Marine Pollution Bulletin 58 (2009) 396–401

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Organohalogen contaminants in striped dolphins (*Stenella coeruleoalba*) from Japan: Present contamination status, body distribution and temporal trends (1978–2003)

Tomohiko Isobe [a], Yoko Ochi [a], Karri Ramu [a], Takahito Yamamoto [b], Yuko Tajima [c], Tadasu K. Yamada [c], Masao Amano [d], Nobuyuki Miyazaki [e], Shin Takahashi [a], Shinsuke Tanabe [a,*]

[a] Center for Marine Environmental Studies (CMES), Ehime University, Bunkyo-cho 2-5, Matsuyama 790-8577, Japan
[b] Department of Natural History, Ehime Prefectural Science Museum, Ohjohin 2133-2, Niihama 792-0060, Japan
[c] Department of Zoology, National Museum of Nature and Science, 3-23-1 Hyakunin-cho, Shinjuku-ku, Tokyo 169-0073, Japan
[d] Faculty of Fisheries, Nagasaki University, Nagasaki 812-8121, Japan
[e] Center for International Cooperation, Ocean Research Institute, The University of Tokyo, Minamidai 1-15-1, Nakano-ku, Tokyo 164-8639, Japan

ARTICLE INFO

Keywords:
Polybrominated diphenyl ethers (PBDEs)
Hexabromocyclododecane (HBCDs)
Striped dolphin
Temporal trend
Archived sample

ABSTRACT

Organohalogen contaminants including PCBs, DDTs, CHLs, HCHs, HCB, PBDEs and HBCDs were determined in striped dolphins (*Stenella coeruleoalba*) found stranded at Gogo-shima ($n = 6$, 2003) and collected from Taiji ($n = 15$, 1978–1992) in Japan. All target compounds were significantly detected in all the specimens, indicating ubiquitous contamination of oceanic cetaceans in northwest Pacific Ocean. Examination of body distribution of organohalogens in the six specimens from Gogo-shima showed no significant difference in concentrations among the analyzed tissues, except for brain, which had lower levels possibly due to the existence of blood–brain barrier. For evaluating temporal trends, archived blubber samples of adult male stripped dolphins collected in 1978, 1979, 1986 and 1992 were analyzed. Concentrations of PCBs, DDTs and HCHs did not change significantly during 1978–2003. In contrast, remarkable increasing trends of PBDEs and HBCDs were observed, suggesting growing consumption in Japan and surrounding countries in recent years.

1. Introduction

Six adult striped dolphins (*Stenella coeruleoalba*) were found stranded at Gogo-shima Island, Ehime, Japan in 2003. Although the main cause of death was undetermined, pathological examination revealed the presence of parasites such as *Monorygma* sp. and *Phyllobothrium* sp. in all the six dissected specimens. Goiter was also observed in three of the six specimens (Makara et al., 2006). We assume that this observation suggests possible disruption of thyroid endocrine system due to accumulation of anthropogenic contaminants in the dolphins. Though not much is known on the potential effects of organic contaminants in marine mammals, recent in vitro toxicological studies have demonstrated that hydroxylated metabolites of lower brominated polybrominated diphenyl ethers (PBDEs) can affect thyroid hormone homeostasis (Meerts et al., 2000). Organochlorines (OCs), including PCBs and organochlorine pesticides such as DDTs were extensively used in Japan until the early 1970s. Because of the persistence and lipophilicity of these compounds, their ubiquitous contamination and extreme biomagnification in marine mammals have been observed (de Wit, 2002; Hites, 2004). Concern over environmental contamination by brominated flame retardants (BFRs), especially PBDEs and hexabromocyclododecanes (HBCDs) has increased in recent years due to their physicochemical and toxicological similarities to OCs. Both PBDEs and HBCDs are used in polymers and textiles as additive flame retardants in a wide range of products, such as printed circuit boards, television and computer housings and other electronic household equipments, automotive parts, thermal insulation foams and furniture upholstery. Statistical data demonstrated that Asian countries shared about 40% and 23% of the global PBDEs and HBCDs consumption in 2001, respectively (BSEF, 2005), suggesting that this region could be a significant source of contamination by BFRs. Although many reports on the environmental behavior and fate of PBDEs have been published (Rahman et al., 2001; Ueno et al., 2004; Tanabe et al., 2008), only limited information on other BFRs, such as HBCDs, is available so far (Covaci et al., 2006). In this regard, OCs and BFRs were analyzed in striped dolphins found stranded at Gogo-shima Island to evaluate the present status and body distribution of organohalogen contaminants. Furthermore, temporal trends of organohalogen contaminants were also evaluated by analyzing archived blubber samples of striped dolphins.

* Corresponding author. Tel./fax: +81 89 927 8171.
E-mail address: shinsuke@agr.ehime-u.ac.jp (S. Tanabe).

0025-326X/$ - see front matter © 2008 Elsevier Ltd. All rights reserved.
doi:10.1016/j.marpolbul.2008.10.008

2. Materials and methods

2.1. Samples

Blubber samples of adult striped dolphins (*S. coeruleoalba*) found stranded at Gogo-shima Island, Ehime (n = 6, 2003) and collected from Taiji (n = 15, 1978–1992) in Japan were analyzed in this study (Fig. 1). Specimens from Taiji were obtained from local fishermen. Biological data of the animals are given in Table 1. For the six specimens (5 males and 1 female) from Gogo-shima, muscle, liver, and brain tissues were also collected for assessing body distribution of organohalogen contaminants. All the samples were stored at −25 °C in *es*-BANK (environmental specimen bank) of Ehime University (Tanabe, 2006) until chemical analysis.

