtP	2	100	49–6800	650	420	280	1	
Propylparaben	PrP	2	100	24–6900	430	250	440	2	420–450
Triclosan	TCS	1	100	1.2–1900	220	91	220	3	87–450
Propiconazole	PrCon	2	65	<2–1600	72	9.2	640	1	
Pesticide (semi-quantified)									
Permethrin	PER	1	100	19–8300	840	290	550	3	550–770
Diethyltoluamide	DEET	1	100	14–6200	280	130	150*	1	
p,p'-DDE	DDE	1	80	<0.4–280	18	3.8	9.1	2	8.4–9.7
Cypermethrin	CyPER	2	28	<2–6800	190	<2	180	3	140–180
Phenolic antioxidant (semi-quantified)									
Butylated hydroxytoluene	BHT	1	100	140–1900	750	670	920	1	520–1300
2,4-Di-tert-butylphenol	DtBPh	2	100	73–7300	380	690	250	1	
Bisphenol (target analysis)									
4,4-Bisphenol A	BPA	1	99	<5–15,000	1400	800	1500	5	590–4700
4,4-Bisphenol S	BPS	1	100	7–1000	170	120	82	2	80–860
4,4-Bisphenol F	BPF	1	95	<0.5–8300	180	32	390	1	
Bisphenol AF	BPAF	1	57	<0.5–31	0.6	0.5	2.5	2	0.4–10
Polycyclic aromatic hydrocarbon (PAH; target analysis))									
Phenanthrene	PHE	1	100	36–4300	590	330	320	4	90–500
Fluoranthene	FLU	1	100	9–3800	330	130	120	3	73–300
Pyrene	PYR	1	100	7–3800	250	95	140	4	42–200
Benzo[b + j + k]fluoranthene	BbjkF	1	100	3–1400	120	47	42	1	
Chrysene	CHR	1	100	2–1200	100	36	23	1	
Naphthalene	NA	1	100	3–240	41	28	21	1	
Fluorene	F	1	100	3–370	40	26	42	3	24–70
Benzo[*a*]anthracene	BaA	1	100	0.8–1200	77	19	17	3	17–100
Anthracene	ANT	1	100	0.2–490	33	15	55	4	13–70
Indeno[1,2,3-*cd*]pyrene	IcdP	1	99	<0.1–720	46	15	14	1	

(*continued on next page*)

Table 2 (*continued*)

Feature	Abbr.	CL	Freq. %	Range (min – max)	Mean	Median	Literature median*	Lit. *n*	Literature range of medians
Acenaphthene	AC	1	100	0.8–140	20	14	16	1	
Benzo[*g*,*h*,*i*]perylene	BghiP	1	92	<0.1–770	42	14	33	1	
Benzo[*a*]pyrene	BaP	1	93	<0.1–810	36	10	8.4	2	7.8–9.0
Dibenzo[*a*,*h*]anthracene	DBahA	1	59	<0.1–110	10	9.9	5.2	1	
Acenaphthylene	Ap	1	100	0.4–73	10	6.0	19	1	
Alkylphenol (target analysis)									
Nonylphenol (brached)	NP(br)	1	89	<80–6900	630	410	520	1	
4-tert-octylphenol	4-t-OP	1	100	1–750	37	17	35	1	
Novel flame retardant (NFR; target analysis)									
Decabromodiphenyl ethane	DBDPE	1	60	<40–37,000	1500	160	150	15	4.3–1100
Bis(2-ethylhexyl) tetrabromophthalate	TBPH	1	81	<5–970	90	46	61	12	13–340
Tetrabromobisphenol A	TBBPA	1	53	<0.5–7400	140	7.4	10	5	6.0–610
2-Ethylhexyl 2,3,4,5-tetrabromobenzoate	EHTBB	1	43	<0.5–390	9.4	<0.5	5.2	10	1.0–10
Dechlorane Plus, anti-	DP, anti-	1	24	<0.5–96	5.2	<0.5	4.5	4	1.7–6.8
Dechlorane Plus, *syn*-	DP, syn-	1	20	<0.5–40	2.2	<0.5	4.2	6	1.0–20
1,2-Bis(2,4,6-tribromophenoxy)ethane	BTBPE	1	20	<0.5–90	2.7	<0.5	3.7	10	0.9–21
PCB (semi-quantified)									
PCB138	CB138	2	87	<0.2–540	15	1.4	6.3	4	0.9–17
PCB153	CB153	2	91	<0.2–490	13	1.2	4,3	3	0.9–7.3
PCB180	CB180	2	60	<0.2–330	8.9	0.6	4.9	3	0.7–13
PCB118	CB118	2	71	<0.1–120	3.9	0.4	6.0	2	0.9–11
Polybrominated diphenyl ether (BDE (semi-quantified)									
PBDE47	BD47	2	95	<0.1–10,000	200	1.3	8.1	19	0.6–130
PBDE99	BDE99	2	63	<0.2–15,000	330	1.0	9.2	19	1.4–170
PBDE100	BDE100	2	33	<0.2–11,000	180	<0.2	2.6	14	0.4–33
PBDE153	BDE153	2	24	<0.2–8400	140	<0.2	2.7	13	0.4–26
PBDE154	BDE154	2	23	<0.2–7800	130	<0.2	1.0	1	0.4–2.6
Perfluoro alkyl substances (PFAS; target analysis)									
Perfluorononanoic acid	PFNA	1	53	<0.05–19	1.2	0.2	1.7	6	0.1–23
Perfluorodecanoic acid	PFDA	1	56	<0.05–22	1.4	0.2	3.7	4	0.2–54
Perfluoroundecanoic acid	PFUnDA	1	43	<0.05–8.8	0.5	<0.05	1.1	3	0.6–2.9
Perfluorooctanoic acid	PFOA	1	36	<0.05–23	2.5	<0.05	14	8	0.7–300
Perfluorooctanesulfonic acid	PFOS	1	31	<0.05–98	3.6	<0.05	5.7	6	0.5–170
Perfluorododecanoic acid	PFDoDA	1	20	<0.05–7.7	0.5	<0.05	2.7	5	0.5–19

* Data not from European house dust contaminant study.

