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Published in:
Environmental Science and Technology

Published: 06/09/2022

Document Version:
Final Published version, also known as Publisher's PDF, Publisher's Final version or Version of Record

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Publication record in CityU Scholars:
Go to record

Published version (DOI):
10.1021/acs.est.2c01161

Publication details:

Citing this paper
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Insights into the Atmospheric Persistence, Transformation, and Health Implications of Organophosphate Esters in Urban Ambient Air

Jia-Yong Lao, Huiju Lin, Xian Qin, Yuefei Ruan,* Kenneth M. Y. Leung, Eddy Y. Zeng, and Paul K. S. Lam*

ABSTRACT: Transformation of organophosphate esters (OPEs) in natural ambient air and potential health risks from coexposure to OPEs and their transformation products are largely unclear. Therefore, a novel framework combining field-based investigation, in silico prediction, and target and suspect screening was employed to understand atmospheric persistence and health impacts of OPEs. Alkyl-OPE transformation products ubiquitously occurred in urban ambient air. The transformation ratios of tris(2-butoxyethyl) phosphate were size-dependent, implying that transformation processes may be affected by particle size. Transformation products of chlorinated- and aryl-OPEs were not detected in atmospheric particles, and atmospheric dry deposition might significantly contribute to their removal. Although inhalation risk of coexposure to OPEs and transformation products in urban ambient air was low, health risks related to OPEs may be underestimated as constrained by the identification of plausible transformation products and their toxicity testing in vitro or in vivo at current stage. The present study highlights the significant impact of particle size on the atmospheric persistence of OPEs and suggests that health risk assessments should be conducted with concurrent consideration of both parental compounds and transformation products of OPEs, in view of the nonnegligible abundances of transformation products in the air and their potential toxicity in silico.

KEYWORDS: novel organophosphate esters, transformation products, suspect screening, particle size, inhalation risk

INTRODUCTION

Air pollution has become one of the biggest environmental threats to human health, causing increased concern worldwide. Approximately 4.2 million deaths are attributable to ambient air pollution every year by inducing noncommunicable diseases related to cardiovascular and respiratory systems. Among atmospheric components, particulate matter, especially fine particles, is a significant contributor to the related mortality. It has been reported that cardiovascular mortality risk can be increased by 8%−18% with an increase in fine particle level of 10 μg m⁻³. Fine particles can penetrate deep into the human’s lungs, blood, or brain and, consequently, transport chemical contaminants into the human body. Rapid urbanization has inevitably aggravated ambient air pollution from abundant particulate matter and associated contaminants, including emerging organic pollutants, and the consequential air pollution potentially poses a higher health risk than ever before.

Organophosphate esters (OPEs) are manufactured and used as flame retardants, plasticizers, or multipurpose agents to meet the wide requirements in various domains, such as construction, transportation, textile, and electronics manufacture. The global annual consumption of OPEs is still growing with increasing demand, especially in emerging electrical and electronic markets. Some novel OPEs have emerged to meet the large market demand, such as t-butylphenyl diphenyl phosphate (BPDP), resorcinol bis(diphenyl phosphate) (RDP), and bisphenol A bis(diphenyl phosphate) (BADP), which are extensively used in electrical and electronic equipment. Since OPEs are additives within host materials without chemical bonding, they can be easily released to the ambient air via volatilization and abrasion and become ubiquitous in the atmosphere. In particular, OPEs are extensively present in the urban ambient air of different parts of the globe, such as in Guangzhou, China, Stockholm, Sweden, and, consequently, health risks related to OPEs may be underestimated as constrained by the identification of plausible transformation products and their toxicity testing in vitro or in vivo at current stage. The present study highlights the significant impact of particle size on the atmospheric persistence of OPEs and suggests that health risk assessments should be conducted with concurrent consideration of both parental compounds and transformation products of OPEs, in view of the nonnegligible abundances of transformation products in the air and their potential toxicity in silico.

KEYWORDS: novel organophosphate esters, transformation products, suspect screening, particle size, inhalation risk
Given the adverse health effects of OPEs, such as neurotoxicity, hepatotoxicity, and immunotoxicity, their prevalence and persistence in urban ambient air certainly deserve broad attention.

Additionally, transformation of OPEs via physicochemical and microbial processes can probably produce new atmospheric pollutants, which can be more toxic and more persistent than the parent OPEs. Therefore, transformation products of OPEs may significantly contribute to ambient air pollution and increased health risk. Although bench-scale experiments and model simulations have been conducted to understand the kinetics and mechanisms of OPE atmospheric transformation, the transformation potential and products of OPEs in ambient air remain largely unknown.