2.2. Chemical analysis

Analysis of OCs and BFRs was performed following the procedures described elsewhere with slight modifications (Ueno et al., 2004; Kunisue et al., 2005; Isobe et al., 2007). Briefly, 2–20 g (wet wt) of the sample was ground with anhydrous sodium sulfate and Soxhlet extracted with diethyl ether/hexane (3:1, v/v) for 7–8 h. Extraction was performed in a laminar flow hood to prevent degradation of target compounds by UV and all the extracts were kept in a shelf with dark condition before further treatments. An aliquot of the extract, after spiking 5 ng of $^{13}C_{12}$-BDEs ($^{13}C_{12}$-BDE-3, 15, 28, 47, 99, 153, 154, 183, 197, 207, 209) and 10 ng of $^{13}C_{12}$-HBCDs (α-, β-, γ-$^{13}C_{12}$-HBCD), was loaded to a gel permeation chromatography (GPC, Bio-Beads S-X3, Bio-Rad, CA, 2 cm i.d., × 50 cm) column for lipid removal. The GPC fraction containing BFRs was concentrated and subjected to an activated silica gel column (Wakogel DX, 4 g, Wako Pure Chemicals, Tokyo) for clean-up and fractionation. The first fraction eluted with 80 ml of dichloromethane/hexane (5:95, v/v) from the silica gel column contained PBDEs, while the second fraction eluted with 100 ml of dichloromethane/hexane (25:75, v/v) contained HBCDs. $^{13}C_{12}$-BDE-139 was spiked as an internal standard to the PBDEs fraction solution prior to GC–MS quantification. The extract was transferred to an amber glass vial and 14 congeners were quantified using GC–MS as described elsewhere (Ueno et al., 2004). The HBCDs fraction was evaporated and spiked with HBCD-d_{18} (α-, β-, γ-HBCD-d_{18}) prior to LC–MS–MS analysis. The diastereoisomer-specific analysis was performed on the basis of the reported analytical method (Tomy et al., 2005; Isobe et al., 2007). Recoveries of spiked internal standard of PBDEs and HBCDs were always in the range of 50–120%. Further, our results were in good agreement with those for reference materials in an inter-laboratory comparison exercise as described in our previous report (Isobe et al., 2007). For analysis of OCs including PCBs, DDTs, HCHs, CHLs, and HCB, another aliquot of the Soxhlet extract was purified and fractionated using a GPC and an activated florisil colum. Identification and quantification of OCs was performed using a GC-ECD. Concentrations of organochlorines were not corrected for recoveries. For quality assurance and quality control, we participated in inter-laboratory comparison exercises several times. In addition, we also analyze standard reference material SRM1945 to confirm analytical performance. All the data from our laboratory were comparable to the certified values or data from other laboratories. Concentrations of analytes are expressed as ng/g lipid weight unless otherwise specified.

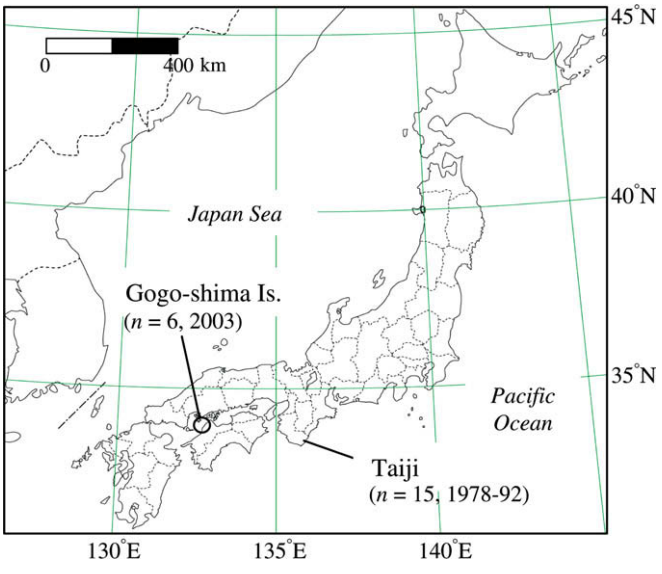

Fig. 1. Sampling locations of striped dolphins from Japan.

3. Results and discussion

3.1. Present contamination status of organohalogen compounds

Concentrations of OCs and BFRs detected in the blubber of striped dolphins are shown in Table 1. Among the organohalogen compounds in the specimens stranded in 2003, concentrations of DDTs ranked first followed by PCBs, CHLs, HBCDs, PBDEs, HCHs, and HCB. Levels of DDTs and PCBs ranged from 3500 to 88,000 and from 3200 to 42,000 ng/g lipid wt., respectively, and were in the same range as those reported for other cetaceans from Japanese and Asian coastal waters (Kajiwara et al., 2006a, 2006b; Ramu et al., 2006; Kajiwara et al., 2008; Tanabe et al., 2008) and from North America and Europe (Johnson-Restrepo et al., 2005; McKinney et al., 2006; Litz et al., 2007). These levels of PCBs and DDTs were still one to two orders of magnitude lower than some of the highest levels ever reported, i.e. concentrations in striped dolphins during mass mortality event at western Mediterranean Sea in 1990 (Kannan et al., 1993). However, concentrations of PCBs detected in the striped dolphins exceeded the threshold levels for disruption of vitamin A dynamics and physiological effects in harbor seals (*Phoca vitulina*) (de Swart et al., 1995; Kannan et al., 2000) suggesting that the striped dolphins from Gogo-shima could be susceptible to the adverse effects by these contaminants.