Texanol: 2,2,4-trimethyl-1,3-pentanediol diisobutyrate; Antiblaze V6: 2,2-bis(chloromethyl)-1,3-propanediyl bis(bis(2-chloroethyl) phosphate).

Three cannabinoids were detected at two orders of magnitude lower concentrations: cannabinol (CBN), delta-9-THC (THC) and cannabicoumaronone (CBCON). Among the biocides, three parabens were found at the highest concentrations, methylparaben (MeP; 0.5 μg/g), ethylparaben (EtP; 0.4 μg/g) and propylparaben (PrP; 0.2 μg/g), with triclosan (TCS) occurring at a slightly lower concentration (0.1 μg/g). The painkiller paracetamol (PARA; 0.5 μg/g) was the most abundant pharmaceutical, followed by the anti-inflammatory drug ibuprofen (IBU; 0.09 μg/g) and the hormone 17α-ethynylestradiol (EE2; 0.07 μg/g), which is used in contraceptives. The most abundant pesticides were the insecticides *trans*- and *cis*-permethrin (tr−/cis-PER; 0.3 and 0.1 μg/g) and diethyltoluamide (DEET; 0.2 μg/g). Benzoyleneurea (BU; 0.2 μg/g), a metabolite of the acaricide Fenazaquin, was found in the same concentration range.

The dust contaminant classes with the lowest median concentrations were, in decreasing concentration order, PAHs, 2-chloroethyl fatty acids, NFRs, PCBs, polybrominated diphenyl ethers (PBDEs), and PFAS. The homologue profile of PAHs was dominated by phenanthrene (PHE; 0.3 μg/g), fluoranthene (FLU; 0.1 μg/g), and pyrene (PYR; 0.1 μg/g), likely originating from road traffic emissions or tobacco smoke. Two fatty acid derivatives, 2-chloroethyl stearate (CESt) and 2-chloroethyl palmitate (CEPa), were detected at similar concentrations (0.2 μg/g). These contaminants probably originate from fumigated food items (Heikes and Griffitt, 1979). Three NFRs were frequently detected: decabromodiphenyl ethane (DBDPE; 0.2 μg/g), bis(2-ethylhexyl) tetrabromophthalate (TBPH; 0.05 μg/g) and tetrabromobisphenol A (TBBPA; 0.01 μg/g), at levels greatly exceeding those obtained for PBDEs (ca. 1 ng/g) and PCBs (ca. 1 ng/g). The least abundant dust contaminant class was PFAS, with perfluorononanoic acid (PFNA; 0.2 ng/g) and perfluorodecanoic acid (PFDA; 0.2 ng/g) being the only compounds that were frequently detected.

The dust contaminant concentrations generated as part of the current study were compared to those of recent (2009–2024) European studies of house dust (Table 2 and Table S3). When European data were not available, house dust data reported for other regions (mainly from North America) were used. These are marked with an asterisk in the "Literature median" column.

From the compilation, it becomes apparent that certain dust contaminant classes have been the subject of intensive scrutiny, i.e. phthalates, OPs, PBDEs and NFRs, and more recently alternative (non-phthalate) plasticizers, bisphenols and PFAS. The remaining compound classes have been much less studied and the benzoic acid esters, dimethyl alkyl amines, PEGs, pharmaceuticals and textile related nitroanilines have barely been investigated at all; at least not using (semi-)quantitative methods. Some of the wax esters (WE), glycols and other low-volatility contaminants have been semi-quantified in house

dust using supercritical fluid extraction and GC–MS and sorted into three concentration ranges: <20 μg/g, 20–200 μg/g and > 200 μg/g (Papadopoulos et al., 2013). In addition, two nitroanilines have been identified (2-chloro-4,6-dinitroaniline; 2Cl46NA) or tentatively identified (2,6-Dichloro-4-nitroaniline; 26Cl4NA) in house dust from North America (Kutarna et al., 2021).

The median contaminant concentrations obtained in this study align well with those reported in other European studies (Table 2). Paired (semi-)quantitative data were available for 100 compounds. More than half of the data obtained in the current study (52 %) differed <2-fold from the data reported in the scientific literature, 65 % differed <3-fold, 78 % differed <5-fold and 91 % differed less than an order of magnitude. Slightly larger deviations (up to 27-fold difference) were noted for phthalic anhydride (PA), di-isobutyl phthalate (DiBP), THC, benzothiazolone (BTON), indigo, bisphenol AF, PCB 118 and one PFAS (PFDA). The largest deviation was observed for propiconazole (70-fold). The observed deviations are comparable to the spread of medians of previously published studies, which generally range one to two orders of magnitude. Consequently, with few exceptions, the median values of the current study fall within the range of medians of other European studies (minimum 3 reference values). The exceptions are primarily found among compounds that occur at low concentration (e.g. dimethyl phthalate) or are detected at low detection frequencies (e.g. NFRs and POPs).

There are, however, a few differences. The concentrations of musk compounds, PCBs, PBDEs and PFAS are generally lower in the current study compared to the median of previous European studies. On the other hand, the concentrations of OPs tend to be higher compared to those studies. The lower bias may be attributable to past and recent voluntary and legislative efforts to reduce the use and emissions of musk compounds and POPs, which would have more effect on the current study data (later in time) than the literature data (earlier in time). Similarly, the positive bias observed for OPs may be due to an increased use of this compound class following the phase-out of other compound classes, such as phthalates and brominated flame retardants. Notably, no difference in agreement with literature median values were observed for compounds that were quantified using target analysis methods and semi-quantified using NTS methodology (often GC × GC-MS). This implies that the variability in (total ion based) MS responses are considerably lower than the variability in dust contaminant concentrations. Consequently, in the following sections, no distinction is made between data that have been generated using the two analytical approaches.

