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Consumer Product Chemicals in Indoor Dust: A Quantitative Meta-analysis of U.S. Studies

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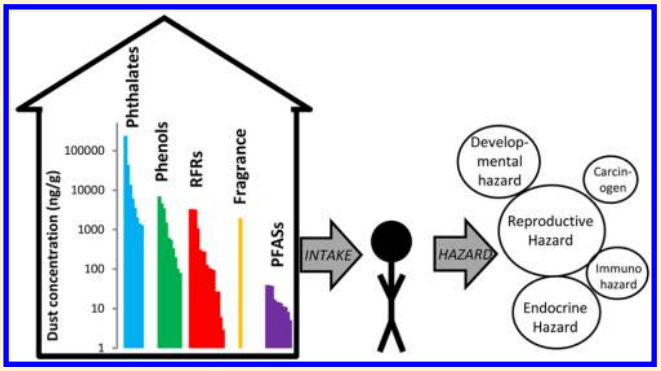
Supporting Information

ABSTRACT: Indoor dust is a reservoir for commercial consumer product chemicals, including many compounds with known or suspected health effects. However, most dust exposure studies measure few chemicals in small samples. We systematically searched the U.S. indoor dust literature on phthalates, replacement flame retardants (RFRs), perfluoroalkyl substances (PFASs), synthetic fragrances, and environmental phenols and estimated pooled geometric means (GMs) and 95% confidence intervals for 45 chemicals measured in ≥3 data sets. In order to rank and contextualize these results, we used the pooled GMs to calculate residential intake from dust ingestion, inhalation, and dermal uptake from air, and then identified hazard traits from the Safer Consumer Products Candidate Chemical List. Our results indicate that U.S. indoor dust consistently contains chemicals from multiple classes. Phthalates occurred in the highest concentrations, followed by phenols, RFRs, fragrance, and PFASs. Several phthalates and RFRs had the highest residential intakes. We also found that many chemicals in dust share hazard traits such as reproductive and endocrine toxicity. We offer recommendations to maximize comparability of studies and advance indoor exposure science. This information is critical in shaping future exposure and health studies, especially related to cumulative exposures, and in providing evidence for intervention development and public policy.



INTRODUCTION

People in developed countries spend more than 90% of their time in indoor environments,[1] creating an important link between indoor environmental quality and public health. Consumer products and building materials including furniture, electronics, personal care and cleaning products, and floor and wall coverings contain chemicals that can leach, migrate, abrade, or off-gas from products resulting in human exposure.[2,3] Consequently, chemicals such as phthalates, phenols, flame retardants, and polyfluorinated alkyl substances (PFASs) are widely detected in the U.S. general population, including vulnerable populations such as pregnant women and children.[4−6]

Exposure to one or more of these chemical classes has been associated with adverse health effects including reproductive toxicity, endocrine disruption, cognitive and behavioral impairment in children, cancer, asthma, immune dysfunction, and chronic disease.[7−9]

Many emerging and current use consumer product chemicals of concern are semivolatile organic compounds (SVOCs), which exist in the gas and condensed phase and redistribute from their original source over time, partitioning between indoor air, dust, and surfaces. Consequently, exposure to SVOC chemicals in the indoor environment may occur from air, dust, and dermal pathways.[10−13] For some phthalate diesters, the use of consumer products and indoor exposures are major contributors to human exposure.[14−17] Similarly, for some flame retardants, dust is a significant contributor to exposure,[18−20] while the contribution of the indoor environment to total exposure of PFASs is less well characterized.[21,22] The chemical properties, sources, exposure pathways and major health effects associated with each chemical class are reviewed in the following sources: phthalates,[16,23−26] flame retardants,[25,27,28] environmental phenols,[25,26,29,30] synthetic fragrances,[29,31,32] and PFASs.[25,33,34]

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Dust can provide critical information on consumer product chemicals in the indoor environment.[35] First, it is a window into which chemicals are present indoors.[36,37] Second, because SVOCs partition between air and dust in the indoor environment, dust concentrations can be used in equilibrium partitioning models to estimate air concentrations and characterize total residential intake with reasonable accuracy.[38,39] Finally, characterizing exposures from indoor dust may have important implications for children's health. Young children are particularly vulnerable to chemical exposures from dust since they crawl, play on the floor, and frequently put their hands in their mouths.[40] Increased dust contact likely plays a role in the higher body burden of flame retardants in young children compared to their parents.[18,19,41−44]

However, it is difficult to obtain a comprehensive assessment of current use chemicals in dust because few indoor dust studies report on a broad range of consumer product chemicals.[26,39,45−47] Most studies measure only one or two chemical classes, and a number emphasize legacy chemicals like PBDEs. Further, the small sample sizes and convenience populations used in most studies make it difficult to assess generalizability to a broader population. Comprehensive estimates of consumer product chemical concentration patterns and common coexposures in environmental media are needed to prioritize chemicals and better understand potential cumulative exposures and impacts.[48−51]

Given these existing data gaps, the objective of our study is to synthesize indoor dust data for a wide suite of consumer product chemicals and assess implications for human exposure and health. Specifically, we aggregate dust data measured in US indoor environments, focusing on the following SVOC classes: phthalates, RFRs (also known as novel FRs), environmental phenols, synthetic fragrance, and PFASs. From this data aggregation, we calculate pooled concentrations, use these pooled concentrations to estimate residential intake, and then describe hazard traits of the chemicals to provide context for the potential health effects. Finally, we make recommendations for future exposure research.

MATERIALS AND METHODS

Systematic Literature Search. We first conducted a screening-level literature search to identify current-use classes of SVOC consumer product chemicals measured in dust. The preliminary search identified five chemical classes: phthalates and phthalate replacements, RFRs, PFAS, synthetic fragrances, and environmental phenols. Other chemical classes in dust that were not included in this analysis include legacy chemicals, combustion byproducts, pharmaceuticals, pesticides, and metals. Legacy chemicals, such as PBDEs and polychlorinated biphenyls (PCBs), are not currently used in U.S. commerce. Combustion byproducts (e.g, polycyclic aromatic hydrocarbons) are not commonly found in consumer products. Pharmaceuticals and pesticides are not "consumer products" as defined in the U.S. Consumer Product Safety Act, and metals are not SVOCs.[3,52,53]

We then conducted a comprehensive literature search for dust analysis studies in February 2015, using PubMed and Web of Science databases for all document types. In order to capture data that would be most informative on contemporary dust composition, we limited our search to studies published during or after the year 2000. The search terms used were [flame retardant*] AND [dust*], [fragrance*] OR [musk*] AND [dust*], [perfluorin*] OR [polyfluorin*] AND [dust*], [phthalate*] AND [dust*], [alkylphenol*] OR [BPA*] OR [paraben*] AND [dust*], and ["semivolatile organic"] OR ["semivolatile organic"] OR [SVOC] AND [dust*]. We also included two unpublished data sets provided by members of our study team.

