

Phthalate Exposure and Risk Assessment in California Child Care Facilities

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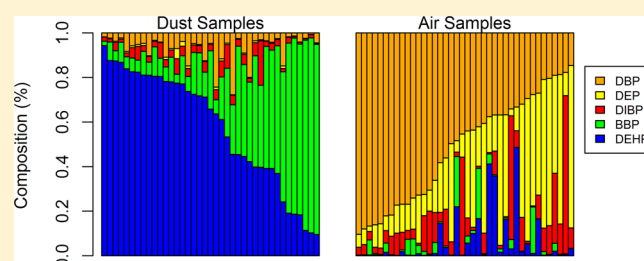
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S Supporting Information

ABSTRACT: Approximately 13 million U.S. children less than 6 years old spend some time in early childhood education (ECE) facilities where they may be exposed to potentially harmful chemicals during critical periods of development. We measured five phthalate esters in indoor dust ($n = 39$) and indoor and outdoor air ($n = 40$ and 14 , respectively) at ECE facilities in Northern California. Dust and airborne concentrations were used to perform a probabilistic health risk assessment to compare estimated exposures with risk levels established for chemicals causing reproductive toxicity and cancer under California's Proposition 65. Di(2-ethylhexyl) phthalate (DEHP) and butyl benzyl phthalate (BBzP) were the dominant phthalates present in floor dust (medians = 172.2 and 46.8 $\mu\text{g/g}$, respectively), and dibutyl phthalate (DBP), diethyl phthalate (DEP), and diisobutyl phthalate (DIBP) were the dominant phthalates in indoor air (medians = 0.52 , 0.21 , and 0.10 $\mu\text{g}/\text{m}^3$, respectively). The risk assessment results indicate that 82–89% of children in California ECE had DBP exposure estimates exceeding reproductive health benchmarks. Further, 8–11% of children less than 2 years old had DEHP exposure estimates exceeding cancer benchmarks. This is the largest study to measure phthalate exposure in U.S. ECE facilities and findings indicate wide phthalate contamination and potential risk to developing children.



INTRODUCTION

Sixty-five percent of U.S. children less than 6 years old ($n = 13$ million) spend some portion of their day in child care or preschool, collectively known as early childhood education (ECE), with some infants and young children spending as much as 10 h per day, 5 days per week, in these facilities.¹ In California, approximately 60% of children attend ECE programs in stand-alone centers, often located in schools, commercial buildings, portable buildings, converted homes, or religious institutions, and 40% attend family-based programs run at an individual's residence.² Early life exposure assessments have primarily focused on exposures occurring in the home, but an increasing percentage of children spend many of their waking hours in ECE facilities.³

Recent studies indicate that ECE environments may contain lead, pesticides, allergens, and other contaminants hazardous to children's health.^{1,4,5} Young children are more likely to be exposed to these environmental contaminants due to their exploratory behavior^{6,7} and they have higher absorbed doses than adults because they breathe, eat, and drink more per unit of body weight.⁸ Young children are also more susceptible to the adverse effects of chemicals and toxins because they are less developed immunologically, physiologically, and neurologically.^{6,7,9} Therefore, children who attend ECE facilities on a

daily basis may be chronically exposed to potentially harmful chemicals during critical periods of development.

Of particular concern for developing children are endocrine disrupting compounds (EDC), which are exogenous compounds that disrupt normal hormonal or homeostatic systems.¹⁰ Phthalate esters are suspected EDCs¹⁰ and ubiquitous in indoor environments, due to their current use as plasticizers in building materials including vinyl flooring, consumer products, and personal care products.^{11–16} Studies have associated phthalate exposures with bronchial obstruction, allergies, and asthma in young children,^{17–23} as well as reproductive outcomes.^{24–26} Phthalates are semivolatile organic compounds (SVOCs) and their presence indoors contributes to a child's phthalate body burden²⁷ via inhalation, nondietary ingestion of dust on surfaces or hands, and dermal absorption through the skin.²⁸

Limited studies, especially within the United States, have investigated phthalate indoor dust^{29–32} and airborne^{30,31,33} contamination in ECE facilities. Whereas these contamination

Received: March 10, 2014

Revised: May 27, 2014

Accepted: May 28, 2014

Published: May 28, 2014

levels indicate a range of exposures and potential health impacts, existing risk analyses have been limited to comparing exposures to the U.S. Environmental Protection Agency's (EPA) Reference Doses (RfD),²⁹ European Food Safety Authority's Tolerable Daily Intake (TDI),^{16,33–38} or the European Chemical Safety's Derived No-Effect Level (DNEL).³⁹ In this study, we report information on phthalate levels in dust and air from 40 ECE facilities in Northern California, including urban, suburban, and rural/agricultural communities. We use this information to carry out a probabilistic health risk assessment comparing exposure estimates to "Safe Harbor" risk levels established for chemicals causing reproductive toxicity and cancer under California's Proposition 65.^{40,41}

MATERIALS AND METHODS

Study Population. We enrolled 40 ECE facilities from the California counties of Alameda ($n = 20$) and Monterey ($n = 20$) to participate in the study. To recruit a diverse sample, we geographically coded center and large (>8 children) home-based licensed facilities by zip code using publicly available databases.⁴² We ultimately recruited 28 child care centers and 12 home-based facilities. All measurement and evaluation procedures were approved by the University of California Berkeley Committee for the Protection of Human Subjects and written informed consent was obtained from each ECE program director or senior administrator.