Levels of PBDEs and HBCDs in the specimens stranded in 2003 ranged from 84 to 850 and 290 to 940 ng/g lipid wt., respectively, which were one to two orders of magnitude lower than DDTs and PCBs, but significantly higher than HCHs and HCB. The observed levels of PBDEs in striped dolphins were higher than those reported for other cetaceans from Japan, with an exception of finless porpoises (*Neophocaena phocaenoides*) from the Seto Inland Sea (Kajiwara et al., 2006a, 2008). Nevertheless, the levels of PBDEs were slightly lower than those from North America and European countries (Lebeuf et al., 2004; Pettersson et al., 2004; Johnson-Restrepo et al., 2005; McKinney et al., 2006; Litz et al., 2007), which could be due to the voluntary termination/restrictions on the usage of lower brominated PBDE commercial mixtures by Japanese industrial associations in the early 1990s, earlier than similar restrictions/ban in other regions. Another possible contamination source of PBDEs in the recent years could be Asian developing countries. Since striped dolphin is an oceanic species migrating from tropical to temperate waters, exposure to contaminants in Asian tropical region could be a factor. In fact, an increasing trend of PBDE levels in finless porpoise from South China Sea was observed in our previous study (Ramu et al., 2006). Similar to PBDEs, levels of HBCDs in the specimens of the present study were also

Table 1
Concentrations of organohalogen compounds in archived blubber samples of striped dolphins analyzed in this study.

Sample ID	Year	Gender	Body length (cm)	Body weight (kg)	Age (yr)	Lipid (%)	Organohalogen concentrations (ng/g lipid)						
							PCBs	DDTs	CHLs	HCB	HCHs	PBDEs	HBCDs
EW04115	1978	Male	227	132	11.5	36.0	26000	75000	3500	770	380	13	12
EW04116	1978	Male	230	160	17.5	49.2	24000	86000	3500	590	320	13	12
EW04157	1979	Male	235	NA	19.5	55.0	31000	130000	5200	560	510	21	16
EW04174	1979	Male	239	NA	18.5	73.5	30000	99000	4400	620	470	20	13
EW04180	1979	Male	230	NA	18.5	91.0	15000	48000	2700	410	240	13	11
EW04186	1979	Male	233	NA	19.5	67.2	24000	76000	3500	570	430	18	10
EW04438	1986	Male	NA	NA	20.5	55.9	28000	79000	5800	420	300	210	29
EW04439	1986	Male	228	NA	9.5	46.8	20000	48000	5300	600	360	270	36
EW04500	1986	Male	251	NA	31.5	54.0	27000	86000	6700	520	280	190	26
EW04534	1992	Male	226	NA	NA	62.3	17000	36000	6000	390	360	550	210
EW04539	1992	Male	231	NA	NA	56.0	23000	54000	8100	690	320	520	200
EW04547	1992	Male	226	NA	NA	64.7	18000	39000	6200	520	340	520	200
EW04548	1992	Male	226	NA	NA	45.9	11000	26000	3900	370	190	420	140
EW04557	1992	Male	235	NA	NA	28.4	26000	70000	9200	760	300	640	310
EW04559	1992	Male	230	NA	NA	71.5	22000	58000	11000	1100	290	660	310
EW04566	2003	Male	231	155	NA	56.2	28000	52000	8700	410	400	520	700
EW04567	2003	Female	223	105	NA	56.2	3200	3500	730	53	40	84	290
EW04568	2003	Male	238	150	NA	58.4	26000	44000	7300	340	300	530	580
EW04569	2003	Male	232	155	NA	49.8	42000	88000	12000	540	490	850	940
EW04570	2003	Male	231	135	NA	44.1	32000	56000	9100	440	380	650	700
EW04571	2003	Male	237	150	NA	54.4	36000	67000	11000	620	520	610	620

NA: Data not available.

higher than those in cetaceans from other locations of Asia–Pacific region and comparable to those from European coasts (Morris et al., 2004; Zegers et al., 2005; Law et al., 2006; Isobe et al., 2007). HBCD levels in biota are generally higher in Europe than North America because of high consumption of these compounds in Europe (Morris et al., 2004; Zegers et al., 2005; Law et al., 2006; Muir et al., 2006). Consumption of HBCDs in Japan was 2,200 tons in 2001, which was more than half of the total consumption among Asian countries (BSEF, 2005). The extensive usage and subsequent discharge of HBCDs into the environment in Japan might have resulted in the high HBCD levels found in the present study. Although individual BFR concentrations were lower than the levels of recorded adverse effects, no information on the effect of exposure to multiple chemicals exists. An interdisciplinary approach involving environmental chemists, toxicologists, and pathologists is essential to elucidate the reasons for mass stranding of striped dolphins.

3.2. Body distribution of organohalogen contaminants

To understand the tissue distribution of organohalogen contaminants, blubber, liver, muscle and brain of striped dolphins stranded in 2003 were analyzed (Fig. 2). Lipid normalized concentrations showed no significant differences between blubber, liver and muscle, which indicates that the distribution of these compounds in body depends on lipid content. Similar distribution pattern like that of PCBs and DDTs was observed for both PBDEs and HBCDs, suggesting similar accumulation kinetics of BFRs and OCs in a dolphin's body. However, concentrations of organohalogens in brain tissue were an order of magnitude lower than those in other tissues due to the existence of the blood–brain barrier. In all the analyzed tissues, concentrations of the analytes were one order of magnitude lower in the female dolphin than those in the males with no exception. It is reported that during lactation OCs are transferred from mother to calf resulting in lower OC levels in the mother (Borrell and Aguilar, 2005). This phenomenon is consistent with previous report (Kajiwara et al., 2008) and was also observed for BFRs, suggesting transfer of PBDEs and HBCDs from mother to calf during lactation. Age-dependent accumulation is also another possible reason for this difference. Although information on age was not available, we assumed that all the specimens