Studies met the eligibility criteria if samples were collected: during or after the year 1999, in the United States, indoors (residential, nonresidential, and vehicle environments), and using a vacuum cleaner (either study vacuumed or from an existing used bag). Studies were excluded if: they collected samples in an international airplane, did not measure chemicals of interest, did not report on primary data, or were not in English. Three additional studies and 1 unpublished data set were excluded during preparations for the meta-analysis because samples measured in the U.S. were not analyzed separately from international samples; they relied on data previously published in another study; or no quantitative data were reported (Figure 1).

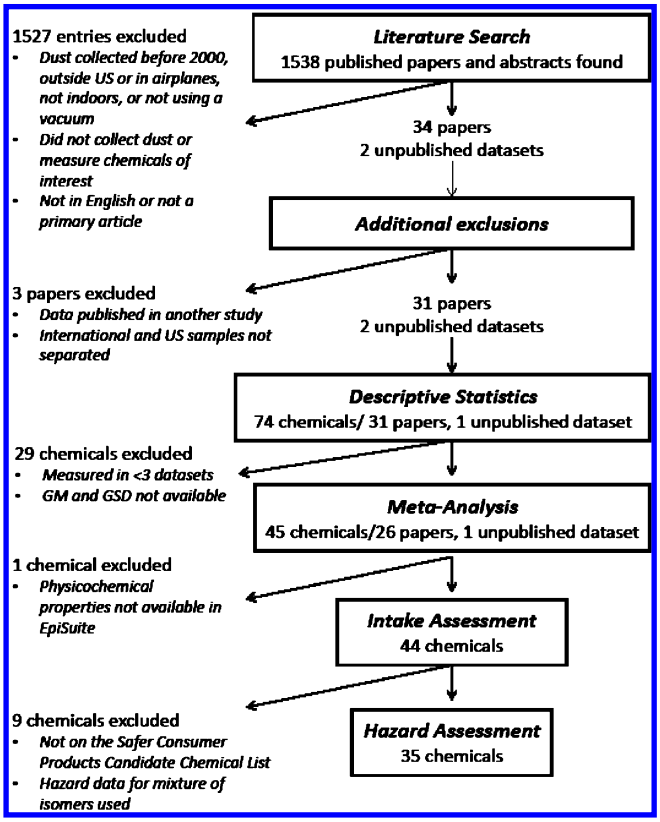

Figure 1. Exclusion criteria and number of studies or chemicals included at each stage of the analysis.

Meta-Analysis. Only chemicals that were measured in ≥3 data sets were eligible for inclusion in the meta-analysis. We collected descriptive information from the eligible published and unpublished studies, including chemicals measured; geographic location; microenvironment sampled, study year; dust collection and storage methods (Supporting Information (SI) Table S1); analytical methods; and quality control measures (SI Table S2). We also collected quantitative information, including sample size, method detection limits (MDL), percent of samples above the MDL, and the following summary statistics: minimum; 10th, 25th, 50th, 75th, 90th, and 95th percentiles; maximum; mean; standard deviation; geometric mean (GM); and geometric standard deviation (GSD). Twenty-nine of 31 papers and the one unpublished data set were missing information determined critical for between-study comparison (collection method, sieve

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Table 1. Chemicals in the Meta-Analysis (45 Chemicals), Intake Assessment (44 Chemicals), and Hazard Identification (35 Chemicals)[a]