Context-specific factors, such as meteorological conditions, radical abundances, and constituents of particles (e.g., microbes and metallic and organic species), can greatly influence the transformation processes in the air. This complexity highlights the need to evaluate the persistence and transformation of OPEs in the natural atmosphere and potential health risks.

To address the above knowledge gap, urban ambient air samples were collected and analyzed to (1) characterize the occurrence of atmospheric traditional and novel OPEs in different urban settings, (2) elucidate the atmospheric persistence of OPEs in the context of deposition and transformation, (3) identify OPE transformation products and their potential toxicity using target and suspect screening methods, and (4) evaluate the inhalation risk of coexposure to atmospheric OPEs and transformation products. The overarching goal of this study was to provide insights into the atmospheric persistence of OPEs and the potential health risk from coexposure to OPEs and their transformation products, assisting in the improvement of health risk assessment of OPEs.

**MATERIALS AND METHODS**

**Materials.** Traditional OPEs have been commonly included in routine environmental monitoring, whereas “novel” OPEs have been recently recognized regarding their occurrence in consumer products and in the environment. Context-specific factors, such as meteorological conditions, radical abundances, and constituents of particles (e.g., microbes and metallic and organic species), can greatly influence the transformation processes in the air. This complexity highlights the need to evaluate the persistence and transformation of OPEs in the natural atmosphere and potential health risks.

To address the above knowledge gap, urban ambient air samples were collected and analyzed to (1) characterize the occurrence of atmospheric traditional and novel OPEs in different urban settings, (2) elucidate the atmospheric persistence of OPEs in the context of deposition and transformation, (3) identify OPE transformation products and their potential toxicity using target and suspect screening methods, and (4) evaluate the inhalation risk of coexposure to atmospheric OPEs and transformation products. The overarching goal of this study was to provide insights into the atmospheric persistence of OPEs and the potential health risk from coexposure to OPEs and their transformation products, assisting in the improvement of health risk assessment of OPEs.

**Sample Extraction and Quantitative Analysis.** The particulate and gaseous samples were extracted with a mixture of n-hexane, dichloromethane, and acetone (2:2:1 in volume) by ultrasonic and Soxhlet extraction, respectively. The gaseous sample extract was further purified using a Florisil cartridge by ultrasonic and Soxhlet extraction, respectively. The gaseous sample was spiked with surrogate standards prior to extraction, and the concentrated extract was spiked with internal standards prior to quantitative analysis. The quantitation of OPEs and their transformation products was performed using an Agilent 1290 Infinity liquid chromatograph coupled to a Triple Quad 5500 mass spectrometer (LC−MS/MS; Sciex, Redwood City, CA). Detailed sample treatment procedures and instrumental analysis are provided in Text S1.

**A Novel Framework to Improve the Understanding of Atmospheric Persistence of OPEs and Related Health Impacts.** A novel framework consisted of field-based investigation, in silico prediction, and target and suspect hydroxylated derivatives. Due to the lack of authentic standards, only 10 OPE transformation products with commercially available authentic standards were obtained, including di-n-butyl phosphate (DNBP), bis(2-chloroethyl) phosphate (BCEP), bis(1-chloro-2-propyl) phosphate (BDCIPP), bis(2-butoxethyl) 2-hydroxymethyl phosphate (BHOEHEP), bis(2-butoxethyl) 2-(3-hydroxybutoxy)ethyl phosphate (3-OH-TBOEP), bis(2-butoxethyl) phosphate (BHOEP), bis(2-ethylhexyl) phosphate (BEHP), diphenyl phosphate (DHP), and di-o-cresyl phosphate (BMPP).

These authentic standards were purchased from Chiron AS, Wellington Laboratories (Guelph, Ontario, Canada), and Toronto Research Chemicals (Toronto, Ontario, Canada). Detailed information on native, surrogate and internal standards is provided in Tables S1 and S2.
screening was employed to comprehensively understand the atmospheric persistence of OPEs and the health impacts of coexposure to OPEs and their transformation products in urban ambient air. The transformation processes of OPEs are complicated in the atmosphere as they can be subject to both abiotic and microbial transformations, and particle sizes may affect their atmospheric transformation efficiencies. These aspects have not been well addressed in the atmosphere. Therefore, it is critical to conduct a field-based investigation to collect size-segregated atmospheric particles and gases in different urban-related settings. Then, in silico methods based on the reaction libraries were applied to predict plausible transformation products. Plausible transformation products via microbial biodegradation were predicted using the EAWAG-BBD pathway prediction system (http://eawag-bbd.ethz.ch/predict/index.html). Besides, the physicochemical properties and potential toxicities of plausible transformation products were predicted in silico using quantitative structure–activity relationship (QSAR) model (https://qsartoolbox.org/) to study their environmental behaviors and health effects. Subsequently, on the basis of the in silico results, a mass-to-charge ratio (m/z) database of plausible transformation products was developed for suspect screening. Suspect screening using a high-resolution mass spectrometer was employed to identify proposed transformation products, while target screening was used to quantify the transformation products with authentic standards in field samples. The novel framework employing in silico prediction and suspect screening can cost-effectively identify OPE transformation products in the field samples. It can also help to prioritize OPE transformation products of concern and conduct to estimate the health risks from exposure to transformation products in urban ambient air.