3.4. Geographical differences in contaminant concentrations and profiles

3.4.1. Differences in total concentrations and contaminant class distribution across Europe

The spatial variation in total concentration (based on the median value of each compound class) among the samples of European house dust is low (Fig. 5), with less than a factor of 2 difference in concentration between the country with the lowest concentration (Finland) and the highest concentration (the United Kingdom, UK). There is a weak spatial trend in the concentrations, increasing from north to south, with the exception of the samples from UK. However, the limited number of samples collected in UK ($n = 3$) might have impacted the UK compound class median values.

The contaminant class distribution is very similar in samples from the three Scandinavian countries (NO, FI, SE), Poland (PL) and Czech Republic (CZ). Samples from the UK have higher levels of phthalates, and samples from UK, France/Belgium (FR/BE), Hungary/Slovakia (HU/SK), and Spain/Portugal (ES/PT) have elevated levels of CPs. In addition, samples from Greece/Cyprus (GR/CY) have elevated levels of PCPs.

3.4.2. Differences in contaminant class concentrations across Europe

The concentrations differ greatly between contaminant classes. In order to spot spatial trends within all the classes, individual compound class graphs were prepared (Fig. 6). The results are organized according to the latitude of the sample origin, with samples collected in the north to the left and samples from the south to the right. In many cases, the samples from the UK show a deviating total contaminant class concentration or contaminant distribution. For example, they contain higher concentrations of di-*n*-butyl phthalate (DnBP; Fig. 6A), long-chain CPs (LCCPs; Fig. 6C) and PAHs (Fig. 6P) and lower concentrations of bisphenols (Fig. 6O) and OPs (Fig. 6E), compared to samples from other countries.

For the other countries (besides UK) geographical trends were observed for several dust contaminant classes. The concentrations (Fig. 6) of CPs (C), stimulants (G), PAHs (P) and PFAS (U) generally increased from north to south. On the contrary, the concentrations of biocides (Fig. 6M) generally decreased from north to south. For most of the remaining contaminant classes, the concentrations did not vary much (<3-fold) across Europe (Fig. 6), viz. phthalates (A), PCPs (B), alternative plasticizers (*D*), UV absorbers (F), industrial chemicals (I), phenols/antioxidants (N) and bisphenols (O). In most cases, the contaminant distribution was also constant between countries. However, relatively large between-country differences in contaminant

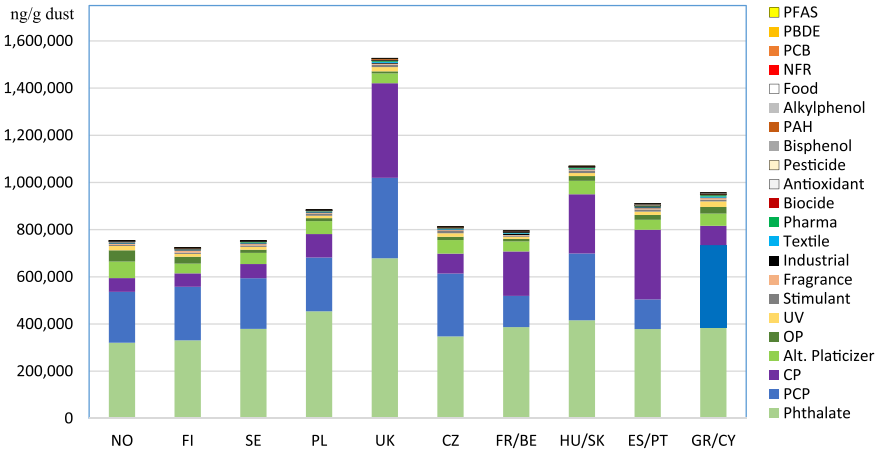

Fig. 5. Concentrations of dust contaminants in the investigated European countries (denoted by their 2-letter code, Table 1). The countries are loosely ordered north (left) to south (right). Results for some neighboring counties were pooled due to a low number of samples. CP: chlorinated paraffin; Alt. plasticizer: non-phthalate plasticizer; OP: organophosphate ester; UV: UV screen; Monomer: plastics monomer; Textile: textile related compound; PAH: polycyclic aromatic hydrocarbon; Food: food related compounds; NFR: novel flame retardant; PCB: polychlorinated biphenyl; PBDE: polybrominated diphenyl ether; PFAS: perfluorinated alkyl substances.

