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Global occurrence of polybrominated diphenyl ethers and their hydroxylated and methoxylated structural analogues in an important animal feed (fishmeal)☆

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ABSTRACT

Polybrominated diphenyl ethers (PBDEs) and their hydroxylated (OH) and methoxylated (MeO) structural analogues have been found widely distributed in aquatic ecosystems, and may exhibit potential adverse effects to humans due to their bioaccumulative behavior through food chain. Fishmeal is an important animal feed applied around the world and is generally of marine origin. However, the levels and sources of PBDEs in fishmeal have not been thoroughly evaluated and their structural analogues have not been reported to date. The present study collected ninety-two fishmeal samples from world main fishmeal producing area to determine 27 PBDEs, 10 MeO-PBDEs and 11 OH-PBDEs. The concentrations of Σ_{27}PBDEs, Σ_{10}MeO-PBDEs and Σ_{11}OH-PBDEs were in the ranges of 0.1–1498 (mean: 75.8), 1.14–881 (37.4) and 1.00–47.5 (8.17) ng/g lipid, respectively. PBDEs were found primarily correlated with the historically commercial production, meaning higher production of certain commercial product in a country, higher corresponding PBDE congeners in local fishmeal. A market shift from penta- and octa-formulations toward deca-formulation was observed. BDE209 was identified as a major congener in fishmeal. Both the MeO-PBDEs and the OH-PBDEs were influenced by fishmeal producing areas ($p < 0.001$). High MeO-PBDEs were identified in the Southeast Asian fishmeal, which might be due to the suitable environmental conditions for the generation of bromoperoxidase-contained algae in local area. The ratio of two major MeO-PBDE congeners, 6-MeO-BDE47/2′-MeO-BDE68, were generally >1 in the northern hemisphere and <1 in the southern hemisphere in the present study, which was consistent with the results obtained from previous published papers. Both MeO-PBDEs and OH-PBDEs were in accordance with the specialties of naturally produced halogenated compounds.

1. Introduction

Polybrominated diphenyl ethers (PBDEs) are among the most pervasive flame retardants. They are added to furniture, textiles, and building materials to improve product fire resistance. Due to the potential of bioaccumulation, long range transportation, and certain toxicities (Covaci et al., 2011), the penta-/octa-BDE mixtures were regulated in Europe in 2004 (EU Directive, 2003) and United States in 2005 (UNEP, 2007), and were officially listed in the Stockholm Convention in 2009 (Stockholm Convention Decision, 2009a; Stockholm Convention Decision, 2009b). Deca-BDE, the largest volumes of commercial PBDEs ever produced and used, is voluntarily withdrawn in Europe (Earnshaw et al., 2015) and United States from the end of 2013 (EPA, 2009), but is allowed in the rest of the world until it is officially listed in the Stockholm Convention in 2017 (UNEP, 2017). After the restriction of penta- and octa- PBDEs, the demand for deca-PBDE increased significantly. Currently, the environmental loading of PBDEs is mainly due to direct manufacturing emissions and secondary release from using and disposal of product stock containing PBDEs (Wiseman et al., 2011). Though the levels were reported to descend after 2009 in marine sediment (Lee and Kim, 2015), PBDEs have shown a still high existence in fish/seafood due to their bioaccumulation from the ongoing sources (Cruz et al., 2015).

The structural analogues of PBDEs, such as the hydroxylated and methoxylated (OH- and MeO-) PBDEs, have been revealed as

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ubiquitous compounds in environmental matrices (Kim et al., 2014; Ueno et al., 2008), and terrestrial and aquatic ecosystems (Covaci et al., 2008; Kelly et al., 2008; Mizukawa et al., 2013) in the past two decades. OH- and MeO-PBDEs are not anthropogenic products. It was reported that OH- and MeO-PBDEs may be primarily produced by marine sponges (Vetter et al., 2002), red algae and cyanobacteria (Malmvärn et al., 2005). Also, potential inter-conversion might occur among PBDEs and their structural analogues (Wan et al., 2010b). Previous studies suggested that OH-PBDEs were potentially more toxic than the parent PBDEs (Athanasiadou et al., 2008) and thus aroused environmental and health concerns. OH-PBDEs are structurally similar to thyroxine (T4) and could compete with T4 for binding to transthyretin and thus cause possible endocrine disruption effects (Kojima et al., 2009; Mizukawa et al., 2016). Even more concerning was that OH- and MeO-PBDEs are able to accumulate in high trophic predators (Barón et al., 2013; Kelly et al., 2008; Montie et al., 2010; Pena-Abaurrea et al., 2009) and bio-magnify through marine food chains (Weijs et al., 2009). Moreover, humans are not immune from exposure to these pollutants. Haraguchi et al. reported that the level of 6-OH-BDE47 was approximately 20-fold higher than BDE 47 in the serum of Japanese women (Haraguchi et al., 2016). Athanasiadou et al. (2008). determined that people preferring fish had higher OH-PBDEs levels in their serum. Wan et al. (2010a). also attributed the relatively high OH-PBDEs in South Korean women's serum to the consumption of seafood. The profiles of OH-PBDEs and MeO-PBDEs in human serum were consistent with those in edible seaweed and fish (Haraguchi et al., 2016), which also implied that seafood might be the major source of OH-PBDEs and MeO-PBDEs in humans.

With a spate of researches, it was still difficult to conclude which were the essential elements manipulating levels and distribution patterns of MeO- and OH-PBDEs. One of the reasons was lack of convincing samples with generality and ubiquity on a large scale. The concentrations of compounds could be orders of magnitude different in even one species when considering different tissues. For example, MeO-PBDEs were detected almost 2000 times higher in shark liver than in serum (200 vs. 0.13 ng/g ww) (Haraguchi et al., 2009; Nomiyama et al., 2011b) from the Japanese Ishimaki coast. Therefore, an easily collected, homogeneous, worldwide distributed sample is needed to evaluate the occurrence of PBDEs and their structural analogues from marine environment. Fishmeal is an excellent protein additive that is one of the most important animal feeds. Fishmeal is generally of marine origin, and it is widely used in the global animal farming industry, especially for pet, fish, poultry and pig feed. Generally, it is a uniform mixture of fish and is able to avoid bias of single sample. Therefore, fishmeal might be used as a good indicator of PBDEs and their structural analogues in marine environment.

Though MeO- and OH-PBDEs were reported ubiquitously in aquatic systems, to the best of our knowledge, there is no study investigating the levels and distribution patterns of them in fishmeal to date. According to the formula of the animal feed, fishmeal could account for 10%–40% in weight. Notably, fishmeal might become a main source of PBDEs and their structural analogues for these domestic animals. These contaminants might further transfer to humans through the feed-to-fork pathway. In the present study, we collected 92 fishmeal samples worldwide to investigate PBDEs and their analogues. The levels, profiles, and geographic distribution of these contaminants in fishmeal samples were identified. Moreover, the essential elements manipulating levels and the distribution patterns of these congeners were also discussed according to the geographical distribution of the fishmeal samples.