Study Visits. Typically, field technicians visited an ECE facility two times. The first visit included administering the consent form and questionnaire, conducting a site inspection, and collecting a dust sample. The second visit involved collecting air samples when children were present. Site visits occurred from May 2010 to May 2011. The facility inspection focused on the primary child care room where air and dust samples were collected, cooking areas, and the bathroom to note items such as building materials and condition. The median difference between the first and second visit was 7 days. The time between sampling dates exceeded 30 days 4 times because of scheduling conflicts or personnel turnover.

Dust and Air Sampling. Dust samples were collected with a High Volume Small Surface Sampler (HVS3) (Envirometrics Inc.) following procedures described in the American Society for Testing Materials Standard Practice D 5438-05.^{43,44} Samples were collected from carpets centrally located in the primary child care room where air sampling would take place during the second site visit. Dust samples were collected from a floor area of at least 1 m² into cleaned, 250-mL amber glass bottles (I-CHEM, item 341-0250). At one facility, dust was not collected because the facility did not have carpet, upholstered furniture, or sufficient bare floor dust.

Indoor air samples were collected at all 40 ECE facilities during the day when children were present (typically 8 h). Samples were collected at the height of a child's breathing zone (0.6–1 m) using a fan-cooled single rotary vane pump installed in a stainless steel box lined with foil-faced fiberglass insulation to reduce noise. The pump's exhaust system included a muffler to reduce noise and a high-efficiency particulate air and carbon filter to eliminate emissions. Air was pulled through a polyurethane foam (PUF) plug at approximately 4 L per minute (LPM) and regulated by an inline taper flowmeter (Key Instruments, part 10710). Bulk dust and PUF samples were shipped on dry ice to the Battelle Memorial Institute in Columbus, OH for analyses.

Dust and Air Analyses. Levels of diethyl phthalate (DEP), diisobutyl phthalate (DIBP), dibutyl phthalate (DBP), butyl benzyl phthalate (BBzP), and di(2-ethylhexyl) phthalate (DEHP) were measured in all samples. Bulk dust was sieved to 150 μm using a stainless steel sieve and a 0.5-g aliquot was spiked with BBzP-d4. The dust was then sonicated in a solution of 1:1 hexane/acetone, centrifuged, and the extract was withdrawn. A 1-mL aliquot of the extract was spiked with internal standards and analyzed using gas chromatography and mass spectrometry in the multiple-ion-detection mode (GC/MS/MID) without further processing to limit laboratory contamination. Phthalates in dust had a method detection limit (MDL) of 20 ng/g. Dust loading was calculated by multiplying the dust concentration by total weight of the sieved dust and dividing by the area vacuumed.

PUF samples were placed in a 22-mL accelerated solvent extractor cell, spiked with 13C4-di-*n*-hexyl phthalate and extracted at 2000 psi and 100 °C through two 5-min cycles using dichloromethane. The extract was concentrated to 10 mL, and 1 mL was removed, spiked with the internal standard dibromobiphenyl, and analyzed using GC/MS/MID. For air measurements, final mass of each phthalate in a sample was calculated by subtracting three times the standard deviation (SD) of field matrix blank levels (see Supporting Information (SI) Table S1). Matrix blank-corrected mass values below the instrument limit of detection (IDL; 0.01 $\mu\text{g}/\text{sample}$ or 0.005 $\mu\text{g}/\text{m}^3$) were replaced with $\text{IDL}/\sqrt{2}$.⁴⁵

Three phthalate dust samples were analyzed in duplicate and the average relative percent difference (RPD) ranged from 5 to 11% for all 5 congeners, showing good precision. Two duplicate indoor air samples were collected with adequate precision observed for phthalates with higher detection frequencies (DEP, DIBP, and DBP; average RPD = 37, 24, and 15%, respectively), while a lack of precision in duplicate air samples was observed for phthalates with lower detection frequencies (BBzP and DEHP; average RPD = 104 and 75%, respectively). Detailed quality control data on field duplicates, field blanks, field spikes, and laboratory spikes are presented in SI Table S1.