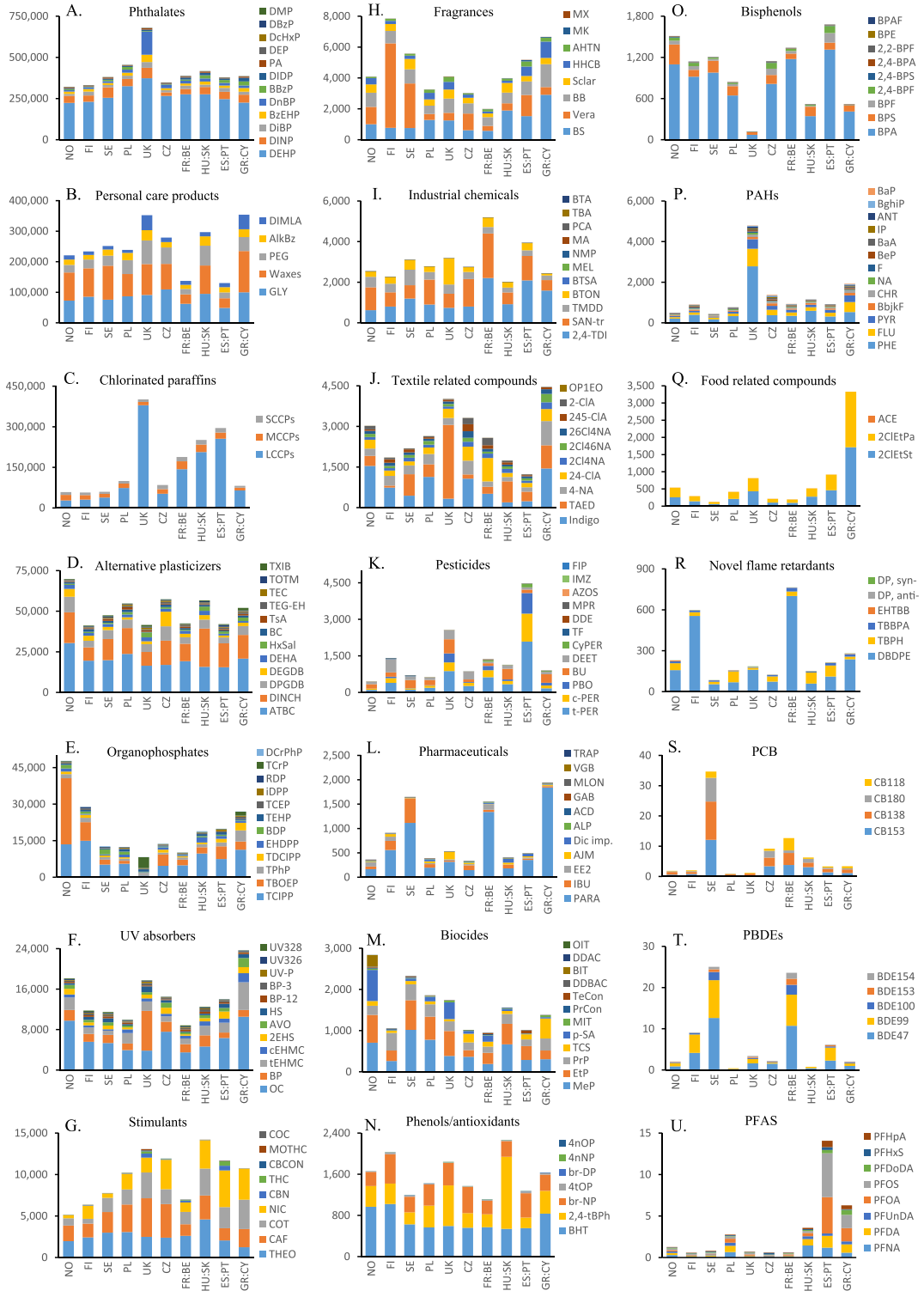

Fig. 6. Median concentrations (ng/g dust) and geographic distribution of investigated contaminants (most abundant compounds at the top left). Countries are loosely ordered north (left) to south (right) and the contaminants from the most (base) to least abundant (top). Results for some neighboring counties were pooled due to a low number of samples. Abbreviations are given in Table 2 and Supplementary material, Table S1. The personal care product chemicals were sub-divided into glycerol (GLY), wax esters (Waxes), polyethylene glycols (PEG), alkyl benzoates (AlkBz), and alkyl amines/amides (mainly dimethyl lauryl amine, DIMLA). To enhance the readability, only the 12 most abundant compounds are included.

concentrations and distribution were observed (Fig. 6) for OPs (E), fragrances (H), textile related compounds (J), pesticides (K), pharmaceuticals (L), food related compounds (Q), NFRs (R), PCBs (S), PBDEs (T) and PFAS (U), which will be further discussed below.

3.4.3. *Geographical differences in concentrations of individual contaminants across Europe*

To find individual compounds exhibiting considerable variations in concentration across Europe, the results of both the country mean calculations (Fig. 6) and the PCA (Fig. 2 and Fig. S1) were scrutinized. The

results of the two data sets were found to be highly complementary. While Fig. 6 clearly illustrates the geographical differences in contaminants with high (relative) abundance, the PCA highlight the contaminants and samples that differ the most within the entire data set. The PCA model of the log-transformed and auto-scaled (mean centered and scaled to unit variance) house dust data had three significant principal components (PCs), which together explained 90.4 % of the total variance. The first PC explained most of the variance (88.5 %) and PC2 and PC3 explained approximately 1 % each. As noted in section 3.2, the three PCs seem to capture variations related to contaminant concentration (PC1), sample type (individual/pooled; PC2) and the latitude of sampling location (north-south; PC3), respectively. A scores 3D-plot, which illustrates the groupings and trends among samples, is shown in Fig. 2. The corresponding loadings, which illustrate the contaminant distributions, are presented as two separate graphs (PC1 vs. PC2 and PC2 vs. PC3) in Fig. S1.

The loading plot of PC2 and PC3, especially the PC3 loadings (Fig. S1), was found to be useful in tracing dust contaminants that show geographical differences across Europe. At the top of the graph, with high PC3 loadings, we find compounds and compound classes that are elevated in samples from southern Europe. These include LCCPs, several stimulants related to tobacco (nicotine, NIC; cotinine, COT) and hemp (THC; cannabinol, CBN; cannabidiol, CBD; cannabigerol, CBG; cannabichromene, CBC), the synthetic hormone EE2, five PFAS (incl. PFOS and PFOA), two aromatic musk fragrances (musk ketone, MK; musk xylene, MX), several PAHs (incl. Fluoranthene, FLU; pyrene, PYR), pyrethroid pesticides (permethrins, PER; cypermethrins, CyPER), and the pyrethroid synergist piperonyl butoxide (PBO). In the lower part of the graph, we find contaminants that are elevated in samples from northern Europe. Those include several pharmaceuticals (incl. Ibuprofen, IBU and gabapentin, GAB), several emollients (PEGs and wax esters), the fragrance veramoss (Vera), several biocides (incl. Parabens; 1,2-benzisothiazol-3-one, BIT and 2-octyl-4-isothiazolin-3-one, OIT), two OPs (TBOEP and bisphenol A bis(diphenyl phosphate), BDP), indigo dye, the insecticide DEET and the fungicide tolylfluanid (TF). Some of these trends were observed already when evaluating the contaminant class concentrations in section 3.4.2 (Fig. 6), but the PCA provided clearer insight into the geographical trends for low abundance contaminants.

The observed distribution of samples along PC2 (Fig. 2), with individual samples at low PC2 values and pooled samples at high PC2 values, was unexpected. However, the loading plots of PC1 vs. PC2 and PC2 vs. PC3 (Fig. S1) revealed that this separation is mainly caused by higher levels of PCBs and PBDEs in pooled samples, as compared to individual samples. The two compound classes, especially PBDEs, appear at low median concentrations and detection frequencies (PBDE: 23–91 %) in the house dust samples (Table S2). Moreover, Fig. 3 shows a huge variability (4-orders of magnitude) in total PCB and PBDE concentrations. Thus, when these contaminants are present, they can be present at very high concentrations, and by pooling sub-samples the probability of including a sample with high concentrations will increase.