2. Materials and methods

2.1. Sample collection

Ninety-two fishmeal samples were collected during the year 2012 and 2013 in the present study. The fishmeal samples covered the most important fishmeal producing areas, and they were grouped into five subgroups according to their origin, as follows: United States (U.S., n = 9), China (n = 28, including 17 from north China and 11 from south China), South America (n = 45; 8 from Chile, 35 from Peru and 2 from Ecuador), Europe (n = 5; 4 from Russia and 1 from Denmark), and Southeast Asia (n = 5; 2 from Vietnam, 2 from Thailand and 1 from Malaysia). Fishmeal is used as an important raw material for the production of animal feed. In this study, all the fishmeals were sampled from the Chinese feed factories' warehouses directly. According to the information of the labels attached on fishmeal bags, we could identify these fishmeals were worldwide collected, such like China, U.S., South America, Europe and Southeast Asia. Related information was collected in situ. All of the samples were wrapped with aluminum foil and sealed in plastic zip bags until they arrived in the laboratory. Next, the samples were kept under −18 °C until analysis. The manufacturing techniques, fish types of raw materials and other information of the fishmeal were recorded in Table S1.

2.2. Reagents and materials

PBDE commercial standards that include twenty-seven congeners (BDE 3, 7, 15, 17, 28, 47, 49, 66, 71, 77, 85, 99, 100, 119, 126, 138, 153, 154, 156, 183, 184, 191, 196, 197, 206, 207 and 209) were purchased from the Wellington Laboratories (Guelph, Canada). The list both covers the most common anthropogenic PBDEs and low/high brominated congeners. Ten MeO-PBDE standards (6-MeO-BDE47, 2′-MeO-BDE68, 6-MeO-BDE85, 4-MeO-BDE42, 4′-MeO-BDE49, 3-MeO-BDE47, 5-MeO-BDE47, 5′-MeO-BDE99, 6′-MeO-BDE99 and 2′-MeO-BDE28) and eleven OH-PBDE (3-OH-BDE28, 2-OH-BDE28, 4-OH-BDE42, 3-OH-BDE47, 4-OH-BDE49, 5-OH-BDE47, 6-OH-BDE47, 2-OH-BDE68, 6-OH-BDE85, 5-OH-BDE99 and 6-OH-BDE99) were purchased from AccuStandard (New Haven, CT, USA). MBDE-MXG ($^{13}C_{12}$ BDE 3, 15, 28, 47, 99, 100, 126, 153, 154, 183, 197, 207, 209) and MBDE-ISS-G ($^{13}C_{12}$ BDE 79, 138, 206) that were purchased from Wellington Laboratories were used as clean-up standards and recovery standards for PBDEs and MeO-PBDEs, respectively. $^{13}C_{12}$-6-OH-BDE47 that was purchased from Wellington Laboratories was used as a clean-up surrogate standard for OH-PBDE. Pesticide-grade dichloromethane (DCM), n-hexane, methyl *tert-butyl* ether (MTBE), 2-propanol and HPLC-grade acetonitrile were purchased from J.T. Baker (Phillipsburg, USA), and the nonane was from Sigma-Aldrich (St. Louis, USA). Silica gel (0.063–0.100 mm) was purchased from Merck (Darmstadt, Germany). Anhydrous sodium sulfate (baked at 660 °C for 6 h), concentrated sulfuric acid and sodium hydroxide were of reagent grade (purity>99.8%) and were purchased from Beijing Chemistry Company (Beijing, China).

2.3. Analytical method and quantification

The fishmeal samples were lyophilized to a moisture content of 12%. Before extraction, the samples were spiked with two surrogate standards (1 ng MBDE-MXG for PBDEs and MeO-PBDEs; 10 ng $^{13}C_{12}$-6-OH-BDE47 for OH-PBDEs). Next, the weighed 1.0 g samples were extracted in an ultrasonic bath for 20 min with 10 mL hexane: MtBE (1:1) and 2 mL 2-propanol twice. After 10 min of centrifuging at a speed of 3500 rpm, the supernatant was transferred to a conical flask. The combined extracts were evaporated by rotary evaporator (Heidolph, Germany) to constant weight to record the fat contents.

Then the extracts were re-dissolved in 30 mL DCM. Acid silica gel (44% in weight) was used to remove fat and macromolecule substances. After using an anhydrous sodium sulfate column to filter, the extract was evaporated until almost dry and was exchanged with 2 mL hexane. A column that was packed with 5 g of 5% water-deactivated silica (in weight) at the bottom and 2 cm Na_2SO_4 on the top was used to separate the PBDEs, MeO-PBDEs and OH-PBDEs. The first component containing PBDEs and MeO-PBDEs was eluted with 12 mL DCM in 48 mL hexane, and the second component containing OH-PBDEs was eluted with 60 mL DCM. The first fraction was concentrated to 20 μL and spiked with 1 ng internal standard (MBDE-ISS-G) before instrument analysis, and the second fraction was concentrated until nearly dry and was exchanged with 100 μL acetonitrile. The PBDEs and MeO-PBDEs were analyzed by high-resolution gas chromatography coupled with high-resolution mass spectrometry (HRGC/HRMS, DFS, Thermo Fisher Scientific, USA). The GC column was a DB-5MS fused silica capillary column (J&W, Scientific, 30 m, 0.1 μm film thickness, 0.25 mm i.d.). The OH-PBDEs were analyzed by liquid chromatography coupled with tandem quadrupole mass spectrometry (TSQ Quantiva, Thermo Fisher Scientific, USA) with the electrospray ionization (ESI) source in negative mode using a C18 column (100 mm × 2.1 mm, 2.2 μm particle size, Dionex, USA).

2.4. Quality assurance and quality control (QA/QC)

The glass ware was washed with target solvents twice before use. The blank samples that were inserted into each analytical batch showed no interferences during the analyses. The method detection limits (MDLs) were defined as three times the signal-to-noise (S/N) ratio, and they were 2.67–65.4 pg/g lw, 344.5 pg/g lw, 0.5–148.3 pg/g lw and 0.067–0.4 ng/g lw for PBDEs (excluding BDE209), BDE209, MeO-PBDEs and OH-PBDEs, respectively. The surrogate standards that were added to each fishmeal showed that the average recoveries of the $^{13}C_{12}$-PBDEs and $^{13}C_{12}$–OH-PBDEs were in the range of 73.0%–109% and 82.7%–108%, respectively. The matrix spiked experiments (1 ng mixture of ten MeO-PBDEs and 5 ng mixture of eleven OH-PBDEs) were used to evaluate the quantification of MeO- and OH-PBDEs because non-$^{13}C_{12}$ labeled standards were available for each congener. The spiked recoveries for each congener were between 90.2% and 105.6%. All of the native compounds' concentrations were recovery corrected. Authentic standards (5 ng/g) were inserted every 10 samples to check the status of the instrument for LC-MS/MS analysis.