The suspect screening of transformation products was performed using an Agilent 1290 Infinity II liquid chromatograph coupled to a high-resolution Sciex X500R quadrupole time-of-flight mass spectrometer (QTOF–MS) in positive (ESI+) and negative (ESI−) electrospray ionization modes. Information-dependent acquisition (IDA) with dynamic background subtraction was conducted for QTOF–MS data acquisition. The detailed QTOF–MS instrumental method is presented in Text S1.

Figure 1. Particulate and gaseous concentrations (pg m−3) of traditional and novel organophosphate esters (OPEs) in urban ambient air of three different urban areas, three sewage treatment plants (STP), and a landfill [bar-chart] and the relative abundance (%) of individual traditional OPEs in the particulate and gaseous phases [pie-charts].

Environmental Science & Technology pubs.acs.org/est Article https://doi.org/10.1021/acs.est.2c01161 Environ. Sci. Technol. 2022, 56, 12003–12013
Quality Assurance and Quality Control. A procedural blank and a field blank were analyzed for each batch of particulate samples, and a procedural blank was analyzed for two batches of four gaseous samples and the related field blanks. The average recoveries of OPE surrogate standards ranged from 72% to 128% and from 52% to 127% in particulate and gaseous samples, respectively. The average recoveries of OPE transformation products ranged from 86% to 111% in particulate samples. Gaseous transformation products were not quantified because of the low recoveries of their surrogate standards in gaseous samples. Concentrations of OPEs and transformation products in field samples were corrected by those in corresponding procedural and field blanks. The lowest calibration concentrations divided by the sample volumes were defined as the reporting limits, presented in Table S4. Detailed information is provided in Text S1. External automated calibration was carried out after 5 sample injections and the mass error should be <2 ppm to ensure the mass accuracy in QTOf-MS.

Evaluation of Inhalation Risk. International Commission on Radiological Protection (ICRP) model was adopted to estimate the regional respiratory deposition of OPEs and their transformation products in the human respiratory tract. The deposition concentrations of size-fractionated particulate OPEs and their transformation products on specific regions, that is, head airway (HA, including nose, mouth, pharynx, and larynx), tracheobronchial region (TB, including trachea and bronchus), and alveolar region (AR, including bronchioles, alveolar duct, and alveoli) were estimated (Text S2). Considering the size-dependent deposition efficiency and combined risk of OPEs and transformation products, the daily intake (DI, pg kg bw⁻¹ d⁻¹), hazard quotient (HQ, unitless), and hazard index (HI, unitless) via inhalation pathway were calculated by

\[
DI_n = \frac{(C_{dp,n} + C_{ga,n}) \times EF \times IR}{BW} \\
HQ_n = \frac{DI_n}{RFD_n} \\
HI = HI_{OPEs} + HI_{TPPs} = \sum HQ_n
\]

where \(C_{dp,n}\) and \(C_{ga,n}\) are the deposition concentration of particle-bound compound \(n\) in the human respiratory tract and gaseous concentration of compound \(n\) in ambient air, respectively (pg m⁻³); EF is the exposure fraction in a specific setting (hours per day, unitless); IR is the inhalation rate (m³ day⁻¹); BW is the human body weight (kg); RFD is the reference dose of compound \(n\) (pg kg⁻¹ d⁻¹); and HI is denoted as the sum of HQ for all investigated OPEs and their transformation products via inhalation pathway. The uncertainty and variability of DI and HI were evaluated by Monte Carlo simulation. The scenario that on-site workers exposed to the ambient air around the sewage treatment plant and the landfill for 8 h per day was regarded as occupational exposure to the urban ambient air, while the general population exposed to the ambient air under the normal settings was regarded as the case of nonoccupational exposure. The detailed human exposure factors are provided in Table S5.