It is important to note that not all observed variations in individual compound concentrations were fully captured or explained by the PCA. For example, the high concentrations of 2-chloroethyl fatty acids (Fig. 6Q) in samples from southern Europe, the high concentrations of DBDPE (Fig. 6R) in samples from Finland (FI), France and Belgium (FR: BE), and the high concentrations of paracetamol (PARA; Fig. 6L) in samples from Sweden (SE), France and Belgium, and Greece and Cyprus (GR:CY) could not be easily explained.

3.4.4. Comparison of contaminant profiles of house dust from Europe and North America

In order to test our main hypothesis, that similar chemicals and products are used in developed countries, leading to similar contaminant concentrations and contaminant profiles in residential house dust, we used the data reported from the first (2016) and second (2021) NORMAN collaborative trial (CT) on suspect and non-target screening of house dust. The first CT used pooled residential dust from North America (Canada) and the second CT pooled residential dust from Europe (Fig. 1). A similar number of organizations joined the exercises (ca. 20) and many of the reporting laboratories joined both exercises. There was a similar distribution of laboratories that reported LC-MS (ca. 2/3) and GC-MS (ca. 1/3) data. The resulting compound lists from both trials were assessed in parallel to ensure comparability.

Biogenic compounds were excluded (manually) during the data evaluation, as they were not within the scope of the study. Only compounds with (at least) tentative structures, i.e. identification confidence level 3 or better, were included. The anthropogenic contaminants were then assigned functional use categories (Fig. 7) as described in the following paragraph. Some of the detected chemicals have no (current) intentional use, but are still present in dust. Those are primarily found in the polycyclic aromatic compound (PAC) and persistent organic pollutant (POP) categories. PFAS can be classified as POPs, but have current uses as water, soil and stain repellents in furniture, rugs and carpets (Savvaides et al., 2021) and were therefore sorted into the textile category. These articles are likely to release textile fibers that end up in floor dust, which was the target matrix. There are also several transformation products and metabolites, mainly in the pharmaceutical and pesticide classes, which were grouped with the parent compound.

Petroleum hydrocarbons, PACs, POPs, phthalates, and OPs were assigned contaminant classes according to chemical class and prior knowledge. In the classification of many of the remaining compound classes the US-EPA CPDat and FUse databases proved valuable (Dionisio et al., 2018). The databases were downloaded and information about main functional use (FU) categories were extracted. Matching was done using CompTox identifiers (DTXSID). In cases when reported FU information was missing, QSAR probability scores were used to find potential FU categories, utilizing an 80 % probability cutoff (Phillips et al., 2017). When multiple FU categories were indicated, priority was given to FU categories that are most relevant for residential dust, e.g. skin conditioner (PCP) would be preferred over lubricant (Industrial). Compounds and compound classes, e.g. pesticides and pharmaceuticals, which were not covered by the databases and were not familiar to the researchers performing the classification, were manually searched for in public databases (mainly PubChem), patent databases, chemical industry databases and market analysis sites.

A comparison of the contaminant class distributions (number of compounds per compound class) for the pooled house dust sample from Europe and North America, respectively, is presented in Fig. 7. For most contaminant classes, the number of reported compounds for European and North American dust were similar. However, a notable disparity was observed in the surfactant class, with three times as many surfactants detected in North American dust (329) as compared to those detected in European dust (122). This difference is particularly pronounced in the high-mass range (compounds with molecular mass greater than PEG-10; 458 g/mol) in which 252 compounds were found in North American dust versus only 30 compounds in European dust. There may be several reasons for this, including differences in household cleaning products used in the two regions or differences in MS libraries or analytical approaches employed in the two CTs. The number of small surfactants (up to PEG-10) were similar in North American (63) and European (92) house dust.

Some moderate differences in substance count between European and North American house dust were observed for some other compound classes. European dust contained a greater number of PCPs (105 vs. 74), textile related chemicals (56 vs. 22), stimulants (33 vs. 12) and biocides (45 vs. 24), whilst North American dust contained a greater number of pharmaceuticals (155 vs. 125) and industrial chemicals (192 vs. 170). However, overall, there were more similarities than differences in compound class distribution.