chemical common abbreviation	common name(s), other abbreviations	CAS RN	meta-analysis	intake assessment	hazard identification
Replacement Flame Retardants (RFRs)[b]					
TDCIPP	chlorinated tris; Tris(1,3-dichloroisopropyl) phosphate; TDCPP	***13674−87−8***	X	X	X
TCIPP	tris(2-chloroisopropyl) phosphate; TCPP	***13674−84−5***	X	X	X
TCEP	tris(2-chloroethyl) phosphate	***115−96−8***	X	X	X
TPHP	triphenyl phosphate; TPP; TPhP	***115−86−6***	X	X	X
HBCDD	hexabromocyclododecane; HBCD; includes alpha-, beta- and gamma- (aHBCD, bHBCD, gHBCD) isomers	*25637−99−4*; ***3194−55−6***[c]	X	X	X
aHBCDD	alpha- hexabromocyclododecane (aHBCD)	134237−50−6; ***3194−55−6***[c]	X	X	
bHBCDD	beta- hexabromocyclododecane (bHBCD)	134237−51−7; ***3194−55−6***[c]	X	X	
gHBCDD	gamma- hexabromocyclododecane (gHBCD)	134237−52−8; ***3194−55−6***[c]	X	X	
BEH-TEBP	bis(2-ethylhexyl) tetrabromophthalate; TBPH	***26040−51−7***	X	X	X
BTBPE	1,2-Bis(2,4,6-tribromophenoxy)ethane	***37853−59−1***	X	X	X
DBDPE	decabromodiphenyl ethane	***84852−53−9***	X	X	X
TBBPA	tetrabromobisphenol A	***79−94−7***	X	X	X
EH-TBB	(2-ethylhexyl)tetrabromobenzoate; 2-Ethylhexyl 2,3,4,5-tetrabromobenzoate	183658−27−7	X		[i]
aDDC−CO	anti-dechlorane plus (aDP)	135821−74−8; ***13560−89−9***[d]	X	X	X
sDDC−CO	syn-dechlorane plus (sDP)	135821−03−3; ***13560−89−9***[d]	X	X	X
Phthalates and Phthalate Alternatives[e]					
BBzP	butyl benzyl phthalate; BBzP	***85−68−7***	X	X	X
DEHA	bis(2-ethylhexyl) adipate; di (2-ethylhexyl adipate)	***103−23−1***	X	X	X
DEHP	di-2-ethylhexyl phthalate; bis(2-ethylhexyl) phthalate; dioctyl phthalate; DOP	***117−81−7***	X	X	X
DnBP	dibutyl phthalate, di-*n*-butyl phthalate	***84−74−2***	X	X	X
DEP	diethyl phthalate	***84−66−2***	X	X	X
DiBP	diisobutyl phthalate; DiBP	***84−69−5***	X	X	X
DnHP	di-*n*-hexyl phthalate; DnHP; DNHP; DHEXP	***84−75−3***	X	X	X
DnOP	di-*n*-octyl phthalate; DOP	***117−84−0***	X	X	X
Environmental Phenols					
BPA	bisphenol A	***80−05−7***	X	X	X
MeP	methyl paraben; Me-PHBA; methyl p-hydroxybenzoate; methyl 4-hydroxybenzoate	***99−76−3***	X	X	X
EtP	Ethyl paraben; Et-PHBA; Ethyl p-hydroxybenzoate; ethyl 4-hydroxybenzoate	***120−47−8***	X	X	X
BuP	butyl paraben; bu-PHBA; butyl *p*-hydroxybenzoate; butyl 4-hydroxybenzoate	***94−26−8***	X	X	X
NP	4-nonylphenol; nonylphenol; 4-NP	*25154−52−3*; **84852−15−3**[f]	X	X	X
NP1EO	nonylphenol monoethoxylate	***9016−45−9***[g]	X	X	X
NP2EO	nonylphenol diethoxylate	***9016−45−9***[g]	X	X	X
2,4-DHBZON	BP-1; 2,4-dihydroxybenzophenone; benzophenone-1; (2,4-dihydroxyphenyl) phenyl methanone	**131−56−6**	X	X	
OP1EO	octylphenol monoethoxylate; 4-tert-octylphenol monoethoxylate	2315−67−5; **9036−19−5**[h]	X	X	
OP2EO	octylphenol diethoxylate; 4-tert-octylphenol diethoxylate	2315−61−9; **9036−19−5**[h]	X	X	
Perfluoroalkyl Substances (PFAS)					
PFOA	Perfluorooctanoic acid; Perfluorooctonoate (C8)	***335−67−1***	X	X	X
PFHxS	Perfluorohexanesulfonate, perfluorohexanesulfonic acid (C6)	***355−46−4***	X	X	X
PFOS	Perfluorooctanesulfonate; Perfluorooctanesulfonic acid (C8)	***1763−23−1***	X	X	X
PFNA	Perfluorononanoic Acid; Perfluorononanoate (C9)	***375−95−1***	X	X	X
PFDA	perfluoro-n-decanoic acid; perfluorodecanoic acid; perfluorodecanoate; PfDeA (C10)	***335−76−2***	X	X	X
PFBS	perfluorobutanesulfonate; perfluorbutanesulfonic acid; nonafluorobutanesulfonic acid; nonafluorobutanesulfonic acid; PFBuS (C4)	***375−73−5***	X	X	X
PFHpA	perfluoroheptanoic acid (C7)	***375−85−9***	X	X	X
PFDoA	perfluoro-*n*-dodecanoic acid; perfluorododecanoate (C10)	***307−55−1***	X	X	X
PFHxA	perfluorohexanoic acid; PFHA (C6)	**307−24−4**	X	X	
PFBA	perfluorobutyric acid; heptafluorobutyric acid; perfluorobutanoic acid (C4)	**375−22−4**	X	X	
8:2 FTOH	1H,1H,2H,2H-perfluorodecanol; 2-(perfluorooctyl)ethanol; 1-Decanol, 3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,10-heptadecafluoro-	***678−39−7***	X	X	X

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Table 1. continued

chemical common abbreviation	common name(s), other abbreviations	CAS RN	meta-analysis	intake assessment	hazard identification
Fragrance					
HHCB	galaxolide; 1,3,4,6,7,8-hexahydro-4,6,6,7,8,8-hexamethylcyclopenta-γ-2-benzopyran	**1222−05−5**	X	X	

[a]CAS RN used for intake assessment is **bolded**, CAS RN used for hazard identification is *italicized*. [b]Abbreviations used are "Practical Abbreviations (PRAB)" according to Bergman, et al. (2012).[85] [c]3194−55−6 is the most accurate CAS RN to use for the HBCD technical mixture. However, it has historically also been referred to with the CAS RN 25637−99−4, and is referenced with this number in a variety of regulatory documents and authoritative lists.[93] Hazards listed for HBCDD in the hazard table (Figure 4) reflect hazards associated with both CAS RNs in the SCP CC list. 3194−55−6 was used for the intake assessment for aHBCDD, bHBCDD and gHBCDD as physicochemical properties for individual isomers were not available in EpiSuite, but are not expected to differ significantly from the technical mixture for the properties under consideration. [d]Hazard information for DDC−CO (Dechlorane Plus; Bis(hexachlorocyclopentadieno)cyclooctane; DP including syn- (sDP and sDDC−CO) and anti- (aDP and aDDC−CO) isomers) is available from the SCP CC list, but studies measured individual isomers (aDDCO−CO, sDDC−CO). Because isomers are measured together in every study, exposure to the mixture is likely and thus we have provided the hazard information for the mixture and added intakes for the individual isomers together in order to rank the mixture by intake in the hazard table. [e]Abbreviations from the Centers for Disease Control and Prevention National Health and Nutrition Examination Survey (NHANES).[94] [f]84852−15−3 is the most accurate CAS RN to use for nonylphenol. However, it has historically also been referred to with the CAS RN 25154−52−3 and is referenced with this number in a variety of regulatory documents and authoritative lists.[95] [g]9016−45−9 is the CAS RN for a mixture of ethoxylated nonylphenols with lower numbers of ethoxylation (EO) units.[96] NPEs with 8 or fewer EO units are typically grouped together as the most toxic forms.[97] Hazard information for NPEO (Mixture of 4-nonylphenol ethoxylates; includes 4-nonylphenol monoethoxylate AND 4-nonylphenol diethoxylate; NP1EO and NP2EO) is available from the SCP CC list, but studies measured individual isomers (NP1EO, NP2EO). Because isomers are measured together in every study, exposure to the mixture is likely and thus we have provided the hazard information for the mixture and added intakes for the individual isomers together in order to rank the mixture by intake in the hazard table. [h]9036−19−5 is the CAS RN for a mixture of ethoxylated octylphenols (OPEO, includes octylphenol monoethoxylate AND octylphenol diethoxylate; OP1EO and OP2EO).[98] 9036−19−5 was used for the intake assessment for OP1EO and OP2EO as physicochemical properties for individual isomers were not available in EpiSuite, but are not expected to differ significantly from the mixture for the properties under consideration. [i]Hazard information for EH-TBB is available from the SCP CC list, but physicochemical properties are not available in EpiSuite.

