**Table 2. Concentrations of Brominated Products in Marine Sponges and Fish Samples Collected between 2005 and 2009 from Palau and the Western Pacific**

| | marine sponge (Palau) | | unicorn fish ($n$ = 7, Palau) | surgeon fish ($n$ = 7, Guam) | grouper ($n$ = 11, Japan) |
|---|---|---|---|---|---|
| | *Haliclona* sp. | *Callyspongia* sp. | *N. lituratus* | *A. xenthopterus* | *Epinephelinae* sp. |
| extracts (%) | 14.5[a] | 21.6[a] | 2.5 ± 1.2[b] | 3.3 ± 1.1[b] | 8.5 ± 4.2[b] |
| | concentration (μg/g EOM) | | concentration (ng/g lipid) | | |
| | | | Neutral Fraction | | |
| 2′-MeO-BDE68 | 43.2 | 19.1 | 130 (20−280) | 110 (18−220) | 290 (65−840) |
| 6-MeO-BDE47 | 8.22 | 5.90 | 80 (20−190) | 120 (14−240) | 330 (40−950) |
| 2,2′-diMeO-BB80 | 2.84 | 2.92 | 25 (10−120) | 28 (15−60) | 120 (75−600) |
| 2′,6-diMeO-BDE68 | 6.72 | 4.51 | 41 (15−110) | 43 (10−80) | 82 (30−250) |
| **sum of four congeners** | **61.0** | **32.4** | **510 (70−1320)** | **301 (57−500)** | **1072 (210−3060)** |
| $\Sigma_4$MeO-triBDE | 4.29 | 4.17 | 31 (10−87) | 20 (5−38) | 46 (15−160) |
| $\Sigma_2$MeO-tetraBDE | 51.5 | 25.0 | 210 (40−470) | 230 (32−460) | 620 (105−1790) |
| ΣMeO-pentaBDE | 0.31 | 1.56 | nd | nd | nd |
| ΣdiMeO-tetraBDE | 7.24 | 5.56 | 41 (15−110) | 43 (10−80) | 82 (30−250) |
| ΣdiMeO-pentaBDE | 0.16 | 0.17 | nd | nd | nd |
| ΣdiMeO-hexBDE | nd | nd | nd | nd | nd |
| **total** | **63.5** | **36.5** | **280 (65−670)** | **290 (47−580)** | **550 (150−2200)** |
| | | | Phenolic Fraction | | |
| 2′-OH-BDE68 | 4.71 | 20.4 | nd | nd | nd |
| 6-OH-BDE47 | 3.41 | 20.5 | nd | nd | 10 (<0.2−20) |
| 2,2′-diOH-BB80 | 0.39 | 1.25 | nd | nd | nd |
| 2′,6-diOH-BDE68 | 18.7 | 37.9 | nd | nd | nd |
| **sum of four congeners** | **27.2** | **80.1** | | | **10 (<0.2−20)** |
| ΣOH-triBDE[c] | nd | nd | nd | nd | nd |
| ΣOH-tetraBDE[c] | 8.12 | 54.8 | nd | nd | nd |
| ΣOH-pentaBDE[c] | 2.59 | 12.9 | nd | nd | nd |
| ΣdiOH-tetraBDE[c,d] | 18.7 | 37.9 | nd | nd | nd |
| ΣdiOH-pentaBDE[c,d] | 9.67 | 9.03 | nd | nd | 10 (<0.2−24) |
| ΣdiOH-hexaBDE[c,d] | 10.7 | 3.65 | nd | nd | nd |
| ΣOH-tetraBDD[c] | 0.21 | 0.45 | nd | nd | 0.2 (<0.2−3) |
| **total** | **50.0** | **119** | | | **10 (<0.2−24)** |
| | | | Ratio | | |
| 6-OH-BDE47/6-MeO-BDE47 | 0.41 | 3.47 | | | 0.03 |

[a] Ethylacetate extractable organic matter (%). [b] Arithmetic mean of hexane extractable lipid (%) ± standard deviation. [c] Calculated by selected ion abundance on the assumption of the same response of target analytes relative to 6-OH-[$^{13}$C]BDE47 (IS). [d] diOH-PBDEs include OH-MeO-PBDEs. Concentrations are medians, along with 10th−90th percentiles in parentheses. nd = not detected (less than LOQ = 0.2 ng/g lipid for 6-OH-[$^{13}$C]BDE47 and 0.5 ng/g lipid for 2′,6-diMeO-BDE68).

(Figure 2). The SIMs at $m/z$ 530, 624, and 594 were indicative of OH-tetraBDDs (peaks **1** and **2**), diOH-pentaBDE (peak **3**), and OH-pentaBDEs (peaks **4** and **5**), respectively. The EI mass spectra of peaks **2** and **3** are shown in Figure 3, and those of peaks **1**, **4**, and **5** are available in Figure S4 (Supporting Information). The spectrum of peak **2** exhibited $M^+$ ($m/z$ 526), $[M\text{-}CH_3]^+$ ($m/z$ 511), and $[M\text{-}CH_3CO]^+$ ($m/z$ 483), which undergoes further fragmentation of a $[M\text{-}CH_3COBr_2]^+$ ion, characteristic of MeO-tetraBDD. The EI mass spectrum of peak **3** exhibited $M^+$ ($m/z$ 620), $[M\text{-}CH_3\text{-}OCH_3]^+$ ($m/z$ 576), $[M\text{-}CH_3Br]^+$ ($m/z$ 526), and $[M\text{-}Br_2]^+$ ($m/z$ 462), characteristic of diMeO-PBDE.[16]

**Profiles from Coral Fishes.** Figure 4 shows the SIM profiles of tri- and tetraBDE products in the neutral fraction of unicornfish (*Naso lituratus*) from Palau Island, surgeonfish (*Acanthurus xanthopterus*) from Guam Island, and groupers (*Epinephelus* sp.) from Okinawa Island, Japan. Major brominated components in fish samples (muscles) were basically the same as brominated products from sponges (MeO-triBDEs, 2′-MeO-BDE68, 6-MeO-BDE47, 2′,6-diMeO-BDE68, and 2,2′-diMeO-BB80). Several fish samples (i.e., kawakawa, (*Euthynnus affinis*) from Palau and yellowfin tuna (*Thunnus albacores*) from Okinawa) also contained these methoxylated analogues with similar profiles to sponge brominated compounds (Figure S5, Supporting Information).

**Levels of Brominated Compounds in Sponge and Fish Samples.** The concentrations of major brominated compounds in sponges (EOM weight basis) and fish (lipid weight basis) are listed in Table 2. The total concentrations of MeO-tetraBDE analogues (sum of four compounds) were 61.0 μg/g EOM in *Haliclona* sp. and 32.4 μg/g EOM in *Callyspongia* sp., whereas the

corresponding OH-tetraBDE analogues were present at the levels of 27.2 and 80.1 μg/g EOM, respectively. The concentrations of MeO-PBDE analogues in nearby fish ranged from 65 to 670 ng/g lipids (median, 280 ng/g lipid), which accounted for 0.4% and 0.8% of ΣMeO-tetraBDE in *Haliclona* sp. and *Callyspongia* sp., respectively. OH-PBDE and diOH-pentaBDE in fish were present in concentrations of up to 40 ng/g lipid. The concentration ratios of ΣOH-PBDE/ΣMeO-PBDE were about 0.4 in *Haliclona* sp. and 3.5 in *Callyspongia* sp., whereas the ratios were smaller than 0.03 in all fish samples.

## DISCUSSION

**Sponge Brominated Products.** Previous chemical studies on the isolation of brominated products from marine sponges have been extensively performed on the genus *Dysidea*,[3,6,21,26] where OH-/MeO-PBDEs may represent as much as 12% of the dry weight.[20] This genus contains a large population of cyanophytes within its tissues,[3,27] and the OH-PBDE analogues are likely produced by the symbiotic filamentous cyanobacterium *Oscillatoria spongeliae*.[20] In the present study, we screened brominated analogues in sponge families other than *Dysidea* sp., of which two genera, *Haliclona* and *Callyspongia*, that are common marine sponges in Palau, had an abundance of OH- and MeO-PBDEs.

In the neutral fractions of both sponge extracts, we isolated four MeO-triBDE congeners (unidentified structures), two MeO-tetraBDEs (2′-MeO-BDE68 and 6-MeO-BDE47), and three dimethoxylated analogues (2,2′-diMeO-BB80, 2′,6-di-MeO-BDE68, and diMeO-pentaBDEs). Although the profiles and amounts of these products in two species seems to be different from those from *Dysidea* sp. in the Indo-Pacific,[6] 2′-MeO-BDE68 and 6-MeO-BDE47 were commonly found in the present study (*Haliclona* and *Callyspongia* sp.) as well as in *Dysidea* sp.[2,4] These products were also isolated from an aquatic sponge (*Ephydatia fluviatilis*) and marine red algae (*Ceramium tenuicorne*) from the Baltic sea[7,27] and marine algae (e.g., *Sargassum* sp.) from the Philippines,[8] indicating that these products are widespread in the marine food web.[28] For other products, 2,2′-diMeO-BB80 isolated in this study has not hitherto been reported in sponges or algae. This compound is likely derived from 2,2′-diOH-BB80 that has been isolated in the bacteria *Pseudoalteromonas phenolica* sp. from the Pacific.[29] Furthermore, MeO-triBDEs (four products) and diMeO-tetraBDE (as 2′,6-diMeO-BDE68) detected in both sponges are also produced by *Dysidea* sp.[27] Most of these products have been seen in whale blubber from Japanese coastal waters[15,16] and in Australia.[13,17]

In the phenolic fraction of marine sponges, 2′-OH-BDE68 and 6-OH-BDE47 were the predominant congeners in both species. Although diOH-penta- and diOH-hexaBDE were determined as methoxylated derivatives, these products included OH-MeO-analogues of tetra-, penta-, and hexaBDEs, some of which were confirmed by direct GC/MS measurement in EI mode (Figure S3, Supporting Information). One of the structures was tentatively identified as 2′-MeO-6-OH-BDE68 due to characteristic $[M\text{-}CH_3Br]^+$ for an ortho-substituted MeO group and *ortho*-bromine in the other ring.[25] Since it seems difficult to directly determine the diOH-PBDEs, the ratios of OH-MeO- and diOH-analogues present in the sponges remain unclear.

Interestingly, tetraBDE products were isolated as a mixture of hydroxy and methoxy analogues (about 1:2 for *Haliclona* sp. and 2:1 for *Callyspongia* sp.). However, the triBDE products were present only as methoxylated analogues, whereas the penta- and hexaBDE products were present primarily as dihydroxylated or hydroxy-methoxy PBDE analogues in both species. These findings suggest that bacterial O-methylation may occur in sponges, depending on the degree of bromination.[30] Alternatively, the biosynthesis process of diOH-PBDEs may be different from that of MeO-PBDEs.

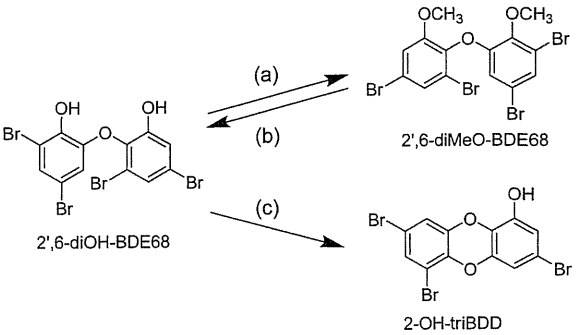


**Figure 5.** Proposed pathway for the biotransformation of dihydroxylated PBDEs. (a) bacterial *O*-methylation (ref 30); (b) microsomal demethylation (ref 37); and (c) bromoperoxidase-mediated condensation (ref 31) in marine biota.