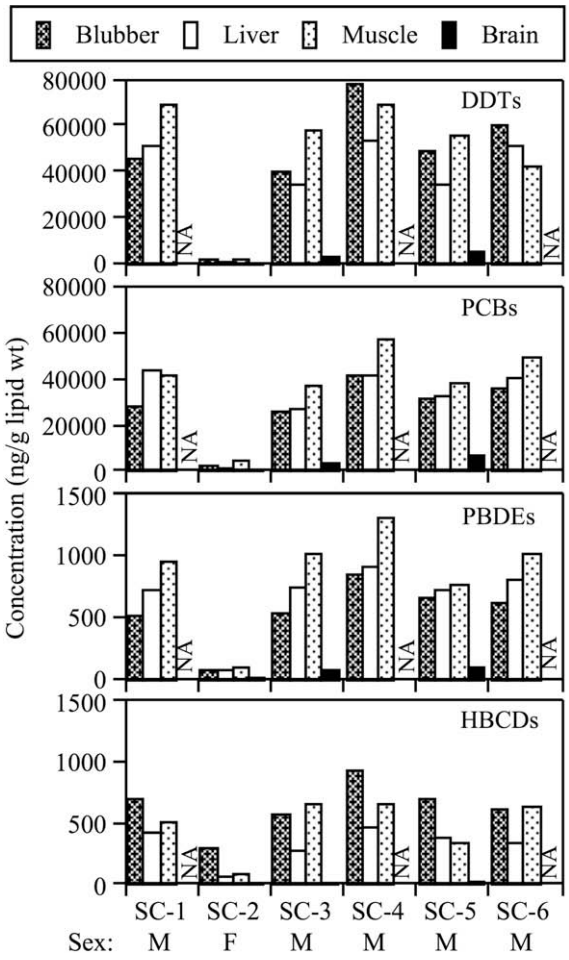

Fig. 2. Concentrations of organohalogens (ng/g lipid wt.) in blubber, liver, muscle and brain of striped dolphins stranded at Gogo-shima, Japan in 2003. NA: Sample not available.

were adults based on their body length and observation of gonads (Makara et al., 2006). The only female specimen had apparently

lower body weight among the analyzed dolphins and, thus, might have been younger than the others. Therefore, the lower contamination levels of organohalogens in the female dolphin could be due to age as well as gender. However, no significant relationships between body weight and concentrations of organohalogens were found with Spearman's rank test. Therefore, although the sample number was small ($n = 6$), we assumed that effect of age on levels of organohalogens was negligible when compared to gender in the present study. Body burden of organohalogens was estimated using concentrations of each analyte in each tissue and reported tissue fraction in striped dolphin (Tanabe et al., 1981) as follows:

$$\text{Burden in tissue} = C(\text{tissue})(\text{ng/g wet wt}) \times W(\text{kg}) \times f(\text{tissue})$$

where C (tissue) is the concentration of analyte in each tissue (blubber, liver and muscle), W is body weight of the dolphin, and f (tissue) is tissue fraction in the body. Tanabe et al. (1981) reported that blubber, muscle and liver constitute 17.2%, 55.6% and 1.25% of total body weight, respectively. It is assumed that burden in bone is negligible because contribution of OCs in bone to total burden was calculated to be 0.4–0.6% of the total body burden (Tanabe et al., 1981). In addition, we assumed that weight of tissue and organs did not change compared to a living dolphin because those dolphins were dissected immediately after they died. Average total body burden of the 6 specimens, which is represented as the sum of burden in blubber, liver and muscle, was 850, 460, 130, 6.0, 6.6, 7.6, and 8.6 mg for DDTs, PCBs, CHLs, HCHs, HCB, PBDEs, and HBCDs, respectively. As shown in Fig. 3, more than 90% of total body burden of all the target compounds accumulated in blubber, suggesting blubber is a suitable indicator to investigate the accumulation and fate of these halogenated contaminants including PBDEs and HBCDs.

3.3. Congener and isomer composition of BFRs

The relative contribution of each congener or isomer to total PBDEs or HBCDs in each tissue is shown in Fig. 4. Of the fourteen PBDE congeners analyzed, a total of thirteen congeners from di- to deca-BDE were identified in all the tissues of striped dolphins. Contributions of BDE-154 and -153 to total PBDEs were higher than those generally reported in biological samples (Ikonomou et al., 2002). It is well known that BDE-47 is usually the predominant congener in higher trophic animals followed by BDE-100, -99, -153 and -154. One of the reasons for higher contribution of BDE-154 and -153 could be the fact that octa-BDE mixture was used more extensively than tetra-BDE mixture (similar in composition to penta-BDE mixture) in Japan. Another possible reason could be the formation of BDE-154 and -153 during debromination of BDE-209, which is the predominant congener of the only in use commercial PBDE mixture in Japan, i.e., Deca-BDE mixture. Only trace levels of BDE-209 were found in the dolphins of this study because of its extremely low bioavailability, consistent with other reports (Ramu et al., 2005, 2006; Kajiwara et al., 2008).