Building Parameters. Temperature and percent relative humidity (RH) were measured indoors ($n = 40$) with the QTrak 8554 from Texas Science Instruments (TSI). QTraks were calibrated in the spring of 2010 by TSI. Monitors were set to log minute averages of temperature and relative humidity and ran for the duration of the phthalate air samples. Temperature and RH measurements were averaged over the entire sampling time for analysis. Air exchange rates (AER) were estimated using both a continuous indoor carbon dioxide (CO₂) mass balance and a CO₂ tracer gas decay method. The CO₂ decay experiment used a single release of medical-grade CO₂ (Praxair, part CD M-10, pharmacopeia grade) when children were not present (usually at lunch or the end of the day) elevating CO₂ levels to approximately 2500 ppm. We then estimated AER based on a continuous mass-balance model, accounting for CO₂ input from outdoors and occupant emissions in our model during occupied periods, and on the CO₂ decay curve when the room was unoccupied.

Statistical Analyses. Statistical analyses were performed with STATA statistical software Version 11.2⁴⁶ and R Version 3.0.1.⁴⁷ When duplicate samples were collected for a facility, the average concentration between the two measurements was calculated and reported for that facility. Indoor to outdoor (I/O) air concentration ratios were calculated by dividing the indoor air concentration by the outdoor air concentration for

Table 1. Summary of Phthalate Dust Concentrations^a

analyte	GM	GSD	min	5th%	25th%	50th%	75th%	95th%	max
DEP	1.7	1.9	0.5	0.7	1.0	1.4	2.7	4.5	7.9
DIBP	10.6	2.2	3.5	3.8	6.1	9.3	13.8	39.6	145.8
DBP	18.2	2.3	2.8	7.3	10.3	13.7	30.2	79.8	138.5
BBzP	68.8	4.6	7.1	8.0	23.1	46.8	236.8	1045.7	1435.4
DEHP	179.5	1.9	51.6	79.9	116.6	172.2	263.4	498.4	1088.1

^aGM = geometric mean; GSD = geometric standard deviation; $\mu\text{g/g}$; $n = 39$).

each facility with paired measurements ($n = 14$). We computed Spearman's rank correlation coefficients to compare the indoor air concentrations with dust concentrations, dust loading, indoor temperature, indoor relative humidity, and air exchange rates. Finally, we tested for differences in phthalate indoor air concentrations, dust concentrations, and dust loading based on presence of vinyl flooring using the Wilcoxon–Mann–Whitney test.

Health Risk Assessment. We conducted a probabilistic health risk assessment for phthalates measured in air and dust using Monte Carlo simulations of exposure to incorporate the uncertainty and variability of measured chemical concentrations and assigned exposure factors.⁴⁸ The calculation of daily doses via inhalation, nondietary dust ingestion, and dermal absorption from dust adhered to skin followed the equations presented in the Agency for Toxic Substances and Disease Registry's *Public Health Assessment Guidance Manual*. Doses resulting from dermal uptake of gaseous phthalates were calculated by adapting relationships developed by Weschler and Nazaroff⁴⁹ and Bekö et al.³³ See the SI for additional information on dose calculations and exposure factor distributions.

Our risk assessment approach followed the U.S. EPA's guidance for performing a human health risk assessment on cancer and noncancer health end points.⁵⁰ Under California's Proposition 65, the Office of Environmental Health Hazard Assessment (OEHHA) has set "Safe Harbor Levels" called Maximum Allowable Dose Levels (MADL) for reproductive toxicants. As of March 2014, OEHHA has set MADLs for DBP, BBzP, and DEHP, which are defined as "the highest level at which the chemical would have no observable reproductive effect assuming exposure at 1,000 times that level."⁴⁰ We did not compare our BBzP exposure estimates to the MADL, as this risk level is for developmental toxicity during the prenatal period.⁵¹ In addition, OEHHA has set a No Significant Risk Level (NSRL) for DEHP, which is defined as the daily intake level posing a 10^{-5} risk of cancer assuming lifetime exposure.⁴⁰ The MADL for DBP and the NSRL for DEHP are applicable for both females and males via inhalation, nondietary ingestion, and dermal routes of exposure.^{52,53} The MADL for DEHP is applicable only for males and for oral exposure only.⁵⁴ We did not perform a risk assessment on DEP or DIBP, as there are currently no MADLs or NSRLs for these compounds.

Uncertainty in our uptake estimates was quantified via a Monte Carlo analysis, in which inputs are randomly drawn from their probability distribution. To estimate risk, we compiled exposure factor distributions (e.g., inhalation rate and body weight) from the U.S. EPA's *Exposure Factors Handbook*⁵⁵ for four age groups: birth to <1 year, 1 to <2 years, 2 to <3 years, and 3 to <6 years. All input parameters including exposure concentrations were assumed to be log-normally distributed except for body weight and surface area to body weight ratio, which were sampled from normal distributions. Next, we performed 100 000 Monte Carlo simulations (enough