In red algae and cyanobacteria from the Baltic Sea, Malmvarm et al.[18] demonstrated the presence of tri-, tetra-, and pentaBDDs. Although we could not confirm the occurrence of such PBDDs in sponges investigated, we studied the occurrence of at least two hydroxy-tetraBDDs (OH-tetraBDDs) in the phenolic fraction. The OH-tetraBDDs isolated in this study may be the same as products from an Australian marine sponge, *Dysidea dendyi*,[4,5] where diOH- or OH-MeO-pentaBDEs have been present as well. As proposed in Figure 5, we hypothesize that sponge-associated organisms produce OH-PBDDs by dehydrobromination of diOH-PBDEs (e.g., 2′,6-diOH-BDE68) present in *Callyspongia* sp. due to the loss of phenolic protons and bromine in the ortho position of the other ring. The levels of OH-tetraBDDs are estimated to be 0.2−0.45 μg/g EOM, which account for about 2−5% of diOH-PBDEs in the both sponges (Table 2). Bromoperoxidase may contribute to the internal cyclization from diOH-PBDEs[31] as it is suggested that chloroperoxidase can be involved in the condensation of chlorophenols to tetrachloro-dibenzo-*p*-dioxin.[32] Alternatively, OH-tetraBDDs could potentially be formed by photolysis of the diOH-PBDEs present in the sponges.[33]

**Brominated Compounds in Fish.** Because fish frequently feed on marine sponges in the coral reef environment, sponge-produced brominated compounds could have been transferred to fish such as unicorn fish included in this study living in the same regions of Palau. Our study shows that sponge-produced MeO-PBDEs are observed not only in Palauan fish, but also in the other fishes (surgeonfish and groupers) from the coastal waters of Guam (Micronesia) and Okinawa (Japan) (Figure 4S, Supporting Information). Similar profiles in fish indicate the wide distribution of MeO-PBDEs produced by marine sponges, likely explaining the higher concentrations observed in whales and dolphins in the Pacific and Oceania.[15,16] The present levels of MeO-tetraBDEs in fish from the western Pacific are comparable to the results from shark liver in Ishigaki Island, in the southern coast of Japan,[12] and blubber of killer whales from northern Japan.[15] In particular, the levels of MeO-tetraBDEs in groupers

from Okinawa seem to be 1 order of magnitude higher than those in bluefin tuna (up to 80 ng/g lipid) from Okinawa Island,[34] farmed tuna (up to 63 ng/g lipid) from the Mediterranean Sea,[11] Baltic salmon (up to 7 ng/g lipid), and Arctic cod liver (up to 17 ng/g lipid).[10] In addition, MeO-triBDEs, 2,2′-diMeO-BB80, and 2′,6-diMeO-BDE68 distributed in marine biota are also major products in the marine sponges investigated.[13,16] Considering that their levels in sponges on an EOM weight basis are about 2 orders of magnitude higher than those observed in fish on a lipid-weight basis (Table 2), it is evident that marine sponges are a potential source of MeO-triBDEs and diMeO-BB/BDEs, which could biomagnify via the food chain to higher trophic organisms.[35]

Total concentrations of 2′-OH-BDE68 and 6-OH-BDE47 are similar to those of the methoxylated analogues in both sponges. Nevertheless, these phenolic products were trace (less than 10 ng/g lipid) or undetectable in the local fish of Micronesian and Japanese coastal waters. A few fish samples (three of eleven groupers) from Okinawa Island contained diOH-pentaBDEs and OH-tetraBDDs, but the levels observed were low (up to 24 ng/g lipid) in this study. These results indicate that sponge-produced OH-PBDE analogues are less transferable to higher trophic animals or accumulate less in the fatty tissues of mammals. In fact, OH-tetraBDEs were present at 2% of the corresponding MeO-tetraBDE levels found in shark livers from Okinawa.[36] These phenolic PBDEs may be more concentrated in fish blood[9] and water-soluble body tissues.[24] In some cases, it is possible that the OH-PBDEs identified in the blood may originate from metabolites of anthropogenic PBDEs[28] via cytochrome P450-mediated hydroxylation in the liver. Such oxidations generally helps organisms make OH-PBDEs more water-soluble and thus excreted more easily as O-conjugates. Another biotransformation process is also possible via bacterial O-methylation of OH-PBDEs,[30] while it has been reported that microsomal demethylation of natural MeO-PBDEs to the corresponding OH-PBDEs may occur in the mammalian liver[37] (Figure 5). Further studies are necessary to elucidate the transformation and biomagnification processes of OH- and MeO-PBDEs in the environment.

## ■ ASSOCIATED CONTENT

**Supporting Information.** GC-EI-MS of peaks **a**−**d** in Figure 1, triBDD or methoxy-triBDD, hydroxy-methoxy analogues of tetraBDE, and pentaBDE; peaks 1, 4, and 5 in Figure 2; and SIM chromatograms of the neutral fractions from several fish materials in the western Pacific. This material is available free of charge via the Internet at http://pubs.acs.org.

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# Analysis of perfluoroalkyl carboxylates in vacuum cleaner dust samples in Japan

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Factor analysis

ABSTRACT

Perfluorooctanoic acid (PFOA) has long been an environmental contaminant of concern owing to its potential health risk. However, exposure to perfluorinated carboxylic acids (PFCAs) other than PFOA is not well understood. In this study, we investigated the concentrations of PFCAs in vacuum cleaner dust in Japan to measure the PFCAs contamination in an indoor environment. Most of the 77 samples contained PFCAs with 6–13 carbon atoms. The median concentration of perfluorononanoic acid (PFNA, 23.2 ng $g^{-1}$) was highest among PFCAs, followed by PFOA (20.8 ng $g^{-1}$) and perfluoroundecanoic acid (PFUnDA, 12.9 ng $g^{-1}$). The 90th percentile concentrations of PFNA, PFUnDA and perfluorotridecanoic acid (PFTrDA) were 948, 283 and 110 ng $g^{-1}$, respectively, and these were detected at greater concentrations than neighboring, even-numbered PFCAs. The proportion of long-chain PFCAs in vacuum cleaner dust from Japan was relatively higher than those reported for other countries. Factor analysis showed three independent factors. Odd-numbered long chain PFCAs (PFNA, PFUnDA and PFTrDA), which can correspond to factor 1, were major components of PFCAs in vacuum cleaner dust. Short chain PFCAs (factor 2) and even numbered long chain PFCAs (factor 3) were also statistically separated. These findings suggest that there are several sources of PFCAs with different origins in indoor environment. Further investigations into the origins of PFCAs are needed to evaluate indoor contamination with PFCAs.



## 1. Introduction

Perfluorinated alkyl acids such as perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA) have been detected in various media in the environment, including wildlife and humans (Houde et al., 2006). In 2002, a major manufacturer, 3M Company, phased out PFOS production (Renner, 2001). Since then, several studies have demonstrated that PFOS and PFOA induce developmental toxicities in animals and humans (Lau et al., 2007; Steenland et al., 2010). Although PFOA was a major component of perfluoroalkyl carboxylates (PFCAs) emission, long chain PFCAs (perfluorononanoic acid (PFNA), perfluoroundecanoic acid (PFUnDA) and perfluorotridecanoic acid (PFTrDA)) have also been detected in discernible concentrations in samples collected from wildlife (Prevedouros et al., 2006; Furdui et al., 2008).

In human biomonitoring studies, PFOS was the major perfluoroalkyl acids (PFAAs) found in human serum. In addition, PFOA was found to be the most prevalent component of serum PFCAs in western countries, followed by PFNA, perfluorodecanoic acid (PFDA) and PFUnDA (Kärrman et al., 2007; Haug et al., 2009; Kato et al., 2009b). Our previous study of Japanese, Korean and Vietnamese adults showed that PFNA and PFUnDA were found in serum at concentrations generally similar to PFOA, and that these levels have continued to increase, even after 2002 (Harada et al., 2011). Factor analysis of PFCAs also revealed two major factors composed of PFNA, PFUnDA and PFTrDA (factor 1) and PFOA and perfluoroheptanoic acid (PFHpA) (factor 2) (Harada et al., 2011). However, the origin of these factors is still not known.

Long chain PFCAs have not been detected in food duplicate samples in Japan at concentrations greater than method detection limits (MDL: 0.1 ng $g^{-1}$ for PFNA, 0.5 ng $g^{-1}$ for PFDA and PFUnDA, respectively) (Kärrman et al., 2009). In addition, the predominance of odd numbered PFCAs has not been reported in aquatic systems in Japan (Murakami et al., 2008, 2009; Zushi et al., 2008; Raj Shivakoti et al., 2011). Fluorotelomer alcohols (FTOHs) are considered to be precursors of long chain PFCAs and have been detected in Japan and Western countries (Martin et al., 2002; Jahnke et al., 2007; Oono et al., 2008a,b; Mahmoud et al., 2009). Although atmospheric degradation of 8:2 FTOH has shown comparable yields of PFOA and PFNA (Ellis et al., 2004), the dominance of odd number PFCAs

*Abbreviations:* PFCAs, perfluorinated carboxylic acids; PFOS, perfluorooctane sulfonate; PFOA, perfluorooctanoic acid; PFHxA, perfluorohexanoic acid; PFHpA, perfluoroheptanoic acid; PFNA, perfluorononanoic acid; PFDA, perfluorodecanoic acid; PFUnDA, perfluoroundecanoic acid; PFDoDA, perfluorododecanoic acid; PFTrDA, perfluorotridecanoic acid; PFTeDA, perfluorotetradecanoic acid; PFAAs, perfluoroalkyl acids; IDLs, instrumental detection limits; LOD, limit of detection; MDLs, method detection limits; RSD, relative standard deviation; SD, standard deviation; GM, geometric mean; GSD, geometric standard deviation.

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in human serum has been observed only in East Asian countries (Harada et al., 2011). A review by Prevedouros et al. (2006) indicated that odd numbered PFCAs have been applied to fluoropolymer manufacturing aids and surfactants that were manufactured *via* oxidation of fluorotelomer olefins. Their application to commercial products might be an exposure source of long chain PFCAs in human serum. Given the indoor use of those products, a portion of them likely disperses and contaminates indoor dusts.

The primary goal of the present study was to investigate PFCAs in house dust. To achieve this goal, we investigated vacuum cleaner dust concentrations of PFCAs in Japan. To evaluate the potential factors influencing the PFCAs level, questionnaires regarding housing conditions and articles were collected. In addition, we conducted factor analysis to elucidate the potential compositions of PFCAs in indoor dust.

## 2. Materials and methods

### 2.1. Sample collection

To evaluate geographical differences in PFCAs in vacuum cleaner dust in Japan, we recruited 77 homes from Osaka, Kyoto, Wakayama and Toyama, Japan and conducted sampling from October to December, 2010. Osaka was selected to evaluate the effects of a local industrial source of PFOA (Saito et al., 2004; Morikawa et al., 2006; Niisoe et al., 2010). Vacuum cleaner dust samples were collected from the used bag of the household vacuum cleaner into a sealable polyethylene bag. Samples were shipped via an overnight delivery service to Kyoto University.

A 500 mg cleaner dust sample was taken out from the sealable polyethylene bag and hairs and plastic garbage were removed from the samples using forceps and a loupe. Samples were then sieved to remove materials greater than 150 μm in diameter and stored at −4 °C until analysis in the Kyoto University Human Specimen Bank (Koizumi et al., 2005, 2009). Information regarding the home condition, articles use and life habits was then collected by questionnaire (Table 1 and Supplemental Table 1). The research protocol for the present study was reviewed and approved by the Ethics Committee of the Kyoto University Graduate School of Medicine on 13 October 2010 (E960). Written informed consent was obtained from all participants.