2.5. Statistical analysis

Pearson correlation coefficients were calculated to assess the bivariate relationships between PBDE homologues and their structural analogues. A one-way ANOVA was conducted to evaluate the correlations between the PBDEs and their structural analogues. Principal component analysis (PCA) was performed on the most abundant PBDEs and their structural analogues in the fishmeal samples to assess their potential different sources. Only the compounds with a detection rate of more than 50% were involved in statistical discussion. The concentrations below the LOD were assigned zero in the level and comparison discussion but assigned half of the LOD in statistical analysis to avoid bias. IBM SPSS Statistics 19.0 (Chicago, IL) and Microsoft Office Excel (2010, U.S.) were employed for statistical analysis. The level of statistical significance was defined at $p < 0.05$ unless otherwise specified.

3. Results

Of the twenty-seven PBDEs, ten MeO-PBDEs and eleven OH-PBDEs congeners monitored, BDE3, BDE126, 6-MeO-BDE85, 4-MeO-BDE42, 5′-MeO-BDE99, 3-OH-BDE47, 5-OH-BDE47 and 6′-OH-BDE99 were under the detection limits in all the 92 fishmeal samples and were thus excluded in later discussion.

3.1. Levels and profiles of PBDEs and their structural analogues in the fishmeal samples

The mean and median values, level ranges, and detection rates of selected PBDEs, MeO- and OH-PBDEs in fishmeal samples were listed in Table 1 on a lipid weight basis. PBDEs, MeO- and OH-PBDEs were detected in all the 92 fishmeal samples. The concentrations of Σ_{27}PBDEs in fishmeal were in the range of 0.1–1498 ng/g lipid, with a notably large concentration span being observed among different samples. The average of Σ_{27}PBDEs was 75.8 ng/g lipid. The concentrations of Σ_{10}MeO-PBDEs and Σ_{11}OH-PBDEs were in the ranges of 1.14–881 ng/g lipid (average 37.4 ng/g lipid) and 1.00–47.5 ng/g

Table 1
Descriptive statistics of concentrations of PBDE, MeO-PBDE and OH-PBDE congeners in fishmeal (n = 92, ng/g lipid).

Analyte	Mean ± SD	Median	Range	DR[a] (%)
BDE7	0.04 ± 0.03	0.04	n.d.-0.06	4.30%
BDE15	0.04 ± 0.04	0.03	n.d.-0.19	92.5%
BDE17	0.08 ± 0.08	0.05	n.d.-0.45	58.1%
BDE28	0.20 ± 0.17	0.16	n.d.-1.02	98.9%
BDE47	1.15 ± 1.89	0.61	0.04–14.0	100%
BDE49	0.49 ± 0.78	0.31	n.d.-6.77	98.9%
BDE66	0.32 ± 0.47	0.14	n.d.-2.28	51.6%
BDE71	0.28 ± 0.17	0.29	n.d.-0.50	5.38%
BDE77	0.18 ± 0.28	0.10	n.d.-1.15	17.2%
BDE85	0.36 ± 0.28	0.36	n.d.-0.56	2.15%
BDE99	0.26 ± 0.23	0.19	n.d.-1.53	82.8%
BDE100	0.40 ± 0.44	0.29	n.d.-2.80	75.3%
BDE119	0.24 ± 0.24	0.19	n.d.-1.75	58.1%
BDE138	0.58 ± 0.90	0.14	n.d.-1.94	4.30%
BDE153	0.33 ± 0.53	0.19	n.d.-3.83	59.1%
BDE154	0.57 ± 0.69	0.46	n.d.-4.36	75.3%
BDE156	0.83 ± 0.46	0.92	n.d.-1.27	4.30%
BDE183	0.59 ± 1.08	0.29	n.d.-6.13	55.9%
BDE184	0.53 ± 0.61	0.27	n.d.-1.73	14.0%
BDE191	2.18 ± 2.20	1.44	n.d.-5.47	8.60%
BDE196	1.29 ± 1.55	0.76	n.d.-7.46	52.7%
BDE197	0.42 ± 0.46	0.25	n.d.-1.89	47.3%
BDE206	6.70 ± 10.9	2.88	n.d.-74.7	62.4%
BDE207	4.29 ± 5.45	2.81	n.d.-32.9	64.5%
BDE209	63.8 ± 174	32.2	n.d.-1382	67.3%
Σ PBDEs	75.8	18.9	0.13–1498	
Σ PBDEs(exclude 209)	12.0	6.52	0.13–115	
6-MeO-BDE47	9.92 ± 19.3	3.81	0.43–121	100.0%
2′-MeO-BDE68	27.3 ± 104	4.25	0.52–754	100.0%
6-MeO-BDE85	0.38	0.38	n.d.-0.38	1.1%
4-MeO-BDE42	0.10	0.10	n.d.-0.10	1.1%
4′-MeO-BDE49	0.11 ± 0.12	0.05	n.d.-0.40	19.6%
3-MeO-BDE47	0.21 ± 0.00	0.21	n.d.-0.21	1.1%
5-MeO-BDE47	0.15 ± 0.21	0.04	n.d.-0.39	3.3%
6′-MeO-BDE99	0.38 ± 0.23	0.39	n.d.-0.64	4.3%
2′-MeO-BDE28	0.11 ± 0.56	0.04	n.d.-5.23	93.5%
Σ MeO-PBDEs	37.4	8.89	1.14–881	
3′-OH-BDE28	3.45 ± 2.29	2.62	n.d.-10.0	26.1%
2′-OH-BDE28	2.22 ± 1.42	2.32	n.d.-3.84	4.3%
4-OH-BDE42	1.29 ± 0.69	1.29	n.d.-1.78	2.2%
3-OH-BDE47	1.93	1.93	n.d.-1.93	1.1%
4′-OH-BDE49	0.99 ± 0.68	0.99	n.d.-1.47	2.2%
5-OH-BDE47	2.04	2.04	n.d.-2.04	1.1%
6-OH-BDE47	4.08 ± 4.43	2.34	0.32–20.9	100.0%
2′-OH-BDE68	2.63 ± 2.49	1.93	n.d.-19.6	95.7%
6-OH-BDE85	0.82 ± 0.42	0.76	n.d.-1.95	29.3%
5′-OH-BDE99	4.25 ± 4.83	1.18	n.d.-11.2	5.4%
6′-OH-BDE99	1.10	1.10	n.d.-1.10	1.1%
Σ OH-PBDEs	8.17	5.51	1.00–47.5	

[a] DR: detection rate.

lipid (8.17 ng/g lipid), respectively.