to reach a stable mean/SD of input parameters) to calculate child phthalate dose ($\mu\text{g/kg/day}$) and exposure estimates ($\mu\text{g/day}$) via inhalation, nondietary ingestion, dermal absorption via dust adherence, and dermal absorption from gas-phase phthalates. Because children are not present in ECE facilities all hours of the day, we calculated an exposure factor (EF) to apportion the dose received only in the ECE facilities. Approximately 22, 41, and 37% of children spent <5 h, 5–8 h, and >8 h per day attending the 40 ECE facilities, respectively. We assigned a lower bound of 3 h per day and an upper bound of 10 h per day for the number of hours spent in child care. On the basis of this distribution, we randomly sampled from a uniform distribution with 22% of our 100 000 simulations between 3 and <5 h, 41% between 5 and 8 h, and 37% between >8 and 10 h. Further, we assumed that children spent 5 days per week and 48 weeks per year attending ECE facilities (which accounts for 4 weeks away from day care for holidays and vacation). To capture the correlation between indoor air and dust concentrations of DBP, values of the DBP air and dust concentrations were sampled using the MASS package `mvrnorm()` function in R, which accounts for the covariance of the variables.⁵⁶ Although the DEHP MADL is applicable only to males, we did not apply a male-only set of exposure factors for that risk analysis due to few physiological differences between the sexes at the ages modeled. Route contributions for DBP and DEHP were calculated by dividing each route of exposure by the total amount of exposure.

We computed child-specific NSRLs based on OEHHA's guidelines to define "Safe Harbor" levels that account for the increased sensitivity of young children, which incorporates an age sensitivity factor of 10 for children <2 years of age and 3 for children between the ages of 2 and 6 years.⁵⁷ Child-specific MADLs and the NSRL (SI Table S7) also accounted for the difference in body weights between children and adults. It should be noted that an age-specific NSRL is the estimated daily intake for that specific age range, which contributes 1/70th (assuming a 70 year lifetime) of the target lifetime cancer risk in that particular year of life (for NSRLs, the "target" lifetime cancer risk is 10^{-5}). Risk quotients (RQ) were calculated by dividing the modeled exposure by the child-specific MADL or NSRL benchmarks for each Monte Carlo simulation. If the RQ was greater than 1, the exposure estimate exceeded health-based exposure limits. Example child-specific MADL and NSRL calculations are presented in the SI.

To assess the impact of model input uncertainties on our exposure results, a sensitivity analysis was conducted on all inputs with probability distributions. The magnitude of the absolute value of the Spearman's rank correlation coefficients between the model inputs and outcomes (exposure estimates) from the Monte Carlo simulations was used to indicate the importance of each input.⁵⁸

Table 2. Summary of Indoor Air Phthalate Detection Frequencies and Concentrations^a

analyte	>MDL (%)	GM	GSD	min	5th%	25th%	50th%	75th%	95th%	max
DEP	97.5	0.24	3.21	<MDL	0.05	0.14	0.21	0.55	1.20	2.81
DIBP	87.5	0.07	5.52	<MDL	<MDL	0.02	0.10	0.21	0.53	2.56
DBP	100.0	0.57	2.67	0.05	0.11	0.29	0.52	1.38	2.34	2.65
BBzP	50.0	0.01	4.17	<MDL	<MDL	<MDL	0.01	0.03	0.17	0.23
DEHP	55.0	0.02	5.83	<MDL	<MDL	<MDL	0.01	0.06	0.34	2.71

^aGeometric means and standard deviations were calculated by replacing values below the MDL with IDL/ $\sqrt{2}$; MDL = method detection limit; GM = geometric mean; GSD = geometric standard deviation; IDL = instrument detection limit; $\mu\text{g}/\text{m}^3$; $n = 40$.

RESULTS

ECE Characteristics. A total of 1764 children were served by the 40 ECE facilities, with 22% spending <5 h, 41% spending 5–8 h, and 37% spending >8 h per day. Of the children enrolled, 1% were <1 year old, 4% were 1 to <2 years old, 19% were 2 to <3 years old, and 76% were 3 to <6 years old. ECE building types consisted of single-family detached homes (37.5%), traditional school buildings (27.5%), portable school buildings (22.5%), office buildings (7.5%), and churches (5%). Vinyl flooring was observed in 63% of the buildings. During the air sampling period, the average indoor temperature was 21.1 °C (SD = 1.7) with a range of 16.0–24.6 °C. Average RH was 49.3% (SD = 6.9) with a range of 34.5–62.8%. Average AER during the air sampling period was 2.01 per hour (h^{-1}) (SD = 1.41) with a range of 0.28–5.63 h^{-1} .

Dust. All 5 target phthalate analytes were above the MDL in all dust samples (Table 1). DEHP and BBzP were the dominant analytes present in floor dust with median DEHP and BBzP dust concentrations (172.2 and 46.8 $\mu\text{g}/\text{g}$, respectively) substantially higher than those of DBP, DIBP, and DEP (medians = 13.7, 9.3, and 1.4 $\mu\text{g}/\text{g}$, respectively). Similarly, loadings of DEHP and BBzP (SI Table S8) were also higher (medians = 361.7 and 135.7 $\mu\text{g}/\text{m}^2$, respectively) than DBP, DIBP, and DEP loadings (medians = 51.1, 26.8, and 3.5 $\mu\text{g}/\text{m}^2$, respectively).