### 2.2. Reagents

Methanol (LC–MS grade) and water (distilled LC–MS grade) were obtained from Kanto Chemicals (Tokyo, Japan). Ammonium hydroxide (25% in water) was purchased from Merck (Darmstadt, Germany). Benzyl bromide was purchased from Wako pure chemicals (Osaka, Japan). A mixture of $^{13}C_2$-labeled perfluorohexanoic acid (PFHxA), $^{13}C_2$-labeled PFOA, $^{13}C_4$-labeled PFOA, $^{13}C_5$-labeled PFNA, $^{13}C_2$-labeled PFDA, $^{13}C_2$-labeled PFUnDA and $^{13}C_2$-labeled perfluorododecanoic acid (PFDoDA) was obtained from Wellington Laboratories (Guelph, Ontario, Canada) and Perkin Elmer (Boston, MA).

### 2.3. Determination of PFCAs in vacuum cleaner dust

PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA and perfluorotetradecanoic acid (PFTeDA) were analyzed by gas chromatography/mass spectrometry (Agilent 6890GC/5973MSD, Agilent Technologies Japan, Ltd., Tokyo, Japan). Perfluorooctane sulfonamides were not included because main purpose in this study was to elucidate a pattern of PFCAs exposure in indoor environment. Vacuum cleaner dust samples were subjected to a clean-up procedure consisting of a weak anion exchange, solid-phase extraction. Briefly, approximately 500 mg of vacuum cleaner dust sample and internal standards (10 ng mixture of $^{13}C_2$-labeled PFHxA, $^{13}C_4$-labeled PFOA, $^{13}C_5$-labeled PFNA, $^{13}C_2$-labeled PFDA, $^{13}C_2$-labeled PFUnDA, and $^{13}C_2$-labeled PFDoDA) were put into a 50 mL polypropylene (PP) centrifugation tube and 20 mL of methanol were added. The samples were then vortexed and shaken on a vertical shaker for 30 min, after which they were placed in an ultrasonic bath for 15 min. Next, the samples were centrifuged (11 000*g*, 15 min), and the supernatant was then reduced to approximately 5 mL by rotary evaporation. Approximately 15 mL of water were subsequently added to the sample, and the solution was put through a WAX solid phase cartridge (6 cc, 150 mg 30 mm, Waters, Milford, MA, USA) that had been previously conditioned with 4 mL methanol and 4 mL water. Subsequent loading of the sample was followed by washing the sorbent with 4 mL 40% (v/v) methanol in water and a second wash using 8 mL methanol. The perfluorinated compounds were then eluted into a tube using 2 mL 2% (v/v) ammonium hydroxide in methanol. The solution was then dried under $N_2$, after which 0.25 mL of 100 mM benzyl bromide acetone and the performance standard $^{13}C_2$-PFOA were added. Next, the solution was transferred to an autosampler vial and heated for 1 h at 80 °C. Extracts were subsequently analyzed by GC/MS in electron impact ionization mode using single ion monitoring. PFCA benzyl esters were separated on a DB-5MS column (30 m length, 0.25 mm i.d., 1 μm film thickness) with a helium carrier gas. Splitless injections (0.5 μL) were performed with the injector set to 220 °C, and the split was opened after 1.5 min. The initial oven temperature was 70 °C for 2 min, after which it was increased to 100 °C at 20 °C min$^{-1}$, and then to 280 °C at 30 °C min$^{-1}$. Ion fragments ($[M]^+$) were monitored and used for quantification (Supplemental Table 2).

Instrumental detection limits (IDL) were defined as the mass of the analyte producing a peak with a signal-to-noise ratio of 3, and ranged from 2 pg (PFTeDA) to 0.5 pg (other PFCAs) (Supplemental Table 2). All samples were quantified using a seven-point calibration curve (range: 0.5–50 000 ng mL$^{-1}$ in methanol). Limit of detection (LOD) was defined as the lowest concentration with a relative standard deviation (RSD) of the relative response factors <15% ($n = 3$ for each concentration). The method detection limit (MDL) was defined as the blank response +3 standard deviations in procedural blank samples. The procedural blank levels using 0.5 mL distilled water were evaluated in every 11 samples. Since blank samples (0.5 mL distilled water) contained no detectable concentrations, MDL was considered to be equal to the LOD

**Table 1**
Study area and population.

| Sampling site | Population density (×10$^3$ km$^{-2}$) | *n* | Building age[a] (yr) | House type | | Building Construction | |
|---|---|---|---|---|---|---|---|
| | | | | Houses | Apartments | Timber | Concrete |
| Osaka | 11.7 | 21 | 31.0 ± 27.3 | 10 | 11 | 7 | 14 |
| Kyoto | 1.7 | 20 | 30.3 ± 21.2 | 13 | 7 | 11 | 9 |
| Wakayama | 0.50 | 16 | 18.1 ± 14.2 | 14 | 2 | 13 | 3 |
| Toyama | 0.86 | 20 | 20.8 ± 9.7 | 19 | 1 | 15 | 5 |

[a] Data are presented as the mean ± standard deviation.

corresponding to 2 ng $g^{-1}$ for PFTeDA and 0.5 ng $g^{-1}$ for the other PFCAs (Supplemental Table 2).

### 2.4. Quality assurance

Quantification was conducted using an internal standard dissolved in acetone. $^{13}C_2$-labeled PFHxA, $^{13}C_4$-labeled PFOA, $^{13}C_5$-labeled PFNA, $^{13}C_2$-labeled PFDA, $^{13}C_2$-labeled PFUnDA and $^{13}C_2$-labeled PFDoDA were used as the internal standard for PFCAs. PFHpA was quantified by $^{13}C_2$-PFHxA, PFTrDA, PFTeDA and $^{13}C_2$-PFDoDA. $^{13}C_2$-PFOA (10 ng) was added prior to derivatization for determination of amount of recovered $^{13}C$-labeled PFHxA, PFOA, PFNA, PFDA, PFUnDA and PFDoDA. The results were corrected for recoveries. The recoveries were evaluated by five replicate fortifications (fortified by 10 times the original concentration of vacuum cleaner dust) of a vacuum cleaner dust sample with low contamination (Supplemental Table 2). To validate variations throughout analysis, duplicates of 5 ng-fortified cleaner dust samples were analyzed in a batch in each sampling sites ($n = 4$). They were compared and indicated good precision with RSDs $\leqslant$ 10.6% (Supplemental Table 2).

### 2.5. Statistical analysis

All statistical analyses were conducted using the JMP software (Version 4; SAS Institute Inc., Cary, NC). Values of $p < 0.05$ were considered to indicate statistical significance. Concentrations lower than the detection limit were considered to be equal to half of the detection limit for statistical analyses. Vacuum cleaner dust levels of PFCAs were assumed to be distributed lognormally because they displayed right-skewed patterns and had geometric means comparable to the medians. Statistical analyses were conducted after logarithmic transformation of the vacuum cleaner dust concentrations. Differences between mean values were tested by Tukey–Kramer's honestly significant difference (HSD) test when statistical tests by ANOVA were significant. Correlations were tested by Spearman's rank correlation coefficient ($\rho$). Factor analysis was used to transform a number of contaminants into a smaller number of potential factors of sources. Factor analysis was conducted *via* a correlation matrix. Eigenvectors were employed through analysis, and the eigenvalues which account for the more than 80% of variations were taken into account. Normalized varimax rotation (an orthogonal rotation of the factor axes) was applied to these eigenvectors to simplify them into a few variables with high correlations.

## 3. Results

### 3.1. PFCAs concentrations in vacuum cleaner dust

The descriptive statistics for PFCAs are presented in Table 2. Most samples contained PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA and PFTrDA. PFTeDA was less frequently observed at concentrations above the MDL. The median concentration of PFNA (23.2 ng $g^{-1}$) was highest among PFCAs, followed by that of PFOA (20.8 ng $g^{-1}$), PFUnDA (12.9 ng $g^{-1}$), and PFHxA (9.2 ng $g^{-1}$). Odd numbered long chain PFCAs showed large variations in concentrations, with geometric standard deviations ranging from 6.1 to 7.2. The 90th percentile concentrations of PFNA, PFUnDA and PFTrDA were 948, 283 and 110 ng $g^{-1}$, respectively, and these were detected at higher concentrations than neighboring, even-numbered PFCAs (PFDA, PFDoDA and PFTeDA).

The geometric mean (GM) of PFOA of dust samples was significantly higher in Osaka than those in Kyoto, Wakayama and Toyama ($p < 0.05$). The GM of PFHpA was also higher in samples from Osaka and Kyoto than those from Toyama ($p < 0.05$). Concentrations of PFHxA in dust samples were higher in Osaka than in Kyoto and Wakayama ($p < 0.05$). In contrast, there was no significant difference in the GMs of PFNA, PFDA, PFUnDA and PFTrDA among the four sampling sites ($p > 0.05$). Samples from Wakayama showed a higher GM of PFDoDA ($p < 0.05$). The mean proportions of PFOA in the total PFCAs were 33.0% ± 22.5%, 21.7% ± 14.7%, 12.3% ± 8.4% and 18.2% ± 12.2% in Osaka, Kyoto, Wakayama and Toyama, respectively.

### 3.2. Correlations among PFCAs levels and factor analysis

The nonparametric correlation coefficients among the PFCAs in the 77 samples are listed in Table 3. PFHpA was more highly correlated with PFOA and PFDA among the PFCAs ($\rho = 0.822$ and 0.544, respectively). PFOA was also significantly correlated with PFNA, PFDA and PFUnDA, although the highest $\rho$ coefficient was 0.429 with PFDA. Among the long chain PFCAs, PFUnDA was strongly associated with PFNA and PFTrDA ($\rho = 0.913$ and 0.888, respectively). PFDoDA also showed high correlation coefficients with PFDA, PFTrDA and PFTeDA ($\rho = 0.719$, 0.554 and 0.535, respectively).

Factor analysis was applied to delineate the relationships among PFCA concentrations. The contributions of factors 1, 2 and 3 to the total variance were 47.7%, 19.7% and 14.5% (with an eigenvalue >1), respectively (Table 4). After varimax rotation, the first factor had greater correlations with odd numbered longer-chain PFCAs (PFNA, PFUnDA, PFTrDA) than short chain and even-numbered long chain PFCAs. Factor 1 represents odd numbered longer chain PFCAs. The second factor had a greater correlation with short-chain PFCAs (PFHxA, PFHpA and PFOA) than with other PFCAs. The second factor may represent PFOA and other short chain PFCAs. The third was negatively correlated with even numbered long-chain PFCAs (PFDA, PFDoDA and PFTeDA). The third factor represents even numbered long chain PFCAs.

Factor scores were compared among the four sampling sites (Table 4). There was no significant difference in factor 1 for samples from each site ($p > 0.05$), suggesting a common emission source of odd numbered longer chain PFCAs in Japan. In contrast, samples collected from Osaka had the highest factor 2 score ($p < 0.05$), suggesting a local emission source in Osaka. Factor 3 had the lowest score in Wakayama ($p < 0.05$), suggesting a possible local emission source specific to Wakayama.