size, storage method, MDL, percent detected, maximum, median, GM, GSD, or study geographic location), so we contacted the corresponding authors to collect that information. The corresponding authors provided the needed information for 20 of the papers and the unpublished data set. In order to use all available data, we included in the meta-analysis all studies that reported the GM and GSD for the chemicals of interest, even if the paper was missing other information. Five studies were excluded at this stage because they did not report GM and GSD, or because they examined chemicals measured by fewer than two other studies (so pooled GM estimates from three data sets could not be calculated).[54−58] In total, we were able to include data from 26 papers and one unpublished data set in the meta-analysis (Figure 1).

We estimated pooled GM dust concentrations for 45 chemicals using GMs and GSDs (Figure 1, SI Table S3). GMs and GSDs were used whenever data were available, including cases in which fewer than 50% of the data were > MDL (12 instances total). Below-MDL values were imputed by each study's author, and most commonly were assigned the value of MDL/2 or MDL/$\sqrt{2}$ (SI Table S4). Pooled GMs and 95% confidence intervals (CI) were generated for all chemicals measured in at least three data sets using random effects models in the R Metafor package (version 1.9−7). In cases where a study contained more than one geographic location (e.g., sampling in two states), each study location was counted as a separate data set. In cases where a corresponding author sent GM or GSD estimates that differed from the published data, we used the estimates sent by the author.

We additionally used random effects models in the R Metafor package to test whether concentrations differed by microenvironment (residential (n = 29 data sets) versus nonresidential (n = 13 data sets)) or sieve size (≤150 μm (n = 15 data sets) versus >150 μm (n = 21 data sets); testing RFRs and PFASs only). Only chemicals measured in at least three data sets collected from each type of microenvironment were compared.

Intake Assessment. To provide context to the pooled dust concentrations, we estimated residential intake (mg/kg-d) for each chemical and used these estimates to rank chemicals from low to high intake. We queried EPA's EPI Suite (v4.11) for physicochemical data, including octanol−water (K_{ow}) and octanol−air (K_{oa}) partitioning coefficient and Henry's law constant, for each chemical, using estimated values rather than empirical values for consistency. One chemical (EH-TBB) was excluded because it lacked physicochemical data in EPISuite (Figure 1). Using the modeled GM dust concentrations and available physicochemical data, we estimated daily residential intake for an adult female and child (3−6 years old) from three exposure routes: dermal uptake from gas phase, inhalation of air, and dust ingestion. We excluded the dermal intake through dust adherence route since it is typically minor[59] and the intake parameters required to estimate this route were not consistently available across all chemicals of interest. For TDCIPP where residential dust concentrations differed significantly from nonresidential concentrations, we excluded nonresidential concentrations from the intake calculations. We estimated indoor air concentrations from pooled GM dust concentrations using partitioning theory first proposed by Weschler and Nazaroff and further empirically validated by Dodson et al., which relies on K_{oa} and assumptions about the organic content of air and dust.[38,39] While this method of estimating air concentrations from dust concentrations may be applied to most chemicals in our analysis, it cannot be applied to the PFASs, which are typically found at lower relative levels in air (pg/m^3 range).[60] Thus, for PFASs, we estimated residential intake only via the dust ingestion pathway. For all other chemicals, we estimated gas-phase air concentrations for the dermal uptake and total air concentrations, that is, gas- and particulate-phase concentrations, for air inhalation intake (SI Table S5). Exposure

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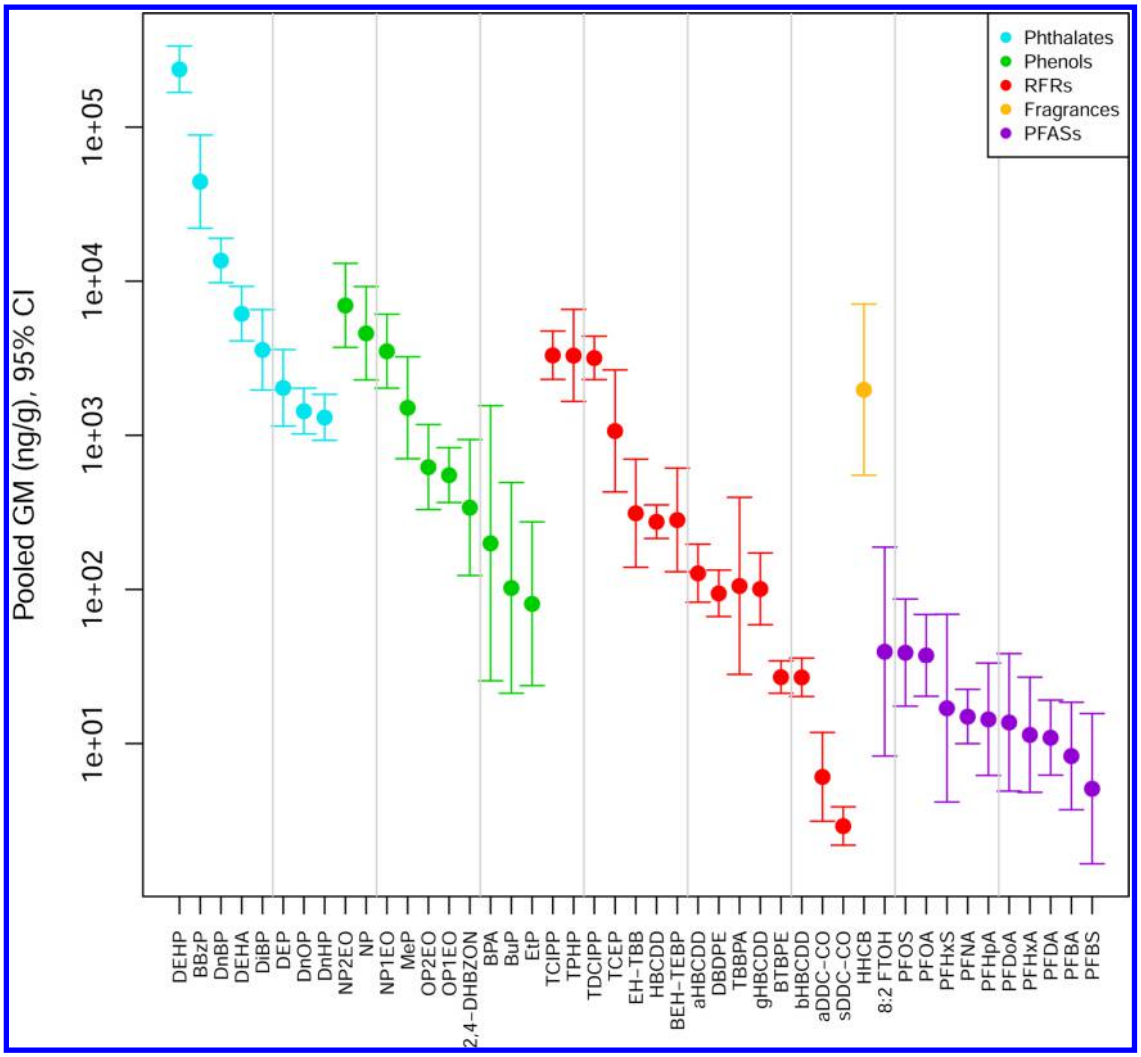