The predominant PBDE congeners were BDE209, BDE47 and BDE28, accounting for 68.2, 8.4 and 2.0% of the Σ_{27}PBDEs in fishmeal respectively. BDE209 was quantified in 62 of the 92 samples, while BDE47 and BDE28 were 100% and 98.9% detected respectively. The detection rates of BDE49, 15, 99, 100, 154, 207, 206, 153, 17, 119 and 183 were also more than 50 percent in fishmeal. The previously mentioned 14 congeners jointly contributed nearly 95% to the total PBDE mass weight in fishmeal. 6-MeO-BDE47 and 2′-MeO-BDE68 were the predominant congeners for the MeO-PBDEs, and both were 100% detected. Another highly identified MeO-congener was 2′-MeO-BDE28, which was found in 86 of the 92 fishmeal samples. The other congeners, such as 4′-MeO-BDE49, 6′-MeO-BDE99, 5-MeO-BDE47, and 3-MeO-BDE47, were sporadically found with detection rates less than 50%. Considering the profiles of the OH-PBDEs, 6-OH-BDE47 and 2′-OH-BDE68 were the main congeners at 100% and 95.7% found in fishmeal, followed by 6-OH-BDE85 (29.3%), 3′-OH-BDE28 (26.1%), 5′-OH-BDE99 (5.4%), 2′-OH-BDE28 (4.3%), 4-OH-BDE42 (2.2%), and 4′-OH-BDE49 (2.2%).

3.2. Geographic distributions of PBDEs and their structural analogues in fishmeal

The geometric means of PBDEs found in fishmeal samples rank as China (78.2 ng/g lipid) > U.S. (22.0 ng/g lipid) > Southeast Asia (9.89 ng/g lipid) > South America (3.98 ng/g lipid) > Europe (2.96 ng/g lipid) (Fig. 1). The highest PBDE level reached 1498 ng/g lipid, and it was found in a fishmeal sample from northern China (S37).

The highest MeO-PBDE levels in fishmeal were found in Southeast Asia (97.0 ng/g lipid) followed by China (23.0 ng/g lipid). The MeO-PBDE levels in fishmeal from Southeast Asia were one order of magnitude higher than from U.S. (8.95 ng/g lipid), Europe (5.97 ng/g lipid) and South America (4.89 ng/g lipid). The contributions of 6-MeO-BDE47 and 2′-MeO-BDE68 to the ΣMeO-PBDEs varied according to the fishmeal producing areas, and they were averagely weighted as 45.8% and 53.6%, 56.7% and 42.9%, 50.9% and 48.8%, 39.8% and 58.4%, 18.4 and 81.2%, corresponding to the ΣMeO-PBDEs in fishmeal collected from U.S., China, Europe, South America, and Southeast Asia, respectively.

The geographic distribution of OH-PBDEs in fishmeal was neither the same as PBDEs nor MeO-PBDEs. Generally, the concentrations of OH-PBDEs in fishmeals from different producing areas were on the same order of magnitude. The OH-PBDE levels were 12.7 ng/g lipid for China, 6.12 ng/g lipid for Southeast Asia, 4.71 ng/g lipid for Europe, 3.59 ng/g lipid for U.S. and 3.49 ng/g lipid for South America. The relative contributions of 6-OH-BDE47 and 2′-OH-BDE68 to total OH-PBDEs were similar among the different areas, showing an average value of 49.3% and 40.5% respectively.

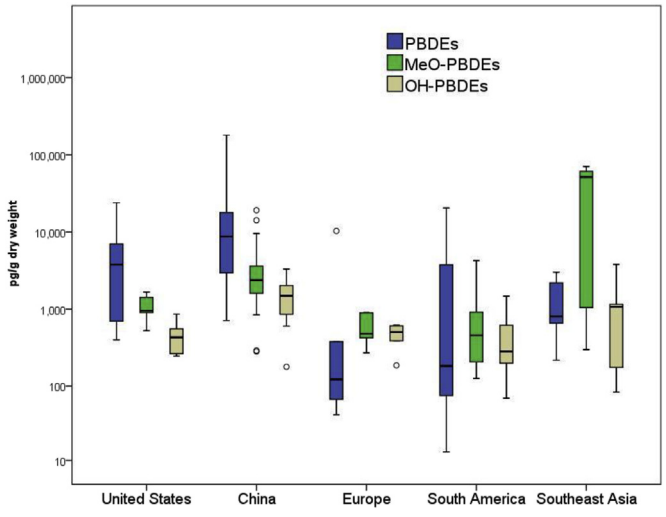

Fig. 1. Geographic distribution of PBDEs, MeO-PBDEs and OH-PBDEs in fishmeal (n = 92). From bottom to top, the five horizontal lines mean the 10th, 25th, 50th, 75th, and 90th percentiles of the concentrations. Open circles are outliers.

4. Discussion

4.1. PBDEs' distribution and influential factors

PBDEs have been identified in all of the fishmeal samples, indicating that this type of animal feedstuff was vulnerable to PBDEs. The PBDEs ranged from 0.9 to 2.2 ng/g dry weight in European fishmeal according to a study by Suominen et al. (2011), which was slightly higher than in the European fishmeal of this study (geomean 0.27 ng/g dry weight). Not too much PBDEs data could be found for fishmeal samples other than Europe recently. Considering the European fishmeal possessed the lowest PBDEs compared to the other areas in our study, the contamination of PBDEs in fishmeal should be taken seriously. To be compared with worldwide fish samples, the average PBDE levels in fishmeal (average 7.4 ng/g dry weight) were similar to Hongkong fish (1.9–16 ng/g ww) (Wang et al., 2011) and Spanish fish (0.97–3.87 ng/g ww) (Pardo et al., 2014). An investigation of the PBDE level between wild and farmed salmon found significantly higher PBDEs in the latter, which could be attributed to the consumption of animal feed (Hites et al., 2004). Additionally, fish consumption is considered to be a major source of PBDEs for humans through diet (Meng et al., 2007; Thomsen et al., 2008). The PBDEs in fishmeal could accumulate through the feed-animal-human chain and might eventually elicit adverse health effects.

It was reported that the production volumes and application categories of commercial PBDEs were different in the United States, Europe and Asia (three main commercial PBDE consuming areas) (Hites, 2004). We conducted a discussion and expected that the PBDEs in fishmeals from different areas were related to the historic production of local area. The sum of BDE (47, 99, 100, 153, 154), BDE (183, 196, 197) and BDE209 t are used to represent concentrations of commercial penta-, octa- and deca-BDE in fishmeal, respectively (La Guardia et al., 2006). The production volumes of commercial PBDE mixtures in United States, Europe and Asia were quoted from Hites et al.'s study (Hites, 2004). No production information for PBDEs was available for South America and Southeast Asia; thus, they were excluded from this discussion. It was reported that penta-BDE products were mainly applied in U.S., accounting for more than 95% to the total penta-production volume (Hites, 2004). The penta-BDEs were the highest in the fishmeals from U.S. (average 0.69 ng/g lw), compared to Asian fishmeal (0.32 ng/g lw) and European fishmeal (0.05 ng/g lw). Deca-BDE was the most produced PBDE product (1.3 million tons in 1970–2010 globally) (Earnshaw et al., 2015). Both U.S. and Asia were the main producers of commercial PBDE products, the two of which jointly contributed 97%, 80% and 85% of penta-BDE, octa-BDE and deca-BDE products to the world annual production (approximately 70,000 tons) in 2001 (Hites, 2004). As BDE209 was voluntarily withdrawn from the U.S. market at the end of 2013 (EPA, 2009), the production and usage of deca-BDE tended to transfer to Asian countries (Li et al., 2016). The fishmeal with the highest PBDE concentration (S37) was from an intensive deca-BDE producing district (Shandong Province, China), where were previously reported with high levels of PBDEs in the local environment (Pan et al., 2011). BDE209 accounted for 92.3% to the total PBDEs in this sample. BDE209 was