RESULTS AND DISCUSSION

Widespread Occurrence of Traditional and Novel OPEs in Ambient Air. The concentrations of TPrP, CDP, and IDDPP were lower than the reporting limits in particulate and gaseous samples collected from all settings and only a trace amount of V6 was detected in atmospheric particles of STP_B1. Atmospheric mean concentrations of traditional OPEs (particulate and gaseous phases combined) were 7.04 × 10⁶ pg m⁻³ in the southern coast, 7.52 × 10³ pg m⁻³ in the urban margin, 6.89 × 10⁵ pg m⁻³ in the urban center, 7.49 × 10⁴ (B1), 7.69 × 10⁴ (B2), and 1.38 × 10⁵ (C) pg m⁻³ in the sewage treatment plants, and 1.13 × 10⁵ pg m⁻³ in the landfill (Tables S8–S9). The present study is the first to report atmospheric OPEs around sewage treatment plants, where wastewaters and sludge have been frequently found to contain considerable amounts of OPEs.⁴⁶ The highest level of traditional OPEs in bulk air occurred in STP_C (Figure 1), implicating wastewater treatment processes as potential emission sources. The bulk air of landfill also exhibited higher atmospheric concentrations of traditional OPEs than those of other settings in Hong Kong (Figure 1) or even landfills in Canada.⁵⁰ Given that much higher concentrations of OPEs were discerned in the atmospheric particles of the 12th size fraction (<0.056 μm) in the present study, but only 11 size fractions of atmospheric particles (≥0.056 μm) were collected in the landfill (Table S3). Therefore, higher gaseous concentrations of traditional OPEs in the landfill compared to other settings should be attributable to the contribution of ultrafine particles of <0.056 μm. Ultrafine particles with large specific surface area can have a great ability to sorb OPEs.⁵¹ Besides, the abundant occurrence of OPEs in ultrafine particles may be ascribed to direct emission from the sources. In this context, high abundance of OPEs in ultrafine particles should raise more concern. Since the landfill under investigation is the largest one in Hong Kong and processes municipal, construction, and special wastes totaling 6400 tonnes per day,⁵₂ fugitive emissions from the immense quantity of solid wastes certainly increase the OPE burden in the air.

Total concentrations (particulate and gaseous concentrations combined) of traditional OPEs in urban ambient air was much higher in Hong Kong than in Dalian, China (mean = 1.63 × 10⁷ pg m⁻³), Toronto, Canada (mean = 1.79 × 10⁷ pg m⁻³),⁴⁵ and Stockholm, Sweden (median = 3.10 × 10⁷ pg m⁻³).⁴⁶ Hong Kong is not only highly urbanized with a high population density but also under the influence of air masses coming from neighboring highly urbanized cities and e-waste dismantling centers in South China (Figure S2). For instance, Yue et al.⁴⁶ found that the neighboring areas of an e-waste dismantling park in Guiyu, South China were subject to high levels of atmospheric OPEs (1.29 × 10⁶–7.09 × 10⁵ pg m⁻³). Luo et al.²¹ also revealed that atmospheric OPE concentrations were comparable in the metropolitan region in Guangzhou (mean = 1.38 × 10⁶ pg m⁻³) and an e-waste dismantling zone in Qingyuan, South China (mean = 1.30 × 10⁵ pg m⁻³). Because of the impacts of high urbanization and regional diffusion from e-waste recycling zones, Hong Kong, Guangzhou, and other major cities in this region may suffer from large amounts of atmospheric OPEs.¹⁶,²¹

In the particulate phase, TBOEP was the predominant constituent among traditional OPEs, accounting for 93.5%–96.9% of the total concentration, followed by TCIPP, while TPHP and TBOEP dominated among traditional OPEs in the gaseous phase (Figure 1). The high abundances of TBOEP, TCIPP, and TPHP should be partially attributed to their wide applications in buildings/construction, electronics/electric devices, automobiles, and textile/adhesives/coatings. Specifi-
physically, TBOEP is extensively utilized in PVC, wax, floor polish, paper coating, and rubber, covering almost all consumer products, while TPHP is usually mixed with other OPEs for broad applications.\(^7\) Wide applications in daily consumer products may contribute to large emissions, particularly in the highly urbanized area. In addition to the direct emissions from the consumer products or waste treatment processes in Hong Kong, continental air masses from the neighboring urbanized cities may also exacerbate the atmospheric burden of these widely used OPEs (Figure S2).\(^{21}\) Apart from the traditional OPEs, novel OPEs including BPDPP, RDP, and BADP were also frequently detected (Tables S8 and S9). The mean concentrations of atmospheric novel OPEs ranged from 152 to 495 pg m\(^{-3}\) among the sampling settings. Since BPDPP, RDP, and BADP are commonly utilized in PC/ABS and HIPS/PPO resins, which are widely embedded in electrical and electronic equipment housings,\(^{11,12}\) a high-tech zone in Shenzhen or the landfill in Hong Kong (Figure S1) may have contributed a considerable amount of novel OPEs to the ambient air, leading to higher concentrations of novel OPEs in the air of the landfill and urban margin close to the high-tech zone than other those in ambient air.