Figure 2. Pooled geometric means (GM) and 95% confidence intervals (CI) for the 45 chemicals whose GM and geometric standard deviation (GSD) were reported in at least three data sets. Gray lines added after every five chemicals are intended to aid legibility. See Table 1 for identification of abbreviations. See SI Table S3 for information on the studies from which data on each chemical was compiled. See SI Table S10 for the values plotted.

factors are listed in SI Table S6 and equations used to estimate intake are further described in the SI.

Hazard Identification. To provide summary information on each chemical's potential hazard(s), we used the California Safer Consumer Products Candidate Chemical (SCP CC) list, which is compiled from existing authoritative lists established by federal and state governmental bodies in North America and Europe, and identifies hazard trait(s) for each chemical.[61] While the particular criteria used by each authoritative body differ, all are science-based consensus processes that require some form of comprehensive review of the evidence in order to support regulatory decision-making (SI Table S7). Hazard traits identified by authoritative bodies fall into broad general categories (e.g., carcinogenic, reproductive toxicity), with each body providing further detailed documents on the particular chemical.

RESULTS AND DISCUSSION

Systematic Literature Search. The literature search identified 1538 published papers and abstracts, of which 34 papers and two unpublished data sets met the inclusion criteria (Figure 1). One hundred seventy-two chemicals from the five classes of interest were measured in at least one study, including 13 phthalates, 24 PFAS, 25 fragrances, 47 RFRs, and 62 phenols. Of these chemicals, fewer than half (74 chemicals) were measured in two or more studies (SI Table S8). The total number of studies available differed by chemical class: phthalates and certain RFRs were the most likely to be measured (up to 10 studies per chemical), while phenols and fragrances were measured less frequently. The large proportion of fragrances (96%), phenols (65%), and RFRs (57%) measured in only one study indicates that further research is needed to characterize these chemical classes in dust.

There was considerable variability in the abbreviations and acronyms used to identify the various chemicals across studies and CAS Registry Numbers (CAS RN) were rarely reported. In order to ensure correct chemical identification, we relied on the full chemical names identified in each paper at the first introduction of the acronym. The inconsistency of acronym use necessitated the manual matching of each acronym to a full chemical name, and precluded quick comparison across studies. We present CAS RNs, full chemical names, and abbreviations in Table 1.

Studies differed in the methods used to collect the dust (study vacuumed or existing used bag), in the size of the sieve used to sift the dust samples collected, and in storage temperature (SI Table

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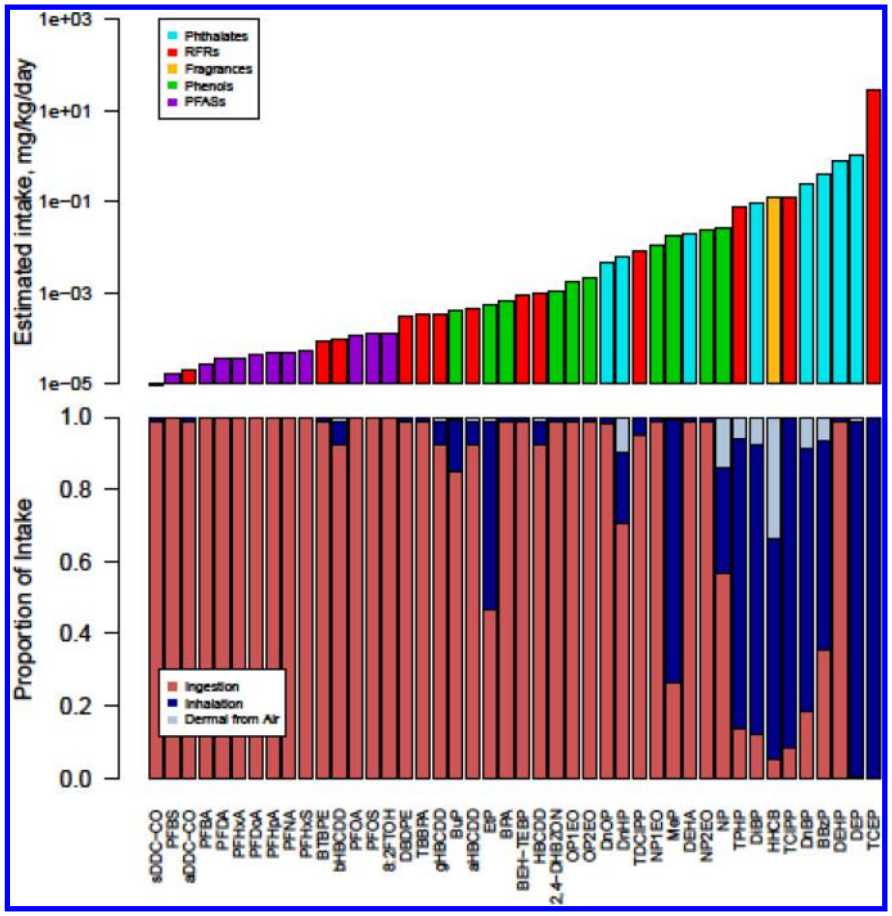

Figure 3. Top panel of the graph shows the estimated daily residential intake of each chemical for a 3−6 year old child (mg/kg/day), based on the pooled GM concentrations of each chemical in dust from the meta-analysis. The bottom panel shows the proportion of intake from three pathways: ingestion, inhalation, and dermal exposure from air. In both panels, PFAS intake estimates were based solely on estimated ingestion.