Air. Phthalates were detected more often in indoor air compared to outdoor air (Tables 2 and SI S9), and concentrations were typically higher indoors for all 5 analytes (median *I/O* ratios = 1.0–27.0, SI Table S10). DIBP, a plasticizer often used in combination with other high molecular weight phthalates as a gelling aid, was only detected indoors. In contrast to the patterns of phthalate analytes in dust, DBP, DEP, and DIBP were present in greater abundance in air than DEHP and BBzP (Figure 1). Median indoor air concentrations of DBP, DEP, and DIBP were 0.52, 0.21, and 0.10 $\mu\text{g}/\text{m}^3$, respectively, while median DEHP and BBzP were both 0.01 $\mu\text{g}/\text{m}^3$.

Factors Associated with Dust/Air Levels. DIBP, DBP, and BBzP indoor air and dust concentrations were significantly correlated ($r = 0.46$ – 0.58 , $p < 0.05$; Table 3), but loading and air concentrations were only significantly correlated for BBzP ($r = 0.64$, $p < 0.05$). Interestingly, average indoor air temperatures were significantly correlated with DEP and DIBP levels in indoor air ($r = 0.43$ – 0.51 , $p < 0.05$) and mildly correlated with indoor air concentrations of DBP ($r = 0.31$, $p = 0.05$). Percent relative humidity and air exchange rate were not associated with phthalate indoor air concentrations.

For indoor air concentrations, only DEP concentrations (Mann–Whitney test, $p = 0.02$) were significantly higher in ECE facilities with vinyl flooring (SI Table S11). In addition, we found no significant differences between dust concentrations and presence of vinyl flooring for any phthalate analyte;

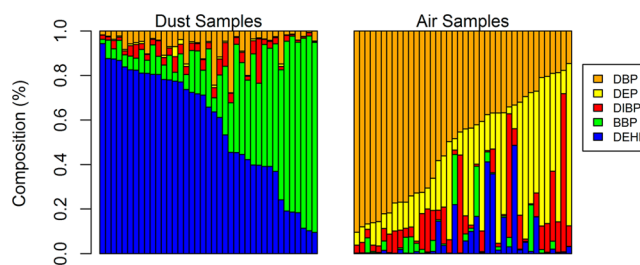


Figure 1. Phthalate analyte contributions to indoor dust and air concentrations. Contrasting profiles are observed with DEHP and BBzP dominating the dust profile (left), while DBP and DEP were more abundant in indoor air (right). Each stacked bar is a phthalate analyte profile for a single ECE facility and sorted by the dominant analyte (DEHP for dust, DBP for indoor air).

however, dust loading levels of DEP, DIBP, DBP, and DEHP were all significantly higher in ECE facilities with vinyl flooring (Mann–Whitney tests, $p < 0.05$).

Results of Cancer and Reproductive Health Risk Assessment. Dose estimates ($\mu\text{g}/\text{kg}/\text{day}$) from the Monte Carlo simulations are presented in Table 4. When converted to exposure levels ($\mu\text{g}/\text{day}$) and compared to Safe Harbor levels, the child-specific MADL for DBP was exceeded for most children (81.5–89.2%) across the 4 age groups (Table 5). Approximately 25% of young children had estimated DBP exposures at least 1 order of magnitude above the reproductive risk level. DEHP exposures (nondietary ingestion only) typically did not exceed the MADL. We found that the child-specific NSRL for DEHP was exceeded in 8.1% of children less than 1 year old and 11.1% of children between 1 and 2 years old, but it was only exceeded for a small number of children ages 2 to <6 years old (0.1%).

Dominant Routes of Exposure. For the four age groups assessed, dermal absorption of gas-phase DBP was the dominant route of exposure, with median percent contribution to exposure between 80 and 86%, depending on the age group. Inhalation was the second most dominant route of exposure (median percent contribution = 11–14%), with nondietary ingestion and dermal absorption via dust adhered playing smaller roles (Figure SI). Unlike DBP, the main route of exposure to DEHP was nondietary ingestion (median percent contribution = 96–98%), then inhalation (median percent contribution = 1–2%). Dermal dust and gas-phase absorption were less significant for DEHP exposure.