found to be the predominant congener with a detection rate of 67.3% and average concentration of 63.8 ng/g lipid in fishmeal. This finding was in accordance with a previous study denoting that the occurrence of BDE209 in marine biota could be associated with local contamination and an ongoing source (Moon et al., 2010). The fishmeal from Europe and South America showed the average BDE209 detection rate to be 20.0% and 48.8% respectively, which inferred relatively less usage or longer elimination time of deca-BDE in these regions. Though other BDE congeners have tended to decrease since the prohibition at approximately 2006 (Sun et al., 2015), the deca-BDE was still produced and used in many countries (especially in developing countries), which might present ongoing sources and cause consistent trend of BDE209 in biota and environment (Li et al., 2016; Cruz et al., 2015). BDE209 was not observed in cetacean blubber between 1995 and 2001 but was determined in the same species between 2003 and 2012 (Zhu et al., 2014). Chen et al. (2008). found that BDE209 in rural American peregrine falcon eggs doubled within a short time span of 3.7 years. What's more, many manufacturing capacities of commercial PBDE products (especially the deca-BDEs) were moved to the less developed nations such as China (Li et al., 2016). And China is recognized as the biggest consumer (27%) with the fastest growth rate (15%) of commercial brominated flame retardants (BFRs) in the World from 2013 (Flame Retardants, 2014), meaning more attention needed to be paid to the Asian countries for deca-BDE was mainly used in this area. Our results showed that penta-, octa- and deca-BDE in fishmeals were in accordance with local production of commercial BDE categories. It means that more production of certain PBDE commercial formulation, more corresponding formation specific congeners would be found in fishmeals.

Relatively low levels were observed for penta- and octa-BDE products in fishmeal. BDE183 was one of the major congeners in octa-BDE commercial mixture; however, it only exhibited an average contribution of 0.6% to the total PBDEs in fishmeal. This result indicated less historical usage of octa-BDE technical products (approximately accounting for 6.0% of the annual PBDE production) (Hites, 2004). Besides, the lower congener preferential bioaccumulation of BDE183 might also be responsible. The contribution of penta-formulation congeners to the total PBDEs in fishmeals was also lower than deca-BDE. The BDE47/99 ratio is less than 1 in two mainly used commercial penta-PBDE products, DE-71 and 70-5DE (La Guardia et al., 2006). Then, this ratio was usually employed to evaluate the historic usage of penta formulation and debromination from BDE99 to BDE47 (Alonso et al., 2012; Tang et al., 2017). In this study, the BDE47/99 ratios were 16.4, 6.28, 4.34, 4.18 and 4.0 for U.S., Europe, Southeast Asia, South America and China, respectively. Previous work showed that BDE47 might tend to bioaccumulate in adipose tissue than BDE99 (Hakk et al., 2002). However, beyond preferential bioaccumulation, debromination of BDE99 to BDE47 might also contribute to the relatively higher BDE47/99 ratio in U.S., especially with consideration of their past large consumption of penta products (Alonso et al., 2012; Tang et al., 2017). Then, the fishmeal, which is a uniform mixture of fish, has been demonstrated to be a very good indicator of the PBDE levels of local areas. As discussed above, an obvious shift of production and usage was observed from penta- and octa-BDE to deca-BDE formulation in tune with the phase-out of penta-/octa-commercial products and continued use of deca-BDE in recent years, especially in the developing countries.

4.2. MeO-PBDEs and OH-PBDEs' distribution and influential factors

The MeO- and OH-PBDEs have been consistently detected in all fishmeal, implying their ubiquitous distribution. Especially, high levels of MeO-PBDEs were observed in fishmeal from Southeast Asia (average 97.0 ng/g lipid), and they were generally orders of magnitude higher than those from other sampling zones. Since no study has reported MeO-PBDEs and OH-PBDEs in fishmeal to date, we summarized the concentration and congener patterns of MeO-PBDEs, OH-PBDEs and PBDEs in published literature, primarily in marine biota, for a comparison and discussion (Table 2). Though the MeO-PBDE levels in fishmeals were lower than in some apex predators, such as dolphin and cetaceans, only those fishes from the Mediterranean Sea contained higher MeO-PBDEs than Southeast Asian fishmeal presented in our study (Ben Ameur et al., 2013; Pena-Abaurrea et al., 2009). The MeO-PBDEs have never been intentionally produced and were generally considered to originate from natural production of algae and sponges in marine environments (Wiseman et al., 2011) or sparsely to be metabolites in biotic and abiotic environments (Wan et al., 2009; Zhang et al., 2012). Therefore, the biomass and activity of algae might be positively related with MeO-PBDE amount. Actually, algae-related eutrophication is one of the environmental problems in Southeast Asia due to leaching of animal manure from farms, fertilizer runoff, waste material from aquaculture (Chua Thia et al., 1989; Todd et al., 2010) and large scale shrimp farming industry discharge (Gräslund and Bengtsson, 2001). It was known that the algal biomass was pertinent to nitrogen and phosphorus availability (Smith, 2003). The local environment will facilitate the generation of algae and further promote the increase of bromoperoxidase content in the algae (Flodin et al., 1999). Together with the fitting tropical water temperature of Southeast Asia, the bromoperoxidase enzymes in marine species exhibit higher activity and make catalysis of the halogenation process of carbon skeleton precursors with bromide much easier (Butler and Walker, 1993). To the best of our knowledge, this report is the first to describe the occurrence of MeO-PBDEs and OH-PBDEs in marine samples from Southeast Asia (details were discussed in the SI). The regional high background level of MeO-PBDEs in Southeast Asian biota might be potential MeO-PBDE sources for ocean ecosystems and should arouse more concern.

Generally, higher MeO-PBDEs were observed in marine fish rather than fresh water fish according to Table 2, which implied more MeO-PBDE exposure risk for marine fish. Similar to PBDEs, MeO- and OH-PBDEs could bio-magnify through the food chain (Weijs et al., 2009) and pose health risks to apex predators (Athanasiadou et al., 2008; Kelly et al., 2008). A study attributed high levels of MeO- and OH-PBDEs in cat serum to the consumption of seafood, indicating their bioaccumulation in predators through food consumption (Norrgran et al., 2015). As is known, fishmeal which was added to pet food could contribute 10–20% to its total weight. Since marine-originated fishmeal is widely and abundantly used as a protein additive in animal feed (e.g., for fish, pets, poultry, and pigs), the MeO- and OH-PBDEs in fishmeal could accumulate in these farmed animals and further biomagnify through the farm-to-fork pathway in the human body. Previous studies reported that people who preferred eating fish had higher OH-PBDEs in their serum compared to others (Athanasiadou et al., 2008; Haraguchi et al., 2016; Wan et al., 2010a). The occurrence of MeO- and OH-PBDE in fishmeal could be a continuous and stable source for human beings.