### 3.3. Association between house conditions and PFCAs factors

To evaluate the influence of the participant's characteristics on PFCAs concentrations in vacuum cleaner dust samples, Pearson's correlation or ANOVA were conducted for the three potential factors (Supplemental Table 1). Only the number of household members was positively correlated with factor 1 (odd-numbered longer chain PFCAs, $p < 0.05$). Samples from apartments and concrete buildings showed higher factor 2 scores (PFOA and other short chain PFCAs, $p < 0.05$). The water repellent use group also showed higher factor 2 scores than the non-use group ($p < 0.05$). The score of factor 3 (even numbered long chain PFCAs) was higher in dust samples from air cleaning device users and lower in those from PTFE cookware users ($p < 0.05$). Frequent cleaning of the house also resulted in a higher factor 3 score ($p < 0.05$). To investigate potential confounding between sampling sites and house condition, analysis of covariance was conducted. After adjusted with sampling sites, house types and building construction did not indicate significant differences on factor 2 (Supplemental Table 1). Effect of frequency of cleaning on factor 3 was also decreased and statistical significance was disappeared. The number of household members,

**Table 2**
Concentrations of PFCAs in vacuum cleaner dust samples

| Sampling site | *n* | Concentration (ng $g^{-1}$) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | PFHxA | PFHpA | PFOA | PFNA | PFDA | PFUnDA | PFDoDA | PFTrDA | PFTeDA |
| Total | 77 | *n* > MDL (%) | 66(85.7) | 76(98.7) | 77(100.0) | 77(100.0) | 77(100.0) | 77(100.0) | 73(94.8) | 74(96.1) | 10(13.0) |
| | | median | 9.2(0.5–7140) | 3.7(0.5–81.2) | 20.8(3.2–340) | 23.2(2.0–37400) | 7.3(1.4–201) | 12.9(2.1–19900) | 4.0(2.1–167) | 5.5(<1–7010) | <2(<2–81.2) |
| | | mean | 1.21 ± 817 | 6.2 ± 10.1 | 42.3 ± 53.4 | 783 ± 4313 | 17.1 ± 29.2 | 394 ± 2230 | 8.4 ± 20.2 | 155 ± 833 | 3.1 ± 9.8 |
| | | GM(GSD) | 7.0(7.0) | 3.7(2.6) | 24.4(2.8) | 41.6(7.2) | 8.8(2.8) | 23.3(6.1) | 4.1(2.8) | 9.2(6.3) | 1.3(2.3) |
| | | P90 | 55 | 11 | 112 | 948 | 43 | 283 | 15 | 110 | 4 |
| Osaka | 21 | *n* > MDL (%) | 19(90.5) | 21(100) | 21(100) | 21(100) | 21(100) | 21(100) | 19(90.5) | 19(90.5) | 4(19.0) |
| | | median | 27.5(<0.5–7140) | 5.5(1.1–81.2) | 78.2(9.7–340) | 28.8(3.3–37300) | 6.5(2.2–93.3) | 15.7(4.1–19900) | 2.7(<1–59.2) | 8.0(<1–701) | <2(<2–23.6) |
| | | mean | 415 ± 1550 | 10.4 ± 17.0 | 94.0 ± 73.0 | 2070 ± 8160 | 17.5 ± 26.5 | 1040 ± 4330 | 7.2 ± 12.7 | 368 ± 1530 | 2.6 ± 5.0 |
| | | GM(GSD) | 23.6(10.8)$^{A}$ | 6.0(26)$^{A}$ | 72.0(2.2)$^{A}$ | 54.4(8.6) | 8.7(30) | 27.6(7.6) | 3.6(3.1)$^{AB}$ | 10.3(8.7) | <2 |
| | | P90 | 720 | 21 | 184 | 4099 | 80 | 1142 | 17 | 388 | 6 |
| Kyoto | 20 | *n* > MDL (%) | 14(70.0) | 20(100) | 20(100) | 20(100) | 20(100) | 20(100) | 18(90.0) | 19(95.0) | 2(10.0) |
| | | median | 8.2(<0.5–27.7) | 3.9(1.4–23.3) | 21.6(5.3–133) | 22.7(5.2–2830) | 5.4(1.5–86.1) | 8.9(2.1–574) | 2.1 (<1–35.2) | 28(<1-125) | <2(<2–14.0) |
| | | mean | 8.2 ± 7.5 | 5.7 ± 4.8 | 28.7 ± 28.4 | 338 ± 740 | 11.5 ± 18.4 | 82.6 ± 164 | 4.9 ± 8.3 | 19.4 ± 35.9 | <2 |
| | | GM(GSD) | 3.3(6.1)$^{BC}$ | 4.6(1.9)$^{A}$ | 21.8(2.0)$^{B}$ | 49.9(7.0) | 6.9(25) | 17.5(5.8) | 2.5(2.8)$^{B}$ | 5.1(5.0) | <2 |
| | | P90 | 19 | 9 | 70 | 1804 | 22 | 470 | 19 | 101 | 3 |
| Wakayama | 16 | *n* > MDL (%) | 13(81.3) | 16(100) | 16(100) | 16(100) | 16(100) | 16(100) | 16(100) | 16(100) | 3(18.8) |
| | | median | 3.0(<0.5–10.5) | 2.8(1.1–29.6) | 13.2(3.2–63.2) | 29.0(2.0-4580) | 11.3(1.4-201) | 21.6(3.7-3500) | 6.0(2.1-167) | 7.8(2.9-2130) | <2(<2-81.2) |
| | | mean | 3.3 ± 2.9 | 4.8 ± 6.8 | 19.8 ± 19.0 | 426 ± 1160 | 30.1 ± 49.3 | 320 ± 879 | 17.6 ± 40.1 | 179 ± 531 | 6.9 ± 20.0 |
| | | GM(GSD) | 2.0(3.4)$^{C}$ | 3.3(2.1)$^{AB}$ | 13.7(2.4)$^{B}$ | 43.7(8.1) | 14.2(3.3) | 38.6(6.7) | 7.9(2.7)$^{A}$ | 18.3(6.8) | <2 |
| | | P90 | 9 | 14 | 63 | 2343 | 107 | 1715 | 64 | 935 | 33 |
| Toyama | 20 | *n* > MDL (%) | 20(100.0) | 19(95.0) | 20(100) | 20(100) | 20(100) | 20(100) | 20(100) | 20(100) | 1(5.0) |
| | | median | 12.4(1.8–54.3) | 2.1(<0.5–14.8) | 12.1(4.1–114) | 15.3(2.7–1580) | 6.5(2.4–64.2) | 11.0(3.6–1040) | 4.1(1.5–26.1) | 5.1(1.7–603) | <2(<2–18.8) |
| | | mean | 17.4 ± 15.2 | 3.3 ± 3.4 | 19.6 ± 24.1 | 161 ± 380 | 11.8 ± 14.4 | 88.5 ± 235 | 6.0 ± 5.9 | 45.6 ± 134 | <2 |
| | | GM(GSD) | 11.3(2.8)$^{AB}$ | 2.1(3.0)$^{B}$ | 13.8(2.2)$^{B}$ | 25.3(5.9) | 7.7(23) | 17.2(4.8) | 4.4(2.1)$^{AB}$ | 8.6(4.6) | <2 |
| | | P90 | 43 | 8 | 41 | 658 | 28 | 216 | 14 | 104 | <2 |

MDL: method detection limit; GM: geometric mean; GSD: geometric standard deviation; P90 90th percentile value
The geometric means within the same columns without bearing the same superscripts differ significantly ($p < 0.05$).
The geometric means within the same columns bearing the same superscripts or without superscripts do not differ significantly ($p > 0.05$).

**Table 3**
Correlation between PFCAs with different chain lengths.

| Variables | PFHxA | PFHpA | PFOA | PFNA | PFDA | PFUnDA | PFDoDA | PFTrDA | PFTeDA |
|---|---|---|---|---|---|---|---|---|---|
| PFHxA | – | | | | | | | | |
| PFHpA | 0.070 | – | | | | | | | |
| PFOA | 0.296** | 0.822*** | – | | | | | | |
| PFNA | −0.018 | 0.470*** | 0.382*** | – | | | | | |
| PFDA | −0.099 | 0.544*** | 0.429*** | 0.421*** | – | | | | |
| PFUnDA | −0.049 | 0.347** | 0.288* | 0.913*** | 0.507*** | – | | | |
| PFDoDA | −0.156 | 0.209 | 0.185 | 0.181 | 0.719*** | 0.397*** | – | | |
| PFTrDA | −0.065 | 0.169 | 0.165 | 0.712*** | 0.411*** | 0.888*** | 0.554*** | – | |
| PFTeDA | −0.151 | 0.344** | 0.352** | 0.271* | 0.489*** | 0.312** | 0.535*** | 0.331** | – |

Numbers indicate Spearman's rank correlation coefficients ($\rho$).
* $p < 0.05$
** $p < 0.01$
*** $p < 0.001$

**Table 4**
Factor analysis among PFCAs

| | Initial solution | | | Varimax rotated | | |
|---|---|---|---|---|---|---|
| | F1 | F2 | F3 | F1 | F2 | F3 |
| Eigenvalue | 4.29 | 1.77 | 1.31 | | | |
| Contribution (%) | 47.7 | 19.7 | 14.5 | | | |
| *Eigenvector* | | | | | | |
| PFHxA | −0.074 | 0.463 | 0.244 | −0.183 | **0.579** | 0.336 |
| PFHpA | 0.269 | 0.490 | 0.092 | 0.203 | **0.804** | −0.243 |
| PFOA | 0.255 | 0.559 | 0.088 | 0.145 | **0.878** | −0.224 |
| PFNA | 0.391 | −0.142 | 0.459 | **0.956** | 0.203 | −0.105 |
| PFDA | 0.377 | 0.211 | −0.305 | 0.239 | 0.437 | **−0.750** |
| PFUnDA | 0.419 | −0.213 | 0.340 | **0.955** | 0.104 | −0.252 |
| PFDoDA | 0.339 | −0.125 | −0.504 | 0.222 | −0.051 | **−0.894** |
| PFTrDA | 0.397 | −0.327 | 0.219 | **0.899** | −0.080 | −0.336 |
| PFTeDA | 0.339 | −0.008 | −0.450 | 0.199 | 0.106 | **−0.841** |
| | | | | Factor score | | |
| | | | | (Mean ± standard deviation) | | |
| | | Osaka | | 0.06 ± 1.02 | $0.89 \pm 1.09^{A}$ | $0.18 \pm 0.91^{AB}$ |
| | | Kyoto | | −0.03 ± 1.05 | $-0.02 \pm 0.65^{B}$ | $0.26 \pm 0.94^{A}$ |
| | | Wakayama | | 0.14 ± 1.08 | $-0.60 \pm 0.64^{B}$ | $-0.64 \pm 1.24^{B}$ |
| | | Toyama | | −0.15 ± 0.91 | $-0.44 \pm 0.79^{B}$ | $0.06 \pm 0.75^{AB}$ |

The factors in bold indicate the greater correlations.The factor scores within the same columns without bearing the same superscripts differ significantly ($p < 0.05$). The facotr scores bearing the same superscripts or without superscripts do not differ significantly ($p > 0.05$). F1: 1st factor; F2: 2nd factor: F3: 3rd factor

air cleaning device use, water repellent use and PTFE cookware use remained significant factors after adjusted by sampling sites.

## 4. Discussion

In the present study, we extracted three major fingerprints (factors 1–3) from mixtures of different PFCAs in vacuum cleaner dust samples collected in Japan. Odd-numbered long chain PFCAs (PFNA, PFUnDA and PFTrDA), which corresponded to factor 1, were likely associated with chemical production processes, were the major components of PFCAs in vacuum cleaner dust. Factor 2, which corresponds to short chain PFCAs, was likely associated with a local emission source of PFOA in Osaka, while even numbered long chain PFCAs (factor 3) were likely associated with another local emission source in Wakayama, were statistically identified. Although these findings suggest that there are several sources of PFCAs with different origins, they also suggest that odd numbered long chain PFCAs are ubiquitous in indoor environments in Japan.