The logarithm of the gas–particle partition coefficient (log \(K_{oa}\)) for individual OPEs (11 in total) ranged from \(-0.43\) to \(-3.19\) (Table S10). Because of the wide range of physicochemical properties of OPEs, log \(K_{oa}\) was not linearly correlated with the logarithm of the n-octanol–air partition coefficient (log \(K_{oa}\)) and vapor pressure. The gas–particle distribution of OPEs might also be largely affected by their n-octanol–water partition coefficient and solubility when log \(K_{oa}\) ≤ 9 as a result of particle hygroscopicity.\(^{55,56}\) Water uptake enables particles to not only change their sizes but also absorb highly soluble OPEs under certain relative humidity.\(^{55,56}\) This may explain why TEP and TCEP had a low log \(K_{oa}\) but still had a high \(K_{oa}\). Overall, the physicochemical properties of OPEs and particles, as well as meteorological conditions greatly influenced the gas-particle distribution of OPEs, which could explain why OPEs predominantly occurred in the particulate phase (Figure 1).

**Particle Size Distribution and Effects of Dry Deposition on Residence Time of Atmospheric OPEs.** The particle size distribution patterns of OPEs varied with individual OPEs (Figure S3). Specifically, TBP, TBOEP, TCEP, TCIPP, and TDCIPP more commonly occurred in ultrafine particles, with an average fraction of ∼40%; while most aryl-OPEs, particularly, TMPP, BPDP, RDP, and BADP, were predominantly adhered to particles in the accumulation mode, with an average fraction of ∼67%. The mass fraction of OPEs was generally higher in fine particles than in coarse particles regardless of sampling setting, suggesting that fine particles are more responsible for the transport and fate of particulate OPEs. To further elucidate the source-related particle size distribution pattern, the geometric mean diameters (GMDs)\(^{17}\) of individual OPEs were calculated (Text S2). The GMDs of OPEs ranged from 0.065 to 1.69 μm, smaller than those observed in urban Guangzhou, especially for TBOEP.\(^{21}\) The GMDs of alkyl- and chlorinated- (Cl-)OPEs were relatively smaller than those of aryl-OPEs (Figure 2). The GMDs of OPEs were greater in the ambient air of the special settings than in the normal settings. Moreover, the GMDs of OPEs were positively correlated with temperature (\(p < 0.05\)), that is, the size distribution of particulate OPEs were strongly affected by source characteristics and meteorological conditions.

As OPEs predominantly occur in the particulate phase, the removal efficiencies of atmospheric particles can greatly affect the atmospheric residence time of particulate OPEs. More importantly, dry deposition dominates deposition flux in dry season. Therefore, dry deposition and transformation processes largely determine the residence time of atmospheric OPEs in dry season. Concerning the dry deposition, the mean dry deposition fluxes of particulate OPEs ranged from 157 to 620 pg m\(^{-2}\) s\(^{-1}\) at the sampling settings (Table S11). Given that the dry deposition velocity is size-dependent,\(^{57}\) the dry deposition fluxes of particulate OPEs in the investigated urban ambient air were generally lower in the accumulation mode particles than in ultrafine and coarse particles (Figure S3). OPEs affiliated with accumulation mode particles would have a longer residence time in the atmosphere and were not readily removed by dry deposition, as compared to ultrafine and coarse particles. The mean dry deposition velocities of individual OPEs ranged from 0.32 to 1.03 cm s\(^{-1}\), relatively
larger than those in urban Guangzhou (0.31–0.38 cm s\(^{-1}\)).\(^{21}\) Compared to other OPEs, TBOEP was less likely to be removed from the atmosphere by dry deposition because of its much lower dry deposition velocity (mean = 0.39 cm s\(^{-1}\)) (Figure 2b). This also explained why TBOEP was abundant in the ambient air of Hong Kong. Particulate TCIPP and TCEP had relatively high dry deposition velocities (Figure 2b); therefore, dry deposition would more effectively shorten their atmospheric residence time in the urban ambient air, as compared to most of the other particulate OPEs.