The various concentrations of PBDEs in fish have been identified relating to the fish type and their collection location (Cruz et al., 2015; Hites, 2004), but the key factors influencing MeO-/OH-PBDE concentrations were not clear. In this study, the fishmeal fishing zones, which were separated into six parts according to the fishing information of the fishmeal (details of each sample were given in Table S1), were taken into consideration as a potential factor that might affect MeO- and OH-PBDEs' levels. The one-way ANOVA statistical analysis denoted that the concentrations of

Table 2
Summation of sampling countries/areas, matrix types/tissues, total concentrations and ranges, primary congener levels, and target congeners of PBDE, MeO-PBDEs and OH-PBDEs in aquatic biota.

Sampling area	Matrix type/Tissue	Concentration				References
		MeO-PBDEs		OH-PBDEs	PBDEs	
		average (range)	47/28[f]	average (range)	average (range)	
fresh						
Canada	fish plasma [c]	0.03 (nd-0.10)	3.43	1.46 (nd-3.58)	5.03 (3.33–9.02)	(De la Torre et al., 2013)
United States	fish plasma [c]	nd [a]	- [b]	0.01 (0.003–0.05)	2.75 (0.93–9.12)	(Valters et al., 2005)
China	fish adipose [c]	0.14 (–)	7.50	0.04 (–)	42.8 (–)	(Zhang et al., 2010)
China HongKong	fish muscle [c]	0.51 (0.09–1.70)	1.05	0.05 (nd-0.24)	4.3 (1.9–14)	(Wang et al., 2011)
China	fish [d]	2.43 (0.80–6.00)	1.41	- (–)	15.4 (4.2–38)	(Zhou et al., 2016)
China	fish [d]	8.53 (0.55–48.0)	18.91	- (–)	18.0 (nd-77.0)	(Su et al., 2010)
marine						
Baltic Sea	blue mussels [d]	43.0 (0.63–220)	8.42	98.3 (0.54–1500)	4.02 (0.10–14)	(Dahlberg et al., 2016b)
Baltic sea	blue mussel [d]	255 (160–420)	1.78	1043 (1600–3500)	- (–)	(Löfstrand et al., 2011)
China	mollusk [e]	0.45 (0.009–2.09)	2.60	0.53 (0.12–2.54)	- (–)	(Sun et al., 2013)
Australia	fish [d]	27 (0.70–110)	0.83	- (–)	51.2 (6.40–115)	(Losada et al., 2010)
Australia	fish muscle [d]	25.8 (0.9–40.7)	0.66	- (–)	115 (51.8–179)	(Losada et al., 2009)
Atlantic sea	fish [d]	2.98 (1.89–4.07)	–	- (–)	12.2 (11.4–12.9)	(Sinkkonen et al., 2004)
Baltic sea	fish [d]	5.18 (3.55–6.81)	–	- (–)	49.8 (49.5–50.0)	
Baltic Sea	fish [d]	96.0 (18–490)	6.84	9.70 (4.4–21)	40.5 (15–85)	(Dahlberg et al., 2016a)
Baltic Sea	fish [c]	0.47 (–)	3.27	- (–)	0.46 (–)	(Kierkegaard et al., 2004)
Canada	fish muscle [e]	9.9 (3.3–30)	2.13	nd (nd)	9.8 (2.6–36)	(Kelly et al., 2008)
Canada	fish muscle [e]	42 (12–150)	5.57	nd (nd)	9.3 (3.0–28)	
China HongKong	fish muscle [c]	1.40 (0.25–6.50)	0.85	0.08 (nd-0.32)	6.30 (2.50–16.0)	(Wang et al., 2011)
Chile	fish [d]	15.7 (6.0–28.0)	0.98	- (–)	45.1 (13.5–122)	(Barón et al., 2013)
Mediterranean sea	fish muscle [d]	325 (284–375)	2.22	- (–)	57.2 (38.0–86.7)	(Ben Ameur et al., 2013)
Mediterranean Sea	farmed fish muscle [d]	63 (42–72)	3.13	- (–)	68 (17–149)	(Pena-Abaurrea et al., 2009)
	wild fish muscle [d]	151 (69–248)	1.36	- (–)	66 (25–219)	
Mediterranean Sea	fish liver [d]	15.5 (4.8–39)	4.09	- (–)	9.82 (3.20–27.3)	(Covaci et al., 2008)
Japan	cetacean blood [c]	0.79 (nd-6.20)	6.51	0.59 (0.05–2.30)	1618 (nd-10000)	(Nomiyama et al., 2011a)
Japan	fish [c]	0.13 (–)	6.98	0.19 (–)	0.01 (–)	(Nomiyama et al., 2011b)
	shark [c]	0.02 (–)	3.09	0.02 (–)	0.02 (–)	
Australia	cetacean [d]	3206 (–)	0.38	- (–)	- (–)	(Melcher et al., 2005)
	crocodile egg [d]	283 (–)	3.50	- (–)	- (–)	
New Zealand	shark liver oil [d]	5.74 (–)	2.30	- (–)	- (–)	
United States	seal blubber [c]	2.87 (1.50–4.39)	2.13	nd (nd)	790 (110–2170)	(Weijs et al., 2014)
United States	blubber [d]	3.68 (–)	4.38	- (–)	- (13.7–80.9)	(Montie et al., 2010)
United States	sea lions blubber [c]	- (nd –12.0)	>1	- (–)	1470 (450–4740)	(Stapleton et al., 2006)
West Africa	seal [d]	39.4 (–)	3.20	- (–)	- (–)	(Vetter et al., 2002)
Antarctic	seal [d]	3.70 (–)	4.30	- (–)	- (–)	
Brazil	cetaceans liver [d]	13540 (26–249000)	0.48	- (–)	647 (3–5960)	(Dorneles et al., 2010)
Brazil	dolphin [d]	1015 (74–4774)	–	- (–)	170 (21.8–572)	(Alonso et al., 2012)
Japan	tiger shark [d]	200 (60–2700)	0.62	- (–)	26 (4–650)	(Haraguchi et al., 2009)
	silvertip shark [d]	190 (100–330)	0.21	- (–)	12 (8.1–30)	
Japan	porpoises [c]	1.36 (0.06–8.60)	≈5.7–14.2	1.13 (0.02–5.20)	0.93 (nd-7.20)	(Ochiai et al., 2016)
Japan	dolphin [d]	1167 (52–2910)	0.65	- (–)	307 (50.8–636)	(Marsh et al., 2005)
	whale [d]	53.1 (48–58.1)	13.20	- (–)	27.4 (7.50–47.3)	
Tanzania	dolphin [d]	74000 (600–210000)	1.07	- (–)	- (–)	(Mwevura et al., 2010)
Belgium and Germany	Harbour porpoise [d]	110 (–)	12.59	- (–)	- (–)	(Weijs et al., 2009)
	Harbour Seal [d]	7.07 (–)	3.79	- (–)	- (–)	
	Cod muscle [d]	7.50 (–)	0.15–0.6	- (–)	12.9 (–)	
	whiting muscle [d]	34.2 (–)	–	- (–)	34.2 (–)	
	Herring muscle [d]	7.10 (–)	–	- (–)	79.6 (–)	
United States	fishmeal [d]	8.95 (4.77–15.1)	0.85	3.59 (2.22–6.16)	22 (3.33–199)	This work
China		23.0 (4.15–177)	1.36	12.7 (2.54–33.0)	78.2 (5.92–1498)	
Europe		5.97 (4.25–9.96)	1.04	4.71 (1.86–6.89)	2.96 (0.67–103)	
South America		4.89 (1.14–47.3)	0.66	3.49 (0.69–21.6)	3.98 (0.13–204)	
Southeast Asia		97.0 (2.98–881)	0.21	6.12 (0.83–47.5)	9.89 (2.17–30.1)	