There are three major chemical engineering processes used to produce PFCAs (Supplemental Fig. 1) (Prevedouros et al., 2006). In the first process, fluorotelomer olefine is used as a starting compound and is then oxidized to produce PFCAs. In this process, odd-numbered PFCAs will be yielded predominately (Daikin Industries, 1998). In contrast, the fluorotelomer iodide oxidation process will yield even numbered PFCAs predominately (Tosoh Corporation, 1990). The electrochemical fluorination process was primarily employed by the 3M Company and yielded PFOA (3M Company Technical Bulletin, 1995).

Although PFOA is considered to be a major component among PFCAs, PFCAs of longer chain lengths than PFOA were frequently detected in vacuum cleaner dust samples in this study. We have also reported that the concentrations of odd numbered PFCAs in human serum have been increasing in Japan, Korea and Vietnam (Harada et al., 2011). The proportion of odd-numbered long-chain PFCAs in serum was estimated to be one order of magnitude higher when compared with reports in western countries (Harada et al., 2011). Importantly, because the serum and dust samples have similar odd-numbered long chain PFCAs profile, long chain PFCAs have not been detected in food duplicate samples in Japan (Kärrman et al., 2009) and the modern individual spends much time indoor, these suggest that dust exposure in indoor may be an important source for the human body burden of PFCAs in Asian. Additionally, the fluorotelomer olefine oxidation process was used primarily to produce the odd numbered PFCAs in Japan (Daikin Industries, 1998; Prevedouros et al., 2006). Collectively, the results of the present study and previous studies indicate the predominance of

**Table 5**
Comparison of concentrations of PFCAs in dust with reported data

| Sampling site | Year | n | | Concentration (ng $g^{-1}$) | | | | | | | | | References |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | PFHxA | PFHpA | PFOA | PFNA | PFDA | PFUnDA | PFDoDA | PFTrDA | PFTeDA | |
| *Japan* | | | | | | | | | | | | | |
| Osaka | 2010 | 21 | median | 27.5 | 5.5 | 78.2 | 28.8 | 6.5 | 15.7 | 2.7 | 8.0 | <2 | This study |
| Kyoto | 2010 | 20 | median | 8.2 | 3.9 | 21.6 | 22.7 | 5.4 | 8.9 | 2.1 | 28 | <2 | This study |
| Wakayama | 2010 | 16 | median | 3.0 | 2.8 | 13.2 | 29.0 | 11.3 | 21.6 | 6.0 | 78 | <2 | This study |
| Toyama | 2010 | 20 | median | 12.4 | 2.1 | 12.1 | 15.3 | 6.5 | 11.0 | 4.1 | 5.1 | <2 | This study |
| Osaka | 2003 | 16 | median | – | – | 165 | – | – | – | – | – | – | Moriwaki et al. (2003) |
| Tokyo | 2006 | 20 | median | – | – | 33.5 | 13.5 | – | – | – | – | – | Katsumata et al. (2006) |
| *China* | | | | | | | | | | | | | |
| Nanchang | 2009 | 5 | mean | | <0.17 | 1.4 | 0.66 | <0.18 | <0.17 | <0.18 | – | – | Zhang et al. (2010) |
| Shanghai | 2009 | 5 | mean | | 43.1 | 718 | 2.95 | 2.72 | 1.12 | <0.18 | – | – | Zhang et al. (2010) |
| Beijing | 2009 | 14 | mean | | 5.09 | 25.2 | 3 | 3.52 | 0.7 | 0.68 | – | – | Zhang et al. (2010) |
| Tianjin | 2009 | 4 | mean | | 194 | 355 | 7.32 | 7.85 | 1.62 | 3.28 | – | – | Zhang et al. (2010) |
| *Kazahkstan* | | | | | | | | | | | | | |
| Almaty and Astana | 2007–2009 | 9 | median | – | – | <0.98 | – | – | – | – | – | – | Goosey and Harrad, 2011 |
| *Thailand* | | | | | | | | | | | | | |
| Bangkok and Nakhonsrithammarat | 2007–2009 | 20 | median | | | 18 | | | | | | | Goosey and Harrad (2011) |
| *Australia* | | | | | | | | | | | | | |
| Brisbane, Newcastle and Sydney | 2007–2009 | 20 | median | | – | 120 | | | | | | | Goosey and Harrad (2011) |
| *USA* | | | | | | | | | | | | | |
| Ohio and North Carolina | 2000–2001 | 112 | median | 54.2 | 50.2 | 142 | 7.99 | 6.65 | 7.57 | 7.78 | – | – | Strynar and Lindstrom, 2008 |
| Colorado | 2007–2009 | 10 | median | | | 240 | | | | | | | Goosey and Harrad (2011) |
| *Canada* | | | | | | | | | | | | | |
| Ottawa | 2002–2003 | 67 | median | | | 19.7 | | | | | | | Kubwabo et al. (2005) |
| Toronto | 2007–200S | 20 | median | – | – | 69 | – | – | – | – | – | – | Goosey and Harrad (2011) |
| Vancouver | 2007–2008 | 132 | mean | | 168 | 97 | 26 | 8.4 | 7.8 | 6.3 | | 7.3 | Shoeib et al. (2011) |
| *UK* | | | | | | | | | | | | | |
| Birmingham | 2007–2009 | 45 | median | | | 190 | | | | | | | Goosey and Harrad (2011) |
| *Norway* | | | | | | | | | | | | | |
| Oslo | 2008 | 41 | median | 28 | 94 | 18 | 23 | 4.1 | | 19 | 6.8 | 3.3 | Haug et al. (2011) |
| *Sweden* | | | | | | | | | | | | | |
| Stockholm | 2006–2007 | 38 | median | | | 93 | | | | | | | Björklund et al. (2009) |
| *Belgium* | | | | | | | | | | | | | |
| Flanders | 2008 | 45 | median | 0.3 | | 0.7 | 0.1 | 02 | | | | | D'Hollander et al. (2010) |
| *Germany* | | | | | | | | | | | | | |
| Augsberg and Michelstadt | 2007–2009 | 10 | median | | | 300 | | | | | | | Goosey and Harrad (2011) |
| *France* | | | | | | | | | | | | | |
| Annecy | 2007–2009 | 10 | median | – | – | 31 | – | – | – | – | – | – | Goosey and Harrad (2011) |
| Germany, UK, Australia and USA | 2004 | 39 | median | <2.6 | 97.3 | 96.5 | <2.6 | <2.6 | <2.6 | <2.6 | - | - | Kato et al. (2009a) |

odd-numbered long chain PFCAs produced by the fluorotelomer olefine oxidation process in East Asia (Harada et al., 2011). However, further studies are needed to confirm such a trend in Asian ecosystems.

Using a questionnaire, we attempted to find an association between house conditions and PFCAs levels in vacuum cleaner dust. Several variables were correlated with factor scores of 1, 2 and 3. The number of household members was correlated with factor 1. Samples collected from apartments, concrete buildings and the water repellent use group were associated with factor 2. Samples from air cleaning device users, PTFE cookware non-users and frequent cleaning of the house group were associated with factor 3. It is possible that the application of PFCAs for domestic articles might not be uniform. The PFCAs contents of specific products should be evaluated in a future study.

Among the long-chain PFCAs, odd-numbered PFCAs accounted for the majority of those found in Japan. Vacuum cleaner dust or house dust levels of PFCAs reported from other countries are summarized in Table 5 (Moriwaki et al., 2003; Kubwabo et al., 2005; Katsumata et al., 2006; Strynar and Lindstrom, 2008; Björklund et al., 2009; Kato et al., 2009a; D'Hollander et al., 2010; Zhang et al., 2010; Goosey and Harrad, 2011; Haug et al., 2011; Shoeib et al., 2011). The PFOA levels in dust samples were relatively low when compared with those in the United States, the United Kingdom, Germany, Australia and several cities in China. A previous study in Osaka showed a higher median concentration of PFOA in cleaner dust than the current study (Moriwaki et al., 2003). However, the major PTFE manufacturer in Osaka reduced emissions of PFOA by 82% in 2006 and 89% in 2007 when compared with emissions in 2000 (Osaka Prefecture, 2007), which might have led to a decline in PFOA levels in dust samples. Although long-chain PFCAs were detected in the United States, Canada and Norway (Strynar and Lindstrom, 2008; Haug et al., 2011; Shoeib et al., 2011), no predominance of odd numbered PFCAs was observed. Additionally, a study conducted in China did not show a similar composition to Japanese samples (Zhang et al., 2010). Therefore, the PFCAs composition of Japan can be considered as a clear fingerprint originating from factor 1. Since we did not define the amount of dust and time that the dust was present in the used bag of the vacuum cleaner, we could not eliminate the possibility that the amount of dust and time could influence the results.

The characteristics of factor 1, namely the predominance of odd numbered long chain PFCAs, is consistent with previous biomonitoring of human serum from East Asian countries (Harada et al., 2011), which implies that indoor PFCAs might be an important route of exposure. In this study, only vacuum cleaner dust was analyzed, therefore, detailed exposure assessment is not possible. In addition, exposure assessment of long chain PFCAs are insufficient in various routes of exposures such as food, drinking water and inhalation. The toxicokinetics of long-chain PFCAs are also unclear, especially in humans. Therefore, more comprehensive toxicological studies on PFCAs are necessary to correlate profiles of PFCAs in serum and those in indoor dust. Further studies are warranted in both exposure assessment and source identification of PFCAs.

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## Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.chemosphere.2011.09.024.

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# Environmental Ecological Modeling of Human Blood Lead Levels in East Asia

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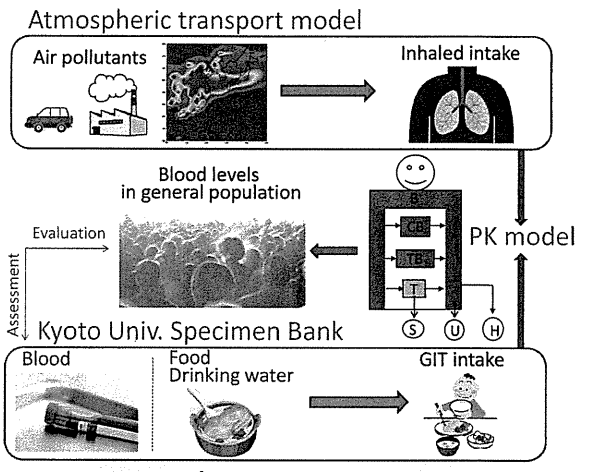
Supporting Information

**ABSTRACT:** Environmental ecological modeling (EEM), which unifies models simulating transport of chemicals and exposure of humans to chemicals, was used to simulate long-term trends of female adult human blood lead levels (BLLs) and historical exposure to the atmospheric lead in four East Asian countries: Japan, Korea, China, and Vietnam. Anthropogenic lead emissions to the atmosphere in Vietnam were estimated from energy statistics to be 1931 t $yr^{-1}$. Calculated BLLs generally agreed with those observed in samples collected in these countries as the error factors were less than 2. The model results revealed that BLLs decreased significantly in Tokyo (by 58%) and Seoul (by 45%) in recent decades and confirmed the effects of efforts to reduce environmental lead in Japan and Korea. The model results also revealed that BLLs in Beijing did not decrease in this decade as much as in Tokyo and Seoul, despite the phasing out of leaded gasoline, and that the contribution from the atmospheric component was increasing (43% in 2009). Finally, we applied EEM to simulate BLLs of children in Hanoi. The probability of children having BLLs greater than 50 $\mu g\ L^{-1}$ was 7.5%, which was greater than those observed in developed countries.