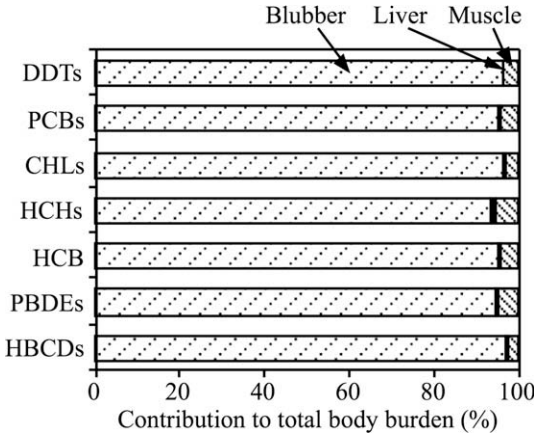

Fig. 3. Percentage contribution of organohalogen compounds in each tissue to total body burden in striped dolphins stranded at Gogo-shima, Japan in 2003.

In the case of HBCDs, α-HBCD isomer was predominant among the three HBCD isomers. Most of the previous studies, which analyzed isomeric profiles of HBCDs in biological samples from higher trophic levels, also reported a higher proportion of α-HBCD as summarized in a review paper (Covaci et al., 2006). This is probably due to the fact that α-HBCD isomer is more persistent and bioaccumulative than the other two isomers. Although γ-HBCD is the predominant isomer in the technical mixture, high proportion of α-HBCD in house dust and electrical appliances was reported (Abdallah Mohamed et al., 2008). Our results were consistent with other reports which showed that higher trophic wildlife do not accumulate γ-HBCD (Morris et al., 2004; Betts, 2005; Zegers et al., 2005; Isobe et al., 2007), possibly due to higher affinity to particles and lower bioavailability and persistency of γ-HBCD than α-HBCD.

3.4. Temporal trends of organohalogen contaminants

To evaluate the temporal trends of contamination by OCs and BFRs, blubber samples of striped dolphins collected in 1978, 1986, and 1992 and archived in *es*-BANK of Ehime University were analyzed. To avoid the effect of gender and age, only adult male specimens were selected. The temporal trends of concentrations for PBDEs, HBCDs, DDTs, PCBs, HCHs, and CHLs are shown in Table 1 and Fig. 5. Concentrations of PCBs, DDTs and HCHs did not change significantly during this period. Relatively steady levels of PCBs suggest continuing discharge of these contaminants from PCB containing equipments in use in the surrounding countries and also possibly in Japan. No significant decreasing trends in DDTs and HCHs concentrations were observed possibly implying use of these chemicals in Asian developing countries. Chlordanes, which had been used as a termiticide for mostly wooden houses in Japan, showed increasing trend (statistically significant with Spearman's rank test, $p < 0.01$), possibly due to still existing residues of the chemicals in the environment even after the ban on these chemicals.

On the other hand, remarkable increasing trends of PBDEs and HBCDs (statistically significant with Spearman's rank test, $p < 0.01$) were observed. Concentrations of PBDEs increased continuously from the late 1970s to 2003. The increasing rate from 1992 to 2003 was slower than that from 1985 to 1992, probably reflecting the effect of voluntary termination of tetra- and octa-BDE mixtures in Japan. This phenomenon was also observed in sediment cores collected from Tokyo Bay (Minh et al., 2007). In addition, the percentage contribution of BDE-154 to total PBDEs concentration continuously increased from 9.5% in 1978 to 24% in 2003, suggesting increasing contribution of octa-BDE mixture or debromination of higher brominated congeners. Concentrations of HBCDs also increased significantly throughout this period ($p < 0.01$). HBCDs have been extensively used since the mid-1980s and consumption is still growing, and subsequent input into the environment is probably increasing. Levels of HBCDs in the striped dolphins showed a dramatically increasing trend after 1992, whereas no apparent increasing trend was observed in finless porpoises from Hong Kong in the previous study (Isobe et al., 2007). This could be the result of differences in the amount of usage of HBCDs in these two regions. In recent years, HBCDs are extensively used as flame retardants in Japan, whereas its usage might be still not so common in other Asian countries. In the case of finless

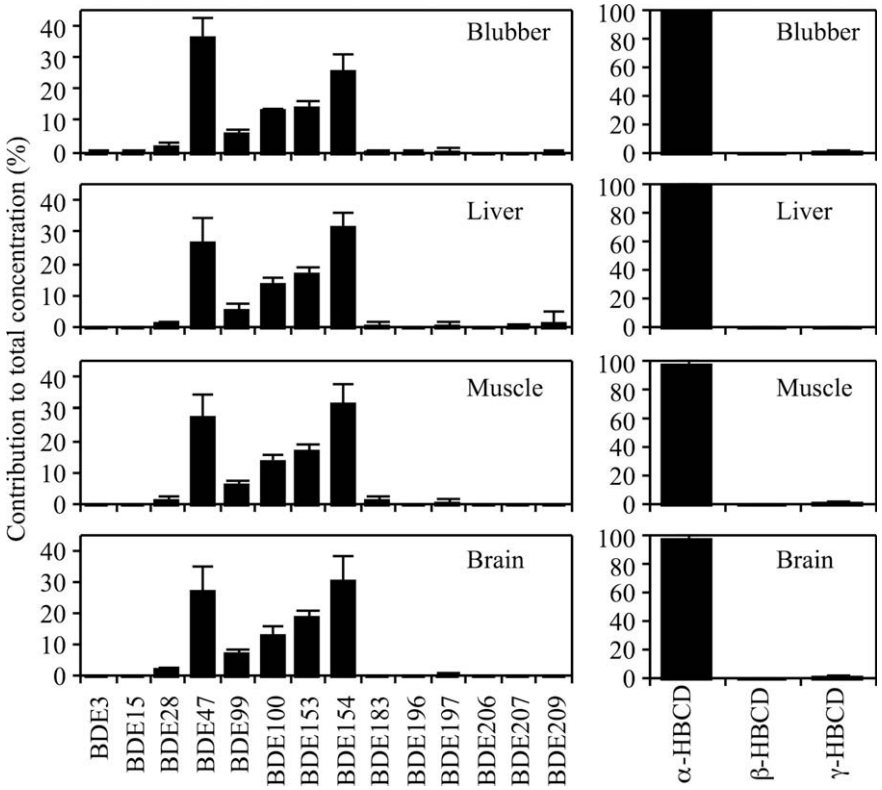

Fig. 4. Profiles of PBDE congeners and HBCD isomers in blubber, liver, muscle and brain of striped dolphins stranded at Gogo-shima, Japan in 2003.