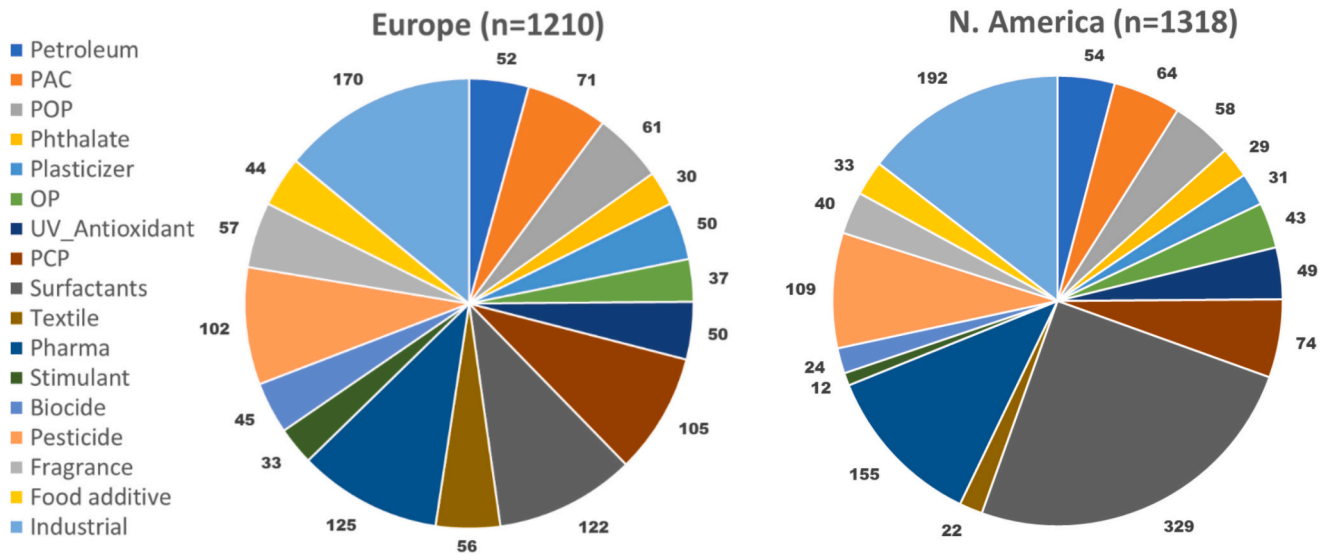

Fig. 7. Comparison of contaminant class distributions (number of compounds per compound class) for house dust from Europe and North America, respectively. PAC: Polycyclic aromatic compound; POP: Persistent organic pollutant; OP: Organophosphate ester; UV: UV absorbers; PCP: Personal care product; Textile: Textile related compound.

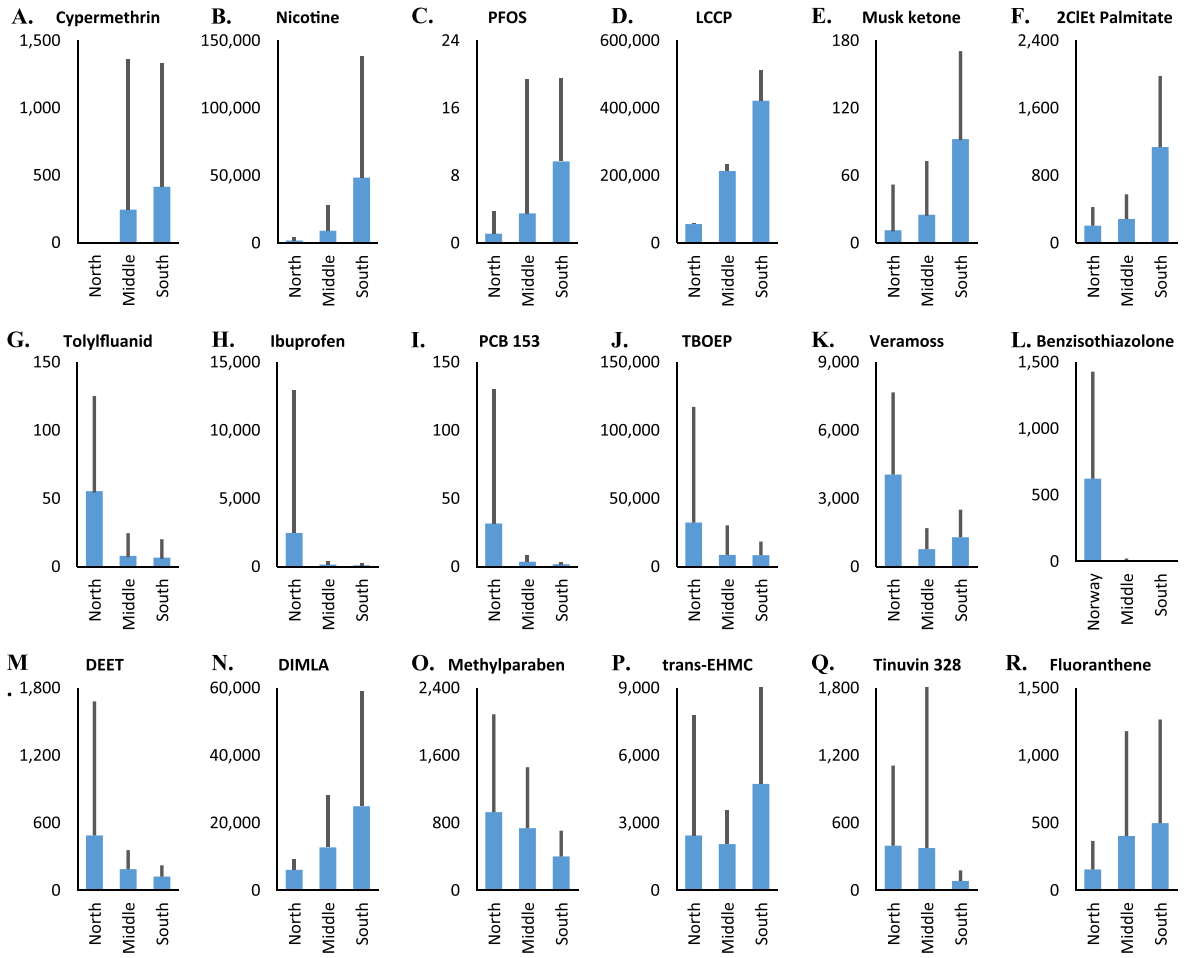

Fig. 8. Average concentrations (ng/g dust) and standard deviations (only positive error bars included) of compounds that exhibits spatial trends in their geographic distribution. Compounds on the top row are generally higher in South Europe than in the other regions and compounds on the middle row are generally higher in North Europe as compared to the other regions. The compounds shown on the lower row show less pronounced trends in their geographic distribution. Abbreviations are given in Table 1; except for DIMLA: dimethyl lauryl amine; 2ClEt Palmitate: 2-chloroethyl palmitate.

3.5. Discussion of contaminant concentrations and geographic distribution in relation to contaminant physicochemical properties, functional use categories and consumer habits and preferences

Most compounds detected in the CT were SVOCs with low or moderate polarity. Among the 1210 identified compounds, the median molecular weight of the 1210 identified chemicals was 260 g/mol (range 74–990 g/mol) and the median log octanol-water partition coefficient (log P) was 3.9 (min −4.1; max 17) (ChemOffice, Ver. 22; ACD/Labs, Toronto, ON, Canada). This can mainly be related to emission and distribution processes in the indoor environment and to the choice of analytical techniques. The most volatile compounds tend to partition into the gas phase in indoor air, and therefore, are less likely to accumulate in dust. On the contrary, low-volatility compounds do not evaporate from building materials and household articles to any major extent. Although surface abrasion could result in release of high-boiling compounds, this emission process seems to be of less importance. Moreover, although the LC-MS/MS and GC-MS techniques used by the participating laboratories cover a wide polarity range, the most polar contaminants are generally not separated by regular reversed-phase HPLC columns and are therefore rarely identified. The coverage of the chemical domain may be expanded into the high-polarity range using HILIC chromatography, but only one of the laboratories used that option.