## INTRODUCTION

Lead is one of the most abundant heavy metals in the environment. It is ubiquitous and toxic to humans. Exposure to lead occurs by two major intake pathways: the gastrointestinal tract (GIT) and the respiratory tract. GIT intake depends on the contamination level of food and water reflecting the ecological system and the infrastructure for the food/water supply. Such intake can be reduced through efforts within each country. On the other hand, inhaled intake depends on air concentrations of contaminants, which are not always easily controlled within each country because of transboundary air pollution. Atmospheric lead exists in fine particulate matter and may be transported to a large extent by air flow.

Atmospheric lead is recognized to originate predominantly from anthropogenic activities such as leaded gasoline use, nonferrous metal production, and fossil fuel combustion.[1] In many developed countries, anthropogenic lead emission has been reduced remarkably in recent years.[2] However, efforts to reduce emissions are insufficient in East Asian countries, several of which are undergoing rapid economic progress. Investigation of human exposure to atmospheric lead is necessary to determine potential impacts in East Asia. It is especially important for children, because the developing nervous system is thought to be far more vulnerable to lead toxicity than the mature brain.[3]

We recently developed an atmospheric transport model of lead that can simulate air concentrations of contaminants in East Asian countries.[4] The model can predict exposure intensity of lead through inhalation. If we measure in addition representative GIT intake levels of lead in a given population, we can predict

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blood lead levels (BLLs) by applying the appropriate pharmacokinetic (PK) model.

In the current study, we applied environmental ecological modeling (EEM). EEM is a novel method that involves simulation of a series of emission, transport, and exposure events. This method was first used to simulate serum levels of perfluorooctanoic acids (PFOA) in Japan.[5] It consists of an atmospheric transport model and a PK model. In previous work, we estimated PFOA emissions from a single plant, transport in the air, and exposure to PFOA by the residents around the plant. Inhaled intake values were derived from surface air concentrations simulated by the atmospheric model. GIT intake values were estimated on the basis of duplicate samples of food and water. EEM enabled us to explain the high serum levels of PFOA in the residents. We here apply the EEM approach to BLL simulations and exposure assessments.

In this work, EEM was used to reproduce BLLs for female adults without occupational exposure and to reconstruct historical exposures to atmospheric lead from 1979 to 2009 in four East Asian countries: Japan, Korea, China, and Vietnam. The BLLs simulated by the PK model using the estimated intake values were compared to those monitored in blood samples collected in these countries.

Finally, we applied EEM to simulate BLLs of children by incorporation of GIT intake variability in Hanoi, Vietnam, where the economy has grown rapidly and few risk assessment studies have been conducted. BLLs exceeding 100 $\mu g\ L^{-1}$ have been accepted internationally to be hazardous to children.[6] Recently, however, evidence of adverse health effects associated with BLLs less than 100 $\mu g\ L^{-1}$ has been presented.[7] The Joint FAO/WHO Expert Committee on Food Additives assessed the health risks of lead and established a provisional tolerable weekly intake of 25 $\mu g\ kg^{-1}$ of body weight to prevent net accumulation in the blood of children above 50 $\mu g\ L^{-1}$.[8] For this reason, we evaluated the probability of children with enhanced BLLs (EBLLs) exceeding two reference values: 50 and 100 $\mu g\ L^{-1}$.

## MATERIALS AND METHODS

**Atmospheric Transport Model.** We developed a global transport model of atmospheric lead.[4] The atmospheric model was an Eulerian transport model driven by 6 h meteorological fields determined from the JRA-25 reanalysis data sets.[9] The computational region was the globe, and the horizontal resolution is 1.25°. The vertical structure consisted of 12 layers stacked from the surface to 100 hPa. Atmospheric lead in the model was emitted, transported, and removed as particulate matter with a diameter of 1.0 $\mu m$. We previously estimated anthropogenic lead emissions to the air in Japan, Korea, and China, and conducted a long-term simulation using the estimated emissions and preexisting data sets for lead emissions from the rest of the world. Model results were compared to observations of air concentrations in the three countries in recent decades, and, in general, good agreement was found.[4]

In the present study, we estimated anthropogenic lead emissions to the air in Vietnam using energy statistics for 2007.[10] The national emission estimate was calculated as fuel consumption multiplied by the corresponding emission factors listed in Table S1, Supporting Information. The estimates were horizontally distributed on the basis of two assumptions (see Figure S1, Supporting Information): (1) 60% of total emissions were allocated to the model grid elements corresponding to the two largest cities, Hanoi and Ho Chi Minh City, based on gross output of industry,[11] and (2) the horizontal distribution of emissions was proportional to the volume of road traffic passengers.[11]

**Air Monitoring in Vietnam.** We investigated daily concentrations of atmospheric lead in an urban area of Hanoi for 3 days beginning on November 20, 2008. Ambient $PM_{10}$ particles were collected over a 24 h period by a low-volume air sampler (AN-200, Tokyo Dylec Co., Tokyo, Japan), which was set inside a shelter 1.5 m above the ground on the campus of Hanoi Medical University. The samples were incinerated and analyzed using an inductively coupled plasma mass spectrometer (Aligent ICP-MS 7500). Our observations, together with published observation results for Hanoi and Ho Chi Minh City listed in Table S2, Supporting Information, were compared to lead concentrations simulated by the atmospheric model.

**PK Model.** Human BLLs in Japan, Korea, China, and Vietnam were calculated using a multicompartmental PK model.[12] The model estimated lead fluxes among four compartments of the human body (blood, soft tissue, cortical bone, and trabecular bone) and lead excretion in urine, sweat, hair, and other materials as follows:

$$\frac{dC_i}{dt} = \frac{E + \sum_j k_{j/i} \cdot C_j \cdot L_j}{L_i} - \left(\sum_j k_{i/j} + k_{i/e}\right) \cdot C_i \qquad (1)$$

where $C$ is lead concentration ($\mu g\ L^{-1}$) and $L$ is pool size (L). The subscripts $i$, $j$, and $e$ represent the $i$th and $j$th compartments and excretion material, respectively. $E$ is lead intake ($\mu g\ day^{-1}$), which is included only for the blood compartment. $k_{i/j}$ is the rate constant ($day^{-1}$) of the flux from "$i$" to "$j$". The rate constants used in the PK model are listed in Table S3, Supporting Information. The Euler-forward time step was set to 1 day.

The pool size of blood for adults was assumed to be 6 L.[12] That for children was scaled to body weight (BW) to the 2/3 power.[13] Assuming BW of 56 kg for adults, BW for children was a function of age (year):[14]

$$BW = 3.5 + \frac{22 \times age}{3 + age} + \frac{34}{1 + 600 \times \exp(-0.017 \times 34 \times age)} \qquad (2)$$

**Inhaled Intake Estimates.** Inhaled intake was estimated from the daily average surface air concentrations predicted by the atmospheric model. It was assumed that 50% of the inhaled lead is absorbed into the blood through the respiratory tract,[14] and that the volume of inhaled air is 20 $m^3\ day^{-1}$ for adults[15] and proportional to the logarithm of BW for children.[16]

**GIT Intake Estimates.** The GIT intake values were derived from contamination levels of lead in food and water multiplied by the portion absorbed by the GIT, which was 0.08.[14] Contamination levels were estimated as a function of time by exponential regression of levels observed in the food and water samples collected in each country, as described later:

$$Pb = Pb_0 \times \exp[k \cdot \{t_y - 1979 + (t_d - 1)/t_t\}] \qquad (3)$$

where $t_y$ is the year, $t_d$ is the day of the year, and $t_t$ is the total number of days in the year. The constants $Pb_0$ and $k$ are listed in Table S4, Supporting Information. The contamination level for Vietnam was assumed to be constant because there was only one sampling date in 2007. GIT intake values were estimated

assuming that the contamination level is uniform in each country, and that consumption levels of both food and water are proportional to BW to the 2/3 power.[17]

**Initial Condition for the PK Model.** For adults, the initial lead burden of cortical bone was set to 36 400 $\mu$g.[12] The initial burdens of other compartments were determined by a preparatory run using fixed lead levels in food/water (27.2 $\mu$g day$^{-1}$) (Table S4, Supporting Information) and atmosphere (100 ng m$^{-3}$),[4] which were representative in Japan in 1979. The burdens under a quasi-steady condition were 250 $\mu$g for blood, 170 $\mu$g for soft tissue, and 1050 $\mu$g for trabecular bone. These values used as the initial burdens in the present study were much lower than those used in the study by Marcus, [12] reflecting intake levels lower by 1 order of magnitude. For children, the initial lead burden of each compartment was set to the simulated adult burden in Hanoi scaled to the BW of neonates (3.5 kg) to the 2/3 power.[13]

**Human Food and Blood Samples.** Efforts have been made by the Kyoto University Human Specimen Bank[18] to collect samples of human blood and duplicate food/water portions to monitor long-term exposure of the nonsmoking and nonoccupational population to heavy metals in Japan since the late 1970s.[19−21] Similar coupled samples in Korea and China were collected in the 1990s through the cooperation of researchers in both countries.[22−26] In 2007, we collected 60 samples in three cities (Kyoto, Sendai, and Takayama) in Japan, 35 samples in Seoul, 25 samples in Beijing, and 22 samples in Hanoi. In 2009, we also collected blood samples from 33 mother−child pairs in Hanoi. All of the children were elementary school students between 7 and 12 years old (see Table S5, Supporting Information). All samples were processed and analyzed using the methods described by Koizumi et al.[18] The observations made by Kyoto University researchers along with published lead concentrations of female adult blood and/or food in Japan, Korea, China, and Vietnam (see Tables S6−S8, Supporting Information) were binned according to the atmospheric model resolution into 23 grid boxes for Japan, 4 boxes for Korea, 14 boxes for China, and 1 box for Vietnam (see Figure S2, Supporting Information). The binned concentrations in Japan are listed in Tables S9−S11, Supporting Information. The food/water contamination levels were used to estimate GIT intake values, and the BLLs were used for comparison to simulated BLLs to evaluate both the atmospheric model and the PK model.

**Risk Assessment of Child BLLs in Hanoi.** We conducted a risk assessment for children with EBLLs of individuals between 7 and 12 years old in Hanoi using Monte Carlo simulations. For each age group, calculations were performed 10 000 times to incorporate GIT intake variability assuming a log-normal distribution of food/water contamination levels with the geometric mean (GM) and geometric standard deviation (GSD) observed in the food/water duplicate samples collected in Hanoi in 2007 (see Table S8, Supporting Information). The time periods were set so that the calculations ended at 2009. Calculated BLLs averaged annually in 2009 were compared to observations in Hanoi in 2009 (see Table S5, Supporting Information) and used to derive the probabilities of children having EBLLs.