**Transformation Potential in Ambient Air and Influence of Particle Size.** As mentioned above, apart from dry deposition, transformation processes can also significantly affect the atmospheric persistence of OPEs.\(^{26-28,30}\) Therefore, well-known transformation products were quantitated to evaluate the transformation potential of OPEs in ambient air and to understand the influence of particle size. Because of the low recoveries of the surrogate standards in gaseous samples, gaseous transformation products were not further discussed. The particulate concentrations of BCEP, BCI IPP, BDCIPP, DPHP, and BMPP were lower than the reporting limits, while DNBP, BBOEHEP, 3-OH-TBOEP, and BEHP were detectable in the particulate phase (Table S8). Previous studies have demonstrated that compared to Cl- and aryl-OPEs, alkyl-OPEs can be more rapidly degraded by microorganisms or more easily oxidized by \(\cdot OH\).\(^{30,35,44,58,59}\) This might be one of the important reasons for higher abundance of transformation products of alkyl-OPEs in the atmospheric particles.

The mean particulate concentrations of transformation products were in a range of 1520–3740 pg m\(^{-3}\) in the ambient air of Hong Kong, and the average mass ratios of DNBP/TBP, BBOEHEP/TBOEP, 3-OH-TBOEP/TBOEP, BBOEP/TBOEP, and BEHP/TEHP were 0.46%, 3.4%, 0.094%, 0.018%, and 21%, respectively. However, some transformation products (e.g., DNBP, BBOEP, and DPHP) are manufactured as plasticizers or for other industrial purposes. Therefore, they can be derived from direct emission sources instead of transformation processes.\(^{42}\) Since no significant correlation was discerned between the concentrations of TBP and DNBP or TEHP and BEHP (Figure S5), the presence of DNBP and BEHP might not result from abiotic/biotic degradation only but also from direct emission. Significant correlations between the concentrations of TBOEP and its transformation products, that is, BBOEHEP (\(r^2 = 0.92\)), 3-OH-TBOEP (\(r^2 = 0.91\)), and BBOEP (\(r^2 = 0.54\)) (\(p < 0.05\); Figure 3a–3c), suggested that these transformation products were possibly derived from transformation processes.

As TBOEP had the greatest \(\cdot OH\) rate constant and the highest abundance in the sampled ambient air, its transformation products were much more abundant than other transformation products of alkyl-OPEs (Tables S8 and S12). Among the transformation products of TBOEP, BBOEHEP was predominant, with a relative abundance of 97% on average, followed by 3-OH-TBOEP (28%). Two isomers of 3-OH-TBOEP, possibly 1-OH-TBOEP and 2-OH-TBOEP, were also detected in the particulate phase but were not included in quantitative analyses due to the lack of authentic standards (Figure S6). In this context, the total transformation ratios (defined as mass ratio of transformation product to the related parent OPE) of TBOEP were in the range of 1.9%–6.3%, which varied with sampling setting, probably because the
transformation of particulate OPEs might be source-related and was largely affected by meteorological conditions (e.g., relative humidity and temperature), the abundance of \( \cdot \)OH, and the physicochemical properties of OPEs and particles (e.g., particle size), among others.\textsuperscript{26–29} Interestingly, the transformation ratios of BBOEHEP/TBOEP and BBOE/TBOEP were significantly size-dependent (\( p < 0.05 \)) and generally decreased with increasing particle diameter within 1 \( \mu m \) (Figure 3d and 3f). It is likely because the reactivity of particulate organics upon the \( \cdot \)OH exposure can be strongly affected by the organic surface area-to-volume ratio (SA/V), and finer particles with higher SA/V can have higher \( \cdot \)OH oxidation rate constants.\textsuperscript{26} Lower transformation ratios were discerned in the size fractions of 0.32–1.0 and >10 \( \mu m \), suggesting that TBOEP sorbed on these particles was more resistant to transformation in the ambient air. Therefore, particle size greatly influenced the atmospheric persistence of OPEs by determining the efficiency of dry deposition and by affecting the transformation efficiency.