S1). Analytical methods also differed across studies (SI Table S2), as well as the statistical treatment of values below the limit of detection (SI Table S4). Because sorption of organic compounds to dust vary by particle size, the differences in sieve size used between studies likely increased interstudy variation in measured chemical concentrations.[62,63] Indeed, we found that DBDPE concentrations were significantly higher in dust sieved with >150 μm sieves, compared to samples sieved with ≤150 μm sieves. No other RFRs or PFASs differed significantly by sieve size. Additionally, because some previous research has suggested that dust collected by a researcher may contain different chemical concentrations than dust in a home vacuum bag,[64] the heterogeneity in dust collection method likely also added variability to our sample.

Meta-Analysis. We identified 45 chemicals to include in the meta-analysis (Figure 1, Table 1). Data collection sites spanned 14 states and tended to cluster around research universities, particularly for RFRs (SI Figure S1). Therefore, the data may not be nationally representative. Though dust was collected from a variety of indoor environments including homes, schools, daycare centers, cars, gymnasiums, and occupational settings, nonresidential environments were less frequently sampled (SI Figure S2). Chemicals and chemical classes also co-occurred in dust: studies that reported sample-specific concentrations consistently found multiple chemicals in each dust sample.[36,65−67]

The detection frequency of each chemical varied between studies, sometimes widely, and did not seem to be solely due to varying detection limits. However, of the 45 chemicals included in the meta-analysis, 10 were detected in over 90% of samples (SI Table S9), indicating that indoor dust contains a mixture of chemicals, and that these particular consumer product chemicals may be ubiquitous. Products or building materials found in almost all indoor environments, such as cables/wires, electronics, and upholstered furniture, may be sources of phthalates and RFRs in a typical U.S. indoor environment.[3,37,68]

Phthalates were measured in the highest concentrations in indoor dust, several orders of magnitude above the other chemical classes. Phenols, RFRs, and fragrances were measured in similar concentrations, while PFAS were measured in the lowest concentrations (Figure 2; SI Table S10). The relative ranking of chemical classes from highest to lowest according to the maximum GM concentrations was phthalates, phenols, RFRs, fragrances, and PFASs.

We compared concentrations of TDICPP, EH-TBB, BEH-TEBP, PFOS, PFOA, PFNA between residential and nonresidential environments. TDCIPP and EH-TBB were significantly higher in nonresidential than residential environments (p = 0.0043 and p = 0.026, respectively), while no differences were found for the other chemicals (SI Table S11). The differences noted for TDCIPP and EH-TBB were likely driven by very high RFR levels in gym and fire station dust. However, no differences by environment were found for BEH-TEBP, which is a component of the Firemaster 550 mixture along with EH-TBB. This is consistent with other studies that report ratios of EH-TBB

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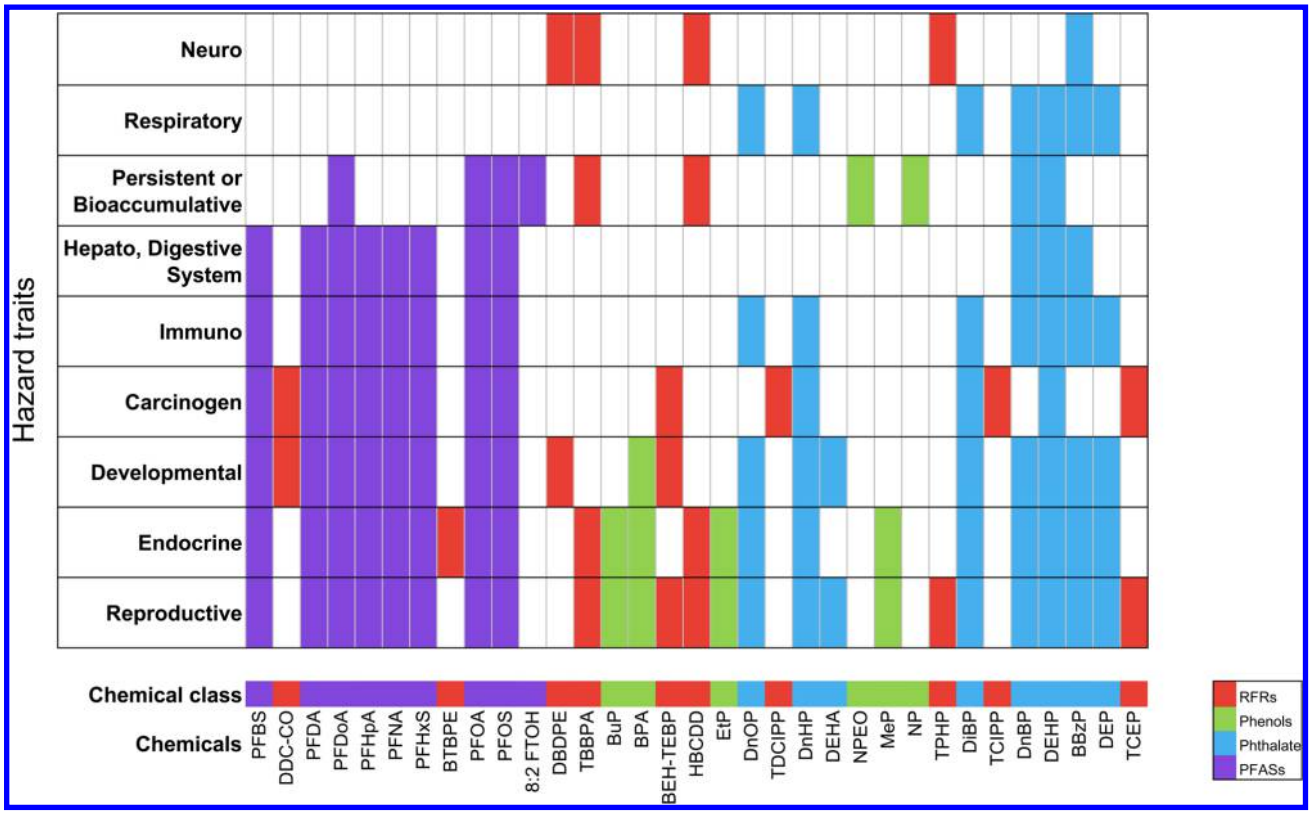