**Evaluation of Model Fit.** The fit of simulated values to observed values was evaluated by the fractional difference $f$[27] averaged for all the samples:

$$f = \frac{V_{mdl} - V_{obs}}{V_{mdl} + V_{obs}} \quad (4)$$

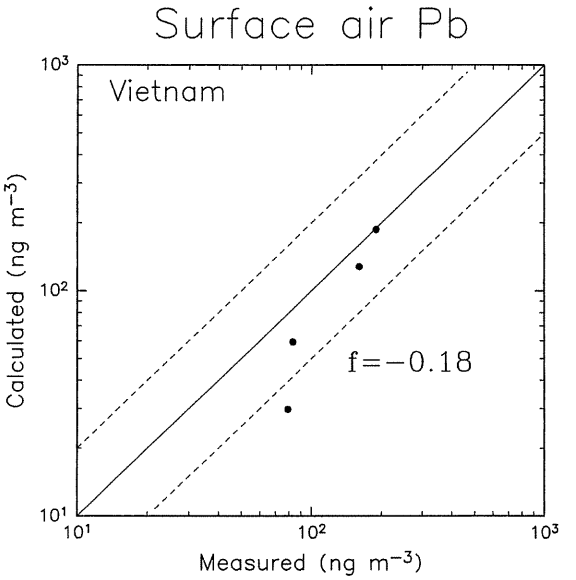


**Figure 1.** Comparison of lead concentrations in surface air (ng m$^{-3}$) in Vietnam between model results and actual observations listed in Table S2, Supporting Information. General agreement is indicated by an averaged fractional difference $f$ of −0.18. Dots on the solid line indicate a perfect fit between observed and calculated values, and those between the two dashed lines indicate error factors less than 2.

where $V_{mdl}$ and $V_{obs}$ are the modeled value and observed value, respectively. $|f| < 0.33$ indicates an error factor less than 2.

## RESULTS

**Lead Emissions and Surface Air Concentrations in Vietnam.** The estimated national lead emission level for Vietnam was 1931 t yr$^{-1}$, which was comparable to the present level of 1631 t yr$^{-1}$ for Korea that was estimated in our previous study.[4] Gasoline use had a dominant contribution (86%) to national emissions because of negligible nonferrous metal production[28] and low consumption of fossil fuels other than gasoline in Vietnam (see Table S1, Supporting Information).

Figure 1 shows comparisons between modeled lead levels in surface air in Vietnam and the actual observations listed in Table S2, Supporting Information. The predicted concentrations were averaged over the same periods as the observations. Three out of the four samples were reproduced with an error factor less than 2. The average fractional difference $f$ equaled −0.18, indicating generally good agreement but slight underestimation of actual levels.

**Lead Contamination Levels in Food and Water.** The lead contamination levels in food/water in Japan, Korea, and China had an obvious descending trend (see Figure S3, Supporting Information). From 1979 to 2009, the inferred contamination levels decreased from 27.2 to 5.3 $\mu$g day$^{-1}$ in Japan, from 36.3 to 12.8 $\mu$g day$^{-1}$ in Korea, and from 35.5 to 20.3 $\mu$g day$^{-1}$ in China. The levels in food and water in Vietnam were constant at 13.8 $\mu$g day$^{-1}$ because of only one sampling date.

**Female Adult BLLs.** Simulated female adult BLLs were compared to observations in Japan, Korea, China, and Vietnam (Figure 2). Fairly good agreement was indicated by the small $f$ value of 0.02. The GM value of BLLs observed in blood samples collected in Japan decreased from 32.8 $\mu$g L$^{-1}$ around 1980 to

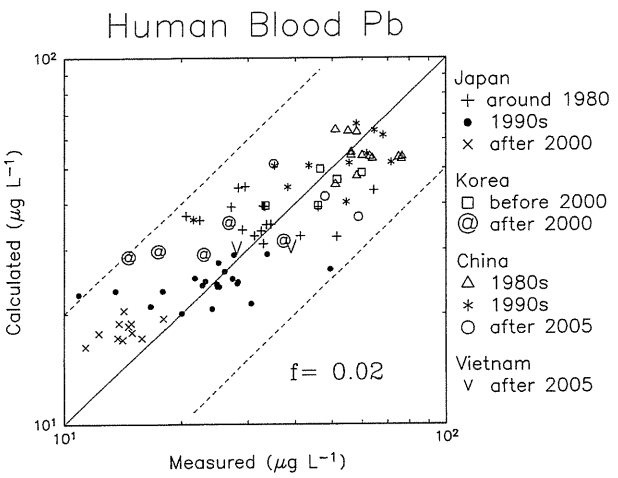


**Figure 2.** Comparison of modeled and observed human BLLs ($\mu g\ L^{-1}$). Good agreement is indicated by a small $f$ value of 0.02. The observations are listed in Tables S6–S11, Supporting Information: (+) ca. 1980 in Japan, (●) 1990s in Japan, (×) after 2000 in Japan, (□) before 2000 in Korea, (@) after 2000 in Korea, (Δ) 1980s in China, (*) 1990s in China, (○) after 2005 in China, and (V) after 2005 in Vietnam.

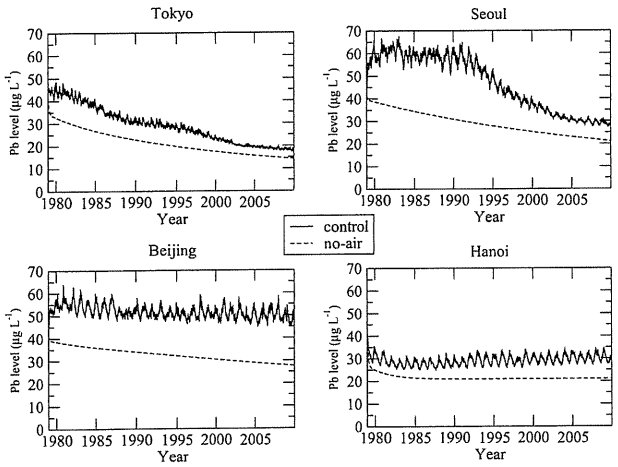


**Figure 3.** Time series of simulated BLLs ($\mu g\ L^{-1}$) from 1979 to 2009 for residents of Tokyo, Seoul, Beijing, and Hanoi. The cases without respiratory exposure (no-air) are also shown in the figures.

24.0 $\mu g\ L^{-1}$ in the 1990s, and then decreased to 15.7 $\mu g\ L^{-1}$ in this decade. The GM value also decreased in Korea from 44.3 $\mu g\ L^{-1}$ in 1994 to 17.7 $\mu g\ L^{-1}$ after 2000. In China, the GM values were 60.5 $\mu g\ L^{-1}$ in the 1980s, 53.6 $\mu g\ L^{-1}$ in 1990s, and 53.9 $\mu g\ L^{-1}$ in the last 5 years. In Hanoi, Vietnam, in 2009, the GM value was 28.0 $\mu g\ L^{-1}$, which is similar to levels in Japan around 1980 and in Korea in the early 2000s. The PK model reproduced almost all of the retrospective human BLLs observed in the samples collected in these countries. The error factors were consistently less than 2.

**Simulated Temporal Trends of Female Adult BLLs and Atmospheric Contributions.** Figure 3 shows the temporal trends of female adult BLLs ($\mu g\ L^{-1}$) simulated for residents in Tokyo, Seoul, Beijing, and Hanoi. Spikes found in the simulated BLLs are the result of seasonal variability in surface air lead concentrations,

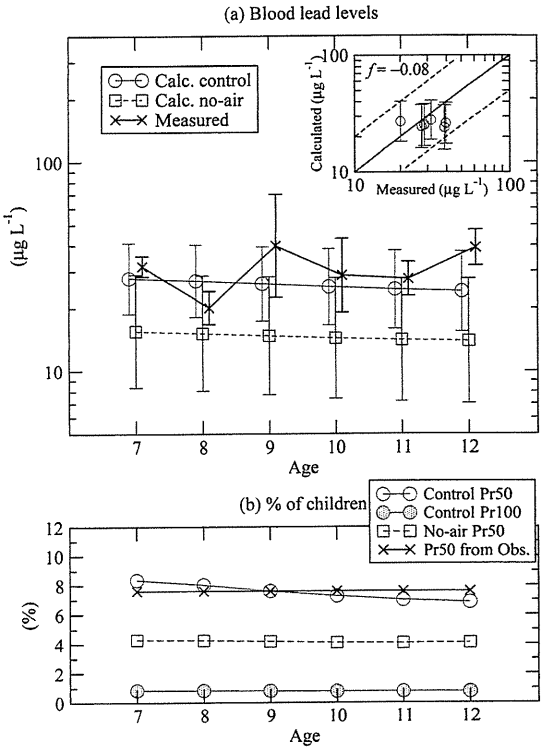


**Figure 4.** Monte Carlo simulations of BLLs for children from 7 to 12 years of age in Hanoi. (a) The modeled BLLs ($\mu g\ L^{-1}$) averaged over the last year (red ○) generally agree with those observed in 2009 (×). Symbols and error bars represent the GM and one GSD. The inset shows that the model reproduced GM values of observed BLLs by error factors less than 2. General agreement is indicated by a low $f$ value of −0.08, which was calculated using GM values of modeled and observed BLLs. (b) Calculated probabilities of children with EBLLs exceeding 50 $\mu g\ L^{-1}$ (Pr50) and 100 $\mu g\ L^{-1}$ (Pr100). The modeled Pr50 agrees well with the Pr50 value estimated from the observed BLLs for all ages (7.6%). Results of the no-air run are shown in both figures (blue □).

which typically peak in winter. BLLs for the case without inhaled intake values (no-air run) are also shown in Figure 3. Differences between the control run and the no-air run represent the atmospheric component of total lead in human blood.

The simulated BLLs in Tokyo decreased steadily by 58% from 44 $\mu g\ L^{-1}$ in 1980 to 18 $\mu g\ L^{-1}$ in 2009. Those in Seoul exceeded 50 $\mu g\ L^{-1}$ until the early 1990s, and decreased rapidly thereafter by 45% to 29 $\mu g\ L^{-1}$ in 2009. The atmospheric component in Tokyo had a minor contribution ranging from 30% in 1983 to 19% in 2003. The atmospheric contribution in Seoul attained a maximum of 48% in 1990 and decreased to 24% in 2005.

Simulated BLLs of about 50 $\mu g\ L^{-1}$ for Beijing were much greater than those for Tokyo in this decade. Although the GIT intake values of lead estimated from the observed contamination levels of food and water decreased steadily in China (see Figure S3, Supporting Information), the simulated BLLs in Beijing decreased much more slowly than in Tokyo and Seoul. This is because the surface air concentrations of lead simulated by the atmospheric model had no temporal trend.[4] The simulated trend of BLLs gently descending in Beijing agreed with that observed in samples collected in China, which was not so clear as in Japan and Korea (Figure 2). The atmospheric contribution increased from 32% in 1980 to 43% in 2009.

In Hanoi, the simulated BLLs were about 30 $\mu g\ L^{-1}$ and the atmospheric contribution was 30%, assuming fixed levels of lead emission to the atmosphere and contamination of food/water. Both BLLs and atmospheric contributions were similar to those in Seoul in recent years. However, the temporal trends cannot be fully assessed without historical observations.

**Child BLLs in Hanoi.** Figure 4 shows simulated BLLs and the probabilities of children with EBLLs exceeding 50 $\mu g\ L^{-1}$ (Pr50) and 100 $\mu g\ L^{-1}$ (Pr100) for each age from 7 to 12 years. The PK model using BW as a function of age, which was highly correlated to the BW values of participants in the 2009 sampling ($R^2 = 0.82$, see Figure S4, Supporting Information), reproduced BLLs with error factors less than 2. The corresponding $f$ value of −0.08 indicated good agreement between simulated and actual levels. Simulated BLLs decreased slightly for older aged individuals because of the larger pool size. The modeled and actual GM values for all ages were 25.9 and 29.6 $\mu g\ L^{-1}$, respectively. Modeled Pr50 values generally agreed with the Pr50 value of 7.6% that was estimated statistically from the observed BLLs for all ages under the assumption of a log-normal distribution. Pr100 values were less than 1% for both the modeled and the observed values. The atmospheric contribution to total BLLs was greater for children (43%) than adults (30% in 2009) because the modeled intake, which is related to inhalation volume, of children approached that of adults more rapidly than the intake by consumption of food/water. The atmospheric component elevated the Pr50 value for all ages from 4.2% to 7.5%.