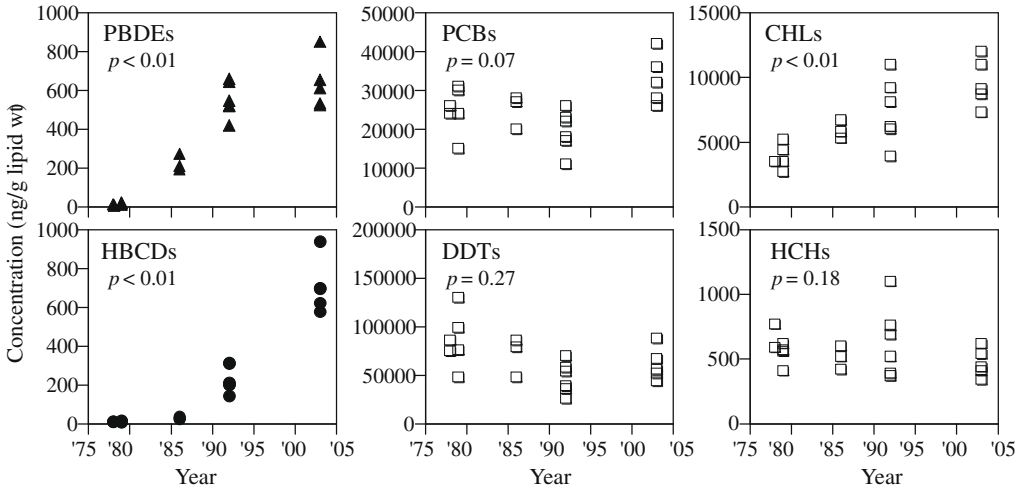

Fig. 5. Temporal trends in concentrations of organohalogen compounds in blubber of striped dolphins from Japan.

porpoises from Hong Kong, concentrations of HBCDs were one order of magnitude lower than those of PBDEs, suggesting relatively less usage of HBCDs in this region. On the other hand, concentrations of HBCDs in the striped dolphins of the present study were almost at the same level as (not significantly but slightly higher than) those of PBDEs in 2003, whereas levels of PBDEs in 1992 were twice that of HBCDs. This is consistent with the fact that HBCDs have been extensively used since the mid-1980s in Japan, while consumption of PBDEs decreased since the early 1990s. Throughout the study period, isomeric composition of HBCDs showed relatively stable proportion of α-HBCD isomer, which was greater than 99%. Temporal variation of HBCDs in biological samples has also been reported elsewhere. Concentrations of PBDEs and HBCDs in blubber samples from stranded male California sea lions (*Zalophus californianus*) demonstrated an increasing trend from 1993 to 2003 (Stapleton et al., 2006). Study on guillemot (*Uria algae*) eggs from the Baltic Sea sampled between 1969 and 2001 showed an increasing trend in concentrations of HBCDs throughout the study period with no significant change during the last 10-yrs, while PBDEs showed a decreasing trend after the late-1980s (Sellstrom et al., 2003). On the other hand, in peregrine falcon (*Falco peregrinus*) eggs from South Greenland, levels of HBCDs decreased between 1986 and 2003, whereas that of PBDEs increased significantly (Vorkamp et al., 2005). The general temporal trend of HBCDs in wildlife is still not clear as it shows either an increasing or a decreasing trend, depending upon various environmental and biological parameters.

However, the present increasing usage of HBCDs in the world may further accelerate contamination of the ecosystems. It has already been reported that the differences in regional production and historical application of PBDEs and HBCDs lead to different temporal trends in the environment and biota (Covaci et al., 2006). Therefore, continuous and detailed monitoring surveys are warranted to investigate the environmental distribution and fate of HBCDs.

As the body burdens of BFRs in marine mammals increase, it is imperative that further research must be directed towards understanding the potential toxicological outcomes of these BFRs and their metabolites for risk assessment.

Acknowledgments

Dissection of stranded dolphins was conducted with the cooperation of many students, scientists in other institutes and volunteers. This research was partly supported by Grant-in-Aid for Scientific Research (S: 20221003, B: 18310046), for Young Scientists (B: 19780239), and Global COE program of the Japanese Ministry of Education, Science, Sports, Culture and Technology and Japan Society for the Promotion of Science. The award of JSPS Postdoctoral Fellowship to K. Ramu (P07153) is acknowledged.

References

Abdallah Mohamed, A.E., Harrad, S., Ibarra, C., Diamond, M., Melymuk, L., Robson, M., Covaci, A., 2008. Hexabromocyclododecanes in indoor dust from Canada, the United Kingdom, and the United States. Environ. Sci. Technol. 42, 459–464.

Betts, K., 2005. More clues to HBCD isomer mystery. Environ. Sci. Technol. 39, 146A–147A.

Borrell, A., Aguilar, A., 2005. Mother-calf transfer of organochlorine compounds in the common dolphin (*Delphinus delphis*). Bull. Environ. Contam. Toxicol. 75, 149–156.