Figure 4. Each row on the chart represents potential chemical hazard traits, and each column represents a chemical. Chemicals are listed in order of estimated adult daily residential intake (lowest intake on the left-hand side). For hazard traits associated with a mixture rather than individual chemicals (HBCDD, DDC−CO, NPEO) the intakes of the chemicals composing the mixture were added to generate the mixture's position in the intake ranking. The hazard traits associated with each chemical, according to the SCP CC list are represented by filled cells; hazard traits not associated with the chemical are left blank. Chemicals not included in the intake assessment and those not listed in SCP CC list are not shown in the chart.

and BEH-TEBP in dust differ from those in the commercial flame retardant mixture.[18,68,69]

Intake Assessment. Intake estimates, like modeled GM concentrations, spanned orders of magnitude (Figure 3). As expected, estimated intakes, normalized by body weight, were higher for the child compared to the adult, although the relative ranking of the chemicals was similar (data not shown).

Those chemicals with the highest dust concentrations also tended to have the highest intake estimates, with the exception of the RFR TCEP, which was driven by its relatively high predicted air concentration (>50 $\mu g/m^3$). TCEP had the highest estimated intake (>1 mg/kg/day) followed by four phthalates: DEP, DEHP, BBzP, and DnBP (>0.1 mg/kg/day) (Figure 3). Exposure to indoor air seemed to drive the intake estimates, since those chemicals with the highest intake estimates also tended to have the largest proportion of intake from inhalation and dermal uptake of indoor air. These modeled results agree with recent evidence of high inhalation exposure to chlorinated RFRs.[70] In contrast, for those chemicals where the dust ingestion route dominated, the intake estimates were lower. Compared to other classes, phthalates generally had the highest intake estimates; RFRs' intake estimates varied widely from approximately 100 ng/kg-day for the Dechlorane isomers to >1 mg/kg-day for TCEP; the fragrance HHCB ranked seventh of 44 in total intake; phenols had midrange estimates; and PFASs, which only rely on the dust ingestion route, had the lowest residential intake estimates (Figure 3).

These residential intake estimates do not account for exposures in microenvironments other than the home. While the data on nonresidential environments were limited (SI Figure S2), the data suggest that other indoor microenvironments will contribute to total exposure. For example, concentrations of the flame retardants TDCIPP and EH-TBB were higher in nonresidential microenvironments (SI Table S11) and we would expect that people spending substantial time in such spaces would have higher exposures than the estimates presented in Figure 3.

In addition to calculating the pooled GM and GSD, we examined the maxima relative to the GM for representative chemicals from four classes (TDCIPP, DEHP, MeP, PFOS) (SI Figure S3). The maxima reported by each data set were, in many cases, at least 1 order of magnitude greater than the reported pooled GM. Thus, focusing only on the central tendency metric could obscure the very high exposures, and potential associated health risks, experienced by a fraction of the population.

We estimated residential intakes in order to provide context to the pooled dust concentrations and link concentration and hazard information. However, this intake assessment is not comprehensive in that it does not quantify total intakes of every chemical, does not account for variability in exposure factors (e.g., dust ingestion rates), and relies on estimated physicochemical data. In addition to the contribution of other microenvironments on intake, for many of these chemicals, particularly the phthalates and parabens, actual intakes may be much higher since these chemicals are widely used in personal care products applied directly to the skin,[29] some medications;[71] and diet has also been shown to be an important route of exposure.[72,73] Exposure factors vary by individuals; for example, the upper percentile dust

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ingestion rate for children is 100 mg/day compared to the central tendency of 60 mg/day used here, which may substantially increase intake of chemicals in dust for some individuals.[74] Finally, we relied on estimated physicochemical properties from EPA's EpiSuite, which provides one data source for all chemicals; however, the algorithms used to estimate properties may vary in accuracy across the chemicals of interest.[38,39] Similarly, the application of the partitioning model to estimate indoor air concentrations may be less reliable for some chemicals since it has not been fully validated across all of the chemicals of interest.

Hazard Identification. The authoritative body listings aggregated in the SCP CC list (SI Table S7) identified nine general hazard traits associated with the 35 chemicals that were both included in the intake assessment and had SCP CC listings (Figure 1). Reproductive toxicity, endocrine toxicity, developmental toxicity, and carcinogenicity were the most common hazard traits (Figure 4). These hazard trait categorizations are broad, but do identify chemicals within and across classes that warrant further investigation for potential cumulative exposures and impacts, as our analysis indicates that coexposure to multiple chemicals is likely (SI Table S9). Previous studies have quantified cumulative risk from a number of phthalates or PFASs, some of which are included in the current study.[16,75−77]

This hazard identification approach and ranking by estimated intake does not account for bioavailability, pharmacokinetics, dose−response and other myriad factors that influence the toxicity of chemicals. However, the ranking of chemicals by intake estimate does point to chemicals with high intake estimates that were associated with multiple hazard traits of concern (DEP, BBzP, DEHP, DnBP, DiBP), suggesting that these chemicals could be prioritized for exposure reduction.

Six chemicals (2,4-DHBZON, OP1EO, OP2EO, PFHxA, PFBA, HHCB) were not found on the SCP CC list (Table 1), which may reflect lack of data, rather than lack of hazard. A search of ACToR (the Aggregated Computational Toxicology Resource, a comprehensive database of in vivo and in vitro chemical toxicity data from U.S. EPA[78]) reveals that while multiple toxicity studies are available for HHCB, toxicity tests for DHBZON, PFBA and PFHxA are limited to lethal dose and/or in vitro studies, and no toxicity testing is reported for OP1EO and OP2EO. A disadvantage of our use of the SCP CC list as the sole source of hazard information is the limited information both on emerging chemicals[54,79] and the emerging toxicities of known chemicals,[80,81] due to the time needed to amass and assess evidence before these are included in authoritative lists.