[a] "nd" means the congener was under detection limit.
[b] "-" means the congener was not analyzed or data cannot be summarized for less information.
[c] ng/g ww.
[d] ng/g lw.
[e] ng/g dw.
[f] 47/28 means ratio of 6-MeO-BDE47/2′-MeO-BDE68.

MeO-PBDEs ($p < 0.001$) and OH-PBDEs ($p < 0.001$) were strongly influenced by the sampling areas. The results implied that different water zones, which had variable natural environments and different biota with various compound production ability and different bioavailability, were the key factors leading to the discrepancy of MeO- and OH-PBDE distribution patterns. This is in accordance to the preceding discussion that the Southeast Asian water zones will facilitate generation of naturally produced halogenated compounds. We further assessed whether the different fishmeal producing methods (semi-skimmed and full-fat) would influence the compound levels. No significant difference was found for PBDEs ($p > 0.05$), MeO-PBDEs ($p > 0.05$) and OH-PBDEs

($p > 0.05$) according to one-way ANOVA analysis, indicating two different fishmeal production modes would not cause concentration variance of PBDEs and their analogues.

Vetter et al. inferred that the ratio of 6-MeO-BDE47/2′-MeO-BDE68 (M47/M28) could aid in evaluating the global distribution variation of MeO-PBDE and identify their origin from the Northern or Southern Hemispheres (Vetter, 2006). We calculated the M47/M28 ratios in the globally collected fishmeal, as well as published literature, and summarized them in Table 2. There is a statistically significant difference ($p < 0.05$) for the M47/M28 ratio between the Northern Hemisphere (>1) and the Southern Hemisphere (<1) in the fishmeal of our study. The summary data in Table 2 also suggest that the M47/M28 ratio in biota generally supports the rule reported by Vetter et al (Vetter, 2006). with the exception of seven species from the 48 species in 33 studies. The fishmeal from China and Europe showed an average ratio of M47/M28 > 1.0, while South America exhibited the ratio <1.0. The U.S. and Southeast Asian fishmeal did not follow this rule, presenting the average ratios of 0.89 and 0.24, respectively. This might be attributed to the difference of individual samples and un-normally high levels of MeO-PBDEs in Southeast Asia. This study was the first to give a global view of the M47/M28 ratio in aquatic samples, the characteristics of which were helpful for evaluating two major MeO-PBDE congeners distribution and for tracing the sample sources.

4.3. A global view of the existence and interconversion of three classes of compounds

From the accumulation histogram of Fig. 2, a large discrepancy in the concentrations and homologue contribution patterns of the PBDEs and their structural analogues was observed among different areas. The total concentrations of PBDEs and their structural analogues are relatively higher in fishmeal from China and Southeast Asia, with the major contributions from PBDEs and MeO-PBDEs, respectively. Higher levels of PBDE structural analogues than PBDEs were not only in our study, but also in bivalves, fish, and marine mammals (Kelly et al., 2008; Nomiyama et al., 2011b; Sun et al., 2013). Principal component analysis (PCA) showed PBDEs clustering separately from MeO- and OH-PBDEs, supporting the conjecture of their unlike sources (Fig. 3). The score plot of PCA (Fig. S1) was grouped by different sampling countries, which also indicated the pattern profiles of major PBDE, MeO-PBDE and OH-PBDE congeners in these countries were different. It is known that PBDEs have been globally put into use as BFRs, while their structural analogues were never produced for industrial purposes. When the total PBDE concentrations in all fishmeal samples were plotted from small to large (Fig. S2), neither MeO- ($p > 0.05$) nor OH-PBDEs ($p > 0.05$) increased according to the PBDE trend, which

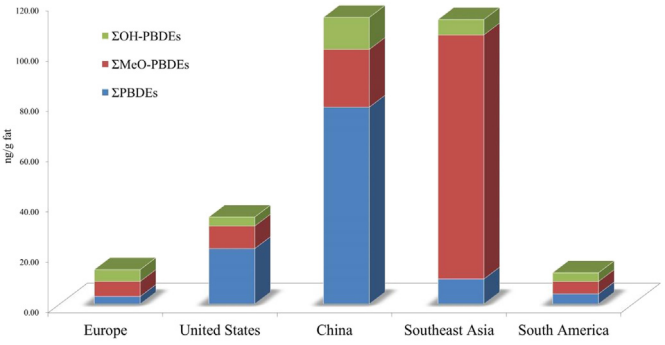

Fig. 2. Relative contributions of PBDEs and their structural analogue concentrations in fishmeal among the fishmeal producing areas.

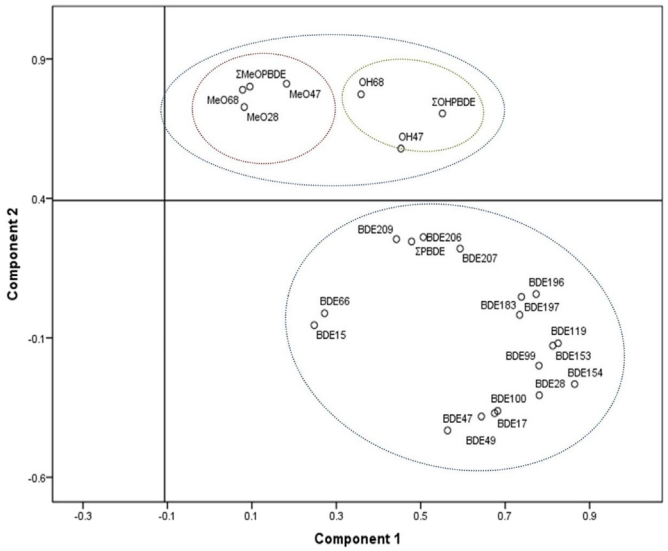

Fig. 3. Loading plot of predominant PBDE, MeO-PBDE and OH-PBDE congener concentrations in all fishmeal samples from the principle component analysis (PCA).

also indicated their disparate sources.