Plausible Transformation Products and Potential Health Impacts. As illustrated in the previous section, OPEs can be somewhat oxidized by \( \cdot \)OH in the atmosphere. For example, alkyl-OPEs can be transformed to diester or monoester derivatives by \( \cdot \)OH addition to P=O bond.\textsuperscript{26,36} Alkyl-OPEs can also be transformed to hydroxylated and carbonylated derivatives by the H-abstraction pathway through the Russell mechanism.\textsuperscript{26,36,61,62} Specifically, \( \cdot \)OH initially attacks the C atom in the alkyl chain to form an alkyl radical, which then rapidly reacts with atmospheric \( \text{O}_2 \) to generate a peroxy radical; the peroxy radicals subsequently decompose to form hydroxylated and carbonylated derivatives through the Russell mechanism.\textsuperscript{26,60–63} These hydroxylated and carbonylated derivatives can further react with \( \cdot \)OH to form carboxylic acids. Therefore, the derivatives can contain \( -\text{CO}^- \), \( -\text{CHO} \), or \( -\text{COOH} \). On the basis of the in \textit{silico} prediction results, microbial biodegradation can also transform OPEs to diester or monoester and hydroxylated derivatives, as well as the derivatives containing \( -\text{CO}^- \), \( -\text{CHO} \), or \( -\text{COOH} \) (Figures S7–S9), while hydrolysis usually generates diester and monoester derivatives and may play a minor role in the transformation processes of OPEs.\textsuperscript{54} Cl- and aryl-OPEs have similar transformation pathways as alkyl-OPEs, but Cl atom in the C–Cl bond of Cl-OPEs can also be substituted by \( \cdot \)OH.\textsuperscript{26,31}

The main plausible transformation products of OPEs from the proposed transformation pathways were screened in the particulate samples by QTOF–MS. In addition to the transformation products mentioned above, four other transformation products, that is, \( \text{C}_9\text{H}_{15}\text{O}_2\text{P} \), \( \text{C}_{11}\text{H}_{17}\text{O}_2\text{P} \), \( \text{C}_7\text{H}_{13}\text{O}_2\text{P} \), and \( \text{C}_6\text{H}_{12}\text{O}_2\text{P} \), were tentatively identified with satisfactory MS intensities and mass errors <5 ppm (Figure S10). It should be noted that none of the Cl-OPE transformation products were detectable in atmospheric particulate samples. It is possible that concentrations of Cl-OPE transformation products were below the detection limits. The tentative transformation products, that is, \( \text{C}_{16}\text{H}_{33}\text{O}_2\text{P} \) and \( \text{C}_{12}\text{H}_{29}\text{O}_2\text{P} \), were inferred to be (OH)\textsubscript{2}-BEHP and OH-BBOEP, respectively; this conclusion needs to be confirmed and quantified by authentic standards. This finding highlights the likelihood of multiple transformations of alkyl-OPEs in the atmosphere. However, probably because of the short time scales of transformation reactions in these urban settings, only early generation transformation products could be detected in the present study.

The QSAR model was utilized to elucidate the environmental behavior, persistence, and potential toxicity of alkyl-OPE transformation products based on their chemical structures. Alkyl-OPE transformation products were predicted to behave similarly to alkyl-OPEs in terms of log \( K_{\text{ow}} \) and the \( \cdot \)OH rate constant, but their hydrophilicity was largely improved by the addition of hydroxyl groups (Tables S12 and S13). With relatively lower log \( K_{\text{ow}} \), some transformation products of hydrophobic alkyl-OPE may be more readily distributed in the human body, such as some of the transformation products of TEHP. Moreover, some transformation products, with the addition of more toxicologically active functional groups, for example, carbonyl groups, may have higher toxicities than parent OPEs.\textsuperscript{27} The alkyl-OPE transformation products are placed at the highest hazard level (Class III) in the Cramer classification scheme (Table S13). Most of these transformation products, especially TBOEP transformation products, were also predicted to potentially induce neurotoxicity, hepatotoxicity, and renal toxicity. Therefore, OPE transformation products are plausible sources of potential human health risks.

Combined Inhalation Risk. Health risk assessments of occupational exposure to the special settings and nonoccupational exposure to the normal settings were evaluated. The deposition concentrations of particulate OPEs in different parts of the human respiratory tract were highly particle size-dependent based on the ICRP model (Figure 4a).\textsuperscript{43} Coarse particles mainly contributed OPEs to the HA region (>79%) (Table S14). Compared to other particles, ultrafine particles not only contributed the highest deposition concentrations of total OPEs to the AR region but also contained relatively higher deposition concentrations in the HA and TB regions (Figure 4a). Ultrafine particles may readily access other human organs, for example, the brain, through the respiratory tract, causing negative effects on the cardiovascular and central nervous systems.\textsuperscript{5,6} In particular, OPE transformation products sorbed on ultrafine particles can be directly inhaled into the human body, but their inhalation exposure and related health effects are largely unknown. Therefore, OPE transformation products were also included to comprehensively evaluate the inhalation risk related to OPEs.