The investigated anthropogenic dust contaminants may be loosely grouped into three categories: i) intentionally used chemicals with relatively fast release (e.g. PCP and household cleaning products), ii) diffusively emitted chemicals with slow release (e.g. plastics additives and textile related chemicals), and iii) unintentionally formed chemicals (e.g. traffic and smoke related compounds, including PAHs, and various transformation products). The chemicals that belong to category ii) are expected to show less temporal and spatial variability than the chemicals that belong to category i). Consequently, we do not observe any strong geographical trends across Europe (Fig. 6) for the following classes of diffusively emitted chemicals: phthalates, alternative plasticizers, UV absorbers, industrial chemicals, textile related compounds, antioxidants and alkylphenols. There are, however, a few exceptions. The LCCP and PFOS concentrations increase from north to south (Fig. 8C and Fig. 8D), whilst those of PCB 153 and TBOEP (Fig. 8I and Fig. 8J) show an opposite trend. The latter may be related to frequent use of PCBs in building sealants (Sundahl et al., 1999) and of TBOEP in floor polish for vinyl and wood flooring (present at 0.5–5 %) (Marklund et al., 2003; US CPID, 2024), which are commonly used in northern Europe. The reasons behind the elevated levels of LCCPs and PFOS in southern Europe are less clear and require further investigation.

Geographical trends in contaminant distributions were more frequently found for compounds belonging to the first category, i.e. PCPs and fragrances, pharmaceuticals, pesticides, and biocides (Fig. 6). These types of chemicals are generally acquired by the user or a family member and used shortly after purchase. Regional differences in user preferences or changes in user behavior are expected to have a greater and faster effect on the concentrations of house dust contaminant. Consequently, opposite trends (north-south) may be expected within the same functional use category, as observed in several cases.

According to the PCA of the dust contaminants Fig. S1, it is evident that different fragrances and pharmaceuticals are preferred by customers and medical doctors in southern and northern Europe. The two traditional fragrances musk xylene (MX) and musk ketone (MK; Fig. 8E) are prevalent in samples from southern Europe, whereas the base note veramoss (Vera; Fig. 8K) is associated with samples from northern Europe, in particular Finland. The former observation aligns with consumer statistics, showing high consumption of fragrances in southern Europe. For instance, three quarters of all women in Spain and France use fragrance at least once a day (Morean, 2009). In contrast, the Finnish personal beauty product market is dominated by young people (16 to 45 years) and they mainly use PCP products such as deodorants, shaving gels and creams, and oral and skin care products (Statistica, 2024). These products often contain masking compounds and base notes, such as veramoss. In the pharmaceutical class, the synthetic hormone EE2 is associated with samples from southern Europe, while ibuprofen (IBU; Fig. 8H), gabapentin (GAB), vigabatrin (VGB) and phenoxybenzamide (PBZ) are more prevalent in samples from northern Europe. The elevated levels of EE2 in samples from southern Europe could be associated to the country birth rates reported in Europe, which are the lowest in Greece, Italy and Spain (Eustat, 2024). There are also regional differences in the prescription of the antiepileptic and pain-relief drug gabapentin across Europe. Usage rates, expressed as defined daily dose per ten thousand inhabitants per day (DDD/TID), increased from Eastern Europe (15 DDD/TID), through Southern and Western Europe (27 and 51 DDD/TID), to Northern Europe (55 DDD/TID) (Chan et al., 2023). This type of region specific information was difficult to find for the other three drugs. However, steep increases in IBU have been reported in Scandinavia, especially in Finland, over recent years (Kristensen et al., 2019) with a peak during the onset of the Covid-19 pandemic (Karlsson et al., 2021). This surge in use was mainly attributed to increased over-the-counter sales of the drug; currently ca. 10 % is prescribed (Kristensen et al., 2019). Finally, no geographic trend is observed for paracetamol (PARA), which is the pharmaceutical that is found at the highest concentration in most countries (Fig. 6), indicating an overall high usage and great variability in consumption between households.

The pyrethroid insecticides permethrin and cypermethrin (Fig. 8A) and the pyrethroid synergist piperonyl butoxide were more abundant in samples collected in southern Europe, whilst the insecticide DEET (Fig. 8M) and biocides tolylfluanid, benzisothiazolone (BIT) and methylparaben (Fig. 8G, L and O) were more abundant in samples from northern Europe. The higher levels of pyrethroids in the south may be explained by more extensive agricultural activities in that region, as compared to the north, which is more dominated by forestry. On the other hand, the cold and wet climate in northern Europe is associated with the risk of condensation and mold formation and the need for biocide use in building materials and paint. This may explain the high abundances of tolylfluanid and BIT in this region. Notably, BIT was only found in Norwegian dust samples, at a detection frequency of 100 %, and in one sample from Poland. According to the usage instructions and safety data sheets of major Scandinavian paint producers, tolylfluanid and BIT are used in both indoor and outdoor products at concentrations up to 0.5 % and 1 %, respectively. The high precipitation in the boreal region is also associated with massive mosquito swarms in summertime, leading to an increased use of DEET (Fig. 8M) as an insect repellant. In all these cases, both indoor and outdoor sources likely contribute to the contaminant levels in house dust.