Sensitivity Analysis. Our sensitivity analysis showed that DBP exposures were highly dependent on indoor air concentrations and the indoor air transdermal coefficient (SI Table S12). These variables are important to dermal exposure via gas-phase absorption, which was the dominant exposure route. For DEHP, dust concentration and ingestion rates were

Table 3. Spearman's Rank Correlation Coefficients (*P*-Values) between Phthalate Indoor Air Concentrations and Dust Concentrations, Dust Loading, Temperature, Humidity, and Air Exchange Rates^a

analyte	indoor air to dust concentration	indoor air to dust loading	indoor air to room temperature	indoor air to room RH	indoor air to room AER
DEP	0.24 (0.15)	0.23 (0.15)	0.43 (<0.01)	-0.13 (0.44)	-0.25 (0.12)
DIBP	0.47 (<0.01)	0.05 (0.76)	0.51 (<0.01)	0.09 (0.57)	-0.15 (0.37)
DBP	0.46 (<0.01)	-0.06 (0.73)	0.31 (0.05)	0.01 (0.98)	0.01 (0.98)
BBzP	0.58 (0.01)	0.64 (<0.01)	0.09 (0.60)	-0.02 (0.92)	0.05 (0.78)
DEHP	-0.22 (0.18)	-0.14 (0.38)	0.03 (0.85)	-0.14 (0.39)	0.09 (0.60)

^aRH = relative humidity; AER = air exchange rate.

Table 4. Summary of Dose Estimates based on Monte Carlo Simulations^a

analyte	age group	GM	GSD	min	5th%	25th%	50th%	75th%	95th%	max
DBP for comparison to the MADL	birth to <1	0.64	3.37	<0.01	0.09	0.28	0.63	1.45	4.89	168.65
	1 to <2 years	0.65	3.32	<0.01	0.09	0.29	0.64	1.47	4.90	151.63
	2 to <3 years	0.46	3.25	<0.01	0.07	0.20	0.45	1.01	3.30	138.14
	3 to <6 years	0.44	3.31	<0.01	0.06	0.19	0.43	0.98	3.28	135.97
DEHP for comparison to the MADL	birth to <1	0.08	3.25	<0.01	0.01	0.04	0.08	0.18	0.55	14.50
	1 to <2 years	0.16	2.19	0.01	0.04	0.10	0.16	0.28	0.57	3.48
	2 to <3 years	0.13	2.19	<0.01	0.04	0.08	0.13	0.23	0.48	3.72
	3 to <6 years	0.10	2.23	<0.01	0.03	0.06	0.10	0.17	0.37	2.75
DEHP for comparison to the NSRL	birth to <1	0.09	3.01	<0.01	0.02	0.04	0.09	0.19	0.58	14.52
	1 to <2 years	0.17	2.15	0.01	0.05	0.10	0.18	0.29	0.60	6.27
	2 to <3 years	0.14	2.15	<0.01	0.04	0.09	0.14	0.24	0.50	9.05
	3 to <6 years	0.11	2.19	<0.01	0.03	0.06	0.11	0.19	0.40	5.67

^aDose estimates for DBP for comparison to the MADL and DEHP for comparison to the NSRL include uptake from inhalation, nondietary ingestion, and dermal routes of exposure. Dose estimates for DEHP for comparison to the MADL only include uptake from nondietary ingestion. GM = geometric mean; GSD = geometric standard deviation; $\mu\text{g}/\text{kg}/\text{day}$.

Table 5. Summary of MADL and NSRL Risk Quotients^a

analyte	age group	% RQ > 1	GM	GSD	min	5th%	25th%	50th%	75th%	95th%	max
DBP for comparison to the MADL	birth to <1	88.6	4.28	3.37	0.01	0.60	1.87	4.19	9.68	32.60	1124.34
	1 to <2 years	89.2	4.36	3.32	0.02	0.63	1.92	4.27	9.77	32.66	1010.89
	2 to <3 years	82.7	3.05	3.25	0.03	0.46	1.36	2.97	6.70	21.98	920.92
	3 to <6 years	81.5	2.94	3.31	0.01	0.42	1.30	2.88	6.55	21.84	906.49
DEHP for comparison to the MADL	birth to <1	<0.05	0.01	3.25	<0.01	<0.01	0.01	0.01	0.03	0.10	2.50
	1 to <2 years	0.0	0.03	2.19	<0.01	0.01	0.02	0.03	0.05	0.10	0.60
	2 to <3 years	0.0	0.02	2.19	<0.01	0.01	0.01	0.02	0.04	0.08	0.64
	3 to <6 years	0.0	0.02	2.23	<0.01	<0.01	0.01	0.02	0.03	0.06	0.47
DEHP for comparison to the NSRL	birth to <1	8.1	0.21	3.01	<0.01	0.03	0.10	0.21	0.44	1.32	32.80
	1 to <2 years	11.1	0.39	2.15	0.01	0.11	0.24	0.40	0.66	1.37	14.17
	2 to <3 years	0.1	0.10	2.15	<0.01	0.03	0.06	0.10	0.16	0.34	6.13
	3 to <6 years	0.1	0.07	2.19	<0.01	0.02	0.04	0.07	0.13	0.27	3.84

^aRisk quotients for DBP for comparison to the MADL and DEHP for comparison to the NSRL include uptake from inhalation, nondietary ingestion, and dermal routes of exposure. Risk quotients for DEHP for comparison to the MADL only include uptake from nondietary ingestion. % RQ > 1 = percent of children with exposures greater than Safe Harbor level; GM = geometric mean; GSD = geometric standard deviation; results are unitless.

the most important variables due to nondietary ingestion being the major route of exposure.