Recommendations for Indoor Dust Studies. While our analysis represents one of the first attempts to aggregate data across multiple studies for consumer product chemicals, it was somewhat limited by heterogeneity of the available data sets, as well as inconsistent or incomplete collection and/or reporting of important information. We present the following recommendations to maximize the usefulness and comparability of future empirical dust studies and advance the field of exposure science.

Dust Collection and Demographic Data. Future studies should consider standardizing methods found to introduce variability including dust sample collection method and sieve size. To collect dust, we suggest following the standards set by ASTM Standard Practice D5438: Standard Practice for Collection of Floor Dust for Chemical Analysis.[82] At the minimum, we suggest that studies should collect fresh dust using a vacuum cleaner with an extraction thimble in the crevice tool, as opposed to sampling from used bags, to improve methodological consistency and ensure thorough sampling. Because the optimal sieve size might vary by chemical of interest, we do not recommend a single sieve size for all studies. The two most common sieve sizes in the studies we collected here were 150 μm (12 studies) and 500 μm (10 studies). In the interest of comparability with prior work, future studies should consider using one of these sieve sizes.

Studies should also collect detailed demographic information on the individuals occupying the indoor environments to enable assessment of demographic factors that are associated with chemical exposures in dust. For example, higher PBDE exposures have been reported in California and in low-income communities.[83,84] This information is critical in shaping future studies of exposure and health, and in providing an evidence base for intervention development and public policy.

Reporting. Many studies included here did not report descriptive information that could be used to pool findings or compare across data sets. Future studies should report: the month, year, and location of sample collection, central tendency measures (e.g., median, GM), measurements of error (e.g., GSD, 95% CI), MDLs, frequencies of detection above the MDL, maxima, CAS RNs for each chemical, and acronyms according to established standards like those developed by Bergman et al. 2012.[85] Because a chemical's dust concentration may change over time, studies should report summary statistics by sample collection year for multiyear studies.

Indoor Environment and Geography. Most studies measured residential environments. Vehicles, workplaces, and other indoor environments where people may spend many hours should be an emphasis of future research. Moreover, important insights could be gained by studying environments with high chemical concentrations in dust, as well as people with elevated exposures, since these studies could help characterize source contributions to chemical exposures.[86,87] Future studies should also target dust sampling in less highly sampled regions of the United States.

Study Design. Future studies should focus on better characterizing emerging consumer product chemicals in the phenol and fragrance classes which were the least studied. Nontargeted analytical methods may provide a more accurate picture of the true landscape of current consumer product chemicals in dust. We were also not able to fully assess impacts of changing product formulations on indoor dust, since most studies were cross-sectional and only included one sample per home. Longitudinal studies with repeated dust measures over time are needed to detect changes in product formulation or use, such as was seen with PBDEs and RFRs in Dodson et al., 2012.[36]

SVOC Dynamics Indoors. Our intake assessment is based on current understanding of the dynamics of SVOCs in the indoor environment, which is largely based on partitioning theory.[38] Future studies should further validate these theoretical models with empirical indoor air, dust, and surface measurements of a range of chemicals. Moreover, partitioning models are most appropriate for nonpolar chemicals and do not appear to apply to chemicals like PFASs.[60] Future research should investigate the indoor dynamics of PFASs and similar chemicals to develop accurate exposure models.

Broader Research Needs and Implications. Cumulative exposures have more often been discussed in the context of ambient outdoor pollution.[88] Our results highlight the co-occurrence of chemicals in the indoor environment that may contribute to common adverse outcomes and thus advance the rapidly emerging field of cumulative exposure assessment. However, quantifying the health risks of chemical mixtures will

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also require new study designs and tools in toxicology, risk assessment, and epidemiology.[89] Our findings inform the next generation of chemical mixture studies by highlighting chemicals found at high concentrations indoors, identifying chemical combinations that reflect typical indoor scenarios, and indicating emerging chemicals of concern for biomonitoring.

We also know that pooling GMs across studies does not adequately capture within-cohort subpopulations that may be highly exposed. Based on the large difference between the maxima and GMs presented here, we suggest that, consistent with recommendations from the National Research Council, risk assessments should quantitatively characterize exposure variability and include central tendency estimates as well as "worst case" (maximum) scenarios.[90] It is likely that some households experience excess exposure due to shared exposure pathways related to building characteristics, occupant behavior or other modifiable factors.[12] A full understanding of these household-level drivers of high exposure is critical to the development of mitigation strategies including regulatory interventions. However, when identifying compounds to remove from consumer products, we also need to ensure that replacements are safer alternatives in order to guard against "regrettable" substitutions that may not reduce risk.[91]

We focused our analysis on indoor dust reservoirs, which reflect long-term exposures that may be the best measure of household exposure to SVOCs, given their partitioning between indoor air, dust, and surfaces. This analysis does not, however, consider other important exposure routes. For example, close contact with flame-retarded products may result in inhalation and dermal exposures that significantly contribute to total exposure for the more volatile organophosphate RFRs.[87,92] Biomonitoring data, combined with environmental measures, could be used to further evaluate the contribution of indoor exposures to total exposures.

In conclusion, this comprehensive analysis of consumer product chemicals in U.S. indoor dust suggests that a wide array of chemicals used in everyday products find their way into indoor environments across the country where people, including vulnerable subpopulations like children, are continuously exposed. In this way, the indoor environment is a haven for chemicals associated with reproductive and developmental toxicity, endocrine disruption, cancer and other health effects. Additionally, despite the observed variability across studies, the existing dust literature has allowed us to identify the chemicals and chemical classes found at the highest levels indoors (phthalates and phenols), with the highest estimated intakes (phthalates and RFRs), and associated with the most hazard traits (phthalates and PFASs). These findings can be used to improve population health by informing future exposure assessment and epidemiologic studies of chemical mixtures as well as individual and regulatory exposure reduction strategies.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.6b02023.

Descriptive statistics, information about data collection and analysis, studies used for meta-analysis data, pooled GMs, chemicals measured in one or two studies, number of samples in various environments, chemicals with highest detection frequencies, comparison of dust levels by environment, input values for intake estimates, identification of authoritative sources used to describe hazard traits, comparisons of geometric mean and maximum values, and exposure factors used for intake estimates. Please contact authors if additional information on the meta-analysis input data or results is desired (PDF)

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Notes

The authors declare no competing financial interest.

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