BSEF, 2005. <http://www.bsef.com/>.

Covaci, A., Gerecke, A.C., Law, R.J., Voorspoels, S., Kohler, M., Heeb, N.V., Leslie, H., Allchin, C.R., de Boer, J., 2006. Hexabromocyclododecanes (HBCDs) in the environment and humans: a review. Environ. Sci. Technol. 40, 3679–3688.

de Swart, R.L., Ross, P.S., Timmerman, H.H., Hijman, W.C., de Ruiter, E.M., Liem, A.K., Brouwer, A., van Loveren, H., Reijnders, P.J., Vos, J.G., Osterhaus, A.D.M.E., 1995. Short term fasting does not aggravate immunosuppression in harbour seals (*Phoca vitulina*) with high body burdens of organochlorines. Chemosphere 31, 4289–4306.

de Wit, C.A., 2002. An overview of brominated flame retardants in the environment. Chemosphere 46, 583–624.

Hites, R.A., 2004. Polybrominated diphenyl ethers in the environment and in people: a meta-analysis of concentrations. Environ. Sci. Technol. 38, 945–956.

Ikonomou, M.G., Rayne, S., Fischer, M., Fernandez, M.P., Cretney, W., 2002. Occurrence and congener profiles of polybrominated diphenyl ethers (PBDEs) in environmental samples from coastal British Columbia, Canada. Chemosphere 46, 649–663.

Isobe, T., Ramu, K., Kajiwara, N., Takahashi, S., Lam, P.K., Jefferson, T.A., Zhou, K., Tanabe, S., 2007. Isomer specific determination of hexabromocyclododecanes (HBCDs) in small cetaceans from the South China Sea-Levels and temporal variation. Mar. Pollut. Bull. 54, 1139–1145.

Johnson-Restrepo, B., Kannan, K., Addink, R., Adams, D.H., 2005. Polybrominated diphenyl ethers and polychlorinated biphenyls in a marine food web of coastal Florida. Environ. Sci. Technol. 39, 8243–8250.

Kajiwara, N., Kamikawa, S., Ramu, K., Ueno, D., Yamada, T.K., Subramanian, A., Lam, P.K., Jefferson, T.A., Prudente, M., Chung, K.H., Tanabe, S., 2006a. Geographical distribution of polybrominated diphenyl ethers (PBDEs) and organochlorines in small cetaceans from Asian waters. Chemosphere 64, 287–295.

Kajiwara, N., Kunisue, T., Kamikawa, S., Ochi, Y., Yano, S., Tanabe, S., 2006b. Organohalogen and organotin compounds in killer whales mass stranded in the Shiretoko Peninsula, Hokkaido, Japan. Mar. Pollut. Bull. 52, 1066–1076.

Kajiwara, N., Kamikawa, S., Amano, M., Hayano, A., Yamada, T.K., Miyazaki, N., Tanabe, S., 2008. Polybrominated diphenyl ethers (PBDEs) and organochlorines in melon-headed whales, *Peponocephala electra*, mass stranded along the Japanese coasts: maternal transfer and temporal trend. Environ. Pollut. 156, 106–114.

Kannan, K., Tanabe, S., Borrell, A., Aguilar, A., Focardi, S., Tatsukawa, R., 1993. Isomer-specific analysis and toxic evaluation of polychlorinated biphenyls in striped dolphins affected by an epizootic in the western Mediterranean Sea. Arch. Environ. Contam. Toxicol. 25, 227–233.

Kannan, K., Blankenship, A.L., Jones, P.D., Giesy, J.P., 2000. Toxicity reference values for the toxic effects of polychlorinated biphenyls to aquatic mammals. Hum. Ecol. Risk Assess. 6, 181–201.

Kunisue, T., Nakanishi, S., Watanabe, M., Abe, T., Nakatsu, S., Kawauchi, S., Sano, A., Horii, A., Kano, Y., Tanabe, S., 2005. Contamination status and accumulation features of persistent organochlorines in pet dogs and cats from Japan. Environ. Pollut. 136, 465–476.

Law, R.J., Bersuder, P., Allchin, C.R., Barry, J., 2006. Levels of the flame retardants hexabromocyclododecane and tetrabromobisphenol A in the blubber of harbor porpoises (*Phocoena phocoena*) stranded or by caught in the U.K., with evidence for an increase in HBCD concentrations in recent years. Environ. Sci. Technol. 40, 2177–2183.

Lebeuf, M., Gouteux, B., Measures, L., Trottier, S., 2004. Levels and temporal trends (1988–1999) of polybrominated diphenyl ethers in beluga whales (*Delphinapterus leucas*) from the St. Lawrence Estuary, Canada. Environ. Sci. Technol. 38, 2971–2977.

Litz, J.A., Garrison, L.P., Fieber, L.A., Martinez, A., Contillo, J.P., Kucklick, J.R., 2007. Fine-scale spatial variation of persistent organic pollutants in bottlenose dolphins (*Tursiops truncatus*) in Biscayne Bay, Florida. Environ. Sci. Technol. 41, 7222–7228.

Makara, M., Shimada, A., Tajima, Y., Yamada, T.K., Tanabe, S., 2006. Pathological study of the mass stranded striped dolphins (*Stenella coeruleoalba*), In: Proceedings of 17th The Cetology Study Group of Japan, Shinjuku, Japan, p. 18.

McKinney, M.A., De Guise, S., Martineau, D., Beland, P., Lebeuf, M., Letcher, R.J., 2006. Organohalogen contaminants and metabolites in beluga whale (*Delphinapterus leucas*) liver from two Canadian populations. Environ. Toxicol. Chem. 25, 1246–1257.