DISCUSSION

This is the first study to report phthalate air and dust levels measured in California ECE facilities, where approximately 1 million children <6 years old spend time each week.² The median phthalate levels measured are similar to those previously reported in studies of ECE facilities that collected dust^{29–32,38} and air,^{16,30,31,38} but appear to be on the lower range of previously reported concentrations (SI Tables S13 and S14). Phthalate levels found in this and other studies conducted in ECE environments are comparable to levels reported in

residential homes,^{16,59–61} suggesting similar phthalate exposures across different indoor environments.

Dust and air concentrations of three phthalates (DIBP, DBP, and BBzP) were moderately to strongly correlated ($r = 0.46–0.58$; $p < 0.05$), suggesting indoor partitioning between floor dust and airborne phases. Previous research, however, has shown inconsistent findings on correlations between phthalate air and dust levels. Oie et al.⁶² found a significant correlation between DEHP and BBzP concentrations in settled dust and airborne particle phase concentrations in a small sample of homes ($n = 6$). Fromme et al.¹⁶ found no correlation between air and dust levels ($n = 30$), but, in a more recent, larger study ($n = 63$), Fromme et al.³⁸ did find significant correlations for DBP, DIBP, and DEHP. Surprisingly, dust loading levels and

air concentrations were only significantly correlated for BBzP and not DIBP or DBP, although dust concentrations for all 3 analytes were significantly correlated with indoor air concentrations. To our knowledge, no study has investigated the relationship between phthalate dust loading and indoor air concentrations. We also found an association with phthalate loading and vinyl floors present in the child care room, but did not find a similar association with dust concentration and floor type. Given the differences in associated characteristics between dust concentration and loading, further research should report loading levels and test for their association with building parameters, especially considering dust loading may be more predictive of children's exposure than concentration.⁶³

Despite finding moderate correlations between DIBP, DBP, and BBzP indoor dust and air concentrations, the analyte profiles between dust and air samples were different, with DEHP/BBzP dominating indoor dust composition and DBP/DEP being more abundant in indoor air. Similarly, Fromme et al.³⁸ found different compositions of phthalates between indoor air and dust with DBP/DIBP accounting for ~55% of the indoor air concentrations and DEHP accounting for ~70% of dust concentrations. Differences in air and dust phthalate composition are likely due to differences in vapor pressures and equilibrium partition coefficients.^{16,64}

We found consistent positive correlations between indoor air levels and room temperature for DEP, DIBP, and DBP. The relationship between air concentrations and temperature was apparently dependent on the vapor pressures of the phthalates or a shift in the equilibrium partition coefficient from adsorbed-phase to gas-phase. For example, DEP, DIBP, and DBP have higher vapor pressures (2.5×10^{-3} , 2.4×10^{-3} , and 2.4×10^{-3} mm Hg at 25 °C, respectively) compared to BBzP and DEHP (4.4×10^{-5} and 2.0×10^{-5} mm Hg at 25 °C, respectively),⁶⁵ which did not show an association with temperature. This relationship suggests that off-gassing from products or dust containing phthalates may increase with temperature for higher vapor pressure phthalates. While this relationship was observed in a controlled experiment,⁶⁶ no field study has reported this association and one study reported no association between phthalate indoor air levels and temperature.¹⁶ Temperature and increased emissions have been noted for other indoor contaminants including formaldehyde,^{67,68} but further studies are needed to confirm this association for phthalates. If temperature does impact indoor air levels, then differences seen within and between indoor air studies may be, in part, due to differences in room temperature while sampling and vary by season. Therefore, temperature may be an important variable to monitor when collecting indoor air phthalate samples. Although the sample size is small, the lack of correlation between the indoor air concentration of phthalates and air exchange rate highlights an important challenge for mitigating indoor SVOC exposures. Common strategies for reducing indoor pollutant concentrations include increasing fresh air ventilation or reducing the pollutant source, but providing additional fresh air ventilation may not significantly reduce SVOC exposure concentrations.

Previous studies have investigated route contribution to indoor phthalate exposure,^{16,39,69,70} with Bekö et al.³³ being the only study to date to evaluate the importance of dermal absorption of gas-phase phthalates. Bekö et al.³³ used household and child care dust samples to model inhalation, nondietary ingestion, and dermal uptake to compare with phthalate metabolites in paired child urine samples. Bekö et al.'s

route contribution estimates for DBP and DEHP are similar to our findings, with dermal absorption from the gas-phase and nondietary ingestion dominating total uptake, respectively.