The deposition pattern of OPE transformation products in the human respiratory tract was similar to that of OPEs. The daily inhalation intakes of total OPEs (including particulate and gaseous OPEs) were 4260 ± 1970 and 1510 ± 663 pg kg \( \text{bw}^{-1} \text{d}^{-1} \) for occupational and nonoccupational exposure, respectively. In contrast, transformation products in the particulate phase contributed a much lower daily intake (Figure 4b), that is, 82 ± 43 and 52 ± 25 pg kg \( \text{bw}^{-1} \text{d}^{-1} \) for occupational and nonoccupational exposure, respectively. Detailed daily intakes of individual OPEs and transformation products are provided in Table S16. The mean HI values for occupational and nonoccupational exposure, respectively. The combined inhalation risk related to OPEs were several orders of magnitude lower than 1 (Figure 4c), with 95% confidence intervals of 1.09 × 10\(^{-4}\)–4.86 × 10\(^{-4}\) and 0.384 × 10\(^{-4}\)–1.62 × 10\(^{-4}\) for occupational and
nonoccupational exposure, respectively. Therefore, both occupational and nonoccupational inhalation risks via inhalation exposure to OPEs and their transformation products in Hong Kong ambient air were low. Although the combined risks were not significantly elevated by considering the transformation products, the health risks posed by transformation products still need to be carefully considered. Because the species and toxicities of transformation products remain largely unclear. Notably, the occupational inhalation risk was ∼3 times higher than the nonoccupational inhalation risk, workers were subject to a higher inhalation risk with regard to OPE exposure. Although workers seem to suffer a much lower risk from the landfill and sewage treatment plants in Hong Kong than from the e-waste dismantling park in South China, more measures are still needed to effectively reduce their occupational exposure.

Implications and Future Perspectives. Given that OPEs affiliated with fine particles are able to undergo longer-range atmospheric transport, the abundances of OPEs in fine particles should draw more attention, especially those released from the landfill and sewage treatment plants. Particle size not only determines the dry deposition velocity of OPEs but also affects the transformation efficiency of OPEs. Both the dry deposition flux and transformation rate of each particle size fraction should be simultaneously considered to better understand the atmospheric persistence of OPEs, which should be the focus of future studies. The present study not only highlights the importance of particle size effects on the atmospheric persistence of OPEs but also emphasizes the need to fully investigate OPE transformation products and carefully consider them in health risk assessment. On the basis of the in silico modeling, some transformation products were predicted to be less toxic than parent OPEs, but others might be more toxic or comparable to parent OPEs. The transformation products containing −CO− or −CHO may potentially exhibit toxic effects on the human body. However, the potential toxicity of OPE transformation products needs to be verified in vitro and in vivo. Future studies should pay attention to the occurrence and species of transformation products in the urban ambient air, especially those with toxic functional groups. The combined health risks posed by OPEs and the transformation products need to be comprehensively investigated to determine whether the transformation processes would aggravate the health risks related to OPEs. The transformation efficiencies of OPEs may be helpful for their source diagnostics; the species of transformation products and the related transformation ratios combined with the air masses trajectories may help to understand the potential emission sources and the transport distances, which can be a direction for future studies. Sewage treatment and landfilling were found to be potential sources of OPEs and might exacerbate the OPE burden in ambient air and increase the occupational inhalation risk. This result calls for more effective management to minimize emissions and potential health risks from urban infrastructures.

ASSOCIATED CONTENT
Supporting Information
The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.2c01161.

Additional tables and figures as mentioned in the main text (PDF)

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Figure 4. (a) Spline curves show the mean deposition concentrations of total particulate OPEs in head airway (HA), tracheobronchial (TB), and alveolar (AR) regions of the human respiratory tract; solid and dotted lines represent occupational and nonoccupational exposure. (b) Box-whisker plots show the estimated daily intake (pg kg bw⁻¹ d⁻¹) of total OPEs and their transformation products (TPs) via inhalation; black dots represent the minimum and maximum values, and dash lines represent the mean values. (c) Cumulative probability (%) of hazard index for inhalation exposure to OPEs and TPs.
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Notes

The authors declare no competing financial interest.

ACKNOWLEDGMENTS

The present study was supported by the Science, Technology, and Innovation Commission of Shenzhen Municipality (JCYJ20190812155805559) and the Research Grants Council, University Grants Committee Committee Theme-Based Research Scheme (T21-602/16-R). Special thanks go to Qi Wang, Rongben Wu, University Grants Committee Theme-Based Research Scheme (JCYJ20190812155805559) and the Research Grants Council, The authors declare no competing financial interest.

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