In category iii), unintentionally formed chemicals, we found traffic related compounds, compounds found in smoke from tobacco and hemp, and compounds formed during food processing. The concentrations of nicotine (Fig. 8B) and PAHs, such as fluoranthene (Fig. 8R) decrease with latitude in European house dust. The highest levels are found in southern Europe, in agreement with expectations – given the higher population density and greater air pollution in southern and central Europe compared to northern Europe. This trend is reflected in the levels of NOx (proxy for traffic-related pollution), which are higher in southern and central Europe and lower in Scandinavia (EFA, 2024). Smoking habits vary across Europe, and the observed nicotine concentrations generally mirror the cigarette consumption, with high consumption in southern Europe and low in Scandinavia (EFA, 2024). Both traffic and smoking are associated with PAH emissions.

The cannabis-related compounds follow the same spatial pattern, with constituents such as THC and cannabinol (CBN) strongly associated with samples from south and south-west Europe in the PCA analysis (Fig. 2 and Fig. S1). The 2-cloroethyl derivatives of fatty acids (2CA-FAs), also follows the same trend (Fig. 8F). These compounds are formed upon ethylene oxide (EO) treatment of food materials and the highest levels of EO residues and 2CA-FAs have been reported in spices and

herbs (Jensen, 1988; Ansari et al., 1995). According to a recent report (Bromham et al., 2021) there is a correlation between temperature and use of spices: the hotter the temperature, the hotter the food. This suggests that the consumption of spices and herbs may be higher in southern Europe than in northern Europe, which could explain the observed spatial trend for 2CA-FAs.

Finally, some compounds have diverse applications, making them difficult to categorize. For example, UV absorbers are used both as UV screens in PCPs (e.g. *trans*-EHMC) and as light stabilizers in polymers (e.g., Tinuvin 328). For these two UV absorbers, spatial trends were less pronounced (Fig. 8P and Fig. 8Q). Surface-active compounds is another group of compounds with multiple uses. One of these, dimethyl lauryl amine (DIMLA), shows a relatively strong spatial trend (Fig. 8N) that is difficult to explain as this chemical is used in plethora of applications, namely as: surfactants in cleaning and disinfecting agents, emulsifiers in paints, coatings, and adhesives, and corrosion inhibitors, antistatic agents, foaming agents, and textile softeners.

4. Potential risks associated with dust contaminants

The calculated EDIs were compared with ADIs via ingestion and the results are presented in Table S5. A graph comparing EDI with ADI values of the investigated dust contaminants is shown in Fig. S2. The lowest ADI values, which corresponds to the compounds with the highest health hazards, were obtained for PFAS (PFOS, PFOA, PFNA, PFHxS – in the range of 0.02 to 0.002 μg/kg/day). However, due to their low concentrations and detection frequencies in the dust, the exposure to these compounds remained well below the respective ADI. The highest ADI were generally observed for PCP compounds and fragrances (typically 10–30 mg/kg/day), for which exposures are high; but calculated EDIs were still below ADI values. However, it is important to note that for many PCP compounds, dermal exposure is expected to be more substantial than dust ingestion.

Moreover, there were a striking lack of data on toxicity for a large number of the compounds detected in indoor dust and, therefore, the resulting risk ratios will have to be treated with caution. Of the 202 compounds for which ADIs were examined, 69 had no available data in CompTox reflecting chronic ingestion exposure to mammals. Furthermore, of the 133 for which values in some form were available, 81 of the values were of limited confidence, e.g., extrapolated based on a single NOAEL (no observed adverse effect level) value, from NOAELs covering multiple order-of-magnitude ranges, and/or with unspecified endpoints. Only 46 compounds had consensus-based ADI values that can be considered reliable.

Of the 50 compounds with the highest concentrations in house dust, and consequently the highest EDIs, only seven have consensus-based safe exposure thresholds: DEHP, DnBP, PA, ethylene glycol, BBzP, styrene acrylonitrile trimers, and TCIPP. A further 20 compounds had NOAEL-type values from which some indication of safe exposure levels can be derived. Twenty-three of the 50 highest concentration compounds in European dust had no toxicity data covering safe levels of ingestion exposure.

According to Table S5, no individual compounds had median EDIs that were close to the ADIs, which is encouraging. When comparing median EDIs and ADIs, only two compounds had EDI/ADI ratios >0.01: 2,4,5-trichloroaniline, a chlorinated aromatic amine used in production of dyes, pigments and pesticides; and DEHP. Considering the 90th-percentile EDI, seven compounds had EDI/ADI ratios >0.01: in addition to 2,4,5-trichloroaniline and DEHP, three PBDE congeners (BDEs 47, 99 and 153), PCB 118, and *N,N*-Dimethyl docosylamine, a PCP compound used in hair products (Table S5).

However, despite the widespread presence of in-use contaminants in indoor dusts, some of the highest risks for highly exposed populations come from compounds that have been restricted, some for several decades. This observation, also previously noted in Central European house dust (Demirtepe et al., 2019), suggests a lag-time between regulatory actions and the subsequent reductions in indoor levels of the regulated agents. Moreover, while comparing individual ADIs and EDIs can be useful to prioritize CECs from extended chemical lists, it does not account for mixture effects. This is important given that there is a clear understanding that dust contains a complex mixture of 1000 or more CECs. Considering the 90th-percentile exposure scenario, five out of the seven compounds with the highest risk are associated with neurodevelopmental effects, such additive exposures have the potential to increase risk, highlighting the need for more studies examining the toxicity profiles of chemical mixtures in indoor dusts (Pinto-Vidal et al., 2024).

The continued presence of restricted chemicals in indoor dust can be attributed to factors such as: the putative large stock of the regulated chemicals in society, increased use of recycled materials and the long lifespan of the articles to which they have been added. It is also likely related to the difficulties associated with efficiently regulating a global market, particularly for chemicals such as DEHP where regulations in Europe are stricter than those in some other jurisdictions. The results highlight the need for proactive measures to prevent hazardous and environmentally harmful chemicals from entering the market and emphasize the importance of carefully selecting substitute chemicals, when such are needed, to avoid regrettable substitutions.

CRediT authorship contribution statement

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Declaration of competing interest

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.177639.

Data availability

A list of European house dust contaminants has been uploaded to NORMAN SLE (S120 - DUSTCT2024) and Zenodo (**doi:**https://doi.org/10.5281/zenodo.13835255**).** Additional data will be made available on request.

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