It has been generally suggested, based on the facts of several studies, that MeO- and OH-PBDEs are possibly natural products of marine biota (Malmvärn et al., 2005; Wiseman et al., 2011). More compelling evidence was that the detectable radiocarbon ($\Delta^{14}C$) contents of 6-MeO-BDE47 and 2′-MeO-BDE68 resembled natural extracted compounds, rather than industrial products, indicating their natural occurrence (Teuten et al., 2005). Moreover, the natural occurrence of MeO- and OH-PBDEs could be distinguished by their substituents' position. Most natural OH- and MeO-PBDEs have been identified as ortho-substituted, while the meta-/para-substituted structural analogues of PBDEs were demonstrated to be metabolites of PBDEs (Malmvärn et al., 2005; Stapleton et al., 2009). In our study, the predominant MeO- and OH-PBDE congeners, including 6-MeO-BDE47, 2′-MeO-BDE68, 6-OH-BDE47 and 2′-OH-BDE68, were all ortho-substituted. Moreover, we concluded the statistical correlations of the predominant PBDE, MeO-PBDE and OH-PBDE congeners in Table 3. In our study, BDE47 was neither correlated with 6-MeO-BDE47 ($r = -0.026$, $p > 0.05$) nor with 6-OH-BDE47 ($r = -0.082$, $p > 0.05$). PBDEs had recalcitrant structures to be methoxylated- or hydroxylated-metabolized (Wan et al., 2009), and it was reported that no MeO-PBDEs could metabolize from PBDEs both *in vitro* and *in vivo* (Wan et al., 2009, 2010b; Zhang et al., 2012). The ΣPBDE was not correlated with the ΣMeO-PBDEs ($r = -0.029$, $p > 0.05$) and the ΣOH-PBDEs ($r = 0.060$, $p > 0.05$) too. The production rate of OH-PBDEs from PBDE was only <0.01–1% (Wiseman et al., 2011). These results supported that major MeO- and OH-PBDE congeners in fishmeal were mainly of natural origin. However, it was noteworthy that a meta-substituted congener, 3′-OH-BDE28, which was considered to be a metabolite of PBDEs, was not detectable in fishmeal from U.S. and Europe. It was found in 6 of 45 samples from South America and was even more detectable in fishmeal samples from China (57.1%) and Southeast Asia (60.0%). This might imply that the origin and formation of OH-PBDEs were distinguishable in fishmeal from different areas. The reason has not been thoroughly elucidated to date.

Compared with transforming PBDEs to MeO- and OH-PBDEs, inter-conversion occurs more easily between MeO- and OH-PBDEs, especially for those congeners with similar structure and Br numbers but different substituents (-OH, -MeO) (Wan et al.,

Table 3
Pearson's correlation for concentrations of selected PBDE, MeO-PBDEs and OH-PBDEs in fishmeal.

	BDE47	6-MeO-BDE47	2′-MeO-BDE68	6-OH-BDE-47	2′-OH-BDE-68	ΣPBDE	ΣMeO-PBDE	ΣOH-PBDE
BDE47	1							
6-MeO-BDE47	−0.026	1						
2′-MeO-BDE68	−0.020	0.933 **[a]	1					
6-OH-BDE-47	−0.082	0.501 **	0.268 **	1				
2′-OH-BDE-68	0.001	0.675 **	0.674 **	0.580 **	1			
ΣPBDE	0.019	0.069	−0.047	0.109	0.165	1		
ΣMeO-PBDE	−0.021	0.952 **	0.998 **	0.308 **	0.681 **	−0.029	1	
ΣOH-PBDE	−0.025	0.595 **	0.478 **	0.854 **	0.839 **	0.060	0.501 **	1

[a] The ** and * represent significant correlation at $p < 0.01$ and < 0.05 levels.

2009, 2010b). OH-PBDEs could be formulated through hydroxylation of MeO-PBDEs, and the reverse process was methylation of OH-PBDEs to MeO-PBDEs (Wan et al., 2009). Significantly positive correlation was found between 6-OH-BDE47 and 6-MeO-BDE47 ($r = 0.501$, $p < 0.01$), and 2′-OH-BDE68 and 2′-MeO-BDE68 ($r = 0.647$, $p < 0.01$) in fishmeal samples, implying co-occurrence or possible inter-conversion between these OH- and MeO-BDE congeners. The PCA plot (Fig. 3) also adds weight to the conclusion. Though MeO-PBDEs clustered close to OH-PBDEs, they only had a resemblance in the PC1 direction. The difference in the PC2 direction revealed that OH-PBDEs might be partly derived from the metabolic transformation of MeO-PBDEs or other sources, rather than totally from natural production from the same source as MeO-PBDEs. Wan el al. also found that OH-PBDEs mainly came from the transformation of MeO-PBDEs, rather than from PBDEs *in vitro* (Wan et al., 2009), and further demonstrated this preponderant mechanism later *in vivo* (Wan et al., 2010b). Therefore, the OH-PBDEs in fishmeal might be partly bio-transformed from MeO-PBDEs in the actual ecosystem of local areas. This finding was in accordance with the results of a previous study (Wan et al., 2009), which noted that metabolism from MeO-PBDEs might be an important contributor to the occurrence of OH-PBDEs found in wildlife from remote areas (Wan et al., 2009).

5. Conclusions

This is a systematic study reporting levels, congener patterns and homologue distribution of PBDEs in an important animal feed (fishmeal), and for the first time reporting MeO-PBDEs and OH-PBDEs in animal feed on a global scale. A worldwide existence of the three classes of compounds was observed. The distributions of PBDEs, OH-PBDEs, and MeO-PBDEs were influenced by the fishmeal producing areas. The PBDE levels in fishmeal were significantly positively correlated with the accumulative production of corresponding commercial PBDEs of local countries. It might be mainly due to direct manufacturing emission and secondary release from usage. BDE209 was recognized to be the predominant congener and implied an ongoing source of the deca-formulation in Asian areas. This result indicated that the fishmeal, which is made by uniform mixture of fish, could be considered a very good indicator of PBDE contamination for local areas. High levels of MeO-PBDEs were observed from Southeast Asian fishmeal samples. The structural analogues of PBDEs were regarded as naturally produced halogenated compounds, which might be preferentially generated in the suitable marine environment of Southeast Asian area where both temperature and eutrophication will facilitate the bromoperoxidase-contained algae's generation. These findings should arouse public concern for that the PBDEs and their analogues ubiquitously distributed in such an important animal feed and might transfer to humans through the farm-to-fork pathway.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.envpol.2017.11.059.

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