Meerts, I.A., van Zanden, J.J., Luijks, E.A., van Leeuwen-Bol, I., Marsh, G., Jakobsson, E., Bergman, A., Brouwer, A., 2000. Potent competitive interactions of some brominated flame retardants and related compounds with human transthyretin in vitro. Toxicol. Sci. 56, 95–104.

Minh, N.H., Isobe, T., Ueno, D., Matsumoto, K., Mine, M., Kajiwara, N., Takahashi, S., Tanabe, S., 2007. Spatial distribution and vertical profile of polybrominated diphenyl ethers and hexabromocyclododecanes in sediment core from Tokyo Bay, Japan. Environ. Pollut. 148, 409–417.

Morris, S., Allchin, C.R., Zegers, B.N., Haftka, J.J., Boon, J.P., Belpaire, C., Leonards, P.E., van Leeuwen, S.P., de Boer, J., 2004. Distribution and fate of HBCD and TBBPA brominated flame retardants in North Sea estuaries and aquatic food webs. Environ. Sci. Technol. 38, 5497–5504.

Muir, D.C., Backus, S., Derocher, A.E., Dietz, R., Evans, T.J., Gabrielsen, G.W., Nagy, J., Norstrom, R.J., Sonne, C., Stirling, I., Taylor, M.K., Letcher, R.J., 2006. Brominated flame retardants in polar bears (*Ursus maritimus*) from Alaska, the Canadian Arctic, East Greenland, and Svalbard. Environ. Sci. Technol. 40, 449–455.

Pettersson, A., van Bavel, B., Engwall, M., Jimenez, B., 2004. Polybrominated diphenylethers and methoxylated tetrabromodiphenylethers in cetaceans from the Mediterranean Sea. Arch. Environ. Contam. Toxicol. 47, 542–550.

Rahman, F., Langford, K.H., Scrimshaw, M.D., Lester, J.N., 2001. Polybrominated diphenyl ether (PBDE) flame retardants. Sci. Total Environ. 275, 1–17.

Ramu, K., Kajiwara, N., Tanabe, S., Lam, P.K., Jefferson, T.A., 2005. Polybrominated diphenyl ethers (PBDEs) and organochlorines in small cetaceans from Hong Kong waters: levels, profiles and distribution. Mar. Pollut. Bull. 51, 669–676.

Ramu, K., Kajiwara, N., Lam, P.K., Jefferson, T.A., Zhou, K., Tanabe, S., 2006. Temporal variation and biomagnification of organohalogen compounds in finless porpoises (*Neophocaena phocaenoides*) from the South China Sea. Environ. Pollut. 144, 516–523.

Sellstrom, U., Bignert, A., Kierkegaard, A., Haggberg, L., de Wit, C.A., Olsson, M., Jansson, B., 2003. Temporal trend studies on tetra- and pentabrominated diphenyl ethers and hexabromocyclododecane in guillemot egg from the Baltic Sea. Environ. Sci. Technol. 37, 5496–5501.

Stapleton, H.M., Dodder, N.G., Kucklick, J.R., Reddy, C.M., Schantz, M.M., Becker, P.R., Gulland, F., Porter, B.J., Wise, S.A., 2006. Determination of HBCD, PBDEs and MeO-BDEs in California sea lions (*Zalophus californianus*) stranded between 1993 and 2003. Mar. Pollut. Bull. 52, 522–531.

Tanabe, S., 2006. Environmental specimen bank in Ehime University (es-BANK), Japan for global monitoring. J. Environ. Monit. 8, 782–790.

Tanabe, S., Tatsukawa, R., Tanaka, H., Maruyama, K., Miyazalki, N., Fujiyama, T., 1981. Distribution and total burdens of chlorinated hydrocarbons in bodies of striped dolphins (*Stenella coeruleoalba*). Agr. Biol. Chem. 45, 2569–2578.

Tanabe, S., Ramu, K., Isobe, T., Takahashi, S., 2008. Brominated flame retardants in the environment of Asia–Pacific: an overview of spatial and temporal trends. J. Environ. Monit. 10, 188–197.

Tomy, G.T., Halldorson, T., Danell, R., Law, K., Arsenault, G., Alaee, M., Macinnis, G., Marvin, C.H., 2005. Refinements to the diastereoisomer-specific method for the analysis of hexabromocyclododecane. Rapid Commun. Mass Spectrom. 19, 2819–2826.

Ueno, D., Kajiwara, N., Tanaka, H., Subramanian, A., Fillmann, G., Lam, P.K., Zheng, G.J., Muchitar, M., Razak, H., Prudente, M., Chung, K.H., Tanabe, S., 2004. Global pollution monitoring of polybrominated diphenyl ethers using skipjack tuna as a bioindicator. Environ. Sci. Technol. 38, 2312–2316.

Vorkamp, K., Thomsen, M., Falk, K., Leslie, H., Moller, S., Sorensen, P.B., 2005. Temporal development of brominated flame retardants in peregrine Falcon (*Falco peregrinus*) eggs from South Greenland (1986–2003). Environ. Sci. Technol. 39, 8199–8206.

Zegers, B.N., Mets, A., Van Bommel, R., Minkenberg, C., Hamers, T., Kamstra, J.H., Pierce, G.J., Boon, J.P., 2005. Levels of hexabromocyclododecane in harbor porpoises and common dolphins from western European seas, with evidence for stereoisomer-specific biotransformation by cytochrome p450. Environ. Sci. Technol. 39, 2095–2100.