## DISCUSSION

In the present study, we used EEM to reconstruct historical human BLLs in four East Asian countries: Japan, Korea, China, and Vietnam. The descending trends modeled for female adult residents in Tokyo and Seoul reflect reductions in environmental lead in Japan and Korea. The decrease in BLLs during 10 years involving phasing out of leaded gasoline was 30.6% from 1980 to 1990 in Tokyo and 37.8% from 1990 to 2000 in Seoul. These values are similar to the reduction rate values of 30.2% observed in BLLs of female adults in the United States from the 1991−1994 survey to the 1999−2002 survey[29] and of 30.5% observed in BLLs of adults in Germany from the 1990/1992 survey to the 1998 survey.[30]

The BLLs simulated for Beijing were much higher than those of the other cities in recent years. The simulated rate of BLLs decrease was only 4.1% from 1999 to 2009. Although leaded gasoline was prohibited nationwide in China in 2001,[31] surface air concentrations of lead observed in Beijing increased gradually in this decade.[32] Apparently, the atmospheric contribution to total blood lead increased as well. Rapid economic growth in China could elevate atmospheric lead levels significantly through increase in coal combustion[32,33] in China and in neighboring countries in the near future. Accordingly, long-term monitoring of environmental contamination levels of lead is needed throughout East Asia.

The daily concentrations that we observed in November 2008 in Hanoi exceeded 100 ng $m^{-3}$. These concentrations still surpassed levels typically observed in Japan and Korea in this decade. The modeled BLLs of 25.9 $\mu g\ L^{-1}$ and Pr100 values less than 1% simulated for children in Hanoi were much lower than those observed in Beijing. It was reported that BLLs and Pr100 values of children less than 6 years old observed in Beijing from 2004 to 2006 were 45.1 $\mu g\ L^{-1}$ and 7.95%, respectively.[31] However, the simulated Pr50 value of 7.5% was greater than those found in developed countries. The Pr50 value in Japan was estimated statistically to be 0.34%, which was derived from the BLLs of children of less than 15 years of age observed in Shizuoka from 2004 to 2005 ($n = 290$, GM = 14 $\mu g\ L^{-1}$, GSD = 1.6).[34] A similar Pr50 value of 0.85% was acquired from the results of the fourth German Environmental Survey in which the subjects were children between 3 and 14 years of age from 2003 to 2006 ($n = 1560$, GM = 16.3 $\mu g\ L^{-1}$, GSD = 1.6).[35] Although leaded gasoline was banned in Vietnam from in 2001,[36] the current model results suggested that strict regulation on leaded gasoline should be made to reduce child BLLs significantly in Vietnam. Recent studies have reported that reducing environmental lead in China has been difficult even after phasing out leaded gasoline primarily because of increasing coal combustion.[32,33] Our estimation of anthropogenic lead emissions to the air in Vietnam, which were primarily from gasoline use (see Table S1, Supporting Information), suggested that they could be reduced significantly in the future through complete phasing out of leaded gasoline. We will follow up BLLs in Vietnam, especially for children.

Uncertainty of input values for the PK model could affect the model results. There is a need to determine the initial burden in each compartment reflecting historical exposure levels before the simulation period. Cortical bone cannot reach any steady state regarding lead due to extremely low clearance (see Table S3, Supporting Information). It acts as a reservoir under high exposure condition and as a source of supply for blood under low exposure condition. Therefore, its initial burden has influence on the model results slightly but over a long period of time, especially after significant reduction of exposure levels. The slight overestimations found in Japanese female adult BLLs in this decade (Figure 2) were mainly attributed to the initial large burden of cortical bone adopted directly from the study by Marcus,[12] where the intake levels were much greater than this study.

Estimation of GIT intakes were based on lead levels observed in food and water. Smaller number of observations tends to greater uncertainty in GIT intakes. A slight overestimation of BLLs by the model in Korea in 2007 is probably the result of an overestimation of lead contamination levels in food and water because of the smaller number of observations than Japan (see Figure S3, Supporting Information).

It was assumed in the present study that GIT intakes were uniform in each country and that inhaled intakes were uniform in each grid element of the atmospheric model. However, there are race-ethnically heterogeneous countries where exposure risks are also heterogeneous depending on the social environment such as income, circumstances, and life style. Differences of BLLs among race-ethnicity groups were reported in the United States.[29] Risk assessment of lead in such countries requires consideration for race-ethnical heterogeneity based on different exposure scenarios.

It should be noted that the global atmospheric transport model used in the current study had a horizontal resolution of 1.25°, which is about 114 km from east to west at 35° N. Consequently, the PK model using surface air concentrations simulated by the atmospheric model could not reproduce BLLs for residents in heavily polluted locales. An atmospheric model with higher spatial resolution should be used to reproduce local distributions of BLLs. It should be also noted that the children who donated blood samples in Hanoi were over 6 years of age

and the sample size for each age group was small. Larger blood sample sizes are necessary for further investigation, especially for samples of younger age groups.

In previous work, we applied EEM to perfluorinated compounds and assessed human exposure to the atmospheric component in Japan.[5] EEM can be applied to other chemicals and other regions, when essential information on contamination levels both in blood and in food/water is available. As demonstrated in this work, we can reconstruct retrospective trends and assess the current risk of chemical contaminants. We can also predict trends and risks in the future using emission scenarios. EEM can contribute significantly to regional environmental assessment and decision making.

## ■ ASSOCIATED CONTENT

**Supporting Information.** Additional figures regarding the model results and detailed information on the observations. This material is available free of charge via the Internet at http://pubs.acs.org.

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# Asian Dust Transport to Kanto by Flow around Japan's Central Mountains

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## Abstract

During the Typical Asian Dust episode from the end of March–early April 2007 (TAD-2007), the mass concentration of suspended particulate matter (SPM) began to increase rapidly on the morning of 1 April in Hokuriku and Tohoku but remained low in Kanto. Ground-level lidar and rawinsonde sounding in Hokuriku observed a dust layer at ~2 km corresponding to the base of the temperature inversion. In Kanto, which is leeward of Japan's central mountain ranges, SPM increase began from the east coast and then advanced westward after 18 JST with easterly winds. Merged CloudSat and CALIPSO datasets indicated that clouds over mainland Japan and the Sea of Japan were located in the upper-level atmosphere (> 6 km). Continuous meteorological observations showed that cloud condensation and rainfall were not observed over the mountains during the daytime of 1 April. These results suggest that the delay in the SPM increase in Kanto was caused by dust being indirectly transported to that region by flowing around the central mountains.

## 1. Introduction

During 29 March–2 April 2007, a massive dust episode referred to as the Typical Asian Dust in 2007 (TAD-2007), was observed over East Asia. The maximum $PM_{10}$ concentration exceeded 1000 μg $m^{-3}$ in Seoul, South Korea (see Fig. 4 of Park et al. 2010) and 700 μg $m^{-3}$ in Banryu, western Japan (see Fig. 5 of Yumimoto et al. 2008). TAD-2007 included extremely high dust concentrations that affected all of East Asia, and its large-scale three-dimensional structure has been intensively documented in previous studies (e.g., Yumimoto et al. 2008; Park et al. 2010). On a regional scale, however, complex spatial and temporal variations of dust concentrations occurred over Japan.

Tsunematsu et al. (2009) examined the regional-scale characteristics of the TAD-2007 using Doppler lidar observations, ground-based meteorological observations, and a regional atmospheric model for the Kanto region on the eastern side of central Japan. They concluded that the surface-based stable layer in Kanto prevented downward transport of elevated dust particles to the ground in the early morning on 1 April. They also noted that the dust layer extended to the ground after breakup of the stable layer. In addition, Takahashi et al. (2010) reported that dust transport from TAD-2007 was delayed over the Kanto plain compared to the highland observatory at Mt. Haruna (1365 m altitude), which is located on the northwestern edge of the plain. In Tsukuba (see Fig. 1b), lidar observations showed that the dust increase started with a southeasterly wind and was restricted to the near surface level (below approximately 1 km) after 16 Japan Standard Time (JST: UTC+9). They noted that the low-level dust layer was advected via a different route to the upper dust layer. Although these studies documented regional-scale features of TAD-2007, there were no sufficient explanations for the delayed dust transport in Kanto on 1 April 2007. From a visible image taken by the Moderate Resolution Imaging Spectroradiometer (MODIS) onboard Aqua, dust plumes and clouds widely covered Japan and surrounding area in the daytime of 1 April (Fig. 1a). Wet deposition processes play an important role in dust scavenging in a downstream region (e.g., Han et al. 2004).

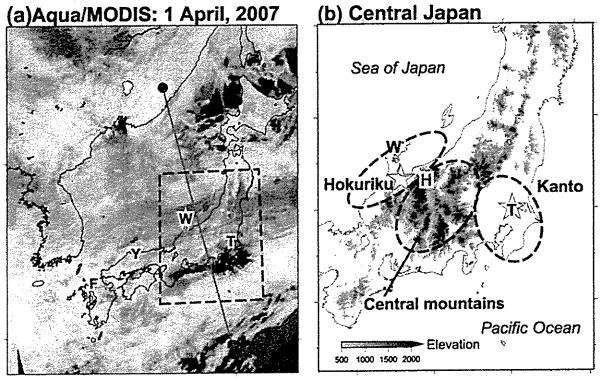


Fig. 1. (a) Visible image from the Aqua/MODIS for 1 April, 2007 (Level-3 surface reflectance; MYD09CMG V005). Rawinsonde sites are labeled as Fukuoka (F), Yonago (Y), Wajima (W), and Tateno (Tsukuba; T). The small domain (Fig. 1b) is indicated by a dashed rectangle. Green line shows the cross section projection for Fig. 5. (b) Star: Lidar sites (Toyama and Tsukuba). The label "H" represents Happo observatory of the Acid Deposition Monitoring Network in East Asia (EANET).

The main objective of this study was to identify the major cause(s) for the delay in dust transport to Kanto, Japan. To investigate the temporal and spatial variation in dust concentrations on a regional scale, we used operational air-quality monitoring data from densely distributed monitoring sites in Japan.

## 2. Data

Aerosol mass concentrations of suspended particulate matter (SPM) were observed at about 1300 ground sites in Japan. Due to the difference in the cut-off property of particles between $PM_{10}$ (50% cut-off at 10 μm) and SPM (100% cut-off at 10 μm), SPM concentration was approximately equivalent to $PM_7$ (Minoura et al. 2006).

The grid-point value meso-scale model (GPV-MSM) data distributed by the Japan Meteorological Agency (JMA) were used in this study. The 3-hourly analysis data had 16 pressure levels (1000–100 hPa) with 0.1° × 0.125° horizontal resolution around Japan (22.4°N–47.6°N, 120°E–150°E).

We used a non-spherical (dust) extinction coefficient from lidar observations by the National Institute for Environmental Studies, Japan. The algorithm and estimation scheme for separating spherical/non-spherical particles were taken from Shimizu et al. (2004).

To identify the vertical profiles of cloud and dust particles, we also used merged datasets containing the Cloud-Aerosol Lidar and Infrared Pathfinder Satellite Observation (CALIPSO) and the CloudSat data provided by the Research Institute for Applied Mechanics (RIAM) of Kyushu University (hereafter merged data-

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