To our knowledge, our study is the first to compare estimated phthalate exposures with MADL and NSRL risk levels. Previous studies have compared exposure estimates to RfDs,²⁹ TDIs,^{16,33–38} or DNELs.³⁹ It should be noted that the MADLs for DBP and DEHP as a dose would be 0.15 and 5.8 $\mu\text{g}/\text{kg}/\text{day}$ (oral exposure only), respectively, while the adjusted NSRL for children <2 years and 2–6 years for DEHP as a dose would be 0.4 and 1.5 $\mu\text{g}/\text{kg}/\text{day}$, respectively. These values are appreciably lower than the RfDs (100 and 20 $\mu\text{g}/\text{kg}/\text{day}$, oral exposure only),^{71,72} TDIs (10 and 50 $\mu\text{g}/\text{kg}/\text{day}$),^{73,74} and DNELs (7 and 24 $\mu\text{g}/\text{kg}/\text{day}$)^{75,76} for DBP and DEHP, respectively. Although more conservative, we chose to compare our exposure estimates to the MADL and NSRL because these standards are set in California; therefore, they are directly applicable to our study population.

Our risk assessment shows that children attending a California ECE facility are exposed to DBP and DEHP at levels that exceed cancer and reproductive health-based benchmarks. However, our risk assessment is an underestimate of a child's overall exposure since children are exposed to phthalates in other indoor environments (e.g., home) likely at levels similar to those reported in this study. In addition, other important sources of phthalate exposure to children such as consumer products, toys, and food were not assessed in this study.^{39,69,70,77} For example, our dose estimates are lower than estimated total daily phthalate doses derived from urinary concentrations of phthalate metabolites.^{33,78–80} Bekö et al.³³ estimated the median child dose (3–6 years) of DBP and DEHP from the urinary metabolites to be 3.3 and 4.4 $\mu\text{g}/\text{kg}/\text{day}$, respectively. These dose estimates are an order of magnitude larger than our median DBP and DEHP exposure estimates for children 3–6 years old (0.43 and 0.11 $\mu\text{g}/\text{kg}/\text{day}$, respectively), which were based on environmental contamination in the ECE facilities.

Our sensitivity analysis showed that DBP exposure was highly dependent on the indoor air transdermal coefficient. Limited research has been completed on air-to-skin transdermal uptake of DBP. Due to its importance in estimating DBP exposure and modeled risk, research is needed to define this critical parameter.

To our knowledge, this is the largest study reporting paired dust and airborne phthalate levels in U.S. ECE facilities. Despite this, the sample size still limits the power to draw inferences. While we report moderate to strong correlations between some phthalates in dust and indoor air, we collected dust and indoor air on separate days. The median time between dust and air sample collections was 7 days, and unlikely to have biased these results. A potential limitation is that we only collected samples at one time-point and the results may not reflect contaminant levels on other days, long-term averages, or seasonal variation. Another important limitation was that participation rate among ECE facilities was less than 5% of those contacted. Low participation rates may have led to a selection bias as directors of enrolled ECE programs typically were interested in environmental risks to children's health and may have implemented prior, exposure-prevention behaviors. However, the ECE facilities enrolled in this study reflect a diverse cross-section of institutions in California including center and home-based facilities, Head Start programs and school districts, private and nonprofit providers, middle class families, and low

income and immigrant communities. Finally, we only estimated exposures for inhalation, nondietary ingestion, and dermal exposures, which did not take into consideration other important sources of exposure including consumer products, toys, and food.

Even with our focus solely on exposures from dust and indoor air contamination, we found phthalate exposures in ECE facilities to be of concern for children <6 years of age. Additional research into the effects of early life exposures to phthalates is needed due to the ubiquity of these compounds in our environment and documented childhood exposure levels.

■ ASSOCIATED CONTENT

■ Supporting Information

QA/QC information, dose calculations steps, exposure factor distributions, outdoor air concentrations, indoor/outdoor air ratios, dust loading levels, sample age-specific NSRL and MADL calculations, route contribution figure, vinyl-flooring Wilcoxon–Mann–Whitney test results, sensitivity results, and literature review results. This material is available free of charge via the Internet at <http://pubs.acs.org>.

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The manuscript was written through contributions of all authors. All authors have given approval to the final version of the manuscript

Notes

The authors declare no competing financial interest.

■ ACKNOWLEDGMENTS

We thank the ECE programs that participated in this study. We thank Dr. Martha Sandy of the Office of Environmental Health Hazard Assessment (OEHHA) for reviewing risk evaluation methods and approaches to incorporate age-specific sensitivity factors when evaluating OEHHA NSRLs and MADLs. We also thank Dr. William Nazaroff for advice on strategies to measure air exchange rates. We thank individuals at the Community Child Care Council of Alameda County and Monterey County Child Care Resource and Referral for help with participant recruitment. Finally, we thank the anonymous reviewers of this manuscript for their insightful comments. This research was supported by the California Air Resource Board, Agreement